Macroplastic Debris Transfer in Rivers: A Travel Distance Approach

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ABSTRACT

Plastic accumulation in the marine environment is a major concern given the harmful effects and longevity of plastics at sea. To understand and mitigate this accumulation, an understanding of plastic transport in rivers and estimates of riverine plastic flux are required (since rivers are potentially significant transport pathways of plastic debris to the oceans). Existing methods to investigate plastic transport in rivers and to estimate riverine plastic flux (from field measurements and modelling efforts) are, however, very crude, subject to a high degree of uncertainty and (perhaps most importantly) fail to consider the processes controlling displacement. Here, a new and fundamentally different approach to investigating plastic transport in rivers is presented, which considers the travel distances of individual items of macroplastic debris (and thus, the processes controlling displacement). This approach combines an experimental component, in which macroplastic tracers were tracked in a small river reach, with the construction of a numerical model, which is intended to act as an interpretive conceptual framework. Macroplastic travel distances were found to be low and variable and the travel distance distribution was systematically controlled by the location and characteristics of ‘trapping points’ (particularly overhanging trees and meander bends). The numerical model was based on ‘the probability of trapping’ and described particle displacement distributions reasonably well. Although significant knowledge gaps remain (which require further investigation), the tracer experiment results and outputs from the numerical model demonstrated the feasibility of using a travel distance approach to understand and predict plastic transfer in rivers. This approach also has the potential to improve riverine plastic flux estimates.
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# TABLE OF CONTENTS

Abstract ........................................................................................................................................... ii  
Acknowledgements ......................................................................................................................... iii  
Table of Contents ............................................................................................................................. iv  
List of Figures .................................................................................................................................. vi  
List of Tables .................................................................................................................................... vii  
List of Abbreviations ....................................................................................................................... viii  
Notation List ..................................................................................................................................... viii  

## 1 Introduction and Literature Review ......................................................................................... 1  
1.1 Introduction ................................................................................................................................. 1  
1.2 Plastic Production, Use, Effects, Transport and Accumulation ............................................... 1  
1.3 Estimating Global Riverine Plastic Flux with Models ............................................................. 3  
1.4 Estimating Riverine Plastic Flux with Field Measurements ............................................... 7  
1.5 Evaluation of Methods Used to Estimate Riverine Plastic Flux ..........................................., 10  
1.6 Macroplastic Debris Displacement in Rivers ............................................................................ 12  
  1.6.1 Sources of Macroplastic Debris in River Channels ......................................................... 12  
  1.6.2 Transport and Degradation of Macroplastic Debris in River Channels ...................... 14  
  1.6.3 Accumulation of Macroplastic Debris at ‘Trapping Points’ ............................................. 16  
  1.6.4 Influence of River Stage on Macroplastic Debris Transport ......................................... 21  
1.7 Chapter Summary, Travel Distance Approach and Justification .......................................... 23  
1.8 Research Aims and Questions ...................................................................................................... 24  

## 2 Methodology ............................................................................................................................... 25  
2.1 Introduction ................................................................................................................................. 25  
2.2 Study Area .................................................................................................................................. 25  
2.3 Tracer Experiments .................................................................................................................... 30  
  2.3.1 Determining the Travel Distance Distribution of Macroplastic Debris ...................... 30  
  2.3.2 Pilot Experiments ............................................................................................................... 30  
  2.3.3 Experimental Design ......................................................................................................... 31  
  2.3.4 Controls on the Travel Distance Distribution of Macroplastic Debris ...................... 32  
  2.3.5 Trap Effectiveness ............................................................................................................. 33  
2.4 Modelling the Transfer of Macroplastic Debris Through a River Catchment ...................... 33  
  2.4.1 Model Introduction ............................................................................................................. 33  
  2.4.2 Probability of Trapping ...................................................................................................... 34  
  2.4.3 Model Framework ............................................................................................................. 37  
  2.4.4 Model Outputs ................................................................................................................. 40  
  2.4.5 Model Calibration and Comparison to Field Data ....................................................... 41
2.4.6 Sensitivity Analysis ................................................................. 42
2.5 Chapter Summary ....................................................................... 42

3 Tracer Experiment Results and Analysis ............................................ 43
  3.1 Introduction .............................................................................. 43
  3.2 Travel Distance Distribution of Macroplastic Debris ....................... 43
  3.3 Controls on the Travel Distance Distribution of Macroplastic Debris ... 46
  3.4 Trap Effectiveness ................................................................... 48
  3.5 Chapter Summary ....................................................................... 50

4 Modelling Exercise Results and Analysis .......................................... 51
  4.1 Introduction .............................................................................. 51
  4.2 Model Calibration ..................................................................... 51
  4.3 Sensitivity Analysis ................................................................. 53
  4.4 Model Outputs ......................................................................... 55
  4.5 Chapter Summary ....................................................................... 58

5 General Discussion and Conclusions ............................................... 59
  5.1 Introduction .............................................................................. 59
  5.2 Travel Distance Distribution of Macroplastic Debris ....................... 59
  5.3 Modelling Plastic Transport Using a Travel Distance Approach .......... 60
    5.3.1 Evaluation of Model Outputs .................................................. 60
    5.3.2 Model Sensitivity .................................................................. 62
    5.3.3 Model Use and Potential ......................................................... 64
  5.4 Variability and Understanding the Controls on Travel Distance .......... 64
    5.4.1 Variability ............................................................................ 64
    5.4.2 Understanding the Controls on Travel Distance ......................... 65
  5.5 Limitations ............................................................................... 66
  5.6 Validation and Scalability .......................................................... 67
  5.7 Chapter Summary and Thesis Conclusions ...................................... 69

References ..................................................................................... 70
LIST OF FIGURES

Figure 1.1 - Mass of river plastic flowing into the oceans in tonnes per year .............. 4

Figure 1.2 – Comparison between modelled and measured estimates of riverine plastic flux from Lebreton et al. (2017) ........................................................................................................ 5

Figure 1.3 – Model framework from Meijer et al. (2019) to estimate riverine plastic flux ........................................................................................................................................... 6

Figure 1.4 – The estimated distribution of plastic inputs to the ocean across rivers from Meijer et al. (2019), Lebreton et al. (2017) and Schmidt et al. (2017) ................................................................. 7

Figure 1.5 – Four-step method proposed by Van Emmerik et al. (2018) to estimate riverine plastic flux ........................................................................................................................................... 8

Figure 1.6 – Processes controlling the fate of macroplastic debris in rivers ............... 13

Figure 1.7 – Transport mechanism of macroplastic debris and degradation processes acting on macroplastic debris in rivers ........................................................................................................ 15

Figure 1.8 – Schematic illustration of the transport of macroplastic debris between sites of temporary storage ................................................................................................................ 16

Figure 1.9 – Flow separation around a boulder on the bed of a river, around an obstruction and at a meander bend. Also the fluid velocity profile parallel to a boundary at a meander bend ........................................................................................................................................... 18

Figure 1.10 - Macroplastic debris tracer items trapped in overhanging vegetation and emergent aquatic vegetation ................................................................................................................ 19

Figure 1.11 – Classifications of aquatic channel vegetation ........................................ 19

Figure 1.12 – Theoretical travel distance distribution where the trapping of macroplastic debris is uniform for each sub-reach ........................................................................................................ 24

Figure 2.1 – Location of the study reach ........................................................................ 26

Figure 2.2 – Photographs of the study reach ................................................................. 28

Figure 2.3 – River channel adjustment following regrade operations along the River Sence, Leicester ................................................................................................................................. 29

Figure 2.4 – The travel distance distributions of three pilot studies ......................... 31

Figure 2.5 – Schematic illustration of the transfer of macroplastic debris downstream from cell to cell ................................................................................................................................. 35
Figure 2.6 – Assumed relationship between (a) the probability of trapping in a flow separation zone and the channel sinuosity index, and (b) the probability of trapping along channel banks and channel width ................................................................. 36

Figure 2.7 – Numerical model decision network ................................................................. 39

Figure 3.1 – Plastic bottle travel distance distributions from six replicate experiments and the combined data ........................................................................................................... 44

Figure 3.2 – Combined travel distance distribution of plastic bottle travel distances ... 45

Figure 3.3 – Plastic bottle travel distance distribution, colour coded according to the presence or absence of overhanging tree and meander bend traps ........................................ 47

Figure 3.4 – Difference in the number of bottles trapped in 10 m sub-reaches between repeat surveys in the same replicate experiment ................................................................. 49

Figure 4.1 – Travel distance distribution for the optimum combination of calibration parameters.......................................................................................................................... 52

Figure 4.2 – Cumulative distribution of predicted and observed plastic bottle travel distances after 24 hours ........................................................................................................ 53

Figure 4.3 – Results from a one-at-a-time sensitivity analysis, showing the change in model goodness of fit ........................................................................................................ 54

Figure 4.4 – Modelled travel distance distributions for second, fifth and tenth time iterations .......................................................................................................................... 55

Figure 4.5 – Example residence time distribution ................................................................ 56

Figure 4.6 – Bottle storage distribution for the diffuse-injection of plastic bottles ....... 57

LIST OF TABLES

Table 1.1 – Definitions of aquatic and riparian vegetation classifications ...................... 20

Table 1.2 – Changes in river width and depth change at a point depending on cross-sectional channel geometry ................................................................. 22

Table 3.1 – Location of overhanging tree and meander bend traps along the study reach ......................................................................................................................... 46
LIST OF ABBREVIATIONS

AD Anderson-Darling
CDF(s) Cumulative Distribution Function(s)
D/S Downstream
KS Kolmogorov-Smirnov
MMPW Mismanaged Plastic Waste
EA Environment Agency
SEPA Scottish Environment Protection Agency

NOTATION LIST

\( a \) Calibration parameter in Equation 2.2
\( b \) Calibration parameter in Equation 2.4
\( B \) Bottle
\( c \) Calibration parameter in Equation 2.4
\( C \) Channel length (m)
\( CV \) Coefficient of variation
\( D \) Downvalley length (m)
\( D_{\text{Stat}}, D_{\text{Crit}} \) Kolmogorov-Smirnov test statistic and critical value
\( i \) Reach index
\( its \) Time iteration
\( j \) Reach index
\( n \) Number of retrieved bottles
\( N \) Number of reaches
\( n_{\text{bottles}} \) Number of bottles
\( n_{\text{its}} \) Number of time iterations
\( n_{\text{OUT}} \) Plastic bottle flux
\( p \) P-value
\( p_{\text{enter}}(i) \) Probability of plastic bottles entering reach \( i \)
\( p(CB) \) Probability of trapping along channel banks
\( p(E) \) Probability of MMPW emission to the ocean (Meijer et al., 2019)
\( p(FS) \) Probability of trapping in a flow separation zone
\( p(i) \) Probability of trapping in reach \( i \)
\( p(j) \) Probability of trapping in reach \( j \)
\( p(M) \) Probability of mobilisation (Meijer et al., 2019)
\( p(R) \) Probability of transport from land to the river (Meijer et al., 2019)
\( p(O) \) Probability of transport from river to the ocean (Meijer et al., 2019)
\( p(T) \) Probability of trapping
PS1, 2 etc. Pilot Study 1, 2 etc.
\( p(V) \) Probability of trapping in overhanging vegetation
\( q(i) \) Probability of being freed from a trapped state in reach \( i \)
\( Q, Q_{bf} \) Discharge and bankfull discharge (m\(^3\)s\(^{-1}\))
R1, 2 etc. Replicate 1, 2 etc.
\( R^2 \) Coefficient of determination
\( S \) Sinuosity Index
\( w, w_{bf}, w_0 \) River channel, bankfull and minimum river channel width (m)
\( xx \) Randomly generated number between zero and one
\( \bar{x} \) Mean travel distance (m)
\( \alpha \) Significance level
\( \sigma \) Standard deviation of the number of plastic bottles trapped in each sub-reach between each replicate
\( \chi^2 \) Chi-squared
1 INTRODUCTION AND LITERATURE REVIEW

1.1 Introduction

Plastic accumulation in the environment (and particularly in the ocean) is a major global concern. The relative contribution of different potential sources of plastic to the oceans remains uncertain but rivers have been recognised as one major transport pathway for marine plastic debris. Despite an increasing body of research seeking to estimate river to ocean plastic fluxes, many aspects of fluvial plastic behaviour are poorly understood. Estimates of riverine plastic flux have been made from modelling efforts and field measurements, but most are subject to a high degree of uncertainty and fail to consider the processes controlling macroplastic debris emission, displacement, storage or degradation. This study attempts to close some of these knowledge gaps by explicitly investigating macroplastic debris displacement empirically and attempting to characterise and model travel distances of individual items. It should be noted that the transport of microplastic particles is also an important issue and significant knowledge gaps also exist for these materials. However, their fate and behaviour differs markedly from that of macroplastic and is, thus, beyond the scope of this thesis.

1.2 Plastic Production, Use, Effects, Transport and Accumulation

Large-scale commercial plastic production began in the 1950s and has increased significantly since. By 2018, global plastic production reached 359 million tonnes per year and continued growth is expected (Geyer et al., 2017; Lebreton and Andrade, 2019; Plastics Europe, 2019). These trends are unsurprising given the considerable benefits and desirable qualities of plastics (Andrady and Neal, 2009). Plastic products are low-cost, lightweight, tough and can be easily manufactured to a range of shapes, sizes and densities (Lebreton and Andrade, 2019). They have a wide variety of uses – the largest being for consumer packaging. Approximately 42% of annual resin production is used for packaging (Geyer et al., 2017). Plastic is also used in construction, textiles, transport, and electronics (Lebreton and Andrade, 2019; Van Emmerik and Schwarz, 2020). In many industries plastic has replaced ‘traditional’ materials (e.g. metal and wood in construction and natural fibres in textiles; Lebreton and Andrade, 2019). A significant benefit of many commonly used plastic whilst they are in use is that they degrade only over very long time periods. However, once they have been discarded, this longevity is a significant problem, resulting in accumulation in landfill and the natural environment (Barnes et al., 2009; Geyer et al., 2017). Some plastic are estimated to take hundreds to thousands of years to completely degrade (Barnes et al., 2009). This issue is exacerbated by the low cost of plastic production which means that many products are
designed as single use items. Although awareness of this issue has increased in recent years, single-use plastics remain ubiquitous and recycling efforts remain low (only 9% of plastics ever made are recycled; Geyer et al., 2017). It is likely, therefore, that plastic accumulation in many environmental systems is likely to continue for at least the next few years (Lebreton and Andrady, 2019; Van Emmerik and Schwarz, 2020).

Contamination of the environment, particularly the marine environment, with mismanaged plastic waste (MMPW; plastic debris that is either littered or inadequately disposed) has been recognised as a major global concern (Jambeck et al., 2015; Kooi et al., 2018). This concern has arisen, in part, due to the discovery that marine plastic pollution is ubiquitous (with evidence of plastic debris found in every major ocean basin; Barnes et al., 2009). The durable nature of plastics in the environment mean they have long-term effects (Welden, 2020). Potentially harmful effects of plastic include ingestion of plastic by organisms and associated physical and toxicological damage, and physical entanglement of fauna in plastic debris (Eriksen et al., 2014; Oehlmann et al., 2009). In addition, plastic pollution is fundamentally aesthetically unacceptable (Sheavly and Register, 2007). Large plastic debris (> 5 mm) is often referred to as macroplastic. This debris can breakdown in the environment into so called microplastic particles (particles < 5 mm). Microplastic particles can be more easily ingested by animals, where they may have harmful effects and they can be transferred into human food chains (Barboza et al., 2018; Wright and Kelly, 2017). However, the extent to which such exposure represents a significant risk for human health remains uncertain. In rivers, macroplastic debris can contribute to the physical clogging of culverts, pipes and open channels, which can increase the likelihood of urban flooding (Van Emmerik and Schwarz, 2020).

Although there are numerous marine-based sources of plastic, land-based emissions (from coasts and via rivers) are believed to be the dominant input of plastic into the oceans (Lebreton et al., 2017; Schmidt et al., 2017). Globally, it is estimated that 0.4-4 million metric tonnes (Mt) of plastic is emitted to the oceans each year by rivers (Lebreton et al., 2017; Schmidt et al., 2017). However, despite the increasing recognition of the importance of land-based sources for ocean plastic debris, surprisingly little attention has been given to understanding plastic litter transport in rivers (Jambeck et al., 2015; Lechner et al., 2014; Rech et al., 2014). Instead, most plastic research continues to focus on the marine environment (Blettler et al., 2018). Riverine plastic flux has been estimated from field measurements and from modelling efforts. Models to estimate riverine plastic flux are based on MMPW generation and calibrated against field measurements (see Section 1.3; below).
### 1.3 Estimating Global Riverine Plastic Flux with Models

Jambeck et al. (2015) developed one of the first global-scale frameworks to estimate terrestrial plastic waste delivery to the oceans. This framework was based on estimates of the amount of MMPW generated < 50 km from the coast and the proportion of MMPW that is converted to marine plastic debris. MMPW generated < 50 km from the coast was calculated from the product of MMPW per capita and the population living within 50 km of the coast, for each country. MMPW is calculated from the mass of waste generated per capita, the proportion of that waste which is plastic and the proportion of that waste which is inadequately disposed of (see Hoornweg and Bhada-Tata, 2012; Lebreton and Andrady, 2019). Jambeck et al. (2015) assumed that between 15% (lower limit) and 40% (upper limit) of MMPW generated within 50 km of the coast is converted to marine plastic debris. These rates are based on one study of San Francisco water quality where plastic uncaptured by streetsweepers and storm drains was assumed to enter the ocean. However, the mechanisms for the assumed conversion of MMPW to marine plastic debris are not explicitly given and could include fluvial transport via rivers, aeolian transport by wind and transport off the coast with tides. In addition, Jambeck et al. (2015) assume that plastic waste emitted to the stream network more than 50 km from the coast is not transported to the oceans. Based on these assumptions, Jambeck et al. (2015) estimate that 4.8-12.7 Mt of land-based plastic enters the world’s oceans each year. They also identify China and the wider Southeast Asia region as the greatest emitters of terrestrial plastic to the ocean globally.

Three studies have estimated global plastic transfers from rivers to the oceans using catchment-based models: Lebreton et al. (2017), Meijer et al. (2019) and Schmidt et al. (2017). In all three cases, global riverine plastic flux is estimated from the sum of fluxes from individual catchments. Lebreton et al. (2017), for example, estimated the flux from 40,760 catchments worldwide. They calculated catchment plastic flux as the product of the MMPW generated in the catchment and monthly average catchment runoff, calibrated using plastic fluxes calculated from measured concentrations and observed discharge data for 13 rivers reported in seven studies. Figure 1.1 shows the annual flux of plastic from land to the oceans from individual rivers predicted by Lebreton et al. (2017). The total predicted global riverine flux was 1.15-2.41 Mt yr⁻¹. Again, the top polluting rivers are mostly located in Asia, with the top being the Yangtze, the Ganges and the Xi. A strong relationship was reported between modelled estimates of riverine plastic flux and plastic fluxes calculated from measured concentrations reported in the literature ($R^2 = 0.93$; see Figure 1.2). There is an urgent need for more field measurements to calibrate and validate new and existing models to estimate river plastic flux.
flux (González-Fernández and Hanke, 2017; Van Emmerik et al., 2018; see also Section 1.5). Schmidt et al. (2017) used a similar approach to estimate global riverine plastic flux to that employed by Lebreton et al. (2017). However, they estimate the flux range to be 0.41-4 Mt yr\(^{-1}\). The difference between the two estimates was due to difference in the model assumptions. In the case of Lebreton et al. (2017) artificial barriers (e.g. dams and weirs) were assumed to act as sinks for plastic debris. The models of Lebreton et al. (2017) and Schmidt et al. (2017) were both empirically derived and did not explicitly consider processes controlling MMPW transport and emission.

*Figure 1.1 - Mass of river plastic flowing into the oceans in tonnes per year (from Lebreton et al., 2017)*
Meijer et al. (2019) used an alternative, grid-based, modelling approach to estimate the catchment flux. MMPW flux for each river flowing into the ocean was calculated as the cumulative flux from 3 x 3 arc-second cells in the contributing catchment. The flux from each cell was estimated as the product of the MMPW generated in that cell (see Lebreton and Andrady, 2019) and a probability of emission \((p(E))\). Computed estimates of MMPW emission to the ocean were calibrated against field measurements. Values of \(p(E)\) were calculated for each grid cell from the probability of intersection of three events: (1) the probability of mobilisation on land, \(p(M)\); (2) the probability of transport from land to river channels, \(p(R)\), and; (3) the probability of transport from the river to the ocean, \(p(O)\). Figure 1.3 illustrates this model framework from Meijer et al. (2019) to estimate riverine plastic flux. \(p(M)\), \(p(R)\) and \(p(O)\) are derived from physical and environmental characteristics including precipitation, wind, land use, slope, distance to river, stream order, river discharge and distance to the river mouth (see Figure 1.3).

Figure 1.2 – Comparison between modelled estimates of riverine plastic flux from Lebreton et al. (2017) and measured estimates of riverine plastic flux (calculated from plastic mass concentrations reported in the literature). Circles indicate midpoint estimates and whiskers represent lower and upper estimates of plastic inputs to the oceans from rivers in kg d\(^{-1}\). The regression analysis was carried out with 30 records from 13 rivers reported in seven studies (see Lebreton et al., 2017 for more details).
Meijer et al. (2019) estimated a global river plastic emission of 0.8-2.7 Mt yr\(^{-1}\). This is of the same order of magnitude as previous estimates (see Lebreton et al., 2017 and Schmidt et al., 2017; above). However, the predicted spatial distribution of plastic inputs to the oceans from different rivers was wider. Figure 1.4 compares the estimated cumulative distribution of riverine plastic emission to the ocean from Meijer et al. (2019) to previous estimates. The distribution predicted by Meijer et al. (2019) suggests that >1000 rivers are responsible for 80% of global riverine plastic emission. Previous estimates suggest that only 47 and 5 major rivers were responsible for the same proportion (Lebreton et al., 2017 and Schmidt et al., 2017, respectively). Meijer et al. (2019) suggest that riverine plastic flux is less dependent on catchment size (contrary to previous modelling studies) and more dependent on physical and environmental factors which control MMPW transport on land and in rivers (see Figure 1.3). Thus, even small coastal rivers can make substantial contributions to marine plastic pollution (see Meijer et al., 2019). Despite the fact that an attempt was made to account for different transport processes and limiting factors in the model of Meijer et al. (2019), the underlying assumptions for how these processes operate remain crude. There is, therefore, an urgent need to improve process understanding of plastic debris transport in rivers and to incorporate this understanding in better models of MMPW behaviour in order to improve estimates of riverine plastic flux.

Figure 1.3 – Model framework from Meijer et al. (2019) to estimate riverine plastic flux. The amount of plastic entering the ocean is calculated from the product of MMPW generated in a grid cell and the probability of emission from that grid cell. MMPW flux for each river flowing into the ocean was calculated as the cumulative flux from 3 x 3 arc-second cells in the contributing catchment. \( p(E) \) is constructed from \( p(M) \), \( p(R) \) and \( p(O) \), which are derived from physical and environmental characteristics accountable for MMPW transport.
Riverine plastic flux can also be calculated from field measurements of plastic concentration and discharge. Sampling methods are, however, not standardised and three main approaches have been employed: (1) active sampling; (2) visual observations, and; (3) passive sampling (Jambeck et al., 2015; Lebreton et al., 2017; Van Emmerik et al., 2018; see also Van Emmerik and Schwarz, 2020). Plastic flux estimates may also be supported or supplemented by crowd-based observations of macroplastic debris (i.e. via citizen science; see Rech et al., 2015 and Van Emmerik et al., 2020). Active sampling refers to the collection of plastic debris from rivers, typically collected with the deployment of nets and trawls from cranes and bridges. The samples collected in this way can be combined with discharge to estimate the flux but can also be used to analyse typical composition, mass and size distributions of the debris and to allow the identification of individual items (Van Emmerik and Schwarz, 2020). Sample collection is relatively simple but dependent on existing infrastructure (e.g. the location of bridges). It is also labour intensive and limited to a small range of flow velocities. Furthermore, active sampling methods do not usually account for plastic transported deeper in the water column because they tend to collect only floating and superficially suspended particles (Van Emmerik and Schwarz, 2020).

Visual observation methods essentially involve counting the number of plastic debris items passing a given point on a river over a certain period of time (e.g. González-Fernandes and Hanke, 2017). With additional plastic statistics (e.g. mean plastic weight), the number of plastic debris items can be converted to an amount of plastic in mass (see Van Emmerik et al., 2018). Although subject to observer bias and other uncertainties...
(visual observation methods typically only allow for the assessment of floating macroplastic litter), the visual observation approach provides a simple and consistent method to estimate riverine plastic flux (Van Emmerik and Schwarz, 2020). Consistent data allows for a comparison of flux estimates between rivers and over time. Visual observation methods may be automated and rapidly scaled up with the use of image processing techniques and cameras mounted to bridges and unmanned aerial vehicles (UAV/drones'; Geraeds et al., 2019; Kataoka and Nihei, 2020; Van Emmerik et al., 2020). Further work is required to accurately, quickly and easily identify and count the number of plastic debris items, using image processing techniques.

Van Emmerik et al. (2018) provided a four-step method to estimate riverine plastic flux from a combination of active sampling and visual observation methods (while also accounting for spatiotemporal variability). The method is summarised by Figure 1.5 and described below:

![Figure 1.5 – Four-step method proposed by Van Emmerik et al. (2018) to estimate riverine plastic flux](image)

- **1. Plastic flux profile**: Graph showing plastic flux vs. river width.
- **2. Plastic statistics**: Graph showing number of plastic particles vs. area covered.
- **3. Relation with hydrology**: Graph showing plastic transport vs. discharge.
- **4. Extrapolation**: Graph showing cumulative plastic emission vs. time.
1. **Determine cross-sectional profiles of plastic flux** from visual observation methods (e.g. plastic piece counting; González-Fernandes and Hanke, 2017). Observations are made across the channel width (not at a point) because plastic flux can be influenced by local hydrodynamic conditions.

2. **Obtain plastic debris statistics** (e.g. average mass per item of plastic debris) from active sampling methods to convert the number of plastic items to an estimate of plastic in mass.

3. **Couple plastic flux to hydrological data.** Plastic flux is typically calculated using river discharge but could also be estimated using river stage and rainfall rate. Long-term monitoring of plastic flux and access to continuous gauged flow data would facilitate this step.

4. **Extrapolate observations.** Estimates of daily, monthly or annual plastic flux can be extrapolated from a time series of hydrological data (Van Emmerik et al., 2018).

Van Emmerik et al. (2019b) detail a successful long-term monitoring effort to estimate plastic flux on the Saigon River, Vietnam, using the methods described above. They estimated $1.1 \times 10^3$ tonnes of floating macroplastic debris is emitted into the oceans each year from the Saigon and also reported a clear seasonality in plastic transport. They were not, however, able to account for the seasonality with a clear relationship between plastic transport and river discharge, rainfall or other factors.

The final method which may be used to estimate riverine plastic flux is passive sampling, which refers to the analysis of plastic debris that has accumulated at existing infrastructure (Van Emmerik and Schwarz, 2020). In various rivers around the world, infrastructure exists which allows plastic to accumulate, be retained and extracted for analysis (Van Emmerik and Schwarz, 2020). One disadvantage of passive sampling (compared to other methods) is that temporal variations and the role of hydrometeorological factors in plastic transport are harder to quantify (Van Emmerik and Schwarz, 2020). Passive sampling has been used to gain information about accumulated plastic (e.g. composition, size, volume and identity) but relatively few studies have used this data to estimate riverine plastic flux. For example, Gasperi et al. (2014) collected floating debris from floating ‘booms’ along the River Seine, France to determine what proportion of floating debris was plastic and the composition of that plastic but did not estimate riverine plastic flux. Vriend et al. (2020) however, did use a combination of visual observation and passive sampling methods to estimate riverine plastic flux from the River Rhine, the Netherlands.
It should be noted that all the above monitoring methods are suitable for estimating macroplastic fluxes but are not appropriate for evaluating microplastic transfers. The determination of microplastic concentrations in river and marine waters is subject to additional challenges which are beyond the scope of this thesis. In addition, the reader should note that the above methods for plastic debris transport provide estimates for concentrations and fluxes in specific rivers over specific periods and cannot be used to estimate continental or global fluxes. Finally, the number of studies which have attempted to quantify riverine plastic fluxes is relatively low and further work is needed to consolidate the data which have been collected thus far.

1.5 Evaluation of Methods Used to Estimate Riverine Plastic Flux

All the methods which have hitherto been used to estimate riverine plastic flux (including modelling and monitoring efforts) are subject to a high degree of uncertainty. This is illustrated by the substantial differences between the lower and upper estimates of global riverine plastic emission from the three main modelling studies. These were 1.26 Mt yr\(^{-1}\) (Lebreton et al., 2017), 3.59 Mt yr\(^{-1}\) (Schmidt et al., 2017) and 2 Mt yr\(^{-1}\) (Meijer et al., 2019; see also Section 1.3). It is important to identify and understand the assumptions, limitations and uncertainties of these methods, especially since the results of these studies are often employed to support subsequent research and used to influence policymaking decisions. Liubartseva et al. (2016), for example, used estimates of total plastic input to the Adriatic Sea from Jambeck et al. (2015) to calculate riverine plastic flux from individual rivers (Van Emmerik and Schwarz, 2020). Global estimates of riverine plastic flux may also be used to identify the most polluting rivers and to prioritise mitigation efforts (Meijer et al., 2019).

Given that most modelled estimates of riverine plastic flux have been generated using some sort of calibration based on field measurements, the quality of these estimates are directly affected by the quality of the field measurements used for calibration. Field measurements often lack consideration of the temporal (daily and seasonal) and spatial (e.g. across a river width and along its length) variability in plastic flux. Early estimates of plastic flux, for example, were based on point measurements that were directly upscaled to daily or annual estimates (Van Emmerik et al., 2018; Van Emmerik et al. 2019b; see also Lechner et al., 2014; Moore et al., 2011). More recent estimates of flux couple observed concentrations with hydrological data (see Figure 1.5; Van Emmerik et al., 2018). There are also very few long-term monitoring studies (see Van Emmerik et al., 2019b). The lack of spatiotemporal information is highlighted by Van
Emmerik et al. (2018), which they consider to be the largest uncertainty in global estimates of plastic flux to the oceans.

Accurate field measurements to calibrate and validate new and existing models of riverine plastic flux are scarce (González-Fernández and Hanke, 2017; Van Emmerik et al., 2018). Moreover, such data tend to be biased toward European and North American rivers (i.e. there is a lack of monitoring studies in Southeast Asia to calibrate modelling efforts; Van Emmerik et al., 2019a). Additionally, sampling methods and reporting units for riverine plastic flux from field measurements are not standardised, which can make it difficult to calibrate models and make meaningful comparisons between rivers (Jambeck et al., 2015; Lebreton et al., 2017; Lechner et al., 2014; Van Emmerik et al., 2018). Van Emmerik and Schwarz (2020) provide a comprehensive (yet non-exhaustive) summary of the sampling methods and reporting units used by 33 studies to estimate riverine plastic flux between 1997 and 2019. Detailed in the summary is the monitoring approach (visual counting, net sampling, passive sampling, or other sampling), measured component (floating debris, suspended debris, bed transport debris, riverbank debris or flood-deposited debris), monitoring period and reporting unit (mass, density, concentration, transport rate, emission or ingestion rate). This summary highlights the need for standardised monitoring methods and reporting units (see Van Emmerik and Schwarz, 2020, for more detail).

Perhaps the greatest limitation of existing methods to estimate riverine plastic flux is the poor consideration of processes which control plastic debris transport in rivers. It is inherently difficult, however, to represent the complicated nature and variability of processes controlling plastic debris displacement in rivers with simple models and equations. The current understanding of plastic debris transport in rivers is not sufficient to be able to confidently draw conclusions about the amount of plastic entering the oceans each year from rivers.
1.6 Macroplastic Debris Displacement in Rivers

The processes controlling plastic debris displacement in rivers are currently poorly understood and not well represented by existing methods to estimate riverine plastic flux. It is important to understand these processes in order to be able to predict plastic accumulation in the fluvial and marine environments and the residence time of plastic debris at sites of temporary storage in rivers (Hoelllein et al., 2014; Hurley et al., 2018; Schwarz et al., 2019). Although a significant proportion of plastic found in the marine environment is macroplastic, most riverine plastic research focuses on the microplastic size fraction (Blettler et al., 2018; Schwarz et al., 2019; see also Lebreton et al., 2019). There is a conspicuous knowledge gap for macroplastic behaviour in rivers. This study, therefore, investigates aspects of the in-stream dispersal of macroplastic debris.

The fate of macroplastic debris (i.e. accumulation in the marine, estuarine or freshwater environment) is strongly dependent on the sources, transport, storage and degradation of macroplastic debris in river channels. Figure 1.6 summarises the processes controlling the fate of macroplastic debris in rivers, including: sources and emissions to river channels; in-channel transport (i.e. processes promoting displacement); in-channel degradation, and; in-channel storage (i.e. accumulation at trapping points). The fate of macroplastic debris can also be affected by human interventions (e.g. removal by clean-up operations or release from ‘trapping points’ by anthropogenic disturbance).

1.6.1 Sources of Macroplastic Debris in River Channels

The input of macroplastic debris to river channels is directly related to human activity (e.g. population density, urbanization and urban point sources; see Best, 2019). Macroplastic debris may enter the river channel through natural processes or direct deposition (Van Emmerik and Schwarz, 2020). Natural processes include wind, overland flow and river flow (the latter can mobilise macroplastic debris on riverbanks and the floodplain; Bruge et al., 2018). Other sources of plastic include direct deposition to channels, urban point sources (e.g. storm drains) and uncontrolled landfill (Jambeck et al., 2015; Mihai, 2018; Willis et al., 2017). In general, waste management infrastructure in high-income countries is more effective at reducing plastic waste inputs to river channels, compared to low and middle-income countries, where plastic waste is more commonly discarded into the environment close to the point of use, rather than being collected and disposed of formally (Jambeck et al., 2015).
Figure 1.6 – Processes controlling the fate of macroplastic debris in rivers, including: the sources, transport, degradation and storage of macroplastic debris in river channels.
1.6.2 Transport and Degradation of Macroplastic Debris in River Channels

The mechanisms promoting macroplastic debris displacement in rivers are similar to those which transport sediment in rivers – suspension, saltation and traction (see Charlton, 2008). However, unlike sediment, many plastic materials are buoyant. This means that macroplastic debris can also be transported on the surface by the current and by wind (see Ivar do Sul, 2014; Kooi et al., 2018). Transport via suspension is through the water column and transport via saltation and traction occur along the riverbed. Figure 1.7 illustrates each transport mechanism and where they operate in the water column. Transport mechanisms lower in the water column (e.g. traction and saltation) tend to be weaker, relative to surface transport mechanisms (e.g. floating and superficially suspended transport; Van Emmerik and Schwarz, 2020). Therefore, the travel distances of macroplastic debris transported on or near the surface tend to be greater than macroplastic debris which is transported closer to the riverbed. In a wide, straight, uniform channel, without any trapping points, the travel distance of macroplastic debris scales with flow velocity ($U$, m s$^{-1}$) – however, in narrow or meandering channels, or in channels where trapping points are present, the travel distance of macroplastic debris may not necessarily increase with an increase in flow velocity. This is because there is a complicated relationship between (1) the hydraulics of flow in a river and the trapping of macroplastic debris (see Section 1.6.3), and (2) river stage and the trapping of macroplastic debris (an increase in river stage usually accompanies an increase in flow velocity; see Section 1.6.4). It is unclear how the travel distance of macroplastic debris changes with fluctuations in wind speed and direction because wind can blow macroplastic debris in the upstream and downstream direction and into and out of sites of temporary storage (trapping points).

The transport mechanisms by which macroplastic debris is transported are dependent on plastic properties (e.g. density, buoyancy, shape and size). Low-density plastics or high-density plastics which contain trapped air (e.g. sealed plastic bottles) tend to float and are transported by the surface flow velocity or wind (Ryan, 2015; Van Emmerik and Schwarz, 2020). Conversely, higher density plastics are transported by suspension, saltation or traction – deeper in the water column. Transport is also affected by biofouling (i.e. the accumulation of algae and other microorganisms as a biofilm). This tends to increase overall density (see Ryan, 2015). Small items (with a high surface area to volume ratio) are more likely to be strongly affected by the impacts of biofouling than larger items.
The degradation of macroplastic debris items, through fragmentation, also affects their transport. Fragmentation is caused by (inter alia) photodegradation by UV radiation, thermal degradation and abrasion (Kooi et al., 2018). This progressively reduces macroplastic particles into smaller fragments, some of which will be microplastic and nanoplastic (Van Emmerik and Schwarz, 2020). Microplastic and nanoplastic particles are likely to behave differently to macroplastic debris in river channels and their displacement may be best considered with existing techniques used to study the transport of sediment and solutes in channels (Van Emmerik and Schwarz, 2020; see also Kooi et al., 2018). Cook et al. (2020), for example, tracked the displacement of microplastic particles using fluorometric techniques (a technique commonly used in studies of solute dispersal). They found that neutrally buoyant microplastic displaced in the same manner as a solute. High-density plastics and smaller plastic particles (microplastic and nanoplastic) can also be deposited in river and floodplain sediments (sedimentation; Van Emmerik and Schwarz, 2020). The fragmentation and biofouling of macroplastic debris promotes sedimentation (Kooi et al., 2018). The properties of macroplastic debris affect their fragmentation, sedimentation and thus, transport. Further work is required to investigate the mechanics of macroplastic debris transport, the fragmentation and sedimentation of macroplastic debris, and how each is affected by macroplastic debris properties.
1.6.3 Accumulation of Macroplastic Debris at ‘Trapping Points’

Both the emission and the transport of macroplastic debris between sites of temporary storage (or ‘trapping points’; see Figure 1.6) is likely to be highly intermittent. Transport is likely to proceed over a series of discrete ‘step’ (the movement phase) and ‘rest’ (the stationary phase) periods, not unlike the ‘step’ and ‘rest’ periods conceptualised for sediment transport (see Hassan et al., 1991). The ability of a trap to capture and retain macroplastic debris is defined here as trap effectiveness. Effective traps capture and retain a high proportion of macroplastic debris (and vice versa). Figure 1.8 is a flow diagram illustrating the transport of macroplastic debris between sites of temporary storage (‘trapping points’) which helps to illustrate capture and retention. In Figure 1.8 macroplastic debris can be either captured by a trap or continue to travel downstream. Captured debris is either retained (entering storage) or released (at which point it can continue to travel downstream). Figure 1.8 provides example amounts of debris that are captured and retained (or not) by two hypothetical traps in series. In Figure 1.8b, 9 out of 20 items of macroplastic debris were trapped in Trap 1 (meaning 11 of 20 items of macroplastic debris were not trapped) and only 4 out of the 9 macroplastic debris items that were trapped were retained (meaning 5 of 9 items were trapped but not retained). In total, 15 out of 20 items of macroplastic debris were either captured but not retained or not captured at all. Trap capture could be represented by the proportion of macroplastic debris which becomes trapped and trap retention may be represented by a residence time distribution, for example. Trap effectiveness is controlled by the size and characteristics of individual trapping points.

![Diagram](image)

**Figure 1.8** – Schematic illustration of the transport of macroplastic debris between sites of temporary storage. Macroplastic debris can be either captured by a trap or continue to travel downstream. Captured macroplastic debris is either retained (entering storage) or released, where it continues to travel downstream. Example amounts of macroplastic debris that are captured and retained (or not) are also provided.
Trapping points of macroplastic debris can be broadly categorised as either hydraulic (trapping by flow separation structures) or physical (trapping by obstacles). Trapping by obstacles has also been observed in studies of large woody debris transport in rivers (see Bocchiola et al., 2006). The trapping of macroplastic debris in hydraulic traps is caused by flow separation, which generates recirculation eddies and “dead zones” of weak reverse flow (Ferguson et al., 2003). Flow separation occurs where there are topographic irregularities (or obstacles) at the channel boundary – at the finest scale, these topographic irregularities include undulations caused by erosion and the slumping of bank material, protruding root balls and lumps of grass sod (Charlton, 2008; Kean and Smith, 2006). At the coarsest scale, these topographic irregularities include channel obstructions (e.g. mid-channel bars) and meander bends (Charlton, 2008). Flow separation refers to the detachment of the boundary layer from the channel boundary (i.e. the area of flow in contract with the channel bed and banks detaches from the channel bed and banks; Charlton, 2008). Flow separation occurs when an adverse pressure gradient is produced downstream of an obstacle, occurring when viscous forces at the channel boundary reduce flow acceleration in front of the obstacle (Powell, 2014). If the adverse pressure gradient is sufficient to decelerate the flow near the boundary to zero and then reverse the flow, the zone of reverse flow forms an obstruction that causes the fluid continuing from the front of the obstacle to separate from the boundary layer (Powell, 2014). The boundary layer reattaches to the channel boundary downstream of the channel irregularity, but the area enclosed by the separated flow (the wake) contains recirculating eddy structures, turbulence and “dead zones” of weak reverse flow (Ferguson et al., 2003, Kean and Smith, 2006; Powell, 2014).

Figure 1.9a-c illustrates flow separation around a boulder on the bed of a river, around an obstruction and at a meander bend (from Charlton, 2008, p.83). Figure 1.9d illustrates the fluid velocity profile parallel to a boundary at a meander bend (from Powell, 2014, p. 308). At the separation point, the flow begins to decelerate and reverse. Recirculating eddy structures, turbulence and “dead zones” of weak reverse flow can cause macroplastic debris to become trapped (or ‘pinned’) against channel banks (i.e. where there does not appear to be an obvious ‘trapping point’). The occurrence and size of flow separation zones in rivers are proportional to the number of meander bends and the curvature of individual bends (respectively) in a given reach (i.e. there are more flow separation zones in reaches with a higher number of meander bends and the size of flow separation zones are larger in sharp, highly curved meander bends; Blanckaert, 2011; Ferguson et al., 2003). Therefore, it is assumed that the trapping of macroplastic debris in hydraulic traps increases with channel sinuosity, commonly represented by the
sinuosity index (see Leopold et al., 1964). Since (small-scale) flow separation also tends to increase with bank roughness, it is also assumed that the trapping of macroplastic debris in hydraulic traps increases with bank roughness. Finally, it is assumed that the trapping of macroplastic debris in hydraulic traps along riverbanks decreases with the increase in channel width because the relative interaction between river flow and channel banks decreases with the increase in channel width. Channel form and materials can therefore, clearly, affect the transport and trapping of macroplastic debris.

![Flow separation diagrams](Figure 1.9)

**Figure 1.9** – (a) Flow separation around a boulder on the bed of a river, (b) flow separation around an obstruction and (c) flow separation at a meander bend (all from Charlton, 2008, p.83), (d) fluid velocity profile parallel to a boundary at a meander bend (from Powell et al., 2014, p. 308)

Physical traps for macroplastic refer to accumulation at obstacles or against rough bank obstructions. Physical traps include vegetation (aquatic, overhanging and riparian), morphological features (e.g. mid-channel and point bars, mature islands, rough banks and exposed boulders and bedrock) and infrastructure (e.g. dams, weirs, bridges, fords, culverts and flow deflectors). Hydraulic and physical traps interact with each other: flow separation can cause macroplastic debris to become trapped in physical traps; physical traps can cause flow separation, and; individual traps may be the result of a combination of hydraulic and physical processes.

Vegetation is a potentially significant trap of macroplastic debris. In a study on the Saigon River, Vietnam, Van Emmerik et al. (2019b) hypothesized that plastic transport appeared to be strongly linked to the presence of water hyacinths in the river, which act as accumulation zones of plastic debris. Similarly, Ivar do Sul et al. (2014) demonstrated that plastic debris can be retained by mangrove forest patches for long
periods of time (months-years). Macroplastic debris may be trapped by aquatic (instream) and riparian (bankside) vegetation and by near-stream vegetation that overhangs the river channel. For illustration, Figure 1.10 shows macroplastic debris tracer items trapped in overhanging vegetation and emergent aquatic vegetation. The ability of a trap to capture and retain macroplastic debris (trap effectiveness) is dependent on vegetation characteristics. Thus vegetation traps can be sub-classified based on these characteristics. Aquatic and riparian vegetation can be sub-classified into submerged, floating-leaved and emergent plants (Bentley et al., 2014). These classifications are illustrated in Figure 1.11 (from Kentucky Pesticide Education Programme, 2016) and defined in Table 1.1 (adapted from Bentley et al., 2014).

![Figure 1.10 - Macroplastic debris tracer items trapped in (a) overhanging vegetation and (b) emergent aquatic vegetation on the River Sence, Leicester, UK](image)

![Figure 1.11 – Classifications of aquatic channel vegetation (macrophytes): emergent, submerged, rooted floating and free-floating plants (from Kentucky Pesticide Education Programme, 2016)](image)
Table 1.1 – Definitions of aquatic and riparian vegetation classifications (adapted from Bentley et al., 2014, pp.10-11)

| Vegetation Classification | Definition |
|---------------------------|------------|
| Submerged plants          | “Species with stems and leaves that grown beneath the surface of the water, although flowers may project above the surface (Barrett and Banks, 1993). They are usually found in deeper water and rooted on the bottom (SEPA, 2003).” “Plants with some or all of the leaves floating on the water surface (Newman, 1995). Members of this group are often found intermingled with emergent and submerged plants in water just over one metre deep, or deeper in some cases (EA, 1998).” This group can be further sub-classified into rooted floating-leaved plants and free-floating species (see Figure 1.1). |
| Floating-leaved plants    | “Plants whose stems and leaves are exposed above the normal water level. They have erect, aerial leaves and can grow both in water and temporarily damp conditions (Newman, 1995; EA, 1998). This category can be sub-divided into tall emergent species with long narrow leaves … [and] the generally smaller, broad-leaved emergent species.” |
| Emergent plants           | |

Further work is required to determine which vegetation types and characteristics are the most effective at trapping and retaining macroplastic debris – it is, however, assumed that both emergent plants and overhanging vegetation, both of which protrude into the flow, are likely to be most effective at trapping and retaining floating macroplastic debris. It is also assumed that the trapping of macroplastic debris increases with an increase in vegetation cover and density. Liro et al. (2020) suggest that vegetation with high surface roughness favours plastic storage – though this is not supported by empirical (or theoretical) evidence. Trap effectiveness is also likely to change with seasonal changes in vegetation characteristics – it is assumed that the trapping of macroplastic debris in vegetation traps increases during the growing season, when vegetation density and cover are greater and vegetation structure is more complex. Liro et al. (2020) suggest that the change in vegetation roughness caused by a seasonal change in vegetation characteristics and floristic composition favours plastic storage – therefore, during the growing season, macroplastic debris is more likely to be trapped in vegetation, relative to the non-growing season. Van Emmerik et al. (2019b) hypothesize that the seasonality of macroplastic debris transport can be attributed to seasonal change in vegetation characteristics. Further work is required to determine how seasonal change in vegetation characteristics affects the trapping and retention of macroplastic debris in vegetation traps.
1.6.4 Influence of River Stage on Macroplastic Debris Transport

The transport of macroplastic debris is likely to be affected by fluctuations in river stage. These fluctuations can cause macroplastic debris to enter into or be released from storage (i.e. the activation and deactivation of trapping points). For example, at high stage, overhanging trees which do not otherwise intersect the water surface, can start to behave as macroplastic traps. Similarly, if overbank flow occurs, floodplain traps can be activated. Conversely, some channel vegetation and some morphological features (e.g. mid-channel and point bars) may become ‘drowned out’ at high stage, reducing their trapping effectiveness. The residence time of macroplastic debris in floodplain traps is likely to be particularly high because (low frequency) overbank flow is required for remobilisation, although it is possible that this floodplain debris could move back to the river by some other mechanism (e.g. overland flow or wind) or be removed (e.g. by clean-up operations) prior to remobilisation by overbank flow. Additionally, changes in river width with changes in stage could affect macroplastic transport. Macroplastic debris is less likely to be trapped when the river width is high due to a lower relative interaction between river flow and channel banks (see Section 1.6.3).

The changes in river depth, width and velocity with changes in river discharge at a single point has been researched extensively and is known as at-a-station hydraulic geometry (see Ferguson, 1986). In theory the change in river width, depth and velocity at a point can be calculated from channel cross-section geometry and a flow resistance equation (see Ferguson, 1986). Table 1.2 details how river width and depth change at a point depending on cross-sectional channel geometry (after Ferguson, 1986, p.6). This helps to explain how cross-sectional geometry could affect macroplastic transport. In channels where a change in river discharge results predominantly in an increase in river depth (e.g. rectangular channels), traps that are activated or deactivated by a change in river depth (e.g. overhanging tree, floodplain, channel vegetation and morphological feature traps; see above) will be highly affected by fluctuations in river stage. In channels where a change in river discharge results predominantly in an increase in river width (e.g. parabolic channels and curved channels), the trapping of macroplastic debris along channel banks is likely to be affected by fluctuations in river stage. Furthermore, macroplastic debris is less likely to be trapped in channels with steep banks and more likely to be trapped in channels with shallow sloping banks. Channel cross-section geometry is, therefore, likely to influence the relationship between river stage and macroplastic behaviour in complex and inter-related ways. This makes it difficult to determine how individual traps are likely to change with river stage.
**Table 1.2** – Changes in river width and depth change at a point depending on cross-sectional channel geometry (after Ferguson, 1986, p.6)

| Channel Shape | Change in width and depth |
|---------------|---------------------------|
| **A rectangular cross-section** | Typifies a rock-walled channel or one with cohesive and vegetated alluvial banks. A rectangular cross-section has a constant width as depth and discharge increase. |
| **A triangular cross-section** | Idealizes banks of non-cohesive material at a constant angle of repose. Since width is directly proportional to depth, any change in width is accompanied by a change in depth by the same proportion. |
| **Since the angle of repose for sand or gravel decreases below the water line, an approximately parabolic cross-section is more realistic than a triangular cross-section.** For a parabolic cross-section, channel width is proportional to the square root of depth – therefore, any change in width is accompanied by a smaller change in depth, compared to a triangular cross-section. |
| **This channel cross-section represents a curved channel with a cut bank and point bar.** The convex shape of the point bar results in a faster increase in width compared to depth, for medium to high discharges. |

Contrary to the general relationships expected for sediment transport, it cannot necessarily be assumed that an increase in river discharge will result in a proportionate increase in the travel distance of macroplastic debris or riverine plastic flux (because it is uncertain how the trapping of macroplastic debris will be affected by an accompanying increase in river stage). Thus far, empirical evidence for these relationships is weak. Van Emmerik et al., (2019b), for example, found a clear seasonality in plastic transport but could not find a clear relationship between plastic transport and river discharge, rainfall or other factors. Macroplastic debris can also be released from trapping points, even under steady stage conditions (due to gusts of wind or random fluctuations in current speed and direction).
1.7 Chapter Summary, Travel Distance Approach and Justification

In summary, plastic accumulation in the environment (particularly the marine environment) is a global environmental concern. Rivers have been recognised as major transport pathways of plastic debris. Although a body of research is emerging which seeks to estimate the amount of plastic entering the oceans from rivers, current estimates remain highly uncertain. Modelled estimates of riverine plastic flux are based on (estimates of) MMPW generation and calibrated against (limited) field measurements. In principle, riverine plastic flux can be estimated from field measurements by coupling observed plastic debris “concentration” data with measured discharge (though it is noted that methods used to estimate riverine plastic flux from field measurements are not standardised). All methods to estimate riverine plastic flux are subject to a high degree of uncertainty and (perhaps most importantly) fail to consider the process mechanisms controlling macroplastic displacement. Such processes include several mechanisms which promote displacement and several which result in storage (accumulation at ‘trapping points’).

In this thesis a fundamentally different approach is taken to investigate macroplastic transport in rivers. It combines an experimental component, in which macroplastic tracers were tracked in a small river reach, with the construction of a numerical travel distance model, which is intended to act as an interpretive conceptual framework. Unlike existing approaches, it also explicitly considers the processes controlling macroplastic displacement, trapping and release. Understanding these processes is important in order to understand plastic accumulation in the marine environment and the residence time of plastic in rivers (Hoellein et al., 2014; Hurley et al., 2018; Schwarz, et al., 2019). Few (if any) studies have used a travel distance approach to understand macroplastic debris displacement in rivers. Travel distance approaches have been used previously in sediment transport research, however, to study the displacement of individual grains of sediment and calculate river sediment flux (e.g. Hassan et al., 1991; Hubbell and Sayre, 1964). These studies typically produce travel distance distributions (in which the concentration of displaced particles are plotted against the distance travelled) and river sediment flux estimates. These serve as alternatives to sediment rating curves derived from field measurements (see Asselman, 2000) and global modelling efforts (which estimate sediment yield for individual catchments from correlations between mean flow conditions and bulk sediment transfer; see Milliman and Meade, 1983; Milliman and Syvitski, 1992; Walling and Webb, 1996).
It is hypothesized that the travel distance distribution of macroplastic debris will be a function of the location and characteristics of trapping points. If macroplastic travel distances are not a function of these, the proportion of bottles trapped in sub-reaches would be equal. Note that the proportion of bottles trapped and the absolute number of bottles trapped are not the same – where the same proportion of bottles trapped is equal for every sub-reach, there is an exponential decrease in the number of trapped bottles with distance downstream. Figure 1.12 illustrates a theoretical travel distance distribution, where the trapping of macroplastic debris is uniform (i.e. does not vary) for each sub-reach. In Figure 1.12 50% of bottles are trapped in each sub-reach, thereby producing an exponential decay in the number of bottles stored in each sub-reach.

Figure 1.12 – Theoretical travel distance distribution where the trapping of macroplastic debris is uniform for each sub-reach. In this example 50% of bottles are trapped in each sub-reach, thereby producing an exponential decay in the number of bottles stored with distance downstream.

1.8 Research Aims and Questions

This thesis has two aims and two specific objectives

Research Aims:

- To characterise experimentally the displacement and trapping of macroplastic debris in rivers, using a travel distance approach
- To model the transfer of macroplastic debris though a river catchment, using a travel distance approach

Objectives

- To determine the travel distance distribution of seeded macroplastic debris in an experimental river reach (and its controls)
- To develop a framework for modelling the displacement and trapping of macroplastic debris in rivers, which predicts the transfer of individual items of macroplastic debris and is informed by the experimental data obtained in the study
2 METHODOLOGY

2.1 Introduction

This study uses a new and fundamentally different approach to investigate plastic transport, which considers the travel distance of individual items of macroplastic debris (thereby explicitly considering the processes controlling displacement in rivers; see Section 1.6). There are two major components to this study: (1) the use of tracer experiments, to determine the travel distance distribution of macroplastic debris, and; (2) the development of a framework for modelling transfer of individual items of macroplastic debris. This chapter describes: (1) the methods used for the collection and analysis of travel distance data from tracer experiments, (2) the approach and framework of the numerical model, and (3) the methods used to calibrate and test the sensitivity of the numerical model.

2.2 Study Area

Travel distance data were collected along a 1 km reach of the River Sence, Wistow, Leicester, UK (injection point at 52°33.498'N, 1°03.092'W) between January and March 2020. The Sence catchment is small (~65 km² at the injection point) of moderate to low relief with a land cover of predominantly arable farming and grassland. Mean annual rainfall is ~650 mm and catchment geology consists of low permeability bedrock, overlain by boulder clay and alluvium deposits. The study reach is representative of a typical small UK agricultural lowland river. Figure 2.1 illustrates the location of the study reach, injection point and catchment area at the injection point. The numbered points in Figure 2.1 correspond to photographs of the study reach in Figure 2.2. Downstream of the study reach, the River Sence flows through Southeast Leicester and joins the River Soar, itself a tributary of the larger Trent basin. Characteristics of the study reach are ideal for the investigation of macroplastic debris ‘trapping’. For example: (1) vegetation overhangs into the river channel, acting as a potential trap for macroplastic debris; (2) channel width is relatively low (~10 m at the injection point), allowing macroplastic debris to interact with, and be potentially trapped along channel banks and; (3) the 1 km reach is meandering (sinuosity index ≈ 1.7) and macroplastic debris may be trapped in flow separation zones as it passes through a meander bend. There is also good access, allowing for the recovery of injected macroplastic debris at points along the reach.
Regrading operations were conducted along the study reach between 1978 and 1980 as part of a wider *River Sence Improvement Scheme*, conducted between 1967 and 1985. The improvement scheme aimed to improve agricultural productivity by increasing field drainage and channel capacity. To increase channel capacity, bankfull width and depth were increased. The regrading scheme caused an extended period of channel instability and rapid siltation immediately following the regrading scheme (Sear, 1993). Rapid siltation was caused by an increased sediment supply from bank failure, locally reduced sediment transport capacity as a result of channel widening and livestock ‘poaching’ of collapsed banks (Sear, 1993). Figure 2.3 summarises river channel adjustment following regrading operations along the River Sence (Sear, 1993, p.12). The long-term effects of the regrading scheme are relevant to this study because channel geometry was significantly altered (as noted in Section 1.6.4 channel geometry affects the influence of river stage on the transport and trapping of macroplastic debris).

*Figure 2.1* – Location of the study reach (River Sence, Wistow, Leicester, UK; injection point [red dot] at 52°33.498’N, 1°03.092’W; catchment area [striped polygon] at the injection point is ~65 km$^2$). Numbered points correspond to locations of study reach photographs in Figure 2.2.
Figure 2.2—Photographs of the study reach. Numbers correspond to locations photographs were taken from in Figure 2.1
Figure 2.3 – River channel adjustment following regrade operations along the River Sence, Leicester (Sear, 1993, p.12)
2.3 Tracer Experiments

2.3.1 Determining the Travel Distance Distribution of Macroplastic Debris

Macroplastic travel distance distributions were determined using tagged plastic bottles. These bottles were injected into the study reach and their location was recorded after 24 hours. The bottles were then retrieved from the river using fishing nets. Temporary fencing was erected at the end of the study reach to prevent littering downstream. Plastic bottles were used as a model litter item to study the displacement of individual items of macroplastic debris. They were chosen because the UK uses 13 billion plastic bottles each year, plastic bottles make up a third of all plastic pollution in the sea and 14% of identifiable plastic litter items found in European freshwater environments were plastic bottles (EAC, 2017; EWI, 2019). Other types of macroplastic debris (e.g. plastic bags, straws or crisp packets) will displace differently to plastic bottles. The displacement of these may be scope for further research. Releasing tagged plastic debris into the natural environment and tracking movement is not a new method. Ivar do Sul et al. (2014), for example, released painted items of plastic debris into a mangrove forest patch to investigate retention and export. However, there are relatively few studies which release and track plastic debris transfer in rivers (see Tramoy et al., 2020).

2.3.2 Pilot Experiments

To guide the development of the experimental protocol, pilot experiments were conducted where plastic bottles were injected and retrieved after only a few hours. Figure 2.4 illustrates the travel distance distributions from three pilot studies (PS1-PS3). Each bar represents a 10 m sub-reach that bottles were retrieved from. A more comprehensive analysis of the results from this study (excluding pilot data) is provided in Chapter Three. Figure 2.4 illustrates that the travel distances of plastic bottles were relatively low, suggesting that macroplastic debris is trapped and retained relatively easily. For example, the maximum and mean travel distances were 699 m and 169 m in PS1, 244 m and 96 m in PS2 and 370 m and 136 m in PS3, respectively. Figure 2.4 also illustrates a high degree of variability in plastic bottle travel distances between pilot studies. As well as the difference in maximum and mean travel distances, the sub-reach where the most amount of bottles were trapped differed (being 100 m downstream (D/S) of the injection point for PS1, 20 m D/S for PS2 and 140 m D/S for PS3). Additionally, the dispersion of the plastic bottle travel distances varied in each pilot study ($0.65 \leq CV \leq 0.97$).
Figure 2.4 – The travel distance distributions of three pilot studies. Each bar represents the number of bottles trapped in a 10 m sub-reach. $\bar{x}$ = mean travel distance (m), CV = coefficient of variation, n = number of bottles recovered

2.3.3 Experimental Design

The key findings from the pilot studies were that travel distances of plastic bottles were low and variable. Given that the travel distances of plastic bottles were relatively low, the pilot data limited the required length of the study reach to ~1 km. Given the variability in travel distances, large sample sizes and high numbers of replicates were required. In this study, six replicate experiments were conducted, each comprising of 90 plastic bottles. To ensure that plastic bottles travelled independently of each other during each replicate, bottles were injected individually at a regular interval (every 5 seconds). Each replicate was also conducted under the same, steady flow conditions. The same flow conditions act as a control variable (otherwise difference between replicates could
be attributed to a difference in river stage or discharge; see Section 1.6.4). River stage and discharge (Q, m³s⁻¹) were measured for each replicate, the latter using the salt dilution method (see Moore, 2005).

2.3.4 Controls on the Travel Distance Distribution of Macroplastic Debris

It is hypothesized that plastic bottle travel distance distributions are systematically controlled by the location and nature of trapping points (otherwise there would be an exponential decrease in the number of bottles with distance downstream; see Section 1.7). To determine if travel distances were exponentially distributed or not, an Anderson-Darling (AD) goodness-of-fit test was conducted (see Stephens, 1974). The AD test computes a test statistic and p-value (p). When p < 0.05, the data is not exponentially distributed. When p > 0.05 it can be assumed that the data follows an exponential distribution.

To investigate the controls on travel distances, it was necessary to map the location and nature of trapping points along the study reach. The following traps were present along the study reach:

- Aquatic (instream), riparian (bankside) and overhanging vegetation
- Rough channel banks
- Meander bends.

If experiments were conducted under variable discharges trapping on the floodplain (which requires overbank flow) would also be possible (see Section 1.6). Trapping in overhanging trees and along meander bends is analogous to point-source pollution in rivers (as both can be identified at a single point). Conversely, trapping along rough channel banks and in aquatic and riparian vegetation is analogous to diffuse pollution (as both operate over the entire catchment and cannot be identified at a single point). Given that they can be identified at a single point, the locations of overhanging trees and meander bends were recorded in a survey of the study reach to investigate the controls on plastic transport. The presence of these trapping points was then related to observed data. It was decided not to map changes in riparian and aquatic vegetation characteristics and channel bank roughness since they cannot be identified at a single point and do not significantly vary along the study reach. Similarly, changes in river channel width and geometry (which also affect plastic transport; see Section 1.6) were not mapped. The assumption that the above characteristics do not significantly vary may not apply for longer study reaches.
2.3.5 Trap Effectiveness

To gain information about the ability of a trap to retain macroplastic debris, repeat surveys were conducted on four replicate experiments. Repeat surveys involve recording the location of plastic bottles multiple times over the 24-hour survey period. In this study the location of bottles were recorded between three and seven times over 24 hours. The difference in the number of bottles stored in each 10 m sub-reach between repeat surveys was used to evaluate trap retention. It is hypothesized that once trapped, bottles mostly remain in storage, unless there is a change in river flow conditions (i.e. increase or decrease in river stage). Trap retention was also related to trap and reach characteristics (see Section 2.3.4; above).

2.4 Modelling the Transfer of Macroplastic Debris Through a River Catchment

2.4.1 Model Introduction

Existing models to estimate riverine plastic delivery to the oceans are based on MMPW generation and calibrated against field measurements (see Section 1.3). However, these models are subject to a high degree of uncertainty and fail to consider the process mechanisms controlling macroplastic debris displacement in rivers, even in a conceptual way (see Sections 1.5 and 1.6). At the same time (and despite a recent increase in interest) there remains a paucity of studies attempting to elucidate macroplastic transport processes and the process mechanisms governing macroplastic behaviour in fluvial systems are not well understood. There is, therefore, an urgent need to: (1) better understand these processes and (2) incorporate this understanding into models of riverine plastic transport. In this study a framework for modelling the transfer of individual items of macroplastic debris along river channels is presented (which is informed by tracer experiments; see Section 2.3, above). The framework explicitly considers a conceptual description of the main processes controlling plastic displacement and retention using a probabilistic travel distance approach. It is intended to be used to: (1) estimate a travel distance distribution for plastic debris; (2) identify sites of temporary storage, and; (3) estimate the residence time distribution of plastic debris at plastic accumulation zones. The effect of macroplastic debris trapping on riverine plastic delivery to the oceans can be evaluated with these (and other) outputs.

It should be noted that although example model outputs are provided in Chapter Four, their validity (and utility) are limited by (inter alia): (1) an imperfect understanding of what controls macroplastic debris transport; (2) a poor understanding of the relationships between reach characteristics and the trapping of macroplastic; and (3) the limited size of the calibration dataset. With an improved understand of the controls on
macroplastic debris displacement and more travel distance data for calibration, the model approach and framework can be refined to produce higher predictive quality outputs.

2.4.2 Probability of Trapping

The simple conceptual approach underpinning the model framework is based on the probability of trapping per unit distance and time, $p(T)$. $p(T)$ is calculated for individual sub-reaches (cells) in a channel network. Macroplastic debris is assumed to transfer downstream from cell to cell as illustrated in Figure 2.6. Example values for $p(T)$ are also provided in Figure 2.6. $p(T)$ is assumed to be proportional to the presence of trapping points (e.g. vegetation, morphological features and existing infrastructure) and inversely proportional to river channel width (see Figure 1.6 and Section 1.6). The principal morphological feature considered is the meander bend. Trapping is related to the number and curvature of meanders along a study reach. In a wide, straight, uniform channel, without any trapping points, $p(T)$ is likely to be low. For simplicity, we assume it is equal to zero in such reaches over a given (short) period of time. Processes which affect $p(T)$ and which are represented in the numerical model are: (1) trapping in flow separation zones, $p(FS)$ (2) trapping in overhanging vegetation, $p(V)$ and (3) trapping along channel banks, $p(CB)$ (all per unit distance and time). Tracer experiment results indicated that a high proportion of macroplastic debris can be trapped in flow separation zones at meander bends and in overhanging tree branches (see Section 3.3). Formally, $p(T)$, is represented by:

$$p(T) = 1 - \left( (1 - p(FS)) \times (1 - p(CB)) \times (1 - p(V)) \right)$$

(2.1)

$p(FS)$ is assumed to be a function of channel sinuosity (represented by the sinuosity index, $S$), where $p(FS)$ increases with $S$ (see Section 1.6.3). The assumed relationship between $p(FS)$ and $S$ is provided by Equation 2.2 and illustrated in Figure 2.6

$$p(FS) = 1 - \frac{1}{(S)^a}$$

(2.2)

where $a$ is an empirical parameter. $S$ is calculated using Equation 2.3:

$$S = \frac{C}{D}$$

(2.3)

where $C$ is channel length (m) and $D$ is downvalley length (m; see Leopold et al., 1964). The assumed relationships for the model are hypothetical and require both validation and further investigation.
Figure 2.5 – Schematic illustration of the transfer of macroplastic debris downstream from cell to cell. Also shown are hypothetical values of the combined probability of trapping, \( p(T) \); probability of trapping in flow separation zones, \( p(FS) \); probability of trapping along channel banks, \( p(CB) \), and; probability of trapping in overhanging vegetation, \( p(V) \) (all per unit distance and time). The equations used to calculate \( p(T) \), \( p(FS) \) and \( p(CB) \) are also shown.

\[
p(FS) = 1 - \frac{1}{(g)^a}
\]

\[
p(CB) = \frac{1}{b(w/g)^f}
\]

\[
p(T) = 1 - \left( (1 - p(FS)) \cdot (1 - p(CB)) \cdot (1 - p(V)) \right)
\]
Figure 2.6 – Assumed relationship between (a) the probability of trapping in a flow separation zone and the channel sinuosity index, and (b) the probability of trapping along channel banks and channel width

$p(CB)$ is assumed to decrease with channel width ($w$, m) because the relative interaction between the bulk river flow and channel banks decreases as width increase (see Section 1.6). This is illustrated in Figure 2.6 and described formally via

$$p(CB) = \frac{1}{b \left(\frac{w}{w_0}\right)^c}$$

(2.4)

where $w_0$ is the minimum channel width (m) and $b$ and $c$ are empirical calibration parameters. $w$ can be either directly measured or estimated from discharge, using the hydraulic geometry relation:

$$w_{bf} = Q_{bf}^{0.5}$$

(2.5)

where $w_{bf}$ is bankfull channel width (m) and $Q_{bf}$ is bankfull discharge ($m^3s^{-1}$). $Q_{bf}$ can be estimated from the discharge with a recurrence interval of 1.5 years (using gauged flow data; see Benson and Thomas, 1966).

A value for $p(V)$ is applied to sub-reaches where an overhanging tree is present as part of the model calibration (Section 2.4.5). Overhanging trees can be identified from field surveys, aerial photographs and satellite imagery but future work may wish to automate tree identification using remote sensing and machine learning techniques.
2.4.3 Model Framework

Figure 2.7 shows a decision network, which describes the travel distance model framework. There are two model scenarios: (1) several particles (bottles) are injected at a single point, representing a tracer experiment and (2) particle emission occurs randomly at multiple points along the river over time, representing ‘real-world’ inputs of macroplastic litter (see Section 1.6.1). In Scenario 1, particles are assumed to enter the river at the most upstream reach during the first time iteration of the model. In Scenario 2, particles enter at multiple points in every time iteration of the model.

When a particle enters at a single point, it is transported downstream. In each downstream cell, it can either pass through or become trapped. This is essentially a Bernoulli trial; a random experiment with exactly two possible outcomes (pass or trap). Particles are assumed to become trapped if a randomly generated number is less than or equal to the probability of trapping for that cell (i.e. \( xx \leq p(i) \), where \( p(i) \) is the probability of trapping in sub-reach \( i \) and \( xx \) is a randomly generated number between zero and one). \( p(T) \), is automatically calculated in the model for each cell from reach/trap characteristics. The use of a randomly generated number to control particle trapping is intended to represent the stochastic nature of plastic debris transport in river systems. When a particle is trapped, the model records the sub-reach in which the trapping occurs and augments the tally of particles trapped there. If the particle passes through all cells without becoming trapped, it is recorded as having left the system. Once the transfer of all bottles has been simulated, the model moves on to the next time iteration (displacement of existing bottles).

In Scenario 2, the model cycles through every cell and runs a Bernoulli trial to determine if any particles (bottles) enter (or not). Particles enter a cell if \( xx \leq p_{\text{enter}}(i) \), where \( p_{\text{enter}}(i) \) is the probability of a particle entering reach \( i \). In this study, \( p_{\text{enter}}(i) \) and the number of particles emitted to each cell were assumed to be fixed (purely for illustration). Both could, however, be derived from physical and environmental characteristics. For example, they may increase in close proximity bridges or footpaths (where there is opportunity for the direct deposition of litter to the channel) and decrease in rural areas (where MMPW generation tends to be low, relative to urban areas). The number of particles that enter a cell could also be selected from a discrete probability distribution or related to MMPW generation data (e.g. Lebreton and Andrady, 2019).
Figure 2.7 – Numerical model decision network for (a) a single injection of plastic bottles at a point source, and (b) the injection of plastic bottles at multiple points in space and time. $B$ = bottle, $i$ = reach index, $t_{its}$ = time iteration, $j$ = reach index, $N$ = number of sub-reaches, $n_{bottles}$ = number of bottles, $n_{its}$ = number of time iterations, $p(i) = \text{probability of trapping in reach } i$, $p(j) = \text{probability of trapping in reach } j$, $p_{\text{enter}}(i) = \text{probability of plastic bottles entering reach } i$, $q(i) = \text{probability of being freed from a trapped state in reach } i$ and $xx = \text{a randomly generated number between 0 and 1}$
When a particle is emitted to a cell, it can either become trapped (when \[ xx \leq p(i) \]) or transported downstream. Particles that are transported downstream follow the same steps as those in Scenario 1 (except that they start in the seeded cell and not in the first cell in the system). Once the transfer of each seeded bottles has been simulated, the model moves on to the next cell (to see if any new particles enter and become trapped, or not) until all cells have been cycled through (see Figure 2.7b).

The stochastic process of displacing trapped particles is the same in both Scenarios. Again, a Bernoulli trial is performed in which particles are released if \( xx \leq q(i) \), where \( q(i) \) is the probability of being freed from a trapped state in reach \( i \). Here, \( q(i) \) was fixed (for illustration) but could be quantified in subsequent research and related to physical, environmental, trap and sub-reach characteristics. Once released from a trapped state, particles are assumed to move downstream and are then subjected to the same stochastic (trap or pass) processes as freshly injected ones. After the displacement (or retention) of existing particles has been simulated, the model moves on to the next time iteration (see Figure 2.7). In subsequent iterations existing particles are assumed to move through the system (or become trapped) and (in Scenario 2) new particles are injected. After all time iterations, model outputs are produced from recordings of: (1) the frequency of tagged particles in each cell, and (2) the number of particles that have left the system.

2.4.4 Model Outputs

Illustrative model outputs are presented in Chapter Four. However, there is potential for additional output options. The following outputs were produced here:

- Travel distance distribution (at different points in time),
- Residence time distributions in plastic accumulation zones
- Differences between travel distance distributions over time (similar to those illustrated in Figure 3.4; see Section 2.3.5)

The number of particles predicted to exit the system is also given (i.e. plastic bottle flux; the number of bottles that pass through the final cell). Particle flux is a potentially significant model output, especially at the catchment scale. At the tidal limit, this is a prediction of the plastic flux entering estuarine and marine environments.
2.4.5 Model Calibration and Comparison to Field Data

For model outputs to be representative of actual macroplastic debris transport, calibration is required. This is performed by adjusting selected parameters to improve the fit between modelled and measured data. $p(V)$, $a$, $b$ and $c$ were adjusted in this study and model predictions were compared with observed travel distance distributions from tracer experiments (see Sections 2.3.1 and 3.2). To match the observed distribution data, $p(FS)$ and $p(CB)$ were calculated for individual sub-reaches at 10 m resolution using Equations 2.2 to 2.4. A non-zero value of $p(V)$ was applied only if an overhanging tree was present in a sub-reach. Although $p(T)$ was calculated at 10 m resolution, the calculation of $p(T)$ is easily scalable to different sub-reach lengths. Note, however, that parameter values are likely to be scale-dependent – i.e. they will change with sub-reach length.

Modelled and measured travel distance distributions were compared using a two-sample Kolmogorov-Smirnov goodness-of-fit test (KS Test). This tests whether two samples come from the same cumulative distribution function (CDF). The KS test statistic ($D_{\text{Stat}}$) is equal to the greatest absolute difference between the modelled and measured CDFs. Optimal model fit is achieved for minimal $D_{\text{Stat}}$. When $D_{\text{Stat}} < D_{\text{Crit},\alpha}$ (where $D_{\text{Crit},\alpha}$ is the critical value at $\alpha$ and $\alpha$ is the significance level) it can be assumed that the samples come from the same distribution (and hence the model is a good fit). A $p$-value ($p$) for $D_{\text{Stat}} < 0.05$ suggests that the samples do not come from a population with the same distribution. Conversely, where $p > 0.05$ it can be assumed that the two datasets do come from a population with the same distribution – i.e. model fit is good (see Berger and Zhou, 2005 for more detail on the two-sample KS test). The two-sample KS Test was used as an alternative to the more popular two-sample chi-squared ($\chi^2$) test because $\chi^2$ does not allow expected frequencies to equal zero. This is important because in some sub-reaches the observed and or modelled frequency of trapped particles is zero.

Calibration was performed using an iterative combinatorial procedure involving multiple model runs and employing a range of parameter combinations. This method is also known as a parameter sweep (see Malleson, 2014) or calibration via factorial analysis (see Hamby, 1994). The calibration parameter space was sampled from $a = 0$ to 1.7 (with an increment of 0.1), $b = 1$ to 10 (with an increment of 1), $c = 1$ to 7 (with an increment of 1) and $p(V) = 0$ to 0.4 (with an increment of 0.05). This gave 11,340 different combinations. $D_{\text{Stat}}$ was recorded during each run. Each parameter combination was only run once. The optimal combination of parameters was that with the lowest value.
of $D_{\text{Stat}}$. However, given the stochastic nature built into the model, future work should involve multiple iterations of the combinatorial procedure to find the average value of $D_{\text{Stat}}$ for each combination. This was not feasible in this project due to limitations on available computing power and project time restrictions. The maximum proportion of bottles trapped in a 10 m sub-reach in the tracer experiments (out of the number of bottles available for trapping) helped to constrain the calibration parameter space of $a$ and $p(V)$. Since the maximum proportion of bottles trapped equalled ~35%, an upper limit of 0.4 was applied to the possible values of $p(\text{FS})$, $p(\text{CB})$, $p(V)$ and $p(T)$. The parameters $b$ and $c$ were constrained via trial and error with preliminary runs of the model (see Refsgaard and Storm, 1990).

2.4.6 Sensitivity Analysis

A one-at-a-time (or ‘local’) sensitivity analysis was conducted to ascertain which parameters exerted the most control over the model predictions. In this analysis one (calibrated) parameter is varied repeatedly while the others are held fixed at their optimal values (Hamby, 1994). Each parameter was varied by the same percentage (+/-10% to +/-100%) away from its optimum and $D_{\text{Stat}}$ was recorded for each run.

2.5 Chapter Summary

In summary, the travel distance distribution of macroplastic debris was determined with the use of tracer experiments along a 1 km reach of the River Sence, Leicester, UK. To investigate what controls the transport of macroplastic debris, a survey of the study reach was conducted, recording the location of overhanging trees and meander bends. It is assumed that once trapped, macroplastic debris remain in stable storage, unless there is a change in river flow conditions. However, to gain information about the ability of a trap to retain macroplastic debris, repeat surveys were conducted on four replicate experiments. The change in location of plastic bottles can then be used to evaluate trap retention. Additionally, a probabilistic travel distance approach for modelling the transfer of individual items of macroplastic through a river catchment is presented. This model explicitly considers the processes controlling plastic debris transport (unlike existing models) and can be used to evaluate the effect of trapping on riverine plastic delivery to the oceans. The modelled travel distance data was calibrated against tracer experiment data via factorial analysis. The goodness-of-fit between modelled and measured data was evaluated with the two-sample Kolmogorov-Smirnov test. Finally, to review uncertainty in the model outputs a local sensitivity analysis was performed.
3 TRACER EXPERIMENT RESULTS AND ANALYSIS

3.1 Introduction

There are two main components to this study: (1) the use of tracer experiments, and (2) the development of a framework for modelling the transfer of individual items of macroplastic debris. The main aim of the tracer experiments was to determine the travel distance distribution of macroplastic debris, and its controls. This chapter presents the results and analyses from the tracer experiments. The results and analyses from the modelling exercise are presented in Chapter Four and a discussion of results are provided in Chapter Five.

3.2 Travel Distance Distribution of Macroplastic Debris

In this study tagged plastic bottles were injected into an experimental study reach and their locations were recorded after 24 hours to produce a travel distance distribution. One experiment with six replicates (R1-R6) was conducted, each injecting 90 plastic bottles. Figure 3.1 illustrates the travel distance distribution for each replicate and for the combined data. The mean travel distance in each replicate ranged from 105 m in R3 to 357 m in R4, with an overall mean travel distance of 231 m. Maximum travel distances ranged from 680 m in R5 to 1,071 m in R6. The coefficient of variation of travel distances (CV) ranged from 0.54 in R4 to 1.41 in R5, with an overall CV of 0.94. The travel distance distributions were all positively skewed with skewness ranging from 0.46 in R1 to 2.13 in R3. In each replicate the tracer recovery rate ranged from 93% in R2 to 99% in R4 with an overall recovery rate of 96%. Bottles not retrieved from the channel were often found on the floodplain, presumably removed by external agencies (e.g. people, dogs and wildlife). These bottles were not included in the calculated travel distance distributions.

Figure 3.1 also illustrates the sub-reaches where most plastic bottles were trapped in each replicate and overall. There were clear ‘hotspots’ where macroplastic debris consistently tended to get trapped (e.g. 20, 30, 140, 250, 340 and 650 m D/S of the injection point). The sub-reaches where most plastic bottles were trapped were 30 m D/S for R5 (trapping 52% of plastic bottles), 40 m D/S for R3 (trapping 24% of plastic bottles), 140 m D/S for R1, R2 and R6 (trapping 20%, 20% and 28% of plastic bottles, respectively), and 340 m D/S for R4 (trapping 25% of bottles). Overall, the sub-reach where most plastic bottles were trapped was 140 m D/S (trapping 15% of plastic bottles).
Figure 3.1 – Plastic bottle travel distance distributions from six replicate experiments (R1-R6) and the combined data. $\bar{x}$ = mean travel distance, CV = coefficient of variation, n = number of bottles recovered.
There is some variability in plastic bottle travel distances between replicates. For example, at the point where most bottles were trapped (140 m D/S) the proportion of bottles trapped ranged from 2% to 28%. Figure 3.2 illustrates the same collated travel distance distribution as Figure 3.1 but with the addition of whiskers, which represent the standard deviation of the proportion of bottles stored in each sub-reach after 24 hours between replicates (σ). σ was particularly high 30 m D/S of the injection point (σ = 20.5) and was also high 10, 40, 140, 250 and 340 m D/S of the injection point (8.3 ≤ σ ≤ 9.9). σ was relatively low in sub-reaches where a low proportion of bottles were trapped – for example, for all sub-reaches where the proportion of bottles trapped was less than 2%, σ ≤ 2.3. Figure 3.2, therefore, illustrates the high degree of variability in plastic bottle travel distances between different replicate experiments.

It should be noted that the proportion of bottles trapped in each sub-reach appeared to be significantly affected by the proportion of bottles that were trapped in transfer. Where a high proportion of bottles were trapped upstream, fewer bottles were available for trapping (and vice versa). This made it difficult to identify consistent ‘trapping hotspots’. Despite the variability in travel distances and complications when sampling without replacement, some ‘trapping hotspots’ are evident in the travel distance distributions in Figure 3.1.

![Figure 3.2 – Combined travel distance distribution of plastic bottle travel distances. Points represent the proportion of bottles trapped in each sub-reach and whiskers represent the standard deviation of the number of bottles trapped in each sub-reach between replicates.](image-url)
3.3 Controls on the Travel Distance Distribution of Macroplastic Debris

It is hypothesized that the travel distance distribution of plastic bottles is mediated by the location and nature of trapping points (otherwise there would be an exponential decrease in the number of bottles with distance downstream; see Section 1.7 and Figure 1.12). An AD goodness-of-fit test confirmed that the data illustrated in Figure 3.1 does not follow an exponential distribution \((p < 0.05)\), thereby supporting the above hypothesis. To determine which trap characteristics were exerting control over bottle transport patterns, the location of overhanging trees and meander bends along the study reach were recorded (see Table 3.1; see also Section 2.3.4). Along the study reach nine overhanging trees and ten meander bends were identified. Figure 3.3 illustrates the same travel distance distribution as Figure 3.1, but with sub-reaches colour coded according to the presence of an overhanging tree or meander bend. Note that some traps extend over multiple 10 m sub-reaches and although each sub-reach is colour coded, some traps do not fill the entire sub-reach (i.e. they act at a specific point).

Table 3.1 – Location of overhanging tree and meander bend traps along the study reach (in distance downstream from the injection point). A reference to a photograph of each trap is provided (if available)

| Trap Type       | Distance Downstream | Photo Reference       |
|-----------------|---------------------|-----------------------|
| Overhanging Trees |                     |                       |
|                 | 8-25 m              | Fig. 2.2 Pic. 1       |
|                 | 80-90 m             | Fig. 2.2 Pic. 4       |
|                 | 125-140 m           | Fig. 2.2 Pic. 6       |
|                 | 238-254 m           | Fig. 2.2 Pic. 8       |
|                 | 635-645 m           | Fig. 2.2 Pic. 15      |
|                 | 678-682 m           | -                     |
|                 | 780-782 m           | -                     |
|                 | 820-840 m           | Fig. 2.2 Pic. 19 (Left)|
|                 | 855-890 m           | Fig. 2.2 Pic. 19 (Right)|
| Meander Bends   |                     |                       |
|                 | 80-90 m             | Fig. 2.2 Pic. 4       |
|                 | 100-110 m           | -                     |
|                 | 120-140 m           | Fig. 2.2 Pic. 6       |
|                 | 210-230 m           | -                     |
|                 | 310-350 m           | Fig. 2.2 Pic. 10/11   |
|                 | 425-460 m           | Fig. 2.2 Pic. 13      |
|                 | 555-570 m           | -                     |
|                 | 630-650 m           | Fig. 2.2 Pic. 15      |
|                 | 700-735 m           | -                     |
|                 | 885-1045 m          | Fig. 2.2 Pic. 20      |
Figure 3.3 – Plastic bottle travel distance distribution, colour coded according to the presence or absence of overhanging tree and meander bend traps

Just over half (59%) of the plastic bottles were trapped in sub-reaches where an overhanging tree was present. Nearly half (43%) were trapped along meander bends. In some sub-reaches both overhanging trees and meander bends were present and these appear to have acted together to determine trapping behaviour. Trapping hotspots always coincided with the presence of overhanging trees (except the trapping hotspot 340 m D/S) or a meander bend (except the trapping hotspots 30 and 250 m D/S; see Figure 3.3). There was, therefore, a clear relationship between the trapping of macroplastic debris and the presence of an overhanging tree or meander bend. No bottles were trapped in the overhanging tree and meander bend furthest from the injection point (855-890 m and 885-1045 m D/S, respectively). This was almost certainly due to the fact that far fewer bottles reach this point and were available for trapping (only three and two bottles passed the overhanging tree and meander bend furthest from the injection point, respectively, over all six replicates).
3.4 Trap Effectiveness

It was assumed that once trapped, bottles mostly remain in storage unless there is a change in river flow conditions (see Section 2.3.5). However, to gain information about trap retention, the locations of plastic bottles were recorded multiple times over the 24-hour survey period of four replicate experiments. Figure 3.4 illustrates the observed changes in the frequency of bottles at different locations between repeat surveys, which were used to evaluate retention. Positive values indicate a net increase and negative values indicate a net decrease in the number of bottles stored at a particular location between repeat surveys. Differently coloured bars illustrate different replicate experiments. In general, Figure 3.4 illustrates a dynamic system where bottles are transported between trapping points even under steady discharge conditions (contrary to the assumed hypothesis above).

Trapping hotspots (overhanging trees and meander bends) generally indicate high trap retention (see Figure 3.3). Figure 3.4, however, illustrates that not all bottles that were captured were retained at these hotspots. For example, 36 bottles were trapped 140 m D/S in R1 after one hour but only 18 bottles were recorded there after 24 hours (i.e. at least 18 bottles were not retained over 24 hours; see Figures 3.1 and 3.3). The release of macroplastic debris after initial trapping was observed during each replicate experiment.

There did not appear to be a decrease in the rate of plastic bottle exchange over time in most traps (i.e. bottles were continually released after initial trapping and did not appear to enter stable storage). For example, although there was no change in the number of bottles trapped 20 m and 140 m D/S between the R3 13:00 and 15:00 surveys, five bottles were released from both sub-reaches between the 15:00 and 08:00 resurveys (see Figure 3.4). It is noted, however, that each replicate experiment was only conducted over 24 hours and there may be a decrease in the rate of exchange over a greater time period (i.e. bottles may enter stable storage beyond 24 hours).
Figure 3.4 – Difference in the number of bottles trapped in 10 m sub-reaches between repeat surveys in the same replicate experiment (R). The start time of resurveys used to calculate the difference is provided in the top right of each graph.
It was difficult to determine the number of bottles that were captured and retained by a trap with distance downstream and without high temporal resolution information about the location of individual bottles. Although resurveys provide information about the number of bottles stored in each trap and change over time, it is not possible to determine which bottles were captured or retained. For example, a bottle can be captured but released very quickly (thus not recorded between repeat surveys) or a bottle may be released from storage but replaced by another bottle that was released upstream (resulting in no net change in the number of bottles recorded between repeat surveys).

In addition, it should be noted that the method used to count the number of bottles in each sub-reach in each resurvey (i.e. without removing them from the river channel) is subject to observer uncertainty and error. If all plastic bottles were accurately accounted for, the sum of the difference between the number of bottles trapped in each sub-reach between repeat surveys should equal zero (i.e. any ‘loss’ of plastic bottles from one sub-reach should be accounted for by a ‘gain’ in plastic bottles in a downstream sub-reach). The actual observed sum of differences ranged from -3 (difference between the R1 09:00 and 09:45 surveys) to 4 (difference between the R2 12:30 and 13:30 surveys). Observer error and uncertainty in visual observation methods has been recognised in other plastic transport studies, but they are also recognised as simple and consistent (see Van Emmerik and Schwarz, 2020 and Section 1.4).

3.5 Chapter Summary

In this chapter the results of tracer experiments were described. This data represents a new and fundamentally different approach to investigating plastic transport in rivers. It also helps to characterise the details of macroplastic debris transport at reach scale for the first time. The objectives of the tracer experiments were to determine the travel distance distribution and elucidate the primary factors controlling this distribution. The data show that travel distances for macroplastic particles are likely to be low and variable. Travel distance distributions were positively skewed and appeared to be controlled by the location and nature of different channel features (particularly by the presence of overhanging trees and meander bends). Repeat surveys illustrated that plastic transport is dynamic (if intermittent) with transient transport between sites of temporary storage even under steady discharge conditions.
4 MODELLING EXERCISE RESULTS AND ANALYSIS

4.1 Introduction

In this study a simulation modelling framework was constructed to predict the travel distances of individual items of macroplastic debris (along with the use of tracer experiments; see Chapter Three). The approach attempts to define a probability of trapping for a given point in a river over a given time, \( p(T) \). This is calculated from a priori assumptions about the relationship between \( p(T) \) and trap and reach characteristics. The injection, transfer and trapping of plastic bottles is simulated by the model (see Section 2.4). This chapter details the results from a calibration exercise and sensitivity analysis with example model outputs.

4.2 Model Calibration

To calibrate the numerical model, predicted travel distance distributions were compared with measured data from tracer experiments (see Chapter Three). The two-sample KS test statistic (\( D_{\text{Stat}} \)) was used as the goodness-of-fit statistic. The calibration process employed was a factorial analysis, in which, \( D_{\text{Stat}} \) was calculated for 11,340 combinations of \( a, b, c \) and \( p(V) \). Note that an upper limit of 0.4 was applied to all possible values of \( p(T) \). The optimum calibration parameters were \( a = 0.3, b = 9, c = 2 \) and \( p(V) = 0.15 \) (\( D_{\text{Stat}} = 0.107 \)). Note, however, that each parameter combination was only run once. Given the stochasticity built into the model, the modelled CDF, and by extension \( D_{\text{Stat}} \), will change with every run. Indeed, several different parameter combinations yielded similar values of \( D_{\text{Stat}} \). For example, \( D_{\text{Stat}} \) equalled 0.110 when \( a = 0.5, b = 8, c = 4 \) and \( p(V) = 0.15 \). The phenomenon where several combinations of parameters result in similar goodness-of-fits is known as equifinality (see Beven and Freer, 2001; Von Bertalanffy, 1968).

Figure 4.1 illustrates the modelled optimum prediction of plastic bottle frequency with distance downstream. The predicted distribution is after one time iteration, which is equivalent to 24 hours in this case (matching the observed data from tracer experiments). Figure 4.2 compares the modelled and observed CDFs after 24 hours. The observed data is the combined data from six replicate experiments (see Chapter Three and Figure 3.1). The model simulates the trapping of macroplastic debris in overhanging trees well. This is because trees represent clearly identifiable and discrete trapping points which often have a relatively high trap effectiveness (i.e. they act as ‘hotspots’). In both the modelled and observed data trapping hotspots relating to overhanging trees were identified 30, 90, 140, 250 and 650 m D/S of the injection point (see Figures 3.3 and 4.1;
see also Table 3.1 and Figure 1.1). However, trapping along meander bends was not well simulated. For example, no bottles were predicted at the meander bend 340 m D/S but several bottles were trapped here during the tracer experiments (7% across all replicates). The average travel distance and dispersal of plastic debris was similar for the modelled and measured data ($\bar{x} = 238$ m and 231 m, respectively; CV = 0.95 and 0.90, respectively; see Figures 4.1 and 3.1). Additionally, modelled bottles were displaced within the same range as observed tracer bottles (only 7 out of 521 simulated bottles travelled beyond the maximum observed travel distance). The modelled travel distance distribution skew was, however, substantially higher for the tracer distribution (skew = 5.16 versus 1.16). Figure 4.2 illustrates a relatively good fit between the modelled and measured CDFs. Nevertheless, there was a statistically significant difference between the two distributions ($D_{\text{Stat}} > D_{\text{Crit}, 0.05}$, $p = 5.11 \times 10^{-3}$) which suggest that formally it should be concluded that the data are not from the same population distribution (see Section 2.4.5).

![Figure 4.1](image_url)

*Figure 4.1 – Travel distance distribution for the optimum combination of calibration parameters ($a = 0.3$, $b = 9$, $c = 2$ and $p(V) = 0.15$) after one time iteration (equivalent to 24 hours). $\bar{x}$ = mean travel distance, $CV$ = coefficient of variation of travel distances, $n$ = number of bottles*
Figure 4.2 – Cumulative distribution of predicted and observed plastic bottle travel distances after 24 hours. The observed data is the combined data from six replicate experiments

4.3 Sensitivity Analysis

To assess the influence of different parameters on model outputs, a one-at-a-time sensitivity analysis was conducted (see Hamby, 1994 and Section 2.4.6). The value of each calibrated parameter was varied by the same percentage away from its optimum (i.e. the best fit value from the calibration) from +/-10% to +/-100%, while other parameters were held fixed at their optimal values. Note, however, that $b$ could not be varied by -100% as this would produce a divide by zero error. $D_{Stat}$ was recorded in each run. The results are shown in Figure 4.3.

The model was most sensitive to parameters $b$ and $c$ (in the negative direction at least). These parameters control the relationship between channel width and the probability of trapping along channel banks ($p(CB)$; Equation 2.4). An increase in both $b$ and $c$ result in a decrease in $p(CB)$, and vice versa. When $b$ and $c$ were varied in the positive direction, however, changes in $D_{Stat}$ were less sensitive and $D_{Stat}$ did not vary by much (see Figure 4.3). The best fit between the modelled and measured data (i.e. when $D_{Stat}$ was low) was when the probability of trapping along channel banks, $p(CB)$, was calibrated to be very low. This might suggest that trapping along channel banks is not the most important process to represent in the model. Further investigation, especially along rivers with varying channel widths, is required.
The model fit was least sensitive to parameter $a$. This parameter controls the relationship between channel sinuosity ($S$) and the probability of trapping in flow separation zones ($p(FS)$; Equation 2.2). An increase in $a$ results in an increase in $p(FS)$, and vice versa. Low changes in $D_{Stat}$ in response to changes in $a$ were largely caused by $S$ being very close to one in every cell ($1 \leq S \leq 1.32$). By extension, $p(FS)$ was close to zero in every cell (see Equation 2.2). This also explains the poor simulation of trapping at meander bends because all sub-reaches were characterised as straight (rather than sinuous or meandering; see Leopold et al., 1964; Mueller, 1968). The use of $S$ as a proxy for the number of meander bends along the study reach, therefore, may not be appropriate (at least at 10 m resolution). At a coarser resolution, $S$ may identify meander bends (making it a good proxy) and simulate trapping well.

Finally, the model fit appeared to be relatively sensitive to $p(V)$, meaning that even small differences in $p(V)$ affected predictive quality. This is because of the important role of vegetation (especially overhanging trees) in the trapping of floating macroplastic. If the probability of trapping at these features was reduced (i.e. reduced $p(V)$), the model tends to underestimate trapping in general (and vice versa if $p(V)$ is increased at these locations).

![Figure 4.3](image_url)

**Figure 4.3** – Results from a one-at-a-time sensitivity analysis, showing the change in model goodness of fit (deviation between the predicted and observed CDFs, $D_{Stat}$) for the parameters $a$, $b$, $c$ and $p(V)$. High change in $D_{Stat}$ for a given percentage increment in the parameter value away from its optimum (0%) indicates a high degree of sensitivity to that parameter.
4.4 Model Outputs

Several key outputs are provided in this study to demonstrate the utility of the model approach. However, there is significant scope for the production of supplementary outputs. Overall, the predictive ability of the model is still limited by an imperfect understanding of the controls on macroplastic debris transport and the small size of the calibration dataset. Figure 4.4 illustrates the predicted travel distance distribution at three different time iterations after injection (the distribution predicted after the first time iteration is illustrated in Figure 4.1). The difference between the number of bottles trapped in each sub-reach between the iterations shown is also displayed in the lower panels (similar to those illustrated in Figure 3.4). The probability of being freed from a trapped state, \( q(i) \), was set to 0.5 for all sub-reaches (for illustrative purposes; see Section 2.4.3). However, this is highly uncertain and more research is required on the residence time of macroplastic debris in storage, which controls \( q(i) \). Figure 4.4 visually illustrates how the dispersal of macroplastic would behave under this assumption. There is a predicted increase in average distance travelled over time and an increase in the spread of bottles with distance. There is also a gradual increase in the number of bottles which have left the study reach, \( n_{\text{OUT}} \) (see Section 2.4.4). For example, \( \bar{x} \) equalled 413, 633 and 779 m and \( n_{\text{OUT}} \) equalled 84, 286 and 471 bottles in the second, fifth and tenth time iterations, respectively.

**Figure 4.4** – Modelled travel distance distributions for second, fifth and tenth time iterations (its). The difference between the number of bottles trapped in each sub-reach between time iterations is also provided in the lower panels.
The difference in the predicted frequency of particles in each sub-reach between iterations in Figure 4.4 (lower panels) is derived from absolute predictions of frequency. However, as with Figure 3.4, Figure 4.4 alone does not allow the user to determine which bottles have been captured or retained in a cell (see Section 3.4). This information is required to determine residence time. To understand plastic accumulation in the marine and estuarine environments, an understanding of the behaviour and associated residence times of plastic in rivers is required (see Section 1.6). A residence time distribution can however be produced in the model from recordings of where individual bottles were in each time iteration. Figure 4.5 illustrates an example residence time distribution that was produced by the model for the sub-reach 30 m D/S of the injection point. In this example, just over half (55%) of the bottles that were trapped had a residence time of one day, 20% had a residence time of two days and 12% had a residence time of three days (and so on; see Figure 4.5). This type of graph cannot be produced from tracer experiments without very high temporal resolution information about the location of individual bottles (see Section 3.4). The approximate exponential decrease in the residence time of plastic bottles is expected given that $q(i)$ was fixed.

Figures 4.4 and 4.5 illustrate model outputs when injecting several plastic bottles as a single injection (i.e. a simulation of the tracer experiments described in Chapters Two and Three). The model can also simulate the behaviour of plastic particles when they are assumed to enter the river system at multiple points in space and time (which is more representative of a 'real-world' input of litter to river channels; see Sections 1.6.1 and 2.4.3). Figure 4.6 shows an example distribution of plastic bottle storage for three
points in time under this scenario. Note that the number of bottles stored in each cell and the travel distance distribution of plastic bottles are slightly different. This distribution is affected by the probability of plastic emission to a cell \( p_{\text{enter}}(i) \) and the number of bottles that are set to enter that cell. Here, \( p_{\text{enter}}(i) \) was set to 0.25 for every cell and the number of bottles that entered each cell during an emission event was set to 5 (for illustrative purposes only; see Section 2.4.3).

The model outputs from this scenario (as illustrated in Figure 4.6) could be compared to ‘real world’ spatial patterns of plastic accumulation. In this case there is an accumulation zone 790 m D/S of the injection point. Long-term surveys could ascertain whether this point really acts as a zone of plastic accumulation or not. This could be used to target mitigation efforts. With realistic estimates of the location, volume and frequency of emission, model predictions in this mode could also be used to estimate riverine plastic flux (with \( n_{\text{OUT}} \)) at the catchment scale.

![Figure 4.6 – Bottle storage distribution for the diffuse-injection of plastic bottles at different points in time. The difference between the number of bottles trapped in each sub-reach between time iterations is also shown in the lower panels. \( \text{its} = \text{time iteration (equivalent to 24 hours in this case), } n = \text{number of bottles in the system, } n_{\text{OUT}} = \text{number of bottles that left the study reach (plastic bottle flux)} \)
4.5 Chapter Summary

In this chapter, example outputs of a new and fundamentally different approach for modelling plastic debris transport have been presented (based on the probability of trapping). This model was calibrated against tracer experiment data. Although there was a statistically significant difference between the cumulative measured and modelled travel distance distributions, the general pattern in the observed behaviour of plastic was represented reasonably well by the model, largely due to the presence of discrete and easily identifiable trapping points at overhanging trees. A sensitivity analysis was performed to identify which aspects of the model exerted the most control over predicted patterns. This showed: (1) goodness-of-fit was greatest when the probability of trapping along channel banks was calibrated to be very low (suggesting that this process may not be the most important to represent in the model); (2) changes in the parameter controlling trapping in flow separation zones did not substantially affect goodness-of-fit (although trapping in said flow separation zones was poorly simulated by the model), and (3) a representative value of trapping in overhanging vegetation is important to the overall predictive quality of the model.

Other potential model outputs were also illustrated, although observed data were not available which these predictions could be compared to. Many questions remain about the nature of macroplastic debris behaviour and further research is needed. Additional empirical evidence will allow the model described here to be refined in future. The model has the potential to improve the predictions of riverine plastic flux at the catchment scale and to identify zones of accumulation (hotspots) which could be targeted for mitigation efforts.
5 GENERAL DISCUSSION AND CONCLUSIONS

5.1 Introduction

This thesis details a new approach to investigate plastic transfer in rivers, which considers the travel distances of individual items of macroplastic debris. It combines an experimental component, in which macroplastic tracers were tracked in a small reach of the River Sence, with the construction of a numerical model, which is intended to act as a conceptual framework to describe and explain the observed phenomena. Unlike many previous approaches to studying fluvial macroplastic transport, the thesis explicitly considers the main processes controlling plastic displacement, trapping and subsequent release. This chapter summarises the key results from the tracer experiments and modelling exercise, evaluates the approaches used and provides recommendations for future research.

5.2 Travel Distance Distribution of Macroplastic Debris

The transfer of macroplastic debris in rivers is complex and is likely to be controlled by many factors. These include plastic properties, the magnitude and variability of river discharge (which is a function of catchment hydrology), hydraulic geometry (the relationships between discharge, velocity and channel shape), transport pathways and degradation mechanisms and reach characteristics, which control the effectiveness of possible plastic traps (see Section 1.6 and Figure 1.6). Unlike the transport of solute and suspended matter (and to some extent microplastic; see Cook et al., 2020), macroplastic items are not well mixed in the flow. Low-density plastic or air-filled items like plastic bottles float on the water surface and this affects their transport characteristics and propensity for trapping. In this study, plastic bottle transport appeared to be systematically controlled by the location and nature of specific channel features (particularly the presence of overhanging trees and meander bends). Typical travel distances over a 24-hour period were very low, suggesting that plastic bottles (and, by extension, floating macroplastic in general) were captured relatively easily. This low distance of movement was initially surprising as floating debris was expected to be transported at approximately the mean velocity of flow.

Although plastic appeared to trap easily, repeat surveys indicated that not all bottles were retained long-term. Indeed, transport was dynamic with intermittent exchange of items between sites of temporary storage, even under steady discharge conditions. This went against the hypothesis that bottles would mostly remain in stable storage unless there is a change in river flow conditions. There was also no evidence
that bottles were intermittently transported until they entered stable storage (though it was noted that the experiments in this study were conducted over a relatively short period of 24 hours). The release of bottles from sites of temporary storage under steady flow conditions was probably caused by random fluctuations in current or wind speed and direction. Perhaps more importantly, the results from the tracer experiments demonstrated the feasibility of using a travel distance approach to understand and investigate plastic transfer in rivers.

5.3 Modelling Plastic Transport Using a Travel Distance Approach

5.3.1 Evaluation of Model Outputs

The understanding gained from tracer experiments of plastic transport controls was used to construct and calibrate a numerical model to simulate the transfer, trapping and release of plastic debris. The calibrated model outputs presented in Chapter Four demonstrate the feasibility of using a travel distance approach to model macroplastic transfer. The general pattern in the observed behaviour of plastic is described reasonably well by the model, largely due to the presence of discrete and easily identifiable trapping points (especially overhanging trees). However, the trapping of bottles at meander bends was less well simulated, which resulted in statistically significant differences between the measured and modelled travel distance distributions. Furthermore, whilst the experimental data was used to calibrate the model, the model was not independently tested (validated) in another reach or using data which were not used for calibration. Nevertheless, the model performance did suggest that there is wider potential to utilise the same approach to model macroplastic transfer in rivers more generally – even up to the catchment scale (see Section 5.6).

At present, the predictive quality of the model is restricted by: (1) an imperfect understanding of what controls plastic transport in rivers; (2) a poor understanding of the relationships between reach characteristics and the probability of trapping, \( p(T) \); (3) the limited size of the calibration dataset; (4) the limited time and spatial scales over which macroplastic was monitored, and; (5) the characteristics (size, shape, material properties) of the plastic items used in the tracer experiments. To improve the fundamental understanding of processes controlling displacement (and thus predictive quality), similar and refined tracer experiments should be used. More details on how these experiments can be refined to understand transport processes are provided in Sections 5.4 and 5.5. Given the paucity of other studies reporting plastic travel distances in rivers, additional work is needed to supplement the preliminary data reported here in
a range of reaches with different characteristics. This would allow the model to be validated. This is discussed in more detail in Section 5.6.

The model outputs reported in Chapter Four illustrate the potential of the model to describe and explain macroplastic travel distance distributions under both experimental (tracer injection) and (by extension) ‘real world’ (multiple emission in space and time) exposure scenarios. One key parameter which needs to be further investigated is the probability of being freed from a trapped state, \( q(i) \). In the work described in Chapter Four, \( q(i) \) was assumed to be constant at 0.5 for all sub-reaches. In reality, a uniform probability of release from all trap types is almost certainly unrealistic. One alternative option could be to select \( q(i) \) from a statistical distribution or relate \( q(i) \) to the characteristics of each trap and environmental conditions (e.g. stage and wind conditions). Again, additional refined tracer experiments could be used to empirically quantify the residence time distribution of different trap types. These experiments would need to be run over longer time periods and the residence time of individual bottles would need to be measured. Further details on how such tracer experiments might be conducted are discussed in Section 5.4.

The predicted travel distance distribution under a ‘real world’ plastic exposure scenario (where plastic bottles enter the river at multiple points in space and time) is sensitive to the probability of plastic emission to a cell \( (p_{\text{enter}}(i)) \) and the number of bottles that enter that cell during an emission event. In the work described in Chapter Four, \( p_{\text{enter}}(i) \) was arbitrarily (for illustrative purposes) set uniformly at 0.25 and the number of bottles entering each cell was uniformly set to 5 (see Sections 2.4.3 and 4.4). Neither of these assumed values represent actual inputs of plastic debris to river channels, which will depend on MMPW generation in the area concerned and on physical and environmental characteristics (see Section 1.6.1). Accurate estimates of the location, frequency and volume of plastic emitted to river channels are essential for estimating real world plastic flux and for identifying plastic accumulation zones. This is because riverine plastic flux is limited by supply (and not by transport capacity; see Charlton, 2008).

Although the empirical evidence base remains weak, there is likely to be a strong link between \( p_{\text{enter}}(i) \) and the probability of MMPW transport from land to rivers (used in Meijer et al., 2019 to calculate the probability of MMPW emission, \( p(E) \), from 3 x 3 arc-second grid cells; see Section 1.3). Meijer et al. (2019) however, do not explicitly consider the input or transport of individual items of macroplastic debris. Instead, \( p(E) \) is
multiplied by the amount of MMPW assumed to be generated in each 3 x 3 arc-second grid cell. Future work should explicitly investigate and quantify the specific locations, frequency and volume of macroplastic debris emission in different contexts. The input of macroplastic debris to river channel is likely to be directly related to human activity (Best, 2019). For example, emission may increase in close proximity to bridges or footpaths (where there is opportunity for the direct deposition of litter to the channel) and decrease in rural areas (where MMPW generation tend to be low relative to urban areas; see Section 2.4.3).

5.3.2 Model Sensitivity

To assess which model parameters exerted the most control over predicted model outputs, a sensitivity analysis was conducted, using the one-at-a-time ('local') method (see Section 2.4.6). Varying degrees of sensitivity were observed but the model fit appeared to be particularly sensitive to changes in the probability of trapping in overhanging vegetation, $p(V)$. This highlights the important role of vegetation in the transport of macroplastic debris and the need to apply a representative value of $p(V)$. Even small changes in $p(V)$ will have substantial effects on the predictive quality of the model (see Section 4.3).

Conversely, model fit was not sensitive to changes in the parameter that controlled the relationship between channel sinuosity and the probability of trapping in flow separation zones. It was noted, however, that plastic accumulation in these zones (along meander bends) was poorly simulated by the model. This may be due to the fact that the sinuosity index ($S$) is a poor proxy for the number of meander bends along the study reach (at least at 10 m resolution; see Section 4.3). At a coarser resolution, $S$ may identify meander bends and better simulate trapping. The sensitivity analysis also highlighted that model fit was greatest when a low value was assigned to the probability of trapping along channel banks, $p(CB)$. This suggests that trapping along banks is not the most important process affecting plastic transport. The formula adopted for $p(CB)$ assumed that trapping is an inverse function of channel width (see Equation 2.4). However, the evidence for this assumption is currently weak and additional research in rivers with varying channel widths is needed to better establish general patterns (see Section 5.6). As overhanging trees appear to act as one of the most important macroplastic debris traps, channel width may also play an inverse role in the effectiveness of these features as traps. Again, this should be investigated with rivers of varying channel width.
Note that calibration of the model is likely to be scale-dependent and parameter values will almost certainly vary at different scales. In this study the probability of trapping, $p(T)$, was calculated for 10 m sub-reaches and over 24 hours. The spatial and temporal resolution defined in the model approach is, however, intentionally broad (i.e. it is defined as $p(T)$ per unit distance and time; see Section 2.4.2) to allow for scalability. The opportunities and challenges in scaling up the model (i.e. to the catchment scale) are discussed in more detail in Section 5.6.

In the sensitivity analysis, the model was run just once when each calibrated parameter was varied from its optimum (see Section 2.4.6 and Hamby, 1994). However, given the stochasticity built into the model, the model predictions and, hence, the KS test statistic ($D_{\text{Stat}}$) change with every run (even when the same parameters are used). Therefore, future work should run the model many times to find an average value of $D_{\text{Stat}}$ and better define the uncertainty envelope. This is essentially a Monte Carlo simulation (see Rubinstein and Kroese, 2016; Whelan et al., 1999). Similarly, multiple iterations of the combinatorial procedure of the model calibration should also be conducted. This was, unfortunately, beyond the scope of this thesis, given limitations in available computing power and the time constraints of the project.

The two-sample KS test was used as the principal quantitative method of measuring goodness-of-fit between predicted and observed distributions in both model calibration and in the sensitivity analysis. Although the KS test is used to determine if two samples come from the same population distribution, $D_{\text{Stat}}$ was used here as a simple method to measure goodness-of-fit only. This is because the threshold to accept the null hypothesis (that both samples come from the same population distribution) is relatively stringent (see Berger and Zhou, 2005 and Section 2.4.5). Future work should compare the predicted and observed distributions with additional goodness-of-fit measures, which might include correlation, as a measure of association (Draper and Smith, 1966) and the root mean square error, as a measure of coincidence (Addiscott and Whitmore, 1987). Note that although, in principle, the two-sample Chi-squared ($\chi^2$) test could be an alternative metric for the proximity of the two distributions, but it does not allow expected frequencies to equal zero and is, therefore, unsuitable (see Section 2.4.5). Similarly, the two-sample Anderson-Darling (AD) test is also probably unsuitable as it is severely affected by ties (many identical values; see Janssen, 1994).
5.3.3 Model Use and Potential

If accurate values for: (1) the probabilities of trapping; (2) the probability of being freed from a trapped state, and; (3) the amount and locations of plastic entering the river channel can be assumed, then the model can be used as is to: (1) predict the travel distance distributions of macroplastic debris over different time periods; (2) identify plastic accumulation zones (‘trapping hotspots’), which can be used to target mitigation efforts; (3) estimate the residence time of plastic debris in rivers, and; (4) eventually calculate riverine plastic flux. There is, however, significant scope for the production of supplementary model outputs. For example, a georeferenced illustration of plastic bottle storage and flux could be produced (similar to Figure 1.1, which illustrated the mass of plastic debris flowing into the oceans each year). This could be useful to visualise areas of plastic accumulation and to target mitigation efforts/clean-up operations. Additionally, tributary inputs of plastic debris could be built into the model. The opportunities and challenges to scaling up the model (i.e. to the catchment scale, which would include tributary inputs) are discussed in more detail in Section 5.6. Another output option which could be calculated is an estimate of the amount of time taken for plastic debris to be naturally ‘flushed’ from a river if supply is cut. Again, this would be helpful to inform policy decisions and target mitigation efforts. The latter would critically depend on a solid empirical basis for \( q(i) \).

5.4 Variability and Understanding the Controls on Travel Distance

5.4.1 Variability

The macroplastic travel distances observed in the tracer experiments were highly variable. Given that stochasticity is an inherent feature of natural and river systems, this was unsurprising. However, to minimise (or at least account for) this variability, carefully controlled (and as far as possible replicated) experiments were conducted. For example: (1) the experiments were conducted under approximately steady flow conditions; (2) bottles were injected individually to ensure they travelled independently of each other, (3) temporary mesh fencing was erected at the end of the study reach to avoid losses at the downstream end and aid a high total recovery rate, and; (4) a large sample size was used in several replicated and independent experiments. Variability did, however, make it difficult to investigate the controls on plastic bottle travel distances (in addition to complications when sampling without replacement; see Section 3.2). The sample size and number of replicates used in this study were largely constrained by logistics. In future work, very large sample sizes indeed may be required to correctly identify the importance of different controlling processes and to unambiguously identify trapping hotspots. Hassan et al. (1991) also concluded that large sample sizes would be required to identify
the form of particle displacement distributions (though this was for gravel – not plastic debris).

The general understanding of the controls on macroplastic transport in rivers has been improved in this study. However, significant knowledge gaps still exist, especially around (1) the relationship between reach characteristics and \( p(T) \) and (2) controls on plastic transport that were not investigated (e.g. different plastic sizes, shapes and properties, flow conditions and vegetation characteristics). These knowledge gaps can be narrowed with additional (and refined) tracer experiments (while also avoiding the need for very large sample sizes). The improved understanding from such experiments would lead to improvements in the numerical model. The proposed refined experiments are described below (through to Section 5.6).

### 5.4.2 Understanding the Controls on Travel Distance

To better understand the controls on macroplastic travel distances, and to better define the relationships between reach characteristics and the probabilities of trapping, tracer experiments should be conducted to investigate trapping at a **single point** (as opposed to trapping over a **series of points**). This would involve injecting plastic bottles immediately upstream of a trap and recording how many bottles are captured. It would also reduce complications when sampling without replacement. The decay in the number of bottles over time could also be recorded to understand trap retention and allow a better quantitative estimate for the probability of release from a trapped state, \( q(i) \). This type of refined experiment would also make it easier to investigate specific trap characteristics (e.g. vegetation roughness and plant architecture).

Alternatively, the experiments described in Chapter Three could be run over longer time periods (long enough for all particles to pass through all traps). Although regular resurveys can be used to record the location of plastic bottles long-term, they are unlikely to be practical (especially over larger spatial scales) given the potentially high residence time of plastic debris in rivers and the resource intensive nature of resurveys. More practically, the location of plastic bottles could be determined by placing GPS trackers inside bottles. Such trackers would provide high-temporal resolution data and could be used to further scrutinise transfer dynamics (e.g. step length and residence time). The residence time of plastic debris in rivers is of primary interest because it directly controls riverine plastic flux and can be used to quantify \( q(i) \) (see above). Disadvantages of GPS trackers include their relatively high procurement and running costs and limited accuracy. There is also a trade-off between battery life and temporal...
resolution. For example, the Logistimatics Micro-299 shipment tracker (Greensboro, NC, USA; a typical GPS tracker that is small enough to fit inside a plastic bottle) has a battery life of one month when its location is reported every hour, but a battery life of just two to three days when its location is reported every 10 minutes.

5.5 Limitations

The observed and predicted patterns of plastic debris displacement along the study reach are only representative of plastic bottles (not other types of macroplastic debris), low and steady flow conditions and winter vegetation characteristics. Tracer experiments with other types of commonly discarded macroplastic debris (e.g. carrier bags, straws and crisp packets) should also be conducted to understand how physical characteristics affect displacement and how the travel distance distributions of these particles compare to plastic bottles. The distributions are likely to be different since transport, trapping and release is probably dependent on these characteristics (e.g. density, buoyancy, shape and size; see Section 1.6.2). Low-density plastics and items which contain trapped air (e.g. sealed plastic bottles) are likely to travel further than high-density plastics because they are transported by surface transport mechanisms (i.e. with the river current and by wind; Ryan, 2015; Van Emmerik and Schwarz, 2020). Surface transport mechanisms are likely to be stronger relative to those lower in the water column (e.g. suspension, saltation and traction; see Figure 1.7), which will be more important for the transport of higher density items (see Section 1.6.2 for more details). The improved understanding of the displacement of other types of plastic debris from tracer experiments will inform their simulation in the numerical model.

In order to understand the role of stage discharge in the transport of plastic debris, tracer experiments should also be conducted along the same study reach (and others) under different flow conditions. These include different (steady) discharges and unsteady discharges (i.e. over storm hydrographs). Again, GPS trackers could facilitate data collection in both cases. GPS would be particularly helpful for unsteady discharges (where high temporal resolution data will be needed). The location of bottles could be compared with high-resolution stage data (either from existing gauging stations or with the installation of a water level data logger, e.g. TD-Diver; van Essen Instruments, Delft, the Netherlands). It is uncertain how travel distances will vary with different discharges. Contrary to the general relationships expected for sediment transport, it cannot necessarily be assumed that an increase in discharge will result in a proportionate increase in travel distance for macroplastic debris. This is because the interaction
between trapping behaviour and river stage is likely to be complicated (see Section 1.6.4 for more details).

Finally, the impact of seasonal changes in vegetation characteristics should be investigated by conducting experiments at different times of year (especially during the summer growing season). This was the original intention in this project but fieldwork beyond March 2020 was cancelled due to the COVID-19 pandemic. It is hypothesized that (cet. par.) the trapping of macroplastic debris by riverine vegetation will be greater (and thus mean travel distances shorter) during the growing season (when vegetation density and cover are greater and vegetation structure is more complex; see Section 1.6.3). Liro et al. (2020) suggest that the change in vegetation roughness caused by a seasonal change in vegetation characteristics and floristic composition favours plastic storage. Therefore, during the growing season, macroplastic debris is more likely to be trapped, relative to the non-growing season. However, this suggestion is not corroborated by any empirical (or even theoretical) evidence. In any case, it is plausible to assume that trends in the travel distance distribution will be complex and challenging to identify, making it difficult to draw firm conclusions. Ideally, teasing out the role of individual factors can be best done via the isolation of that factor (see Blair and McPherson, 2009). However, this is difficult for vegetation characteristics in the field because other factors controlling displacement may also change seasonally. Thus, the impact of changes in vegetation (and other) characteristics may be easier to evaluate using flume experiments (where conditions can be controlled, although somewhat abstract and idealised).

5.6 Validation and Scalability

This study explored the behaviour of macroplastic in a specific reach of the River Sence. Given that plastic transport appears to be systematically controlled by the location and nature of specific channel features, plastic transport and retention is likely to differ in different reaches. To compare patterns of displacement and to derive a more general understanding of macroplastic transport, additional tracer experiments should be conducted along reaches with: (1) similar characteristics (i.e. UK agricultural lowland rivers), and; (2) different characteristics (e.g. mountain or wide rivers). These experiments will deepen the understanding of controlling processes and inform the structure and parameterisation of the numerical model. They will also allow the numerical model to be independently validated.
The numerical model has significant potential to be scaled up, for example to the catchment scale. At this scale, it would probably not be practical to estimate \( p(T) \) at 10 m resolution. Instead, a representative value of \( p(T) \) could be applied to longer reach segments based on aggregate characteristics. These values could be determined empirically from experiments conducted along different study reaches (described above) with model parameters derived from statistical relationships (e.g. between quantitative estimates of riparian vegetation density and \( p(T) \)). To assist with scaling up the model, reach characteristics (used to estimate the probabilities of trapping) could be automatically calculated. For example, channel sinuosity can be automatically calculated in a geographical information system (e.g. ArcMap or QGIS) from existing channel network datasets (e.g. the WWF’s HydroSHEDS data; see Lehner et al., 2008). Channel width could be estimated from bankfull discharge \( (Q_{bf}) \) using hydraulic geometry (see Equation 2.5). \( Q_{bf} \) itself can be estimated from the discharge with a recurrence interval of 1.5 years (using readily available gauged flow data, if available; see Benson and Thomas, 1966). If gauged flow data are not available (as is the case in many parts of the world) mean annual discharge could be estimated from a combination of modelled runoff fields (e.g. from water balance calculations; see Fekete et al., 1999) and accumulated area. Additionally, it may be possible to automatically identify overhanging trees using remote sensing and machine learning techniques (see Tomsett and Leyland, 2019).

A large number of long-term and refined experiments are likely to be needed to get a good quantitative understanding of the different controls on macroplastic transport and release (e.g. reach characteristics, flow conditions and plastic properties). This represents a significant challenge going forward, which needs to be met in order to scale up the model described in this thesis. In addition, accurate estimates are also needed of the location, frequency and volume of plastic emitted to channels (essential for real-world estimates of plastic retention and flux). Nevertheless, the approach presented in this thesis is feasible as an alternative method for modelling riverine plastic transport to those which have hitherto been described (e.g. Lebreton et al., 2017; Meijer et al., 2019; Schmidt et al., 2017). The ability of this approach to better estimate riverine plastic flux remains to be seen.
5.7 Chapter Summary and Thesis Conclusions

To conclude, plastic accumulation in the marine environment is a major concern, given the longevity of plastics at sea and their potentially harmful effects. Despite the recognition that rivers probably act as major transport pathways for plastic debris delivery to the oceans, and recent efforts to estimate the amount of plastic entering the oceans from rivers, many aspects of plastic transport remain poorly understood and flux estimates remain uncertain. Existing methods to estimate flux (from field measurements and coarse-scale modelling efforts) are very crude and fail to consider the actual processes controlling macroplastic displacement (even in a conceptual way). The work described in this thesis took a fundamentally different approach to investigate plastic transport. This considered more explicitly the displacement of individual items of plastic debris and the processes controlling displacement. It combined an experimental component, in which macroplastic tracers were tracked along a small river reach, with the construction of a numerical model to describe the phenomena observed.

The main objectives of the thesis were: (1) to determine the travel distance distribution of seeded macroplastic debris in an experimental study reach (and to identify the principal controls affecting this distribution), and; (2) to develop a model framework for describing macroplastic debris transport in rivers. The experiments showed clearly that seeded macroplastic debris displayed rapid dispersion and was prone to “trapping” at different points. No seeded items were observed to leave the 1 km study reach over 24 hours in six replicate experiments. Although there was a pronounced random component to the behaviour of individual macroplastic items, the observed distribution of travel distances immediately after injection and after 24 hours appeared to be systematically controlled by the location and nature of specific channel features (particularly the presence of overhanging trees and meander bends). Travel distances were also found to be low and variable. The model that was developed described the transport of individual items of macroplastic debris using a stochastic approach, employing a series of Bernoulli trials, with probabilities linked to channel features such as the presence of vegetation, channel width and sinuosity. It was able to describe the observed patterns reasonably well, suggesting that the approach has potential as an interpretive tool for investigating fluvial plastic transport. Significant knowledge gaps remain at both the reach and catchment scales. However, if these gaps can be filled, the approach described has the potential to be scaled up and make a genuine step-change in the prediction of riverine plastic flux to the oceans.
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