Water Air Soil Pollut (2024) 235:361
https://doi.org/10.1007/s11270-024-07168-4

Sewage Derived Microplastic and Anthropogenic Fibre Retention by Integrated Constructed Wetlands

Richard J. Warren · Richard J. Cooper · Andrew G. Mayes · Stefanie Nolte · Kevin M. Hiscock · Jonah Tosney

Received: 1 February 2024 / Accepted: 14 May 2024

Abstract High loads of microplastics and anthropogenic fibres can be discharged from wastewater treatment plants (WWTPs) into surface water bodies. Integrated Constructed Wetlands (ICWs) are potentially well suited to provide a cost-effective mitigation solution at small WWTPs where conventional treatment is prohibitively expensive. This study aimed to assess the microplastic and anthropogenic fibre retention efficiency of two ICWs (Northrepps and Ingoldisthorpe) in Norfolk (UK) over a 12-month period (2022–2023). Analysing a total of 54 water and 23 sediment samples, the findings revealed that Northrepps ICW received on average 349,920 (±763,776) anthropogenic fibres day^{-1}, with a retention rate of 99.3%. No seasonal variation was observed in retention efficiency. Ingoldisthorpe ICW intermittently received anthropogenic fibres in low concentrations, with an average of 9504 (±19,872) day^{-1} and a retention rate of 100%. Microplastics and anthropogenic fibres were prevalent in sediment samples of the first cell of Northrepps ICW, averaging 10,090 items kg^{-1} dry sediment, while none were found at concentrations above the limit of detection in the second or third cell. Of the 369 fibres analysed by ATR-FTIR, 55% were plastic (dominated by polyester). Of the 140 suspected microplastic fragments analysed by ATR-FTIR, 73% were confidently identified as plastic (mostly polystyrene, polyethylene, or polypropylene). This study demonstrates how ICWs can effectively retain sewage effluent derived microplastics and anthropogenic fibres. However, the accumulation of plastic waste in ICWs may complicate long term management and their cost-effectiveness.

Keywords Microplastic · Constructed wetlands · Sewage · Wastewater treatment · Anthropogenic fibres

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11270-024-07168-4.

R. J. Warren (✉) · R. J. Cooper · S. Nolte · K. M. Hiscock
School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich NR4 7TJ, UK
e-mail: Richard.J.Warren@uea.ac.uk

A. G. Mayes
School of Chemistry, University of East Anglia, Norwich Research Park, Norwich NR4 7TJ, UK

S. Nolte
Centre for Environment, Fisheries and Aquaculture Science, Pakefield Rd, Lowestoft NR33 0HT, UK

J. Tosney
Norfolk Rivers Trust, Bayfield Brecks, Holt, Norfolk NR25 7DZ, UK

1 Introduction

High microplastic and anthropogenic fibre loads can be discharged into rivers from wastewater treatment plants (WWTPs) with primary, secondary, and tertiary treatment (Blair et al., 2019; Napper et al., 2023; Ziajahromi et al., 2017). Generally, microplastics in WWTP influent are derived from a variety of sources (Prata, 2018), while washing machine emissions are a dominant source of anthropogenic fibres (Browne et al., 2011). Assuming working operation, an average of 72% of microplastics are removed after primary treatment, 88% after secondary treatment, and 94% after tertiary treatment (Iyare et al., 2020), although removal rates can be as high as 99.9%, as reported by Carr et al. (2016). Tertiary treatment is expensive and is typically used in WWTPs discharging into sensitive waterbodies and serving population equivalents > 10,000 (Bunce et al., 2018), thus limited cost-effective measures are available to resolve the problem of microplastic and anthropogenic fibre release from WWTPs. Additionally, WWTPs do not always perform as they are intended to (Hammond et al., 2021) and releases of untreated wastewater contribute a significant source of microplastics and anthropogenic fibres to waterbodies, particularly from combined sewer overflows during rainfall events (Woodward et al., 2021). Hence, WWTP discharge exports significant microplastic loads to the sea (Siegfried et al., 2017). Risk assessment for microplastic particles is complicated (Koelmans et al., 2023), especially given the diversity of microplastics (Rochman et al., 2019). However, microplastics act as vectors of other pollutants enhancing their transport (Wagstaff et al., 2022) and leach chemical additives from within the plastic itself, such as endocrine disrupting plasticizers (Meeker et al., 2009).

In addition to microplastics, WWTP discharges can elevate nutrient levels in rivers (Cooper et al., 2022) and increase eutrophication risk (Jarvie et al., 2006a). This can be particularly detrimental where WWTPs discharge into sensitive waterbodies, such as rare chalk streams and lacustrine environments that would naturally be oligotrophic (Edo et al., 2020; Jarvie et al., 2006b). As a result, integrated constructed wetlands (ICWs) have been applied to enhance the water quality of WWTP effluent prior to release into surface waterbodies (Scholz et al., 2007). ICWs generally consist of a series of connected surface flowing ponds containing shallow water year-round supplied entirely from WWTP effluent (Fig. 1). Their implementation balances ecological aims of wetland restoration and engineering targets for economically and consistently enhancing water quality (Babatunde et al., 2008). The two ICWs investigated in the present study, Northrepps and Ingoldisthorpe, Norfolk (UK), have been shown to effectively retain nutrients and reduce eutrophication risk from WWTP discharges: mean nitrate and phosphate concentrations were reduced by ~ 63% and ~ 30% across the Northrepps and Ingoldisthorpe ICWs respectively (Cooper et al., 2020). The dense stands of emergent vegetation in ICWs decrease water velocities and promote sedimentation of suspended material, meaning ICWs may also be well placed to cost-effectively reduce microplastic loads in receiving waters.

Few studies have assessed microplastic retention by constructed wetlands, and those that do are mostly subsurface flow constructed wetlands (Xu et al., 2022). In a surface flow constructed wetland (SFCW) in Northern China, the Lingang Ecological Wetland Park, average microplastic removal rates were 29.4% from September to October 2020 (Zhou et al., 2021). In a nearby SFCW, Konggang, microplastic removal rates were 43.7% (Zhou et al., 2021). These removal rates are based on the numbers of microplastics (including fibres) down to a reported size of 20 μm. In the Lingang SFCW, larger particles (> 100 μm) were better retained than smaller (20–100 μm) particles. Fibres were most well retained, and fragments least well retained in both the Lingang and Konggang SFCW. However, the surface flow wetlands studied by Zhou et al. (2021) are not comparable in design to those in the present study: they are part of a much larger wetland system combined with subsurface flow constructed wetlands. The most similar work to the present study identified in the literature is that of Bydalek et al. (2023). Their study aimed to assess microplastic fate in a surface flow constructed wetland at the Cromhall ICW, Gloucestershire, UK. Here the loading rate was 790 m^3 day^{-1} from a secondary treatment WWTP serving 2000 people (Bydalek et al., 2023). Although the ICW is relatively similar to those in the present study, the sampling campaign by Bydalek et al. (2023) occurred only during summer months in July and August 2021, so seasonal variation in ICW performance was not addressed. Constructed wetland plant biomass (including underground biomass) is lowest during winter months (Zhang et al., 2022), meaning the microplastic filtering capacity of these wetlands may be reduced in winter when loading rates from WWTP effluent

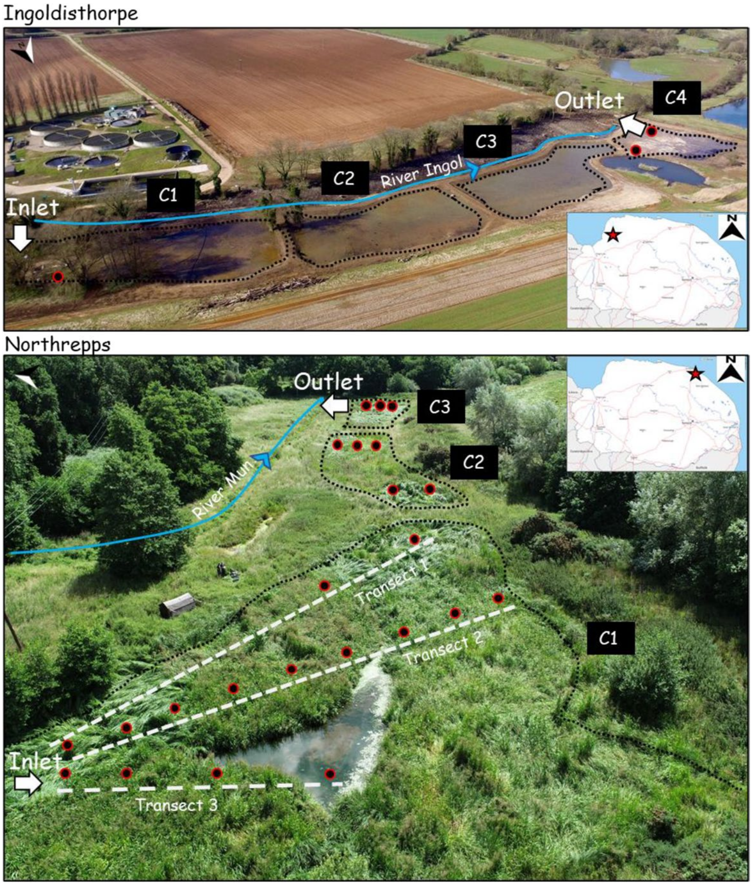


Fig. 1 Aerial view of Ingoldisthorpe and Northrepps ICWs. Black dots with red outlines indicate approximate sediment sampling locations. Location of approximate transects highlighted in the first cell. Ingoldisthorpe image (top) taken before vegetation was planted (Credit: Norfolk Rivers Trust). Northrepps image (bottom) taken 6 July 2023. Cells labelled as 'C1' etc. Cells are outlined with dashed black line. 'Inlet' shows location where the wetland is supplied by WWTP effluent. 'Outlet' shows location of wetland outflow

are higher due to increased rainfall. Additionally, the material composition of suspected microplastics and fibres found was not investigated in detail by Bydalek et al. (2023), with only 12 particles > 1 mm verified by FTIR. Furthermore, microplastic fragments and anthropogenic fibres have not to date been quantified in fine bed sediment of ICWs treating WWTP effluent.

The present study addresses these research gaps and aims to assess the microplastic and anthropogenic fibre retention efficiency of two ICWs receiving treated WWTP effluent (Ingoldisthorpe and Northrepps) over a long duration by:

1. Quantifying anthropogenic fibre retention in both ICWs over a 12-month period (May 2022 to June 2023) by analysing approximately monthly water samples from the inlet and outlet;
2. Assessing how the concentration and size of microplastic fragments and anthropogenic fibres in fine bed sediment samples changes with distance from the inlet in the Northrepps ICW; and
3. Using ATR-FTIR to ascertain the material composition of microplastics and anthropogenic fibres entering and within ICWs.

2 Methods

2.1 Study Location

The Ingoldisthorpe ICW lies on the River Ingol (52°51′53″N 0°31′18″E, Norfolk, UK), a 10.3 km long, predominately groundwater fed lowland calcareous river with a 35.3 km^2 catchment. The area experiences a temperate maritime climate, with a mean annual temperature of 10.5 °C and a mean annual precipitation total of 684 mm (1991–2020) (Meteorological Office, 2023). Parts of the river have Special Area of Conservation (SAC), Site of

Special Scientific Interest (SSSI), and Ramsar status, although it is being degraded by WWTP effluent (Cooper et al., 2020). Ingoldisthorpe ICW was operational in April 2018 and covers 10,788 m^2 across four shallow (20–30 cm) unlined cells with areas of 1972 m^2 (cell 1), 2450 m^2 (cell 2), 3560 m^2 (cell 3) and 2806 m^2 (cell 4) (Fig. 1). Emergent vegetation cover in cell 1 is > 90%, and 30–50% in cells 2–4. The wetland was planted with 25,000 native aquatic plants, including *Glyceria maxima, Iris pseudacorus, Juncaceae sp., Cyperaceae sp., Typha latifolia,* and *Caltha palustris*. Prior to discharge into the ICW, sewage effluent is treated with primary settlement tanks, trickling filters, humus tanks, and a nitrifying sand filter. Effluent discharge rate into the wetland was approximately 950 m^3 day^{-1} from May 2022 to June 2023. Cooper et al. (2020) reported an outflow rate from the wetland of 129 m^3 day^{-1}. Total capital costs were £194,000 for Ingoldisthorpe ICW, equating to £31 per person served (Cooper et al., 2020).

The Northrepps ICW lies on the River Mun, (52°53′46″N, 1°20′41″E, Norfolk, UK) a 7.9 km long groundwater dominated lowland river with a 22 km^2 catchment. The ICW was operational in October 2014 and covers 2900 m^2 across three shallow (~ 20 cm) unlined cells with areas of 1600 m^2 (cell 1), 700 m^2 (cell 2), and 600 m^2 (cell 3) (Fig. 1). Emergent vegetation cover is > 95% in each cell. The wetland was planted with 15,000 native aquatic plants including *Carex riparia, Iris pseudacorus, Alisma plantago-aquatica, Sparganium erectum, Veronica beccabunga, and Mentha aquatica.* Prior to discharge into the ICW, sewage effluent is treated with activated sludge, an aeration tank, and a final settlement tank. Effluent discharge rate into the wetland was approximately 64 m^3 day^{-1} from May 2022 to June 2023. Cooper et al. (2020) reported an outflow rate from the wetland of 187 m^3 day^{-1}. Total capital costs were £30,021 for the Northrepps ICW, equating to £39 per person served (Cooper et al., 2020).

2.2 Field Campaigns

Water samples were collected over a 12-month period at Ingoldisthorpe and Northrepps ICWs at each wetland's inlet and outlet (Supplementary Fig. 1), sampling at approximately monthly intervals between July 2022 and June 2023 at Ingoldisthorpe, and between June 2022 and May 2023 at Northrepps (Table 1).

At the Northrepps inlet, water samples ($n = 14$) were taken from an inspection point approximately halfway along the 150 m pipe supplying the wetland from the WWTP (Supplementary Fig. 1). At the Northrepps outlet ($n = 14$), Ingoldisthorpe inlet ($n = 13$), and Ingoldisthorpe outlet ($n = 13$), samples were collected by holding a plastic bucket (8 L capacity with 500 mL graduations) beneath the pipe, meaning the entire flow of the pipe was sampled (note: only fibres were analysed in these water samples, thus use of a non-fibrous plastic bucket was deemed low-risk). Known volumes of water samples were then poured from the bucket through a 38 µm stainless steel sieve (200 × 50 mm) until the sieve began to lightly clog (this point was determined when water would take ~ 30 s to completely filter through). Sampling volume therefore varied considerably at each location, depending on the turbidity of the water sample. The average sample volume at the Ingoldisthorpe ICW inlet was 436 (standard deviation (SD) = 176) L and 300 (SD = 176) L at the outlet. The average sample volume at the inlet of Northrepps ICW was 44 (SD = 15) L and 314 (SD = 57) L at the outlet. After sampling, the sieves were sealed with a stainless-steel lid and base and transported back to the laboratory. A new sieve was used for each sampling location. Sampling was performed during dry weather only, although loading rates (from WWTP effluent) were variable (Supplementary Table 1).

Fine bed sediment (FBS) samples were collected on 11 and 20 December 2022 from Northrepps ICW and on 22 November 2022 from Ingoldisthorpe ICW. Three FBS samples were collected from Ingoldisthorpe ICW: one in cell 1 (10 m from the inlet) and two in cell 4 (at the beginning and end of the cell). In the first cell at Northrepps ICW, 51 samples were collected at approximately 2 m intervals (by stride) along three transects, although 15 of these were analysed for microplastics (Fig. 1). Eight samples were collected from cells two and three at Northrepps (Fig. 1). FBS samples were collected using an isolation agitation method based on Woodward et al. (2021). A stainless-steel cylinder (300 × 900 mm) was pushed down as firmly as possible approximately 5 cm into

Table 1 Anthropogenic fibre concentrations and removal performance of the Northrepps and Ingoldisthorpe ICW, derived from the water sampling campaign. NA indicates no available data

Wetland	Sampling date	Inlet		Outlet		Fibre retention (%)
		Volume (L)	Fibres L^{-1} (LOD subtracted)	Volume (L)	Fibres L^{-1} (LOD subtracted)	
Northrepps	06/06/2022	40.5	3.46	228	< LOD	100
Northrepps	24/06/2022	49.5	1.07	375	0.06	94.8
Northrepps	12/07/2022	39	2.90	225	< LOD	100
Northrepps	12/07/2022	39	3.92	NA		NA
Northrepps	12/07/2022	39	4.28			
Northrepps	02/08/2022	54	3.74	300	< LOD	100
Northrepps	31/08/2022	45	4.36	200	0.08	98.3
Northrepps	03/10/2022	40	2.23	300	< LOD	100
Northrepps	04/11/2022	60	2.37	315	0.03	98.9
Northrepps	04/01/2023	60	2.90	350	< LOD	100
Northrepps	04/01/2023	NA		350	< LOD	NA
Northrepps	13/02/2023	69	3.28	350	< LOD	100
Northrepps	13/03/2023	6	39.00	350	0.04	99.9
Northrepps	26/04/2023	30	1.10	350	0	100
Northrepps	26/04/2023	NA		350	0	NA
Northrepps	09/05/2023	45	2.16	350	0	100
Ingoldisthorpe	11/07/2022	600	0.01	252	< LOD	100
Ingoldisthorpe	25/07/2022	630	0.06	NA		NA
Ingoldisthorpe	01/08/2022	750	0.01	300	< LOD	100
Ingoldisthorpe	20/09/2022	350	< LOD	350	< LOD	NA
Ingoldisthorpe	17/10/2022	350	0.06	175	< LOD	100
Ingoldisthorpe	22/11/2022	140	< LOD	140	< LOD	NA
Ingoldisthorpe	29/11/2022	700	< LOD	350	< LOD	NA
Ingoldisthorpe	09/01/2023	400	< LOD	350	< LOD	NA
Ingoldisthorpe	09/01/2023	NA		350	< LOD	NA
Ingoldisthorpe	20/02/2023	350	< LOD	350	< LOD	NA
Ingoldisthorpe	27/03/2023	350	< LOD	350	< LOD	NA
Ingoldisthorpe	23/04/2023	350	0.01	350	< LOD	NA
Ingoldisthorpe	21/05/2023	350	< LOD	350	< LOD	NA
Ingoldisthorpe	12/06/2023	350	< LOD	350	< LOD	NA

the sediment, sometimes requiring vegetation to be carefully pulled apart (supplementary Fig. 2). The FBS samples were agitated into suspension using a stainless-steel saucepan for 30–60 s. Turbid water samples were poured (with the stainless-steel saucepan) through a sieve stack of 2 mm and 38 μm until enough sediment could be extracted from the 38 μm sieve to approximately fill a glass jar (480 mL). The contents of the 2 mm sieve were extracted into separate jars to be analysed for macroplastics.

2.3 Laboratory Analysis

2.3.1 Water Samples

Upon return to the laboratory the same day as sample collection, sieves were rinsed (with MilliQ water) into glass beakers in a laminar flow cabinet. MilliQ water was added to the 100 mL mark, and the same volume of sodium hypochlorite (minimum 14% free chlorine) was applied to create a 1:1 dilution (v/v). This beaker was then sealed with a glass petri dish and placed in a shaker incubator (Orbital Shaker Incubator ES-80) at 40 °C and 90 rpm for 16–20 h. After digestion, the solution was vacuum filtered onto 47 mm diameter cellulose nitrate filters (pore size 3 μm). A density separation step was not required because the suspended particulate matter within these constructed wetlands consisted almost entirely of low-density organic material.

Each filter was transferred to a microscope (Leica CMA) and fibres were identified by visual inspection (with 4 × objective). Fibres were categorised based on their colour and estimated length grouped into three categories: small (38–250 μm), medium (250–800 μm), and large (> 800 μm). These measurements represent an estimation because often a portion of a fibre was hidden beneath another layer on the filters and attempting to excavate each one would risk dislodging other material (including other fibres) from the filter.

2.3.2 *Sediment Samples*

FBS samples were freeze dried (Scanvac Coolsafe) until a constant dry weight was achieved. To remove much of the organic material, 400 mL of sodium hypochlorite (50% dilution in water) was added and left for 16–20 h at 40 °C in a shaker incubator (Orbital Shaker Incubator ES-80) at 90 rpm. The solution was then filtered through a 38 μm stainless steel sieve and rinsed thoroughly. A second digestion was then performed for 16–20 h at 40 °C in a shaker incubator at 90 rpm. The contents were then rinsed into a glass beaker (ensuring as little water as possible entered the beaker) and placed in a vacuum oven at 40 °C until the water level was less than ~ 25 mL. Completely drying the sample was avoided at this stage to prevent crisps of sediment forming that would disrupt the density separation process. Zinc chloride (1.5 g mL^{-3}) was added to the same glass jar containing the dried sample and left on a shaker at 90 rpm for at least 30 min to better separate any sediment agglomerates. The homogenised sample was then added to the density separator and topped with $ZnCl_2$ (1.5 g mL^{-3}). The density separator (supplementary Fig. 3) used was custom made and based on the design of Vermeiren et al. (2020). The units were filled close to the top and left for a minimum of three hours before $ZnCl_2$ (1.5 g mL^{-3}) was added to overflow the units. The overflow was stopped after approximately 100 mL of $ZnCl_2$ had overflowed into the glass beaker. The density separator was then left for a minimum of three hours following agitation with a magnetic stirrer for 60 s before another 100 mL was overflowed. After density separation, the sample (combined overflow) was vacuum filtered onto a 47 mm cellulose nitrate filter (0.45 μm pore size) before analysis of the filter with a microscope (Leica CME). Anthropogenic fibres were identified as for the water samples, but in addition microplastic fragments were identified by the following criteria:

- Fragment appearing artificially coloured or shiny (resembling glitter).
- Fragment dark in colour with sharp edges and smooth surface.

2.3.3 *ATR-FTIR*

Selected fibres and suspected microplastic fragments were extracted with either tweezers or a 33-gauge syringe needle into a glass beaker containing water. An attempt was made to extract approximately every tenth fibre and fragment found (on each water and sediment sample filter, respectively) for chemical identification. In total, 369 fibres and 140 suspected microplastic fragments were validated by ATR-FTIR (5.3% and 11.6% of total identified in all samples, respectively). The samples were then vacuum filtered onto 25 mm silver coated filters (0.45 μm pore size). Filters with only fibres were lightly coated with a spray-on glue ('Crafter's Companion Stick and Spray') before filtration to prevent fibres blowing away when handling the filter and performing FTIR (supplementary Fig. 4). No glue was used with the fragments because they generally remained in place. All particles and fibres found on each of these filters were analysed by a micro-ATR-FTIR microscope (Bruker Hyperion 2000, 20 X ATR objective, resolution = 4.0 cm^{-1}, 64 scans $sample^{-1}$). The spectra acquired were analysed using Open Specy (Cowger et al., 2021) to determine the best library match. Default pre-processing settings were used for threshold signal–noise, smoothing, intensity adjustment, baseline correction and flatten region (removing CO_2 peaks) options. Wavenumber range selection of 0–3500 cm^{-1} was applied. Identification was performed using the 'Cor: FTIR Deriv' option. A spectral hit quality score (Pearson correlation coefficient) of 0.7 was set as the threshold, below which all samples were considered unknown to avoid bias in spectral interpretation.

2.4 Quality Control

During sampling a 100% woollen jumper was worn (woollen fibres were removed in sample processing by chemical digestion with NaClO) or a closed weave shirt in warm weather to prevent contamination from clothing fibres. All sieves and glass jars used to collect samples were pre-cleaned in the laboratory with MilliQ water and sealed (either with a sieve lid or aluminium foil) prior to sampling. Sediment sampling equipment was rinsed with MilliQ water between sampling locations.

Unless otherwise stated, all solutions used in sample processing were pre-filtered through 0.45 µm polycarbonate filters. Laboratory work was undertaken within a laminar flow cabinet that was vacuumed and wiped down with paper towel before use. Microscope analysis was done in a room with managed airflow to minimise airborne contamination and was regularly cleaned.

2.5 Positive Controls

To assess the recovery rate of the water sampling method, 30 individual pink polyester (PET) fibres were peeled from a sewing thread and cut to approximately 2–5 mm in length. These fibres were stored in a glass beaker in water and then poured through a 38 µm sieve, following which the standard water sampling method was followed. This was performed three times to achieve a more reliable average recovery rate.

To assess the recovery rate of the sediment sampling method, a spiked field control sample was collected. Pink polyester fibres were prepared in the same way as for the water samples, while microplastic fragments were generated by filing macroplastic items to generate small fragments that were then sieved to 250–750 µm for use in recovery experiments. In total, 30 pink polyester fibres, 30 blue polyvinyl chloride (PVC) fragments and 30 yellow polypropylene (PP) fragments were mixed in with 10 g of fine bed sediment from the third cell of the Northrepps ICW (no particles or fibres resembling the spiked ones were identified in this sample). PP (~0.9 g cm^{-3}) was used to represent low density plastics and PVC (~1.48 g cm^{-3}) was used to represent high density plastics. After spiking the sediment, the standard sediment sampling method was then followed. This was performed three times. The size of the microplastic fragments found after recovery was recorded by measuring the longest dimension of each particle using ToupView software.

For the water sampling method, an average recovery rate of 86.7 (SD = 12.0) % of polyester fibres was achieved. For the sediment sampling method, recovery rates were 61.1 (SD = 10.2) % for polyester fibres, 92.2 (SD = 8.4) % for PP fragments, and 81.1 (SD = 6.5) % for PVC fragments. The average size (longest dimension) of all microplastic fragments recovered after the sediment sampling procedure was 399 (SD = 144) µm. The recovery rates are not fully representative of the diversity of microplastics and anthropogenic fibres found in samples (regarding their material type, shape, and size), meaning the data are not corrected based on recovery to avoid introducing an unknown bias, as Simon et al. (2018) recommended. The anthropogenic fibre numbers reported in the present study are therefore likely an underestimate of actual values due to losses during sample processing and should be regarded as minimum estimates.

2.6 Negative Controls

A total of 19 procedural blank samples were taken over the course of the 12-month water sampling campaign (supplementary Table 2). To do so an empty sieve was placed beside the sampling location to assess airborne fibre contamination while sampling. This sieve was then sealed, transported back to the laboratory, processed, and analysed following the standard water sampling method. Anthropogenic fibres were found in every procedural blank: averaging 4.5 (SD = 2.6) fibres $sample^{-1}$. A total of 85 fibres were found in the procedural blanks: most were clear (68%) or dark (25%). Additionally, 32% of fibres found were approximately > 800 µm, 58% were 800–250 µm, and 10% were 38–250 µm. The limit of detection (LOD) was calculated as 12 fibres $sample^{-1}$, and this value was therefore subtracted from each sample fibre count (as done by Dawson et al. (2023)).

A total of five procedural blank samples were taken for the sediment samples. Fibres were found in every blank sample (Supplementary Table 3), with an average of 11.2 (SD = 8.6) fibres $sample^{-1}$. A total of 56 fibres were found in these blank samples: most were clear (57%) or dark (29%). Additionally, 30% of fibres found were approximately > 800 μm, 52% were 800–250 μm, and 18% were 38–250 μm. The LOD was calculated as 37 fibres sample^{-1}, and this value was therefore subtracted from each sample fibre count. Suspected microplastic fragments were found in three out of five blank samples, with an average of 2 (SD = 2.5) fragments sample^{-1}. A total of 10 suspected microplastic fragments were found in these blank samples (Supplementary Fig. 5). The LOD was calculated as 10 fragments sample^{-1}, and this value was subtracted from the sample counts.

2.7 Data Analysis

The limit of detection (LOD) was calculated for both fibres and fragments separately (for sediment samples) and subtracted from the total value of each sample.

$$LOD = Meanblank + (3 \times standarddeviationblank)$$

This correction method was chosen based on the findings by Dawson et al. (2023), where LOD methods were recommended for microplastic studies.

Fibre retention rates were calculated as:

$$Retention(\%) = \left(1 - \frac{Conc.outlet}{Conc.inlet}\right) \times 100$$

Areal removal (AR) rates were calculated as:

$$AR\left(itemsm^{-2}day^{-1}\right) = \left(\frac{conc.inlet - conc.outlet}{S} \times Q\right)$$

where Q is the discharge at the inlet (m^3 day^{-1}) and S is the surface area (m^2). Discharge data provided by Anglian Water.

Data analysis was performed in Microsoft Excel. A Shapiro–Wilk test was used and indicated that the data did not meet the normality assumption for parametric tests. Consequently, a Mann–Whitney U Test was used to compare mean fibre concentrations at the inlet and outlet of Northrepps ICW. Standard deviation is reported in parenthesis after average values.

Error propagation was applied to fibre loading rate (Z) calculations using the equation below (Fantner, 2013):

$$\frac{\sigma_Z}{Z} = \sqrt{\left(\frac{\sigma_x}{x}\right)^2 + \left(\frac{\sigma_y}{y}\right)^2}$$

where x is fibre concentration (fibres L^{-1}) and y is discharge (L s^{-1}). The propagated error is denoted with '±' in parenthesis.

3 Results

3.1 Ingoldisthorpe Anthropogenic Fibre Retention

Mean fibre concentrations at the inlet of Ingoldisthorpe ICW across the entire sampling period were 0.01 (SD = 0.02) fibres L^{-1}, thus fibres appeared to be passing through the treatment plant and entering the wetland in low concentrations (Table 1). Average discharge from WWTP effluent entering the wetland for the period May 2022 to June 2023 was 11.01 (SD = 7.25) L s^{-1}, equating to an average loading rate of 0.11 (± 0.23) fibres s^{-1}, or 9504 (± 19,872) fibres day $^{-1}$. However, fibres were not continuously released in significant numbers from the WWTP into the wetland because in seven out of 13 of these samples, fibres were not detected above the LOD. In no outlet samples were microplastic concentrations detected above the LOD. Fibre retention at the Ingoldisthorpe wetland therefore appears consistently 100%. However, the low fibre concentrations at the inlet show that it was the WWTP that was highly effective at retaining fibres, meaning the wetland was not overloaded with high fibre numbers.

3.2 Northrepps Anthropogenic Fibre Retention

Fibres were found in concentrations above the LOD in all samples from the Northrepps inlet (Table 1), with mean fibre concentrations across the entire sampling period of 5.48 (SD = 9.70) fibres L^{-1}. Fibres were therefore consistently passing through the treatment plant and entering the wetland. Average discharge from WWTP effluent entering the wetland over the period May 2022 to June 2023 was 0.74 (SD = 0.94) L s^{-1}, equating to an average loading rate of 4.05 (± 8.84) fibres s^{-1}, or 349,920 (± 763,776) fibres day^{-1}. Fibres were clear (62%), dark (29.9%), red (4.2%), blue (2.4%), and light (1.5%) in colour (light includes white and cream), while 27.2% were approximately > 800 μm, 56% were 800–250 μm, and 16.8% were 38–250 μm at the Northrepps inlet.

The highest fibre concentration was observed on 13 March 2023 at 39 fibres L^{-1}: a clear outlier in the dataset (Table 1). On this date there was a blockage at the WWTP that was cleared approximately 30 min before sampling; a deliberate attempt was made not to sample the initial pulse after the blockage was cleared. It was assumed after 30 min the flow in the pipe would become normal, and indeed it was when sampling commenced. However, after 2 L were sampled, the flow increased to a level significantly higher than normal levels observed in the pipe (supplementary Fig. 6) and the water was also more turbid than usual, reflected in the low sample volume (Table 1). Excluding this sampling date, mean concentrations at the inlet were 2.90 (SD = 1.08) fibres L^{-1}. It is highly likely that the fibre concentrations reported for the Northrepps inlet are an underestimate because (not including losses during sample processing), in two inlet samples bundles of fibres were found (supplementary Fig. 7) and the number of fibres making up each bundle was not counted.

The mean fibre concentration at the outlet of the Northrepps ICW across the entire sampling period was 0.01 (SD = 0.02) fibres L^{-1}, lower (Mann Whitney U Test, $U = 4$, $p < 0.01$), than the mean inlet fibre concentration. The average retention efficiency was 99.3% (SD = 1.5%) across the entire sampling period. Only in four of the 14 outlet samples were fibres detected above the LOD (Table 1), thus fibres appear to be released intermittently from the wetland at low concentrations. Owing to the high workload of sample processing, sample replicates were not attempted in each month, thus statistical comparisons cannot be made. However, Table 1 indicates no clear change in fibre retention by month or season.

3.3 Suspected Microplastics and Anthropogenic Fibres in Northrepps ICW Sediment Samples

A total of 1203 fragments and 4540 anthropogenic fibres were found in the 23 sediment samples at the Northrepps wetland. In the first cell, average concentrations were 8152 (SD = 7022) anthropogenic fibres kg^{-1} and 1938 (SD = 991) suspected microplastic fragments kg^{-1} dry weight sediment. Anthropogenic fibre concentration declined with increasing distance from the inlet pipe (Fig. 2), although the highest concentration of 22,602 fibres kg^{-1} was 22 m from the inlet. The proportion of large, medium, and small size fibres did not change significantly with distance from the inlet (supplementary Fig. 8). As stated earlier, the size categories may not be completely accurate given the limitations of measuring fibre length when part of the fibre is buried under other material on filters. Most of the fibres found were clear (76%) and dark (17%), while 51% were approximately > 800 µm, 40% were 800–250 µm, and 9% were 38–250 µm. These values are proportionally similar to those for the inlet water samples at Northrepps which is expected because the WWTP effluent is the dominant source of anthropogenic fibres to this wetland. However, proportionally more large (> 800 µm) fibres were identified in sediment samples than inlet water samples, possibly because of the breakdown of fibrous macroplastics in the wetland into longer length fibres.

Microplastic fragment number decreased with increasing distance from the outlet in cell 1 (Fig. 2). The size (longest dimension) of suspected microplastic fragments varied little with increasing distance from the inlet (Fig. 3). A fibre bundle with a longest dimension of approximately 8 mm was found 2 m from the inlet in cell 1. This fragment was omitted from Fig. 3. Most suspected microplastic fragments found were < 100 µm or 100–200 µm (35.7% and 37.4%, respectively) (Fig. 4).

Most of the suspected microplastic fragments were blue or green (Fig. 4). However, this does not reflect the actual colours of microplastics within the wetland because clear and white microplastics were not identifiable with the method applied here. Additionally, only dark fragments that were obviously suspected microplastic were counted: those with sharp edges and a smooth texture. Therefore, tyre wear particles dark in colour could have been missed, although the catchment for both WWTPs was rural with generally low speed traffic so these were not likely to occur in high concentration in the Northrepps ICW. 26 microplastic fragments were spherically shaped resembling microplastic beads (all pink or blue colour), probably deposited in the wetland before the 2018 ban of microbeads in cosmetics in the UK (Department for Environment, Food & Rural Affairs, 2018), or from leftover products containing them. Seven of these spherical beads were analysed by ATR-FTIR: all were polyethylene (PE).

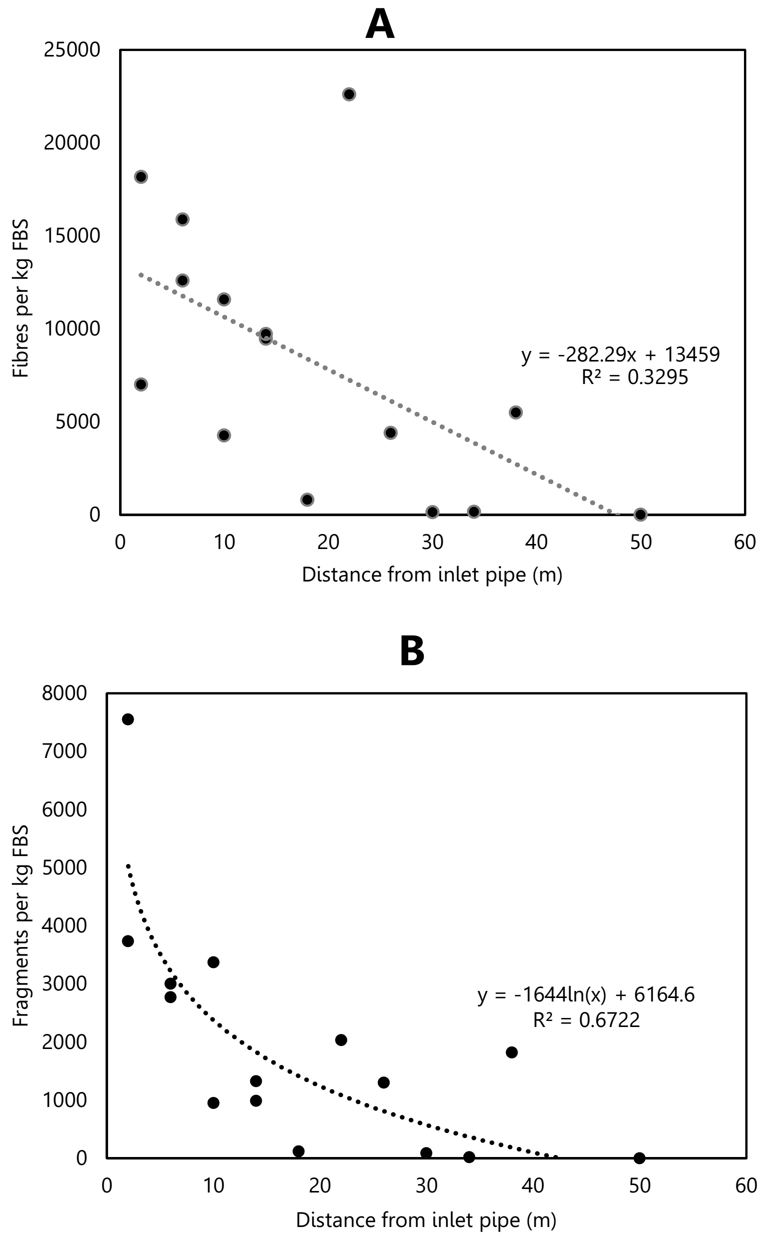


Fig. 2 Anthropogenic fibre concentration (**A**) and suspected microplastic fragments (**B**) in fine bed sediment (FBS) samples and distance from the inlet in the first cell of the Northrepps ICW

In cells 2 and 3 at Northrepps, fibres and suspected microplastic fragments were not detected at numbers above the LOD in any of the sediment samples.

Microplastic and anthropogenic fibre accumulation at the Ingoldisthorpe ICW is unlikely to be significant because the treatment plant has been shown to be highly effective at removing fibres (and so it is reasonable to assume it is equally effective at retaining microplastic fragments). Three sediment samples were collected from the Ingoldisthorpe ICW: one in cell 1 (10 m from the inlet) and two in cell 4 (at the beginning and end of the cell). In none of these samples were anthropogenic fibres and microplastic fragments detected above the LOD. The microplastics and anthropogenic fibres that enter the wetland from WWTP effluent probably are all retained in the first cell given that the area and vegetation cover are similar to the first cell at Northrepps ICW.

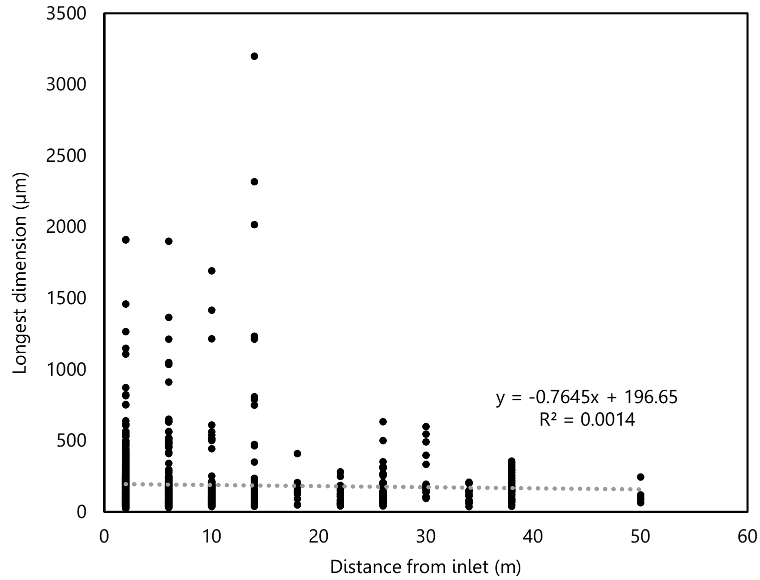


Fig. 3 Longest dimension of suspected microplastic fragments in wetland sediment against distance from the inlet in the first cell of the Northrepps ICW

3.4 Material Composition of Anthropogenic Fibres and Suspected Microplastics

To confirm the chemical composition of fibres identified, spectra were acquired for 369 fibres by ATR-FTIR. These fibres were sampled randomly for chemical identification, although FTIR validation was not performed on a filter-by-filter basis because the sample was pooled. However, the proportion of each fibre colour and size category in the FTIR validated samples and the entire sample pool is similar (supplementary Fig. 9): in both almost 60% of fibres were clear, 32% were dark, and over 50% were in the medium size category of approximately between 250 and 800 µm. These proportions suggest that the samples validated by FTIR are sufficiently representative of the entire sample pool. Additionally, the proportion of FTIR validated fibres that are cellulosic, plastic, and unknown generally plateau after ~ 200 samples (supplementary Fig. 10). Therefore, the 369 fibres validated appears sufficiently high and equates to 14% of the total fibres found in all water samples.

Of the 369 fibres validated by FTIR:

- 54.7% are plastic, of which 90% are PET, 4% acrylic, 3% PP, 2% PE, 1% polyamide.
- 35.5% are cellulosic fibres, of which 89.3% are clear or dark and 10.7% artificially dyed.
- 9.8% of fibres are unknown, of which 3% are unidentifiable and anthropogenic (fibre artificially dyed), 30% are unidentifiable and not artificially dyed, and 67% are the glue that was sprayed onto the filter (to ensure that fibres did not blow away during operation of the ATR-FTIR).

As a result of this FTIR validation, approximately 55% of the fibres reported in water samples are plastic (dominated by PET), approximately 36% are cellulosic and the remainder are ambiguous. No fibres were sampled from the sediment samples, although it is reasonable to assume that the fibres found are proportionally similar in their material composition because there are no other sources than the WWTP.

To confirm the chemical composition of suspected microplastic fragments identified, spectra were acquired for 140 fragments (38 µm to 2 mm) by ATR-FTIR. The FTIR validated samples appear sufficiently representative because a variety of fragment colours, sizes and materials were identified (Fig. 5).

Of the 140 suspected microplastic fragments analysed by ATR-FTIR: 73% were plastic, 6% non-plastic, and 21% unconfirmed (hit quality score < 0.7). Most of the fragments that were confirmed plastic were either polystyrene, polyethylene, or polypropylene (34%, 20%, and 23%, respectively) (Fig. 6). Common anthropogenic fibres and microplastic fragments are shown in supplementary Fig. 11.

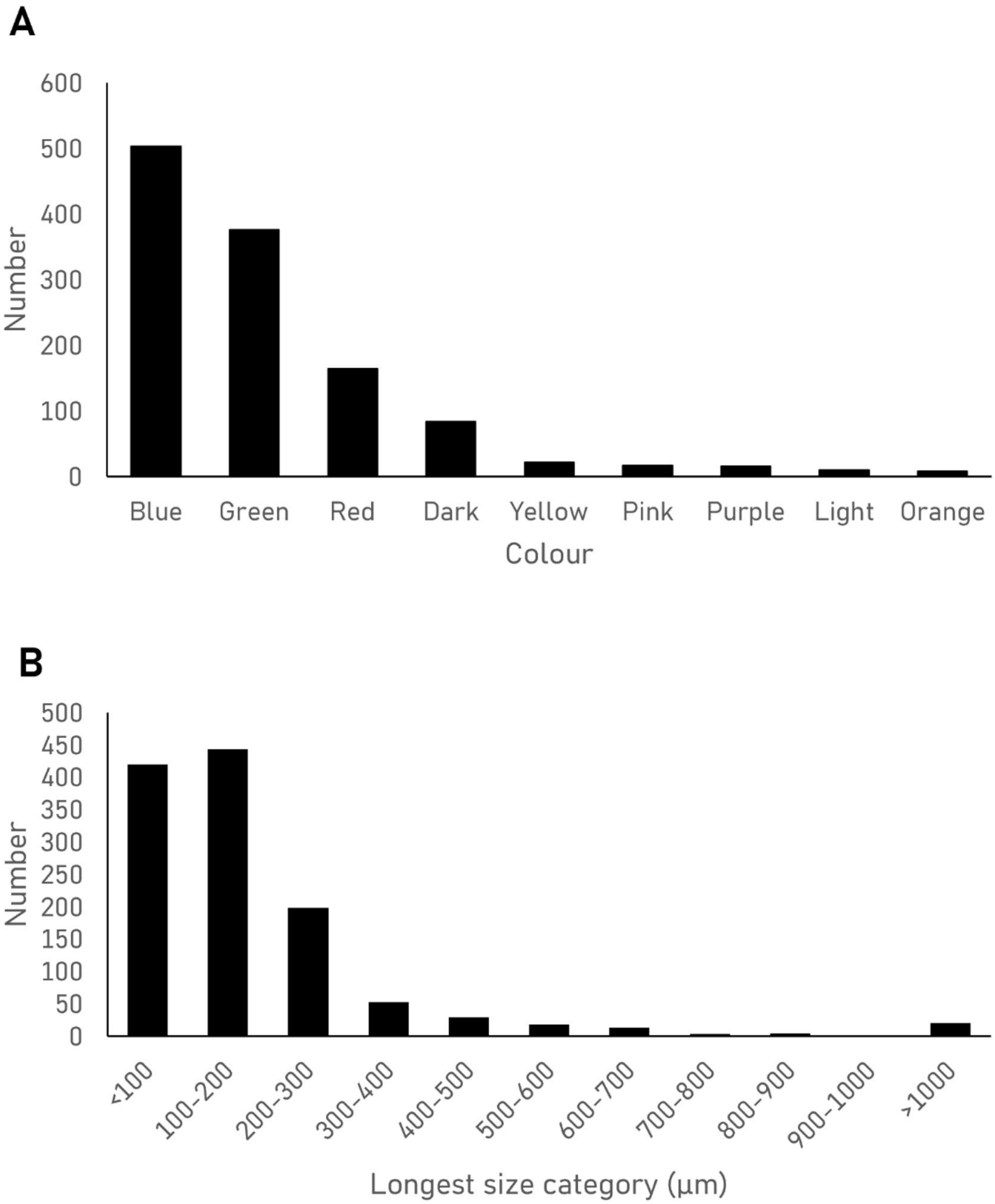


Fig. 4 Colour and size category (longest dimension) of suspected microplastic fragments identified in sediment samples from the Northrepps ICW

3.5 Plastics > 2 mm in Northrepps ICW

Plastic material was searched for in the 2 mm aperture sieve (in the laboratory after drying) for each sediment sampling location in Northrepps ICW. In total, 132 suspected plastic pieces were found. Of these, 97 were a white material, all visually appearing as though they were from the same source. Ten of these white plastic pieces were tested using a benchtop ATR-FTIR and revealed to be PE, likely from sanitary products: indeed, whole sanitary towels were found in the Northrepps wetland within 4 m

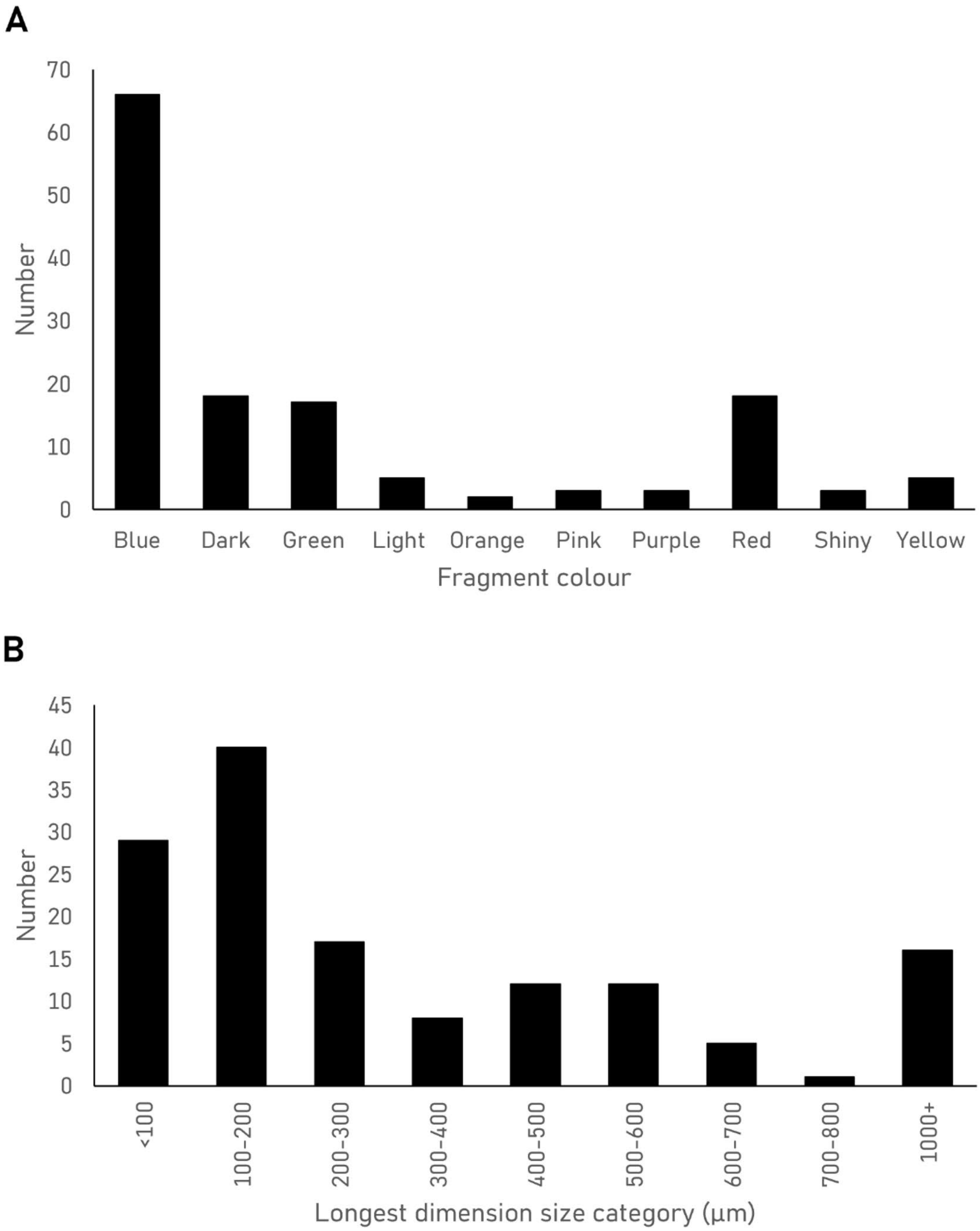
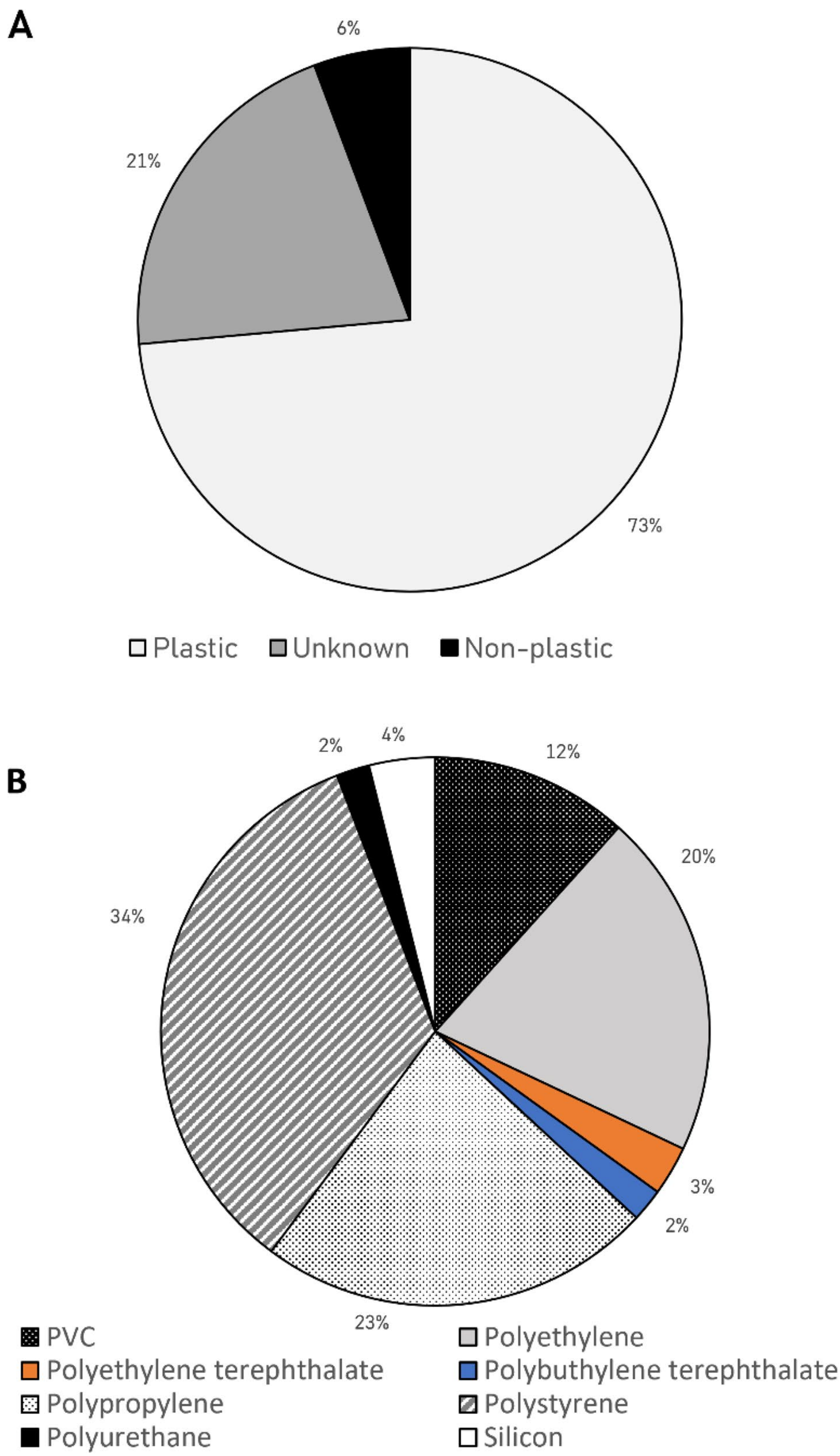


Fig. 5 Colour (**A**) and size category (longest dimension) (**B**) of suspected microplastic fragments found in wetland sediment and analysed by ATR-FTIR

of the inlet (supplementary Fig. 12), thus providing evidence that untreated sewage entered this wetland. Other plastic material found included clear PP films, orange PP fragments, blue and dark fragments, and white PET fibrous material probably also from sanitary products (such as wet wipes). The approximate size of plastic found varied from 3 mm to fully intact sanitary towels. Most plastic found (103 of 132


A
6%
21%
73%
Plastic
Unknown
Non-plastic
B
2%
4%
12%
20%
34%
3%
2%
23%
PVC
Polyethylene
Polyethylene terephthalate
Polybuthylene terephthalate
Polypropylene
Polystyrene
Polyurethane
Silicon

◂**Fig. 6** **A**) Material composition of the 140 suspected microplastic fragments found in wetland sediment and analysed by ATR-FTIR. **B**) Types of confirmed plastics in wetland sediment

pieces) was within 10 m of the inlet pipe along transect 1 (Fig. 1), indicating that there may have been a preferential flow pathway here. The furthest distance from the inlet in cell 1 where plastic was found was 40 m on transect 1: a 14 mm (longest dimension) piece of white plastic (resembling the same PE as found elsewhere). The prevalence of white plastic (73.5% of total) highlights the extent to which microplastic fragment numbers are underestimated in the present study because white fragments were excluded due to method limitations.

4 Discussion

4.1 Retention of Microplastics and Anthropogenic Fibres by Vegetation

This study provides evidence in support of the hypothesis that dense vegetation in constructed wetlands acts as an efficient barrier to anthropogenic fibre and microplastic fragment transport (Helcoski et al., 2020). Microplastic fragment and anthropogenic fibre concentration declined rapidly with increasing distance from the inlet in Northrepps ICW, and entrapment occurred as soon as 2 m from the inlet pipe (no samples were taken closer than this). Anthropogenic fibres and microplastic fragments were detected up to a maximum of 38 m from the inlet pipe (along transect 1) in the first cell of Northrepps ICW, and none were detected at levels above the LOD in cells 2 and 3. Emergent linear-leaved vegetation planted in constructed wetlands slows water velocities encouraging sedimentation of microplastics and anthropogenic fibres. For ICWs designed with shallow water depths of 20–30 cm, emergent plants and their litter form a non-homogeneous 'filter' that takes up most of, or all, the depth of the water column (also based on observations at Northrepps ICW), encouraging interception of suspended particles. Microplastic fibres and fragments have been shown to stick to biofilms on submerged vegetation (Goss et al., 2018), as well as other sediment particles as small as 0.5–2.5 μm (Kadlec, 2019). Microplastics and fibres may also aggregate with flocculent suspended particulate matter in constructed wetlands, thus increasing the relative size of the particle and the likelihood of entrapment (Leiser et al., 2021).

It is not possible to accurately determine the vegetated area that the main flow pathway contacts in the first cell of Northrepps ICW. However, given that most macroplastics were identified along transect 1, there may be a preferential flow pathway there. Additionally, during sampling at the wetland, it was observed that flow dead zones existed where there was only wet mud. It is unlikely then that the flow pathway contacts the full 1600 m^2 of the first cell at Northrepps, potentially reducing microplastic and fibre retention efficiency.

Zhou et al. (2021) found that fibres were better retained by surface flow constructed wetlands than microplastic fragments, and that larger microplastics and anthropogenic fibres were better retained than smaller ones. In the first cell of Northrepps ICW, although there was little change in microplastic fragment size (longest dimension) with increasing distance from the inlet pipe, no fragments > 1000 μm were detected beyond 20 m of the inlet, suggesting that these size fragments are better retained in dense vegetation. Bydalek et al. (2023) found a higher proportion of large fibres > 1000 μm at the Cromhall wetland outlet (21%) than at the inlet (8.3%) and suggested that this difference could be explained by the higher buoyancy of larger fibres causing slower sedimentation. In the present study at the Northrepps wetland the proportion of fibres > 800 μm were also higher at the outlet (35%) than at the inlet (27%). However, when subtracting the LOD from the fibre counts at the outlet, merely 32% are then counted, thus 68% of the datapoints making up the proportional size figures at the outlet are likely from contamination. The data are therefore not robust enough to conclude that larger fibres are more prevalent in the outlet and that vegetated wetlands preferentially retain shorter length fibres.

4.2 Factors Influencing Anthropogenic Fibre Retention

This study shows strong evidence that the Northrepps and Ingoldisthorpe ICW consistently retains anthropogenic fibres throughout a 12-month period. However, at Ingoldisthorpe the loading rate of anthropogenic fibres is low (9504 ± 19,872 fibres day^{-1}), meaning the areal removal rate is also low at 0.88 ± 1.84 fibres/m^2/day. At Northrepps ICW both the loading rate (349,920 ± 763,776 fibres day^{-1}) and areal removal rate (119 ± 261 fibres m^{-2} day^{-1}) are higher than at Ingoldisthorpe. The total area covered by plants is approximately 2755 m^2 at Northrepps ICW and 4420 m^2 at Ingoldisthorpe ICW (based on minimum estimated % plant cover values). Ingoldisthorpe ICW may therefore be able to effectively retain anthropogenic fibres when loading rates are similar to those at Northrepps, if it is assumed that the total vegetation cover is the dominant factor influencing anthropogenic fibre retention. Additionally, at the Cromhall ICW, Bydalek et al. (2023) reported calculated anthropogenic fibre loading rates of 2,616,754 fibres day^{-1} (assuming a reported wetland inflow flow rate of 9.15 L s^{-1} and concentrations of 3.31 fibres L^{-1}), with 92.2% of these retained, equating to areal removal rates of 310 fibres m^{-2} day^{-1} (Bydalek et al., 2023). The total area of plant cover at Cromhall ICW is approximately 3957 m^2. Therefore, Ingoldisthorpe ICW may potentially retain anthropogenic fibres at loading rates similar to those reported at Cromhall ICW, assuming the total area of plant coverage is the dominant factor controlling retention efficiency.

Other wetland features also inevitably play a role in microplastic and anthropogenic fibre retention. The residence time is probably of high importance: the greater the residence time, the greater the time available for microplastic and anthropogenic fibre sedimentation. The residence time for Northrepps and Ingoldisthorpe ICWs were reported as 3.1 and 16.8 days, respectively, not accounting for preferential flow pathways (Cooper et al., 2020).

Zhou et al. (2021) showed that the removal efficiency of a surface flow constructed wetland was 32.7% lower on a rainy sampling day compared to a dry one. In the present study, no sampling was conducted while it rained, although on several occasions it had rained in the morning or day(s) prior to sampling. On average, WWTP effluent discharges during the period 08:00 to 12:45 (coinciding with the time when sampling was undertaken) for both wetlands were calculated on each sampling day (supplementary Table 3). During this sampling period, the highest discharge from the Ingoldisthorpe WWTP was on the 22 November 2022, with an average loading rate of 26.2 L s^{-1} and the lowest on 29 November 2022, with an average loading rate of 13.7 L s^{-1}. On both these days anthropogenic fibres were not detected above the LOD in the inlet and outlet samples. At Northrepps ICW, the highest WWTP discharge into the wetland during the sampling period was recorded on the 6 June 2022 (4.7 L s^{-1}), and the lowest on 12 July 2022 (1.1 L s^{-1}). The loading rates were therefore variable over the course of the sampling period at both wetlands (supplementary 3), meaning the only aspect that may have been missed in the 12-month sampling campaign is the effect of rain droplet impact on re-suspension. However, rainfall droplets will not directly hit the water surface causing sediment (and potentially microplastic) re-suspension at the Northrepps wetland because the emergent vegetation percentage cover is so high (> 95%) in each cell. At Ingoldisthorpe, larger areas of the wetland are unvegetated (30–50% cover in cells 2–4), meaning rainfall may disturb sediment more, although given the low concentrations of anthropogenic fibres entering the wetland, few would be expected to be released by this mechanism.

Anthropogenic fibres were detected in water samples from the outlet of the Northrepps ICW, despite no fibres being detected above the LOD in sediment samples in cell 3 (from which the outlet flows). However, fibres may be present in fine bed sediment in concentrations below the LOD. Bioturbation may result in the movement of microplastics and anthropogenic fibres into the water column (Xue et al., 2020). At the Northrepps ICW, on several occasions large mammals (deer) were observed resting in or running through the second and third cell of the wetland. Their activity may be the most likely cause of re-suspension of entrained microplastics and anthropogenic fibres at the Northrepps ICW. Additionally, the high vegetation percentage cover at the Northrepps ICW means waterfowl are unlikely to cause bioturbation.

4.3 Implications for Wetland Design

In the present study, anthropogenic fibre concentrations were not recorded at the end of each cell. However, given that anthropogenic fibre and microplastic fragment concentrations were not detected above the LOD in cells 2 or 3 at Northrepps, most is, therefore, likely retained in cell 1. Hence, a single cell with an area of 1600 m^2 and > 95% emergent plant cover appears sufficient to retain microplastics and anthropogenic fibres from WWTP effluent. Bydalek et al. (2023) showed that the highest areal removal rate at Cromhall ICW was in the first cell at 10,066 fibres m^{-2} day^{-1} (calculated based on supplementary material in Bydalek et al., 2023). This cell had a surface area of 150 m^2, a depth of 1.5 m, was unvegetated and had a hydraulic retention time of 150 min and a loading rate of 9.5 L s^{-1}. If this is the maximum achievable areal removal rate (which may not be the case given this cell was unvegetated), then the first cell at Northrepps ICW could be much smaller and still retain most microplastics and anthropogenic fibres from the WWTP. Having a smaller first cell may be beneficial for wetland management, particularly regarding disposal of accumulated micro and macro plastic waste, if such a process were required in future site decontamination actions.

4.4 Implications for Wetland Management

To date, no studies appear to have assessed microplastic concentration in free surface flow constructed wetland (FSCW) sediment. The combined concentration of anthropogenic fibres and suspected microplastic fragments in cell 1 of Northrepps ICW was 10,090 (SD = 8519) items kg^{-1} dry sediment. This figure is comparable to the global average microplastic concentration in sewage sludge samples of 12,800 (± 5200) items kg^{-1} (Rolsky et al., 2020). Given that the present study did not include white and clear microplastic fragments, and especially since a large amount of white polyethylene macroplastic was found in cell 1, the actual concentrations in sediment are probably much higher than reported here. Additionally, Ren et al. (2020) found that 58% of microplastics in sewage sludge were white. The high microplastic and anthropogenic fibre concentrations found in the Northrepps wetland may have significant impacts on longer term management. Above-ground plant material can be harvested in constructed wetlands for the purpose of enhancing nutrient removal, although it is questionable how effective this practice is (Vymazal et al., 2010). Above ground plant harvesting would presumably have minimal impact on microplastic retention in constructed wetlands because microplastics will be retained by submerged vegetation and debris only. However, Zheng et al. (2015) reported that the density of plants in a FSCW increased by 7.4% a year after above surface harvesting to 175 shoots m^{-2}, compared to a 16.1% decline in plant density without harvesting over the same time period. The slightly higher plant density may aid in microplastic retention by having more area for plastics to attach to and by increasing residence times for enhanced sedimentation (Helcoski et al., 2020).

Dredging is a long-term practice in constructed wetland management (Hernández-del Amo et al., 2020). Dredging in FSCWs is recommended to a depth of 25 cm by mechanical excavation (Zhu et al., 2022), thus significant microplastic and anthropogenic fibre loads will be present within this, presenting similar problems as wastewater sludge in terms of land application (Liu et al., 2021), including uptake of nano-plastics into crops (Li et al., 2020). Although it was not quantified in the present study, the Northrepps ICW contains a large amount of macroplastic that will also be removed during dredging.

ICWs are highly effective at retaining microplastics and anthropogenic fibres from sewage effluent, although they may be ineffective as a long-term retention mechanism: the problem of how to manage the plastic that has built up in the wetland sediment is significant. Incineration of the dredged material destroys microplastics (Vuori & Ollikainen, 2022), although this technique is prohibitively expensive (Milojevic & Cydzik-Kwiatkowska, 2021). ICWs have been reported to be cost effective measures to reduce nutrient pollution from WWTP discharges (Cooper et al., 2020), but this may be reduced when costs to legally dispose deposited plastic waste are accounted for. To avoid spreading a significant amount of plastic waste to land when wetland sediment is dredged, it may be beneficial for future ICW designs to include a small first cell that is densely vegetated to retain most of the microplastic and anthropogenic fibre loads from WWTP effluent. This way, a smaller amount of material that is highly concentrated in microplastics could be disposed of at controlled landfill, and the dredged material from the remaining cells of the ICW that are less contaminated with microplastics could be safely applied to land as regulated practice dictates. Further research should be carried out to determine the optimum depth, vegetation type and hydraulic conditions of this recommended first cell to retain microplastic fragments and anthropogenic fibres most effectively in the smallest possible area.

5 Conclusions

The key findings of this research are summarised as follows:

1. Northrepps ICW consistently retained anthropogenic fibres over a 12-month period, with average removal efficiencies of 99.3% (with an average of 349,920 fibres entering the wetland day^{-1}).
2. Ingoldisthorpe ICW consistently retained 100% of anthropogenic fibres over a 12-month period, although the wetland received low and intermittent anthropogenic fibre loads from WWTP effluent (averaging 9504 fibres day^{-1}).
3. There was no evidence of seasonality in anthropogenic fibre removal performance at either wetland.
4. Microplastic fragments and anthropogenic fibres were prevalent in the fine bed sediment of Northrepps ICW: averaging 10,090 (SD = 8519) items kg^{-1} dry sediment in the first cell. No microplastic fragments or fibres were detected in sediment samples in cells 2 and 3 at Northrepps ICW.
5. Approximately 54% of anthropogenic fibres entering the ICWs were plastic, dominated by polyester. Of the suspected microplastic fragments in sediment samples from the Northrepps ICW, 73% were confidently identified as plastic (mostly polystyrene, polyethylene, or polypropylene).
6. Future ICW design may include a smaller first cell to retain most of the sewage effluent derived microplastics and anthropogenic fibres to improve long-term management prospects.

Acknowledgements With thanks to Hayley McIlwraith for her assistance with method development, Benjamin Grover for piloting the drone used to take aerial photos of Northrepps wetland, Maria Staines for assistance with collecting sediment samples, Eddie Anderson for allowing site access to the Northrepps ICW and Anglian Water for access to the Ingoldisthorpe ICW and providing discharge data.

Funding Funding was provided by the SCI Faculty at the University of East Anglia and by Norfolk Rivers Trust.

Data availability Data sets generated during the current study are available from the corresponding author on reasonable request.

Declarations

Competing interests The authors have no competing interests to declare that are relevant to the content of this article.

References

Babatunde, A.O., Zhao, Y.Q., O'Neill, M. and O'Sullivan, B. (2008). Constructed wetlands for environmental pollution control: A review of developments, research and practice in Ireland. *Environment International*, [online] 34(1), pp.116–126. https://doi.org/10.1016/j.envint.2007.06.013.

Blair, R. M., Waldron, S., & Gauchotte-Lindsay, C. (2019). Average daily flow of microplastics through a tertiary wastewater treatment plant over a ten-month period. *Water Research, 163*(1), 114909. https://doi.org/10.1016/j.watres.2019.114909

Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T., & Thompson, R. (2011). Accumulation of Microplastic on Shorelines Woldwide: Sources and Sinks. *Environmental Science & Technology, 45*(21), 9175–9179. https://doi.org/10.1021/es201811s

Bunce, J.T., Ndam, E., Ofiteru, I.D., Moore, A. and Graham, D.W. (2018). A Review of Phosphorus Removal Technologies and Their Applicability to Small-Scale Domestic Wastewater Treatment Systems. *Frontiers in Environmental Science*, [online] 6(1). https://doi.org/10.3389/fenvs.2018.00008.

Bydalek, F., Ifayemi, D., Reynolds, L., Barden, R., Kasprzyk-Hordern, B., & Wenk, J. (2023). Microplastic dynamics in a free water surface constructed wetland. *Science of The Total Environment, 858*, 160113. https://doi.org/10.1016/j.scitotenv.2022.160113

Carr, S. A., Liu, J., & Tesoro, A. G. (2016). Transport and fate of microplastic particles in wastewater treatment plants. *Water Research, 91*, 174–182. https://doi.org/10.1016/j.watres.2016.01.002

Cooper, R. J., Hawkins, E. T., Locke, J. W., Thomas, T., & Tosney, J. (2020). Assessing the environmental and economic efficacy of two integrated constructed wetlands at mitigating eutrophication risk from sewage effluent. *Water and*

Environment Journal, 34(4), 669–678. https://doi.org/10.1111/wej.12605

Cooper, R. J., Warren, R. J., Clarke, S. J., & Hiscock, K. M. (2022). Evaluating the impacts of contrasting sewage treatment methods on nutrient dynamics across the River Wensum catchment UK. *Science of the Total Environment, 804*(1), 150146. https://doi.org/10.1016/j.scitotenv.2021.150146

Cowger, W., Steinmetz, Z., Gray, A., Munno, K., Lynch, J., Hapich, H., Primpke, S., De Frond, H., Rochman, C., & Herodotou, O. (2021). Microplastic Spectral Classification Needs an Open Source Community: Open Specy to the Rescue! *Analytical Chemistry, 93*(21), 7543–7548. https://doi.org/10.1021/acs.analchem.1c00123

Dawson, A. L., Santana, M., Nelis, J. L. D., & Motti, C. A. (2023). Taking control of microplastics data: A comparison of control and blank data correction methods. *Journal of Hazardous Materials, 443*(1), 130218–130218. https://doi.org/10.1016/j.jhazmat.2022.130218

Department for Environment, Food & Rural Affairs (2018). *World leading microbeads ban comes into force*. [online] GOV.UK. Available at: https://www.gov.uk/government/news/world-leading-microbeads-ban-comes-into-force.

Edo, C., González-Pleiter, M., Tamayo-Belda, M., Ortega-Ojeda, F. E., Leganés, F., Fernández-Piñas, F., & Rosal, R. (2020). Microplastics in sediments of artificially recharged lagoons: Case study in a Biosphere Reserve. *Science of the Total Environment, 729*, 138824. https://doi.org/10.1016/j.scitotenv.2020.138824

Fantner, G. (2013). *A brief introduction to error analysis and propagation*. [online] Available at: https://www.epfl.ch/labs/lben/wp-content/uploads/2018/07/Error-Propagation_2013.pdf.

Goss, H., Jaskiel, J., & Rotjan, R. (2018). Thalassia testudinum as a potential vector for incorporating microplastics into benthic marine food webs. *Marine Pollution Bulletin, 135*, 1085–1089. https://doi.org/10.1016/j.marpolbul.2018.08.024

Hammond, P., Suttie, M., Lewis, V.T., Smith, A.P. and Singer, A.C. (2021). Detection of untreated sewage discharges to watercourses using machine learning. *Clean Water*, 4(1). https://doi.org/10.1038/s41545-021-00108-3.

Helcoski, R., Yonkos, L. T., Sanchez, A., & Baldwin, A. H. (2020). Wetland soil microplastics are negatively related to vegetation cover and stem density. *Environmental Pollution, 256*, 113391. https://doi.org/10.1016/j.envpol.2019.113391

Hernández-del Amo, E., Ramió-Pujol, S., Gich, F., Trias, R., & Bañeras, L. (2020). Changes in the Potential Activity of Nitrite Reducers and the Microbial Community Structure After Sediment Dredging and Plant Removal in the Empuriabrava FWS-CW. *Microbial Ecology, 79*(3), 588–603. https://doi.org/10.1007/s00248-019-01425-4

Iyare, P.U., Ouki, S.K. and Bond, T. (2020). Microplastics removal in wastewater treatment plants: a critical review. *Environmental Science: Water Research & Technology*, [online] 6(10), pp.2664–2675. https://doi.org/10.1039/D0EW00397B.

Jarvie, H. P., Neal, C., & Withers, P. J. A. (2006a). Sewage-effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus? *Science of the Total Environment, 360*(1–3), 246–253. https://doi.org/10.1016/j.scitotenv.2005.08.038

Jarvie, H. P., Neal, C., Jürgens, M. D., Sutton, E. J., Neal, M., Wickham, H. D., Hill, L. K., Harman, S. A., Davies, J. J. L., Warwick, A., Barrett, C., Griffiths, J., Binley, A., Swannack, N., & McIntyre, N. (2006b). Within-river nutrient processing in Chalk streams: The Pang and Lambourn. *UK. Journal of Hydrology, 330*(1–2), 101–125. https://doi.org/10.1016/j.jhydrol.2006.04.014

Kadlec, R. H. (2019). *Treatment marshes for runoff and polishing*. CRC Press.

Koelmans, A.A., Berte, Mintenig, S.M., Nor, M., Redondo-Hasselerharm, P.E. and Vera (2023). Towards a rational and efficient risk assessment for microplastics. *Trends in Analytical Chemistry*, 165, pp.117142–117142https://doi.org/10.1016/j.trac.2023.117142

Leiser, R., Schumann, M., Dadi, T. and Katrin Wendt-Potthoff (2021). Burial of microplastics in freshwater sediments facilitated by iron-organo flocs. *Scientific Reports*, 11(1). https://doi.org/10.1038/s41598-021-02748-4.

Li, L., Luo, Y., Li, R., Zhou, Q., Peijnenburg, W.J.G.M., Yin, N., Yang, J., Tu, C. and Zhang, Y. (2020). Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. *Nature Sustainability*, [online] 3(11), pp.929–937. https://doi.org/10.1038/s41893-020-0567-9.

Liu, H., Wang, Z., Nghiem, L. D., Gao, L., Zamyadi, A., Zhang, Z., Sun, J., & Wang, Q. (2021). Solid-Embedded Microplastics from Sewage Sludge to Agricultural Soils: Detection, Occurrence, and Impacts. *ACS ES&T Water, 1*(6), 1322–1333. https://doi.org/10.1021/acsestwater.0c00218

Meeker, J.D., Sathyanarayana, S. and Swan, S.H. (2009). Phthalates and other additives in plastics: human exposure and associated health outcomes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, [online] 364(1526), pp.2097–2113. https://doi.org/10.1098/rstb.2008.0268.

Meteorological Office (2023). *Weybourne (Norfolk) UK climate averages*. [online] Met Office. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/u12v1xhdm [Accessed 8 Aug. 2023].

Milojevic, N. and Cydzik-Kwiatkowska, A. (2021). Agricultural Use of Sewage Sludge as a Threat of Microplastic (MP) Spread in the Environment and the Role of Governance. *Energies*, [online] 14(19), p.6293. https://doi.org/10.3390/en14196293.

Napper, I. E., Parker-Jurd, F. N. F., Wright, S. L., & Thompson, R. C. (2023). Examining the release of synthetic microfibres to the environment via two major pathways: Atmospheric deposition and treated wastewater effluent. *Science of the Total Environment, 857*, 159317. https://doi.org/10.1016/j.scitotenv.2022.159317

Prata, J.C. (2018). Microplastics in wastewater: State of the knowledge on sources, fate and solutions. *Marine Pollution Bulletin*, [online] 129(1), pp.262–265. https://doi.org/10.1016/j.marpolbul.2018.02.046.

Ren, X., Sun, Y., Wang, Z., Barceló, D., Wang, Q., Zhang, Z., & Zhang, Y. (2020). Abundance and characteristics of microplastic in sewage sludge: A case study of Yangling, Shaanxi province, China. *Case Studies in Chemical and Environmental Engineering, 2*, 100050. https://doi.org/10.1016/j.cscee.2020.100050

Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K., Athey, S., Huntington, A., McIlwraith, H., Munno, K., Hannah De Frond, Kolomijeca, A., Erdle, L., Grbic, J., Bayoumi, M., Borrelle, S.B., Wu, T., Santoro, S., Werbowski, L.M. and Zhu, X. (2019). Rethinking microplastics as a diverse contaminant suite. *Environmental Toxicology and Chemistry*, [online] 38(4), pp.703–711. https://doi.org/10.1002/etc.4371.

Rolsky, C., Kelkar, V., Driver, E., & Halden, R. U. (2020). Municipal sewage sludge as a source of microplastics in the environment. *Current Opinion in Environmental Science & Health, 14*, 16–22. https://doi.org/10.1016/j.coesh.2019.12.001

Scholz, M., Harrington, R., Carroll, P., & Mustafa, A. (2007). The Integrated Constructed Wetlands (ICW) concept. *Wetlands, 27*(2), 337–354. https://doi.org/10.1672/0277-5212(2007)27[337:ticwic]2.0.co;2

Siegfried, M., Koelmans, A. A., Besseling, E., & Kroeze, C. (2017). Export of microplastics from land to sea A Modelling Approach. *Water Research, 127*, 249–257. https://doi.org/10.1016/j.watres.2017.10.011

Simon, M., van Alst, N. and Vollertsen, J. (2018). Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. *Water Research*, [online] 142, pp.1–9. https://doi.org/10.1016/j.watres.2018.05.019.

Vermeiren, P., Muñoz, C., & Ikejima, K. (2020). Microplastic identification and quantification from organic rich sediments: A validated laboratory protocol. *Environmental Pollution, 262*, 114298. https://doi.org/10.1016/j.envpol.2020.114298

Vuori, L. and Ollikainen, M. (2022). How to remove microplastics in wastewater? A cost-effectiveness analysis. *Ecological Economics*, [online] 192, p.107246. https://doi.org/10.1016/j.ecolecon.2021.107246.

Vymazal, J., Kröpfelová, L., Švehla, J., & Štíchová, J. (2010). Can multiple harvest of aboveground biomass enhance removal of trace elements in constructed wetlands receiving municipal sewage? *Ecological Engineering, 36*(7), 939–945. https://doi.org/10.1016/j.ecoleng.2010.04.010

Wagstaff, A., Lawton, L. A., & Petrie, B. (2022). Polyamide microplastics in wastewater as vectors of cationic pharmaceutical drugs. *Chemosphere, 288*, 132578–132578. https://doi.org/10.1016/j.chemosphere.2021.132578

Woodward, J., Li, J., Rothwell, J. and Hurley, R. (2021). Acute riverine microplastic contamination due to avoidable releases of untreated wastewater. *Nature Sustainability*, 4. https://doi.org/10.1038/s41893-021-00718-2.

Xu, D., Yin, X., Zhou, S., Jiang, Y., Xi, X., Sun, H. and Wang, J. (2022). A review on the remediation of microplastics using constructed wetlands: Bibliometric, co-occurrence, current trends, and future directions. *Chemosphere*, [online] 303, p.134990. https://doi.org/10.1016/j.chemosphere.2022.134990.

Xue, B., Zhang, L., Li, R., Wang, Y., Guo, J., Yu, K., & Wang, S. (2020). Underestimated Microplastic Pollution Derived from Fishery Activities and 'Hidden' in Deep Sediment. *Environmental Science & Technology, 54*(4), 2210–2217. https://doi.org/10.1021/acs.est.9b04850

Ziajahromi, S., Neale, P. A., Rintoul, L., & Leusch, F. D. L. (2017). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research, 112*, 93–99. https://doi.org/10.1016/j.watres.2017.01.042

Zhang, Q., Huang, J., Dzakpasu, M., Gao, Z., Zhou, W., Zhu, R., & Xiong, J. (2022). Assessment of plants radial oxygen loss for nutrients and organic matter removal in full-scale constructed wetlands treating municipal effluents. *Bioresource Technology, 360*, 127545–127545. https://doi.org/10.1016/j.biortech.2022.127545

Zheng, Y., Wang, X. C., Ge, Y., Dzakpasu, M., Zhao, Y., & Xiong, J. (2015). Effects of annual harvesting on plants growth and nutrients removal in surface-flow constructed wetlands in northwestern China. *Ecological Engineering, 83*, 268–275. https://doi.org/10.1016/j.ecoleng.2015.06.035

Zhou, X., Zhao, Y., Pang, G., Jia, X., Song, Y., Guo, A., Wang, A., Zhang, S., & Ji, M. (2021). Microplastic abundance, characteristics and removal in large-scale multi-stage constructed wetlands for effluent polishing in northern China. *Chemical Engineering Journal, 322*, 132752. https://doi.org/10.1016/j.cej.2021.132752

Zhu, L., Zhang, H., Li, Y., Sun, W., Song, C., Wang, L., Du, G., Qiao, S., Sun, J. and Nuamah, L.A. (2022). Dredging effects on nutrient release of the sediment in the long-term operational free water surface constructed wetland. *Journal of Environmental Management*, [online] 322, p.116160. https://doi.org/10.1016/j.jenvman.2022.116160.