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Degradation of cellulose-based wet wipes marketed as 'biodegradable' in their receiving urban rivers☆

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ABSTRACT

The environmental degradation and fate of cellulose-based 'biodegradable' wet wipes under real-world conditions remain underexplored, particularly in urban freshwater systems where they are frequently discharged via toilet flushing. Building on previous mesocosm-based experiment, this study quantified in-situ degradation of two commercially available cellulose-based wipes labelled 'biodegradable' across ten urban rivers and streams and identified key environmental drivers of degradation. Tensile strength loss was used as a proxy measurement for wipe degradation alongside cotton strip bioassays as ecological benchmarks. Wipes rich in natural cellulose degraded substantially faster (Brand A: 6.69 ± 3.19 % per day) than those dominated by regenerated cellulose (Brand B: 3.12 ± 1.93 % per day) or cotton bioassay controls (2.22 ± 1.00 % per day). Degradation rates were shaped by microbial biomass, total dissolved solids, temperature, and river-level fluctuations, although exposure duration had the largest effect – suggesting complex interactions between physical and biological processes. Despite early-stage degradation, most wipes persisted after five weeks, challenging their biodegradability claims. These findings highlight potential ecological risks from persistent textile fibre pollution, emphasising the need for updated labelling and biodegradability standards that appropriately reflect real-world freshwater conditions, as well as greater scrutiny of plastic-free alternative products and their environmental fates in general.

1. Introduction

Cellulose-based wet wipes, often marketed as 'biodegradable', have been introduced as plastic-free alternatives and are intended to break down quickly in natural environments (Allison et al., 2023). Yet their in-situ degradation behaviour and environmental fate in freshwater – particularly urban rivers and streams – remain poorly understood. These environments frequently receive flushed wipes via sewage discharges (McGoran et al., 2017; Thames21, 2019; McCoy et al., 2020; Besley and Cassidy, 2022; Fortibuoni et al., 2025), historically dominated by plastic-containing wipes with poorly degradable fibres but now increasingly by cellulosic varieties (see Allison et al., 2023, for further details on material properties). Moreover, eco-friendly branding of these alternatives as biodegradable may even encourage flushing (Kachef, 2024; Kargar and Joksimovic, 2024), increasing their presence from source to sea.

In these environments, cellulose-based wipes also shed large quantities of microfibres (Kwon et al., 2022) – a form of pollutant that travels through wastewater infrastructure and into receiving waters (UKWIR, 2022; Tserendorj et al., 2024; Bach et al., 2025). Both macro- and micro-scale pollution from these wipes can harm aquatic wildlife via ingestion, chemical additive leaching, and contaminant or pathogen 'trojan horse' effects (Windsor et al., 2019; McCoy et al., 2020; Ó Briain et al., 2020; Zimmerman et al., 2020; Courtene-Jones et al., 2024; MacAulay et al., 2024; Smith et al., 2024), with potential risks to human health as well (e.g. Metcalf et al., 2024). These concerns underscore the need to quantify both the degradation and fate of biodegradable wipes under real-world conditions, in order to better understand their ecological and human health impacts.

Most existing research focuses on lab-based wastewaters (Joksimovic et al., 2020; Lee et al., 2021; Kwon et al., 2022; Pedersen et al., 2022; Kargar and Joksimovic, 2024) or beach mesocosms (Metcalf et al., 2024), assessing dispersibility or microfibre release rather than microbial or molecular processes. These studies often report slow or minimal

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breakdown: Pedersen et al. (2022) found cellulosic wipes retained >93 % structure after 48 h in simulated wastewater, while Kwon et al. (2022) reported greater microfibre shedding from natural cellulose wipes than regenerated or synthetic ones. Even under high flow velocities in sewer pipe models, many remain intact (Kargar and Joksimovic, 2024), and fail disintegration tests such as the IWSFG 'Slosh box' (Joksimovic et al., 2020). Outside wastewater, Metcalf et al. (2024) found compostable wipes persisted in beach sand for 8–15 weeks, with viable *E. coli* for at least 14 weeks; however, the drivers of degradation were not examined.

Our recent mesocosm study (Allison et al., 2025a) simulated river conditions to explore wipe degradation in controlled field settings. Degradation arose from physical fibre loss, likely driven by continual flow and turbulence (Kargar and Joksimovic, 2024), and from environmental conditions (temperature, pH, total dissolved solids [TDS]) that likely supported microbial activity. Yet overall degradation remained limited over five weeks, possibly due to low microbial availability typical of upland stream mesocosms (Zancarini et al., 2017). These findings align with evidence that microbial colonisation and enzymatic activity govern cellulose breakdown in freshwater, shaped by pH, light, temperature, and nutrient availability (Chauvet et al., 2016; Griffiths and Tiegs, 2016; Burdon et al., 2020; Pingram et al., 2020). However, mesocosms did not capture the complexity of urban rivers - where wipes are more likely to accumulate - and where variable chemistry, hydrology, microbial assemblages, and pollution may strongly influence degradation (Paul and Meyer, 2001; Imberger et al., 2010; Burdon et al., 2020; Tiegs et al., 2024).

The mesocosm study also demonstrated the utility of tensile strength loss (TSL) as a reliable proxy for cellulosic degradation. TSL is widely used in freshwater ecology through cotton strip assays to track microbial and invertebrate activity and infer ecosystem functioning (Tiegs et al., 2013; Colas et al., 2019; Carballeira et al., 2020; Hill et al., 2022). Applied to wipes, TSL offers a practical field method to assess degradation, though it cannot distinguish between mechanical, chemical, or biological pathways (see Allison et al., 2023). Additional chemical or structural analyses are needed to differentiate these degradation pathways.

Here, we investigate the degradation of commercially-available cellulose-based wipes marketed as 'biodegradable' in urban rivers and streams - environments where they are known to accumulate due to incorrect toilet disposal. Cotton strip assays served as ecological controls to benchmark degradation. Building on mesocosm findings, we examine how water chemistry, biological activity, and local conditions influence wipe degradation under dynamic, real-world flows. The study aims to clarify the fate of biodegradable alternatives to plastic wipes and assess their environmental risks in freshwater ecosystems.

2. Methods

2.1. Study area

To assess degradation of biodegradable wet wipes in environments where they commonly accumulate, samples were deployed in ten urban river and stream sites in Cardiff, Wales, UK (Fig. 1). Cardiff covers 141 km^2, has a population of ~383,500 (ONS, 2024), and a temperate

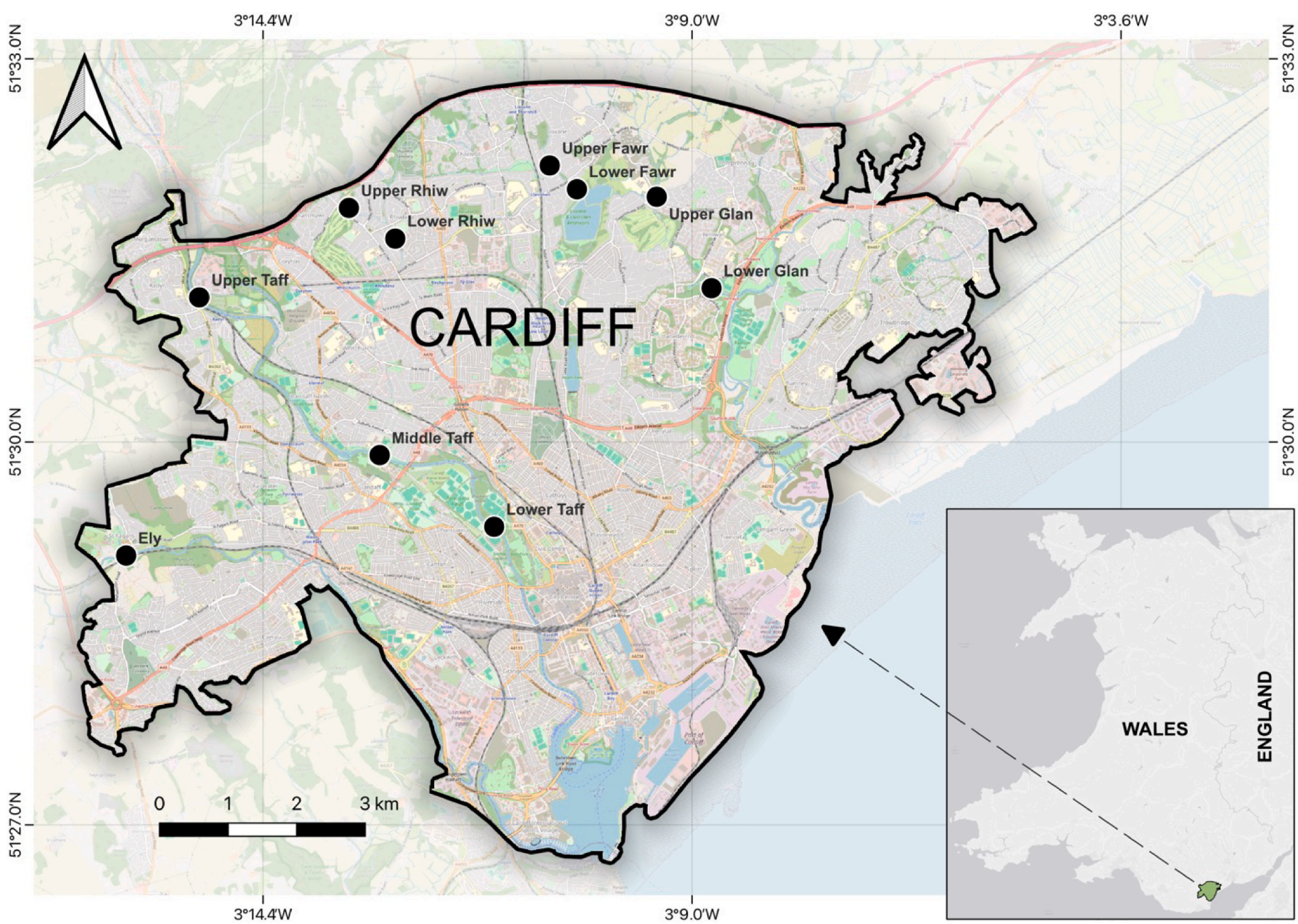

Fig. 1. Locations of urban river and stream study sites in Cardiff, Wales, UK. Rhiwbina and Glandulais sites are abbreviated here. Map generated using the QGIS 3.36 software. OSM data provided by © OpenStreetMap contributors URL: https://www.openstreetmap.org/.

maritime climate with river temperatures of 2–25 °C (Farr et al., 2017), solar radiation of 8.5 MJ/m^2/day (Solargis, 2025), mean annual rainfall of ~1203 mm, and air temperatures of 7–15 °C (Met Office, 2025). Sites included three reaches of the River Taff, one on the River Ely, and six residential streams (upper and lower Rhiwbina [Rhydwaedlyd Brook], Nant Fawr, and Nant Glandulais). Selection was based on likely wipe accumulation, prior pollution research (Hadley et al., in prep), accessibility, and efforts to minimise public interference. Sampling ran for five weeks in October–November 2024.

2.2. Materials and study design

Two commercial wet wipe brands (Brand A and Brand B), both marketed as biodegradable and 100 % bio-based with similar listed additives, were selected. Brand B packaging claimed that it can be disposed of in a garden compost bin and will disappear in under 2 months, citing ISO 14855 aerobic controlled composting test conditions, while Brand A did not specify any certification scheme, standard, or test conditions. No specific cellulosic fibre types (natural or regenerated), coatings or modifications were disclosed on either product. It should be noted that laboratory composting/aqueous standards (e.g., ISO/EN/ASTM) differ fundamentally from flowing freshwater (see Allison et al., 2025a); our study evaluates actual in-situ river behaviour rather than compliance with any laboratory standard. Standardised cotton strips (8 × 2.5 cm) served as controls due to their high cellulose content (>95 %) and established sensitivity to aquatic biodegradation (Tiegs et al., 2013). Wipes were deployed in their original sizes (Brand A: 20 × 16 cm; Brand B: 18 × 16.5 cm) and as cut strips matching cotton controls. Preparation methods follow Allison et al. (2025a).

At each site, five full-sized wipes and fifteen strips per brand, plus five cotton strips, were enclosed in fine mesh bags (100 μm aperture, 11 × 15 cm; iQuatics), which prevented macroinvertebrate interference and allowed microbial degradation to occur in isolation. Mesh bags were colour-coded with cable ties for identification, secured to bricks, submerged, and anchored to the streambed with metal pegs to prevent loss and standardise exposure across sites. Saturated wipes generally sink and/or snag; accordingly, this design reflects primarily bed-contact conditions. However, at shallow stream sites some bags floated partially, and wipes intermittently occupied the upper water column, providing a mix of benthic and near-surface exposure states.

Samples were retrieved weekly over five intervals. Each collection included one full-sized wipe and three strips per brand, and one cotton strip. Retrieved samples were sealed in 1 L zip-lock bags (24 × 17 cm) with 5 g desiccant, chilled, and kept in the dark to prevent further degradation during transport. In the laboratory, samples were rinsed with deionised water to remove debris, dried at 40 °C for 24 h, and stored in desiccators until analysis. Across all sites, 50 full wipes and 150 strips per brand were deployed, along with 50 cotton strips, enabling comparisons of degradation rates across materials and environmental conditions.

2.3. Quantifying degradation using tensile strength loss

In this study, we use the term degradation operationally as loss of mechanical integrity in situ, quantified via tensile strength loss (TSL) – a field-robust mechanical proxy measure widely used in cellulose decomposition assays (e.g. Tiegs et al., 2013; Colas et al., 2019). We measured TSL on strips using a tensiometer (Zwick/Roell Z050) with self-tightening roller grips, pulling at 20 cm/min (preload = 1 N; preload speed = 5 cm/min; initial grip separation = 11.61 cm) until failure. Reference strength values came from non-incubated strips (n = 10 per material) processed identically to incubated samples.

TSL was calculated as the percentage of initial tensile strength lost per day of incubation, using a linear degradation model from established methods (Tiegs et al., 2013; Colas et al., 2019; Tiegs et al., 2019):

$$TSL = \left[1 - \left(\frac{Tensile\ Strength_{treatment\ strips}}{Tensile\ Strength_{reference\ strips}}\right)\right] \times 100 / Incubation\ Time \tag{1}$$

where $\text{Tensile Strength}_{\text{treatment strips}}$ is the maximum tensile strength of incubated strips, $\text{Tensile Strength}_{\text{reference strips}}$ is the mean tensile strength of the ten non-incubated reference strips for that respective material type, and incubation time is the number of days the strip was exposed for.

To account for temperature variation across sites and sampling intervals, TSL was also expressed per degree-day, where incubation time was replaced with cumulative degree-days, calculated by summing mean daily temperatures above 0 °C for each site.

2.4. Environmental variables

To monitor environmental drivers of degradation, pH and total dissolved solids (TDS) were recorded weekly at each site using a handheld test (HANNA instruments® pH/EC/TDS Combo Tester, HI98129). Hourly temperature and light intensity were measured with automated loggers (HOBO Pendant® Temp/Light) attached to each site's anchor brick.

To characterise hydrological conditions at each site, both static and time-varying proxy metrics were incorporated. The discharge equation,

$$Q = A\ x\ v \tag{2}$$

where Q is the mean flow rate (m^3/s), A is cross-sectional area (m^2), and v is velocity (m/s), structured our approach. In practice, mean discharge values (Q) were obtained from National River Flow Archive (NRFA) gauging stations (https://nrfa.ceh.ac.uk), while site-specific cross-sectional areas (A) were measured in the field; average velocities (v) were then derived as Q/A. These estimates are proxies, as NRFA mean flows and field cross-sections were not measured simultaneously.

For River Taff sites, long-term mean flow data were taken from NRFA Station 57005 (Pontypridd). Although upstream of Cardiff, this gauge (mean = 21.175 m^3/s) closely matched historical flows at the decommissioned downstream Cardiff station (57003; 21.373 m^3/s, 1965–1972), supporting its use as a representative proxy. Velocities for Upper, Middle, and Lower Taff sites were calculated by dividing this discharge by each site's cross-sectional area.

For the River Ely, mean flows from NRFA Station 57009 (St Fagans) were combined with field cross-sections to calculate average velocity (0.45 m/s). This Ely velocity was then scaled for the ungauged streams according to their cross-sectional area relative to the Ely, assuming smaller channels with lower area would sustain proportionally reduced velocities. This scaling provides differentiated but physically plausible flow estimates where no direct gauging data exist.

Temporal variability was captured using daily river stage (water level) data from NRW gauges on the Ely (St Fagans) and Taff (Western Avenue). As no stream gauges exist, Taff stage data were also applied to streams as a regional rainfall-driven proxy. These values were not used to calculate discharge or velocity but instead provided a consistent indicator of temporal fluctuations in flow conditions across sites.

2.5. Microbial biomass determination

To assess microbial activity, a terracotta tile (12 x 25 × 5 cm) was deployed at each site (n = 10) to allow biofilm accumulation over the five-week period. After five weeks, biofilm from both faces was scraped into 50 mL Corning tubes, stored on ice and in darkness. Microbial biomass was estimated gravimetrically using ash-free dry mass (AFDM). Biofilm samples were filtered, dried to constant weight, combusted in a muffle furnace, and reweighed. Calculations followed Steinman et al. (2007):

$$Dry\ Mass = \left(W_a - W_f\right)\ /\ A_t \tag{3}$$

$$AFDM = (W_a - W_{ash}) \ / \ A_t \quad (4)$$

where W_a represents the dried biomass weight on the filter before ashing (mg), W_f is the filter weight alone (mg), W_{ash} is the biomass weight after ashing (mg), and A_t is the total tile area scraped (cm^2). In Equation (4), the filter weight is not subtracted explicitly because it is present in both W_a and W_{ash} and thus cancels when calculating organic fraction.

2.6. Scanning electron microscopy

To characterise wet wipe fibre composition and degradation patterns, scanning electron microscopy (SEM) was used on both pristine and incubated wipe samples. Samples were air-dried for 24 h, mounted on aluminium stubs with conductive carbon tape, and coated with gold-palladium (Quorum Q150T ES sputter coater) to improve conductivity and image resolution. SEM imaging (TESCAN MAIA-3; Cardiff Catalysis Institute Electron Microscope Facility, UK), at 10.0 kV accelerating voltage captured magnifications from 250 × to 2500 × to examine fibre arrangement and surface features. Integrated energy-dispersive X-ray spectroscopy (EDS; Oxford Instruments X-Max 80^{NT} silicon drift detector) was used to identify elemental profiles of cellulose fibres, additives, and environmental deposits.

2.7. Statistical analysis

All statistical analyses were conducted in R (version 4.3.1), using the following packages: "*tidyverse*", "*ggstatsplot*", "*patchwork*", and "*broom*" (Wickham et al., 2019; Patil, 2021; Pedersen, 2024; Robinson et al., 2025). Data normality and homoscedasticity were checked with Shapiro-Wilk tests and Q-Q/residual plots.

Negative TSL values (~3–4 % of data) and extreme light intensity outliers (top 5 %) were removed to improve modelling and visualisation. Differences in temperature, light, and square-root transformed TSL among materials, sites, and timepoints were tested using Aligned Rank Transformation (ART) ANOVA and post-hoc pairwise comparisons (Kay et al., 2021). ART was selected for its robustness in factorial, non-parametric designs with nested or hierarchical structures.

Environmental influences on TSL were assessed initially using correlation matrices, then through a modelling workflow adapted from Hill et al. (2022). Environmental variables were first scaled and screened for multicollinearity using variance inflation factors (VIFs; <7 removal threshold) from the '*car*' package (Fox and Weisberg, 2019). Outliers exceeding three times the interquartile range (IQR) were removed. Two response variables were modelled: raw TSL (% d) and temperature-adjusted TSL (% dd).

Each remaining environmental predictor was then fitted to four candidate model forms – linear, exponential, logarithmic, and quadratic – with lowest Akaike Information Criterion (AIC) to identify the best fit. Exposure duration (Day) was included in additional models as an additive covariate to account for influences on degradation over time. The full model selection process was repeated for each material. R^2 and p-values from best fit models were extracted for model evaluation, with Benjamini-Hochberg (BH) correction applied to control for false discovery rate. Significant models (adjusted $p < 0.05$) were further filtered by diagnostic checks for normality (Shapiro–Wilk test), homoscedasticity (Breusch–Pagan test), and high-leverage points (Cook's distance ≥4/n). Final plots used 95 % confidence intervals and unscaled predictor values for interpretability.

Linear and non-linear time effects on TSL were tested independently using regressions and generalised additive mixed models (GAMMs; '*mgcv*' package) with smooth terms for Day (k = 5) and random intercepts for Site (Wood, 2011). Effective degrees of freedom (edf) values were used to determine the complexity (non-linearity) of the smoothed terms, and residual and basis dimension diagnostics confirmed model validity.

3. Findings

3.1. Environmental variation at urban sites

Water chemistry and environmental conditions varied across the ten sites (Table 1; Supplementary Fig. 1) Mean pH was consistent (7.94–8.09), though Upper Rhiwbina was more variable (SD = 0.51) than other sites (SD = 0.12–0.26). Mean total dissolved solids (TDS) ranged from 114 to 260 mg/L, with higher values in non-Taff streams (187–211 mg/L). Within the River Taff, TDS increased progressively downstream.

Temperature and light data were available for only seven sites due to early sample washouts at Upper and Middle Taff after week 1 and datalogger failure at Upper Rhiwbina. Hourly logger readings were averaged to daily means, then summarised as site-level mean ± SD for Table 1. Both variables differed significantly between sites (ART: temperature $F_{6, 195} = 27.29$, $p < 0.001$; light $F_{6, 196} = 17.62$ $p < 0.001$). Ely recorded the highest mean values, Lower Fawr the lowest temperature, and Lower Glandulais the lowest light (Table 1; Supplementary Fig. 1). Mean temperature declined during the study, especially between 17 and 23 November. Despite filtering extreme light values (>300 lux), outliers remained, likely from water depth or shading changes.

Hydrology varied between streams and rivers. Flows ranged from 0.02 m^3/s (Upper Glandulais) to 22.46 m^3/s (Lower Taff). Taff sites carried the greatest flows (19.97–22.46 m^3/s), Ely was intermediate (4.72 m^3/s), and streams were <0.2 m^3/s. Non-Ely sites shared a constant mean water level (0.86 ± 0.54 m) due to reliance on one available Taff gauge, but rainfall peaks were still distinguishable, especially around 23 November (Supplementary Fig. 2).

Biofilm biomass (AFDM) also varied across sites (Supplementary Fig. 3). Greater biomass was found in smaller streams (Upper Fawr = 0.107 mg/cm^2, Lower Rhiwbina = 0.099 mg/cm^2), while the River Ely had the lowest (0.025 mg/cm^2).

3.2. Tensile strength loss across urban flowing water sites

We quantified material degradation as mean tensile strength loss (TSL) per day (% d) and temperature-normalised TSL per degree day (% dd). Due to washouts, Week 1 data were the only measures for Upper and Middle Taff. Across all materials and sites, TSL ranged from 0.019 to 13.33 % d and 0.002–0.95 % dd (Supplementary Table 1). Material type significantly affected degradation rates (% d: $F_{2, 139} = 74.5$, $p < 0.001$; % dd: $F_{2, 112} = 57.76$, $p < 0.001$; Fig. 2a). Brand A degraded fastest (6.69 ± 3.19 % d; 0.5 ± 0.23 % dd), followed by Brand B (3.12 ± 1.93 % d; 0.26 ± 0.17 % dd), and cotton controls (2.22 ± 1.00 % d; 0.19 ± 0.09 % dd). Post-hoc ART pairwise comparisons confirmed that Brand A degraded significantly faster than both Brand B and cotton ($p < 0.001$), while Brand B degraded faster than cotton (% d: $p < 0.01$; % dd: $p < 0.05$).

Degradation rates varied among sites (% d: $F_{9, 137} = 7.89$, $p < 0.001$; % dd: $F_{6, 111} = 11.38$, $p < 0.001$; Fig. 2b). Lower Taff differed most from other sites, particularly Lower Fawr and Lower Glandulais ($p < 0.001$). Middle and Upper Taff showed the highest raw TSL but lacked statistical power due to limited data and high variability. Still, their similarity to Lower Taff suggests a consistent rapid degradation pattern across the Taff sites.

Site-specific trends were also clear within materials (Fig. 2c; see Supplementary Table 1). Brand A degraded fastest at Lower Rhiwbina (11.01 ± 0.28 % d; 0.94 ± 0.02 % dd) and slowest at Lower Glandulais (4.51 ± 2.32 % d; 0.36 ± 0.18 % dd). Cotton followed a similar pattern, degrading fastest at Lower Rhiwbina (3.19 ± 0.39 % d; 0.28 ± 0.03 % dd) and slowest at Lower Fawr (1.12 ± 0.73 % d; 0.10 ± 0.07 % dd). Brand B peaked at Lower Taff (6.57 ± 2.30 % d; 0.55 ± 0.19 % dd) and was slowest at Upper Glandulais (1.83 ± 1.06 % d; 0.15 ± 0.08 % dd). Notably, Brand A strips at Lower Taff fully degraded before first-week collection. Based on Brand B data, this site likely had the highest

Table 1
Environmental and hydrological conditions measured and calculated across each urban river site (Mean ± SD). Flow rate (mean; rounded). Blank values reflect missing dataloggers at sites.

Site	pH	TDS (mg/L)	Temperature (°C)	Light (Lux)	Water Level (m)	Flow rate (m^3/s)
Ely	7.95 (0.12)	208.00 (38.65)	11.31 (1.55)	59.93 (72.01)	0.55 (0.4)	4.72
Upper Fawr	8.09 (0.22)	209.80 (22.39)	10.46 (1.65)	7.40 (8.74)	0.86 (0.54)	0.02
Lower Fawr	8.08 (0.26)	211.40 (20.31)	10.22 (1.58)	32.59 (55.69)	0.86 (0.54)	0.19
Upper Glandulais	8.02 (0.20)	200.00 (17.54)	10.63 (2.45)	31.29 (25.09)	0.86 (0.54)	0.02
Lower Glandulais	8.01 (0.15)	203.20 (26.04)	11.11 (1.75)	1.79 (4.42)	0.86 (0.54)	0.15
Upper Rhiwbina	7.95 (0.51)	204.80 (26.28)	–	–	0.86 (0.54)	0.04
Lower Rhiwbina	8.07 (0.29)	186.80 (32.46)	10.71 (1.01)	39.24 (65.93)	0.86 (0.54)	0.03
Upper Taff	8.01 (0.10)	127.80 (18.43)	–	–	0.86 (0.54)	19.97
Middle Taff	8.04 (0.13)	144.80 (32.62)	–	–	0.86 (0.54)	20.71
Lower Taff	7.94 (0.13)	161.60 (38.95)	10.77 (1.89)	34.70 (70.58)	0.86 (0.54)	22.46

degradation for all wipes. Largely significant post-hoc contrasts confirmed strong site-specific drivers of degradation.

Temporal models highlighted important degradation patterns. Linear regressions showed that Brand A's TSL declined strongly over time for both metrics ($R^2 = 0.64$–0.65, $p < 0.001$), whereas Brand B and cotton showed no trends (Fig. 2d). GAMMs revealed non-linear declines for Brand A (edf = 2.4–2.6; $R^2 = 0.71$–0.73, $p < 0.001$), indicating rapid early loss followed by plateauing (Fig. 2e). No significant temporal patterns occurred for Brand B or cotton.

3.3. Environmental drivers of degradation

Spearman's rank correlations across all materials and sites showed that raw TSL rates increased slightly with temperature ($r = 0.29$, $p < 0.05$) but declined with exposure time for both raw and temperature-adjusted metrics (d %: $r = -0.32$; dd %: $r = -0.29$; both $p < 0.05$). These trends were driven by Brand A, which showed strong positive correlations with temperature (% d: $r = 0.68$; % dd: $r = 0.65$; both $p < 0.05$) and river level (both $r = 0.71$, $p < 0.05$), and strong negative correlations with exposure time (% d: $r = -0.88$: % dd: $r = -0.89$; both $p < 0.05$). Brand B and cotton showed no significant trends.

Environmental variables also covaried: pH rose with AFDM ($r = 0.64$) and TDS ($r = 0.56$); AFDM decreased with light ($r = -0.23$) and temperature ($r = -0.28$); and temperature also fell sharply over time ($r = -0.92$). All correlations were significant at $p < 0.05$.

We modelled the effects of temperature, light, TDS, river level, flow rate, and AFDM on raw and temperature-adjusted TSL for each material, with and without exposure time as a covariate. pH was excluded due to high multicollinearity (VIF = 20.67).

Without controlling for time (Supplementary Fig. 4), Brand A degraded exponentially at higher temperatures ($R^2 = 0.14$, $p < 0.05$) and AFDM ($R^2 = 0.29$, $p < 0.05$) but showed a quadratic decline at higher TDS ($R^2 = 0.31$, $p < 0.001$). Adjusted TSL showed a strong quadratic link with AFDM ($R^2 = 0.56$, $p < 0.01$). Brand B showed only a weak TDS exponential relationship ($R^2 = 0.11$, $p < 0.05$). Cotton controls displayed strong quadratic AFDM effects ($R^2 = 0.72$–0.75, $p < 0.05$) and a unimodal, quadratic TDS effect ($R^2 = 0.19$, $p < 0.05$), with peak degradation near 170 mg/L.

When controlling for exposure time (Fig. 3), Brand A's degradation remained closely linked to environmental factors, with stronger model fits. Both TSL metrics increased with AFDM ($R^2 = 0.9$, $p < 0.01$) and river level ($R^2 = 0.42$–0.56, $p < 0.01$) while temperature effects reversed, showing quadratic declines ($R^2 = 0.75$–0.76, $p < 0.01$). Adjusted TSL also declined quadratically with TDS ($R^2 = 0.72$, $p < 0.05$). For Brand B, no predictors were significant. Cotton controls retained AFDM and TDS effects consistent with time-excluded models, suggesting time was a weaker driver for this material type. Quadratic model fits should be interpreted cautiously at predictor extremes due to sparse data. Mixed-effect models showed negligible site-level variance, indicating local environmental conditions, discharge events (via mean water level), and time drove degradation more than site identity.

3.4. Photographic and SEM analysis of degradation

Photographs showed clear differences in degradation between the two full-sized wet wipe brands across urban sites (Supplementary Fig. 5). Brand A degraded faster and more extensively than Brand B. By Week 5, Brand A wipes had fragmented or deteriorated at Ely, Upper and Lower Fawr, and Upper Rhiwbina, while Brand B often retained structural integrity, notably at Lower Glandulais, Lower Fawr, and Upper Rhiwbina.

At Ely, rapid degradation (Weeks 2–3) coincided with *Gammarus pulex* presence in full-sized wipe bags, though strips were unaffected. Downstream loss of samples prevented direct comparison at Upper and Middle Taff, but complete wipe degradation at Lower Taff by Week 4 implied similar upstream dynamics.

SEM confirmed fibre differences between wipe brands (Fig. 4). Brand A comprised mostly wide, ribbon-shaped fibres natural cellulose fibres, with some smooth, cylindrical fibres typical of regenerated cellulose (e.g. viscose), while Brand B displayed the reverse pattern. EDS detected titanium (~1.67 % by weight) in Brand B, likely from titanium dioxide, a whitening or UV-protective additive (Supplementary Fig. 6).

SEM imaging of incubated wipes retrieved from Lower Nant and Upper Glandulais sites showed Brand A natural fibres roughening, fraying, and detaching by Week 3, with biofilm and organic debris coatings that thickened by Week 5 (Fig. 5). At Upper Glandulais, Brand A degradation seemed milder, with some pitting, cracking, and tearing, but less advanced structural decay. Brand B regenerated fibres stayed largely intact: by Week 5, they were embedded in biofilm/debris but with little fraying or fragmentation. Fine filaments here appeared environmental rather than wipe degradation products. EDS spectra of incubated wipes confirmed elevated aluminium, silicon, iron, magnesium, potassium, and calcium, consistent with mineral sediment deposition (Supplementary Fig. 7).

4. Discussion

Escalating concern over plastic pollution, coupled with new regulatory pressure such as the recent UK legislation restricting plastic-containing wipes (DEFRA, 2024), has accelerated the adoption of plastic-free, cellulose-based wet wipes. These products are marketed as biodegradable and environmentally friendly, yet their degradation in freshwater environments – particularly urban rivers and streams where they commonly accumulate – remains poorly understood. This study provides the first multi-site, in-situ assessment of biodegradable wet wipe degradation in flowing urban waters using tensile strength loss (TSL) as a degradation proxy and cotton strip bioassays as ecological baselines.

4.1. Degradation behaviour in urban rivers and streams

Cellulose-based wipes showed distinct material- and site-specific degradation patterns, with Brand A degrading more than twice as fast

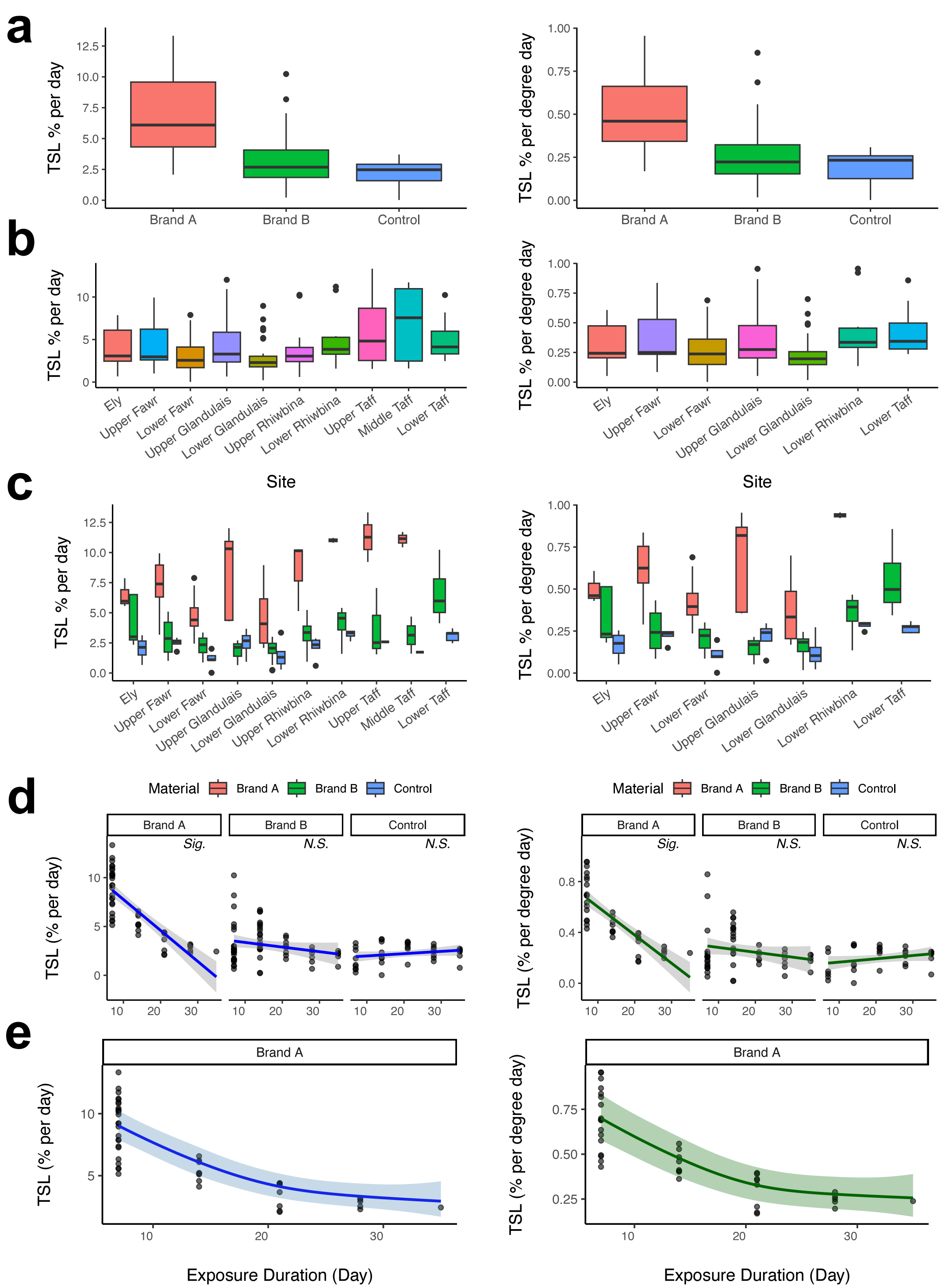

Fig. 2. Mean degradation rates (TSL % per day and TSL % degree day) for each: a) material; b) site; c) material between sites; d) material over time via linear regressions; and for e) Brand A over time via non-linear GAMMs. Upper and Middle Taff raw TSL represent Week 1 findings. Sites with missing temperature data were excluded for TSL % degree day. Boxplots show median, interquartile range and outliers. Sig. = significant models, N.S. = non-significant models. Linear and non-linear models show 95 % confidence intervals.

as Brand B and both outperforming cotton controls. This pattern, consistent with earlier mesocosm work (Allison et al., 2025a), underscores the strong role of intrinsic material composition in driving degradation.

SEM confirmed these compositional contrasts, which Fourier-Transform Infrared Spectroscopy was unable to do effectively in our previous work (Allison et al., 2025a; Supplementary Fig. 8). Brand A contained mostly wide, ribbon-shaped fibres characteristic of natural cellulose (e.g. wood pulp), with only minor regenerated cellulose likely from viscose (Salama et al., 2021; Harter et al., 2022; Ziklo et al., 2024). Brand B displayed the reverse: more smooth, cylindrical regenerated fibres and fewer natural ones. In blended wipes, regenerated fibres typically provide tensile strength through fibre entanglement, while shorter natural fibres promote absorption and break down more readily (Tipper, 2016; Harter et al., 2022). Their larger surface area and rougher texture also support microbial colonisation (Ziklo et al., 2024). These

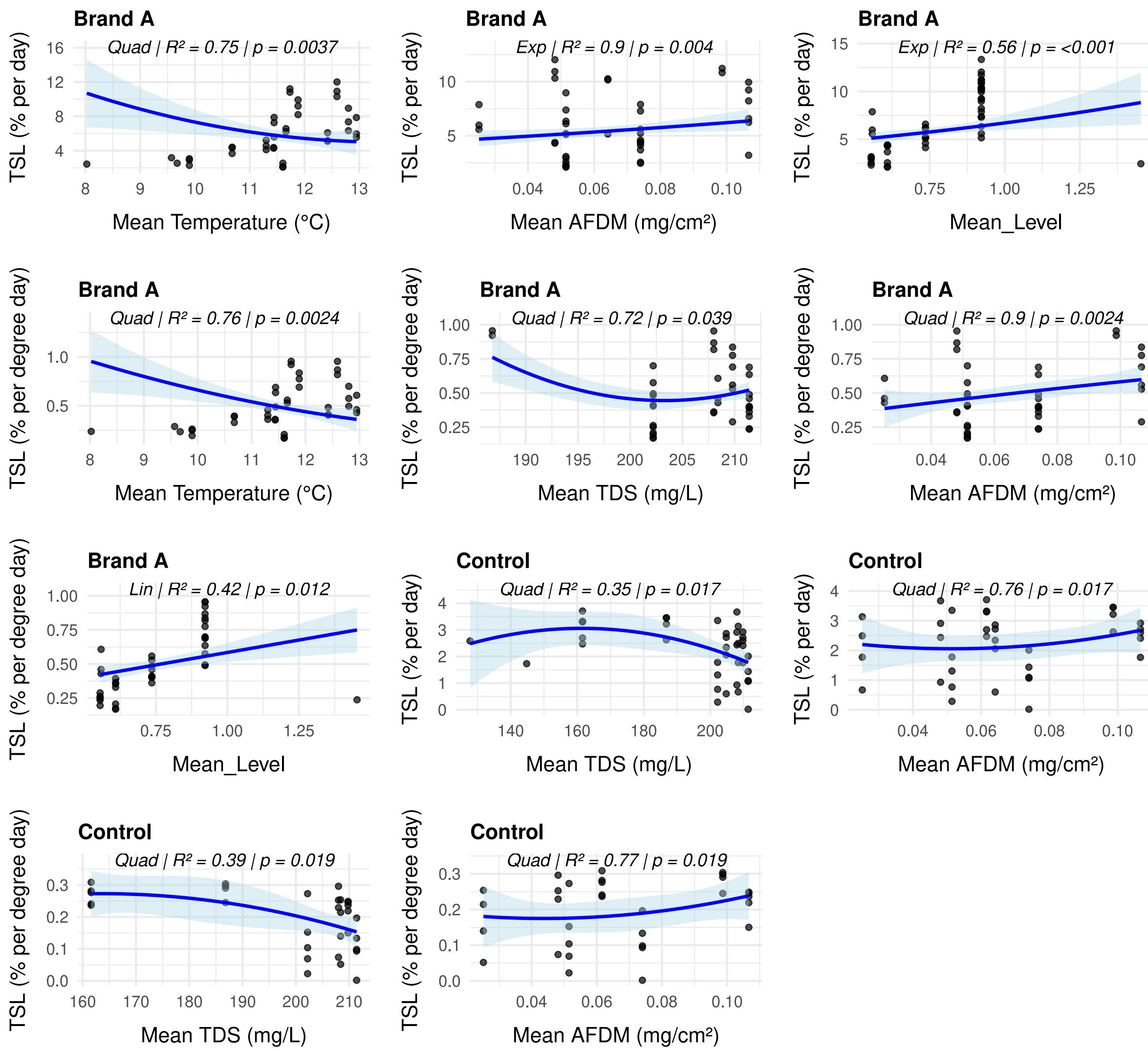

Fig. 3. Relationships between statistically significant environmental predictors and TSL metrics for each material type with exposure duration (Day) held constant. Each plot shows 95 % confidence interval, linear (lin), exponential (exp), or quadratic (quad) function, R^2 value, and Benjamini-Hochberg adjusted p-value.

differences align with broader evidence that natural cellulose textiles degrade faster than regenerated forms in aquatic systems (Durukan and Karadagli, 2019; Zambrano et al., 2020; Kwon et al., 2022; Kwon et al., 2023; Smith et al., 2024).

Chemical additives may have further shaped degradation dynamics. Both brands listed antimicrobial preservatives (e.g. benzoic acid, phenoxyethanol, potassium sorbate, sodium benzoate), which could initially suppress microbial colonisation until leached (Windler et al., 2013; Malis et al., 2019; Sait et al., 2021). Furthermore, cellulosic wipes often include binders that enhance wet strength (Tipper, 2016), although these were not disclosed on the product packaging nor identified in our EDS analysis. Rates of leaching likely vary with water chemistry, fibre structure, and exposure time. These potential effects were not directly measured here, and future work should isolate additive influence by comparing pre-washed and untreated wipes under controlled microbial exposures.

Urban waters accelerated degradation relative to our prior mesocosm study (Allison et al., 2025a). Brand A degraded more than three times faster, while Brand B and cotton strips degraded at roughly double their mesocosm rates. These differences, consistent across raw and temperature-adjusted TSL metrics, suggest that urban rivers could provide more favourable conditions for breakdown than rural upland streams (see Section 4.2). Comparable beach mesocosm data from Metcalf et al. (2024) show even lower daily TSL rates (2.06 ± 1.36 % d and 3.04 ± 1.72 % d for home and commercially compostable wipes, respectively) when applied to Equation (1), highlighting how environmental context shapes degradation trajectories.

Several methodological constraints temper these degradation findings. TSL is a practical, ecologically relevant proxy for cellulose degradation (Colas et al., 2019; Jabiol et al., 2020; Hill et al., 2022), but it cannot disentangle physical disintegration from microbial or chemical breakdown. For instance, elevated TSL values may partly reflect mechanical fibre loss from turbulence or sediment abrasion rather than complete microbial decay (Kwon et al., 2022; Kargar and Joksimovic, 2024). SEM and EDS revealed surface wear and debris accumulation but limited direct evidence of enzymatic activity; such debris likely restricted microbial or hydraulic access over time, slowing TSL. More targeted assays - such as microbial respiration or biological oxygen demand tests, as well as more advanced chemical analyses (e.g., NMR Spectroscopy), could apportion mechanisms (Tiegs et al., 2013; Zambrano et al., 2020; Nagamine et al., 2022), though consistent field application of the former assays is challenging (Mulholland et al., 2001; Jouanneau et al., 2014; Irving et al., 2024). Because samples were anchored, results represent an upper bound for near-benthic water-column exposure; freely transported or intermittently suspended wipes would have shorter residence times and reduced microbial contact. Prior work tracking wipe position provides complementary context (Allison et al., 2025a). Finally, despite fine mesh bags excluding most

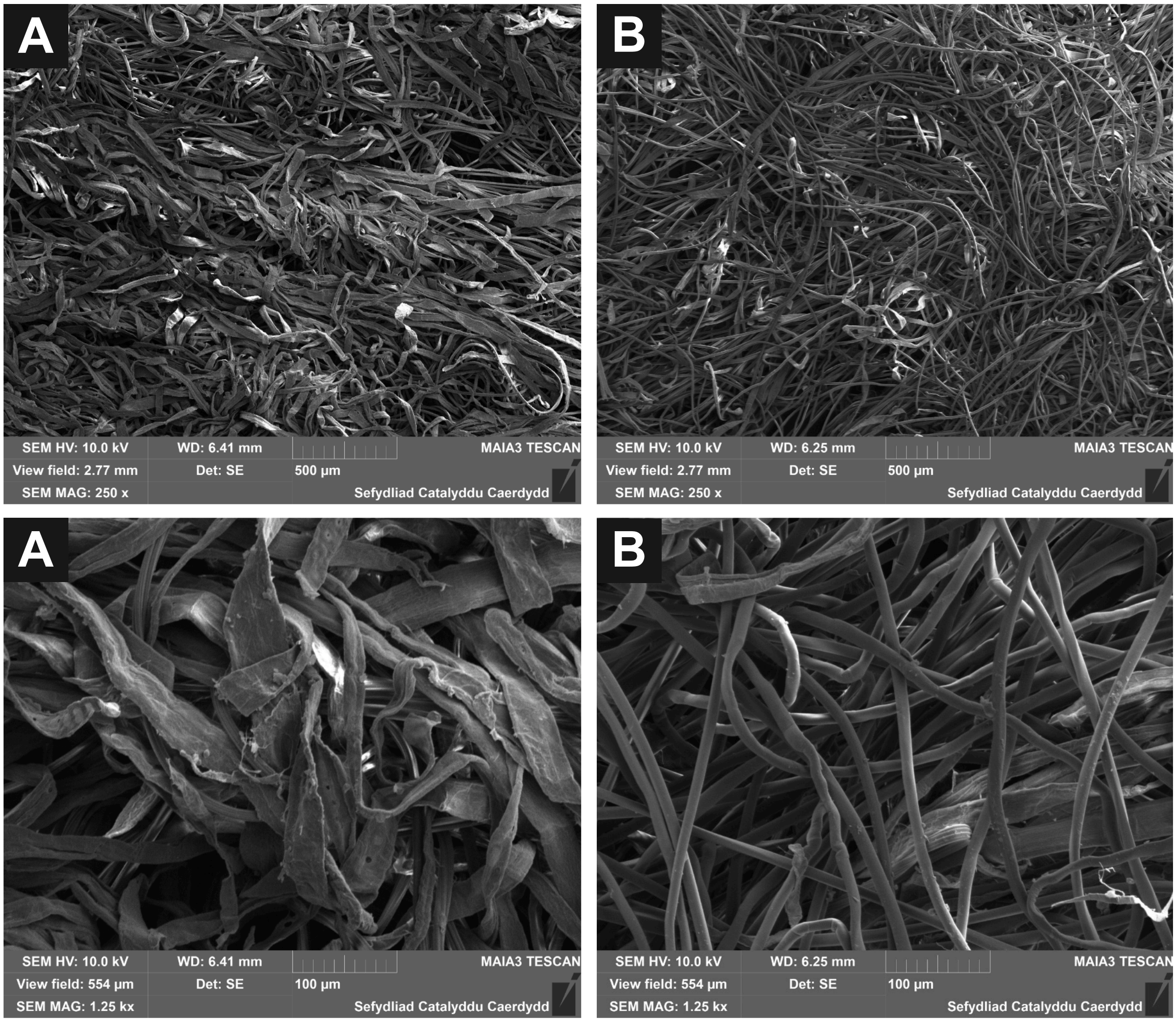

Fig. 4. SEM images of non-incubated cellulosic wet wipe brands (A and B) at different magnifications.

macroinvertebrates, full-sized wipes at the River Ely showed signs of *Gammarus pulex* interference, indicating the potential role of shredders in accelerating wipe breakdown (Tiegs et al., 2007).

Importantly, these methodological constraints do not alter the central finding that wipe fragments clearly persisted: after five weeks in both mesocosm and urban river studies, and up to 15 weeks for compostable wipes in the beach mesocosm study (Metcalf et al., 2024). While our use of fine mesh bags excluded most macroinvertebrate activity - likely producing a conservative degradation estimates - this persistence nonetheless challenges the environmental relevance of 'biodegradable' claims when for products frequently entering freshwaters via toilet flushing. Many such claims rely on industrial composting standards (e.g. ISO 14855 for Brand B; OK compost HOME; EN 13432), which involve elevated temperatures and controlled microbial conditions not representative of flowing surface waters (Liao and Chen, 2021; Allison et al., 2025a). While these standards are appropriate for their intended disposal route, they do not ensure degradation in aquatic environments. The issue therefore lies less with the standards themselves and more with the mismatch between certified disposal routes and actual consumer behaviour. Producers should clarify and focus efforts on the limitations of compostability or biodegradability claims, especially for products likely to enter wastewater systems. Future research should focus on strategies to better align disposal guidance with environmental realities.

4.2. Environmental, temporal, and hydrodynamic drivers of degradation

Degradation trajectories of cellulose-based wipes in urban rivers also depended on environmental conditions, hydrodynamics, and exposure duration. Among predictors, AFDM (a proxy for microbial biomass) showed the strongest and most consistent positive association with degradation, especially for Brand A and cotton. This matches well-established ecological evidence that microbial abundance and diversity are key to successful cellulose decomposition in freshwaters (Burdon et al., 2020; Polman et al., 2021; Hayer et al., 2022). Unlike our mesocosm study – where low nutrient levels likely limited microbial growth and AFDM was confounded by dominant algal biomass (Winterbourn et al., 1992; Ledger and Hildrew, 2008) - urban sites here had higher AFDM and stronger TSL correlations. This underscores the value of field-based testing and supports TSL as a reliable proxy for in-situ biodegradation (Tiegs et al., 2013; Colas et al., 2019; Hill et al., 2022). However, because AFDM measures total organic biomass rather than cellulolytic microbial groups specifically, its explanatory link to degradation is indirect. Future research coupling TSL with eDNA or metagenomic approaches could help identify the functional taxa driving wipe degradation in urban rivers.

TDS also influenced degradation, but with a non-linear, negative curve. High TDS reduced degradation, especially in Brand A and cotton, contrasting earlier cotton strip findings (Colas et al., 2019). This indicates possibly inhibitory thresholds for microbial growth and biodegradation under nutrient rich and poor conditions. While

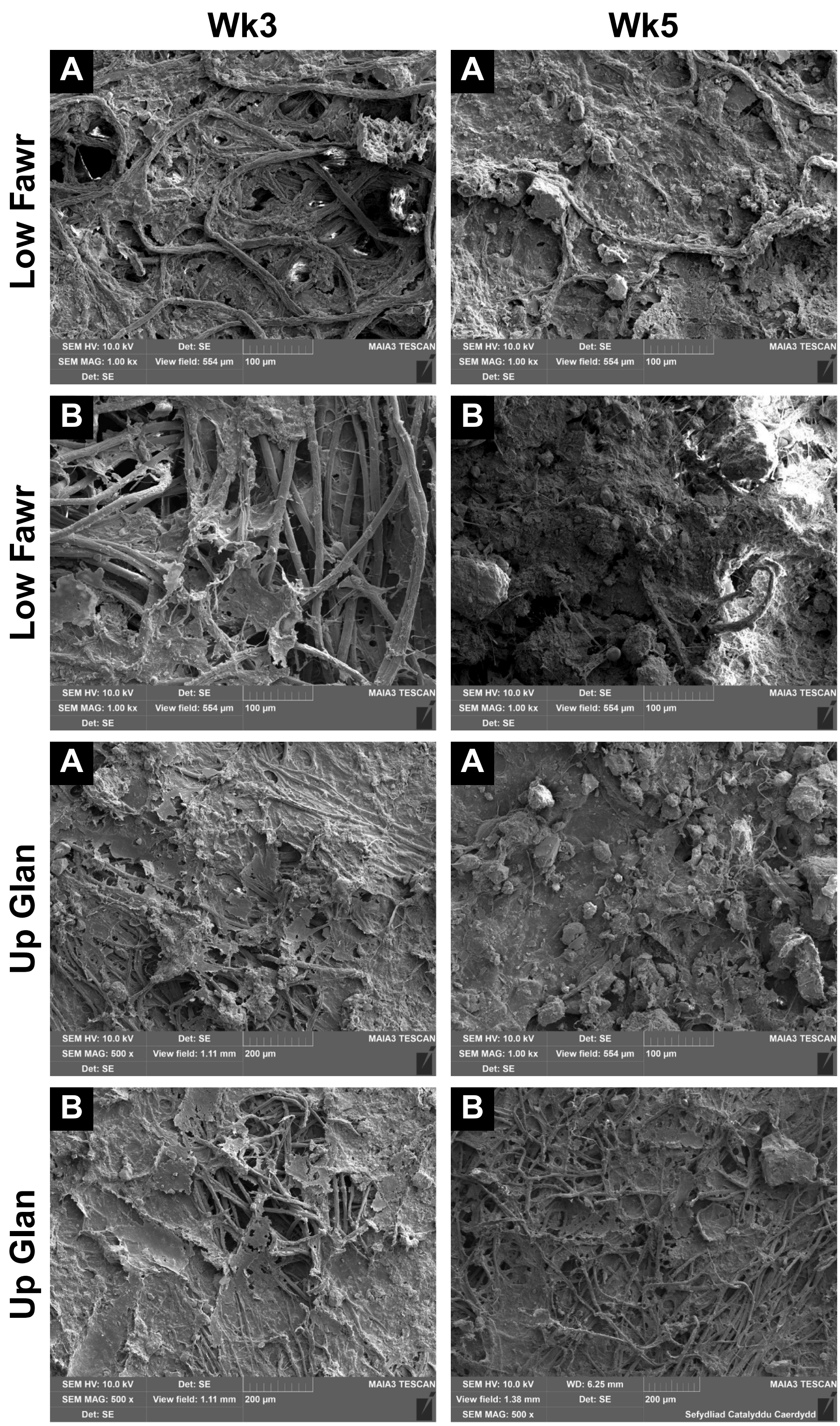

Fig. 5. SEM images of Brand A and B cellulosic wet wipes over time for the Lower Fawr and Upper Glandulais sites.

moderate TDS may aid microbes by supplying nutrients and ionic stability, high TDS - common in polluted urban waters - likely reflects stressors that inhibit microbial growth or enzyme activity (Woodward et al., 2012; Griffiths and Tiegs, 2016).

Temperature emerged as a positive driver for Brand A in simple correlations and regressions excluding time. This fits with earlier mesocosm findings and wider evidence that higher temperatures enhance microbial metabolism and enzyme activity (Griffiths and Tiegs, 2016; Zambrano et al., 2020; Nagamine et al., 2022; Allison et al., 2025a). Yet, when time was included, temperature effects reversed: Brand A showed negative associations. This apparent contradiction reflects strong collinearity between temperature and time ($r = -0.92$). In other words, temporal degradation patterns - especially early physical breakdown - absorbed explanatory power otherwise attributed to temperature. Linear regressions confirmed that exposure time alone strongly predicted Brand A degradation (Fig. 2d), suggesting time-driven processes outweighed temperature effects.

Hydrodynamic influences were mixed. Mean flow rates ranged from <0.2 m^3/s in streams to ~ 22.5 m^3/s in the Taff, but showed no significant link to degradation, likely because static averages lack temporal variability. By contrast, river level - a proxy for discharge dynamics -showed a strong positive association with Brand A degradation, even after accounting for time. This suggests that discharge fluctuations, such as rainfall events, may accelerate breakdown through abrasion, turbulence, and shear stress (Kwon et al., 2022; Kargar and Joksimovic, 2024). Unlike static flow rates, continuous river-level data captured these short-term dynamics. As wipes remained submerged throughout, wet-dry cycling was unlikely, and instead, enhanced shear during storm-driven flows is the more plausible driver.

Temporal patterns of degradation varied markedly between the three material types, reflecting their contrasting compositions and structures. Brand A degraded most rapidly, with TSL occurring primarily within the first three weeks before plateauing. This two-phase pattern, also observed in the earlier mesocosm experiment (Allison et al., 2025a), points to an initial phase dominated by physical disintegration followed by slower biological breakdown. SEM imagery corroborated this trajectory: by Week 3, Brand A fibres at both Lower Nant and Upper Glandulais displayed surface erosion, fraying, and partial detachment, consistent with hydrodynamic abrasion and fibre shedding of natural cellulose components (Kwon et al., 2022; Kargar and Joksimovic, 2024). At the same time, thin filamentous matrices interwoven with fibres suggested early biofilm colonisation. Yet by Week 5, Brand A wipes were heavily embedded in mineral-organic debris, which may have obscured further fibre decay and restricted microbial access, explaining the plateau in TSL.

By contrast, Brand B degraded much more slowly, with TSL proceeding in a steady, almost linear fashion and with limited change in rate over time. SEM images revealed smoother, cylindrical fibres with more limited signs of erosion or fraying, consistent with its higher proportion of more resistant regenerated cellulose (Tipper, 2016). While faint pitting and scarring was observed by Week 3, features indicative of microbial activity (Nagamine et al., 2022), fibres remained structurally intact through Week 5. Although web-like coatings, possibly linked to microbial colonisation, were observed at Lower Nant in Week 3 samples, they did not appear to progress in tandem with overall degradation. Instead, Brand B samples became increasingly encrusted with mineral and organic debris, particularly by Week 5, which likely restricted microbial access or masked subtle surface decay. Its relative resistance therefore appears to stem from a combination of intrinsic fibre properties, regenerated cellulose's inherent durability, and external shielding effects from debris or additives. Unmeasured water uptake (absorption) and fibre swelling - likely greater in regenerated cellulose with lower crystallinity - may also have modulated early mechanical weakening and apparent TSL (see Allison et al., 2023), yet these effects were not identifiable from SEM of dried samples and would require future gravimetric moisture-uptake/swelling assays to test.

Cotton controls followed a third trajectory, characterised by a gradual but steady increase in TSL across the study period. Their woven structure and higher physical integrity limited debris entrapment and buffered against early disintegration, allowing microbial and environmental processes to act more consistently over time. This performance aligns with results from standardised bioassays (Tiegs et al., 2013; Colas et al., 2019; Hill et al., 2022) underscores cotton's value as a reliable ecological comparator in TSL-based biodegradation assessments.

Taken together, this study highlights the potential for cellulose-based wipes to generate substantial, albeit short-term, macro- and microfibre loads in urban rivers (Allison et al., 2025b), reinforcing the need for more rigorous, environmentally realistic biodegradability assessments. Such persistence also creates a temporary but ecologically relevant exposure window in which aquatic organisms may ingest or interact with wipe fibres, as evidenced for *Gammarus pulex* in this study, with potential dietary or toxicological effects that require further investigation in follow-up studies (Lusher et al., 2013; see also Courtene-Jones et al., 2024). Beyond direct ingestion, these fibres could also act as vectors for other pollutants or pathogens (Caruso, 2019; Metcalf et al., 2024), compounding risks to freshwater ecosystems and potentially human health. Future work on cellulosic wipes and their fibres should therefore include leachate and particle bioassays (e.g., cell-based and standard aquatic tests) to evaluate ecotoxicity and sorbed-contaminant effects.

This study provides the first in-situ assessment of biodegradable wet wipe degradation in urban rivers and streams, building on prior mesocosm research. In these systems - where wipes most often accumulate - degradation depended strongly on material composition, environmental conditions, and exposure time. Natural cellulose-rich wipes degraded faster than regenerated cellulose-dominant wipes and cotton bioassays, likely due to greater fibre shedding and microbial biodegradation. While microbial biomass and total dissolved solids influenced degradation, exposure duration explained the most variation across materials, highlighting the role of physical breakdown under hydrodynamic stress. Yet substantial wipe fragments and sediment-laden fibres persisted after five weeks, raising concerns about the environmental credibility of eco-friendly labels.

These findings have direct implications for ecological risk, product regulation, and consumer behaviour. Biodegradability standards and labelling criteria must better reflect the real-world freshwater conditions in which products typically end up, and consider hydrodynamics, debris accumulation, and microbial community variation. Future work should extend study durations, characterise cellulolytic microbial communities, and track the fate of released fibres and additives to fully evaluate the environmental impact of these 'biodegradable' alternatives.

CRediT authorship contribution statement

Thomas Allison: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Benjamin D. Ward:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Funding acquisition, Conceptualization. **Michael Harbottle:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Conceptualization. **Isabelle Durance:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2025.127376.

Data availability

Data used for the research described in this article are available here: https://doi-org.cardiff.idm.oclc.org/10.17035/cardiff.29986186.v1

References

Allison, T., Ward, B.D., Harbottle, M., Durance, I., 2023. Do flushed biodegradable wet wipes really degrade? Sci. Total Environ. 894. https://doi.org/10.1016/j.scitotenv.2023.164912.

Allison, T., Ward, B.D., Harbottle, M., Durance, I., 2025a. Cellulose-based wet wipes undergo limited degradation in river environments. Environ. Pollut. 384. https://doi.org/10.1016/j.envpol.2025.126971.

Allison, T., Ward, B.D., Durance, I., Harbottle, M., 2025b. Predicting flushed wet wipe emissions into Rivers. Water Res. 268. https://doi.org/10.1016/j.watres.2024.122733.

Bach, L., Strand, J., Salame, H., Simon, M., Fritt-Rasmussen, J., Jensen, P.E., 2025. Wet wipes in untreated wastewater are a source of litter pollution to the arctic marine environment - a case study on the loads of litter and microplastics in wastewater effluents in Greenland. Environ. Sci. Adv. 4 (223). https://doi.org/10.1039/D4VA00233D.

Besley, C.H., Cassidy, M., 2022. The composition of gross pollutants contained in wet weather overflows for different locations, spill frequencies and discharge volumes. J. Environ. Manag. 303. https://doi.org/10.1016/j.jenvman.2021.114256.

Burdon, F.J., et al., 2020. Stream microbial communities and ecosystem functioning show complex responses to multiple stressors in wastewater. Glob. Change Biol. 26. https://doi.org/10.1111/gcb.15302.

Carballeira, C., Villares, R., Mata-Rivas, B., Carballeria, A., 2020. The cotton-strip assay as an environmental surveillance tool for ecological integrity assessment of rivers affected by WWTP effluents. Water Res. 169. https://doi.org/10.1016/j.watres.2019.115247.

Caruso, G., 2019. Microplastics as vectors of contaminants. Mar. Pollut. Bull. 146. https://doi.org/10.1016/j.marpolbul.2019.07.052.

Chauvet, E., et al., 2016. Litter decomposition as an indicator of stream ecosystem functioning at local-to-continental scales: insights from the European RivFunction Project. Adv. Ecol. Res. 55. https://doi.org/10.1016/bs.aecr.2016.08.006.

Colas, F., et al., 2019. Towards a simple global-standard bioassay for a key ecosystem process: organic-matter decomposition using cotton strips. Ecol. Indic. 106. https://doi.org/10.1016/j.ecolind.2019.105466.

Courtene-Jones, W., De Falco, F., Burgevin, F., Handy, R.D., Thompson, R.C., 2024. Are biobased microfibers less harmful than conventional plastic microfibers: evidence from earthworms. Environ. Sci. Technol. 54 (46). https://doi.org/10.1021/acs.est.4c05856.

DEFRA, 2024. UK-wide ban on wet wipes containing plastic to be put into law. Available at: https://www.gov.uk/government/news/uk-wide-ban-on-wet-wipes-containing-plastic-to-be-put-into-law. Accessed: 10/12/24.

Durukan, S., Karadagli, F., 2019. Physical characteristics, fiber compositions, and tensile properties of nonwoven wipes and toilet papers in relevance to what is flushable. Sci. Total Environ. 697. https://doi.org/10.1016/j.scitotenv.2019.134135.

Farr, G.J., Patton, A.M., Boon, D.P., James, D.R., Williams, B., Schofield, D.I., 2017. Mapping shallow urban groundwater temperatures, a case study from Cardiff, UK. Q. J. Eng. Geol. Hydrogeol. 50. https://doi.org/10.1144/qjegh2016-058.

Fortibuoni, T., Di Vito, S., Scocchera, E., 2025. Preliminary assessment of waste accumulation along riverbanks in Italy. Environ. Sci. Pollut. Control Ser. 32. https://doi.org/10.1007/s11356-025-35899-2.

Fox, J., Weisberg, S., 2019. An R companion to Applied Regression, Third Edition. Sage, Thousand Oaks, CA.

Griffiths, N.A., Tiegs, S.D., 2016. Organic-matter decomposition along a temperature gradient in a forested headwater stream. Freshw. Sci. 35. https://doi.org/10.1086/685657.

Harter, T., Steiner, H., Hirn, U., 2022. Investigating mechanisms deteriorating the dispersibility of biodegradable and flushable wet wipes. Cellulose 29. https://doi.org/10.1007/s10570-022-04816-y.

Hayer, M., Wymore, A.S., Hungate, B.A., Schwartz, E., Koch, B.J., Marks, J., 2022. Microbes on decomposing litter in streams: entering on the leaf or colonizing in the water? ISME J. 16 (3). https://doi.org/10.1038/s41396-021-01114-6.

Hill, M.J., et al., 2022. Organic-matter decomposition in urban stream and pond habitats. Ecol. Indic. 142. https://doi.org/10.1016/j.ecolind.2022.109232.

Imberger, S.J., Thompson, R.C., Grace, M.R., 2010. Searching for effective indicators of ecosystem function in urban streams: assessing cellulose decomposition potential. Freshw. Biol. 55. https://doi.org/10.1111/j.1365-2427.2010.02469.x.

Irving, D., Bakhshandeh, S., Tran, T.K.A., McBratney, A.B., 2024. A cost-effective method for quantifying soil respiration. Soil Secur. 16. https://doi.org/10.1016/j.soisec.2024.100162.

Jabiol, J., Colas, F., Guerold, F., 2020. Cotton-strip assays: let's move on to eco-friendly biomonitoring. Water Res. 170. https://doi.org/10.1016/j.watres.2019.115295.

Joksimovic, D., Khan, A., Orr, B., 2020. Inappropriate disposal of 'flushable' consumer products- reasons for concern. Water Sci. Technol. 81, 102–108. https://doi.org/10.2166/wst.2020.087.

Jouanneau, S., et al., 2014. Methods for assessing biochemical oxygen demand (BOD): a review. Water Res. 49. https://doi.org/10.1016/j.watres.2013.10.066.

Kachef, R., 2024. A ban on plastic wet wipes may lead to more river pollution, not less. The Conversation. https://doi.org/10.64628/AB.sdwm9569q.

Kargar, K., Joksimovic, D., 2024. Blockage dynamics in sewer pipes: investigating snagging, accumulation, and dissipation rates of wet wipes. Water Sci. Technol. 90 (5). https://doi.org/10.2166/wst.2024.291.

Kay, M., Elkin, L., Higgins, J., Wobbrock, J., 2021. ARTool: Aligned Rank Transform for Nonparametric Factorial ANOVAs. R package version 0.11.1. Available at: https://cran.r-project.org/web/packages/ARTool/ARTool.pdf. Accessed: 24/10/2024.

Kwon, S., Zambrano, M.C., Venditti, R.A., Frazier, R., Zambrano, F., Gonzalez, R.W., Pawlak, J.J., 2022. Microfiber shedding from nonwoven materials including wipes and meltblown nonwovens in air and water environments. Environ. Sci. Pollut. Control Ser. 29, 60584–60599. https://doi.org/10.1007/s11356-022-20053-z.

Kwon, S., Zambrano, M.C., Venditti, R.A., Pawlak, J.J., 2023. Aerobic aquatic biodegradation of bio-based and biodegradable polymers: kinetic modeling and key factors for biodegradability. Int. Biodeterior. Biodegrad. 185. https://doi.org/10.1016/j.ibiod.2023.105671.

Ledger, M.E., Hildrew, A.G., 2008. Growth of an acid-tolerant stonefly on epilithic biofilms from streams of contrasting pH. Freshw. Biol. 46. https://doi.org/10.1046/j.1365-2427.2001.00764.x.

Lee, J., Jeong, S., Chae, K.-J., 2021. Discharge of microplastics fibres from wet wipes in aquatic and solid environments under different release conditions. Sci. Total Environ. 784. https://doi.org/10.1016/j.scitotenv.2021.147144.

Liao, J., Chen, Q., 2021. Biodegradable plastics in the air and soil environment: low degradation rate and high microplastics formation. J. Hazard Mater. 418. https://doi.org/10.1016/j.jhazmat.2021.126329.

Lusher, A.L., McHugh, M., Thompson, R.C., 2013. Occurrence of microplastics in the gastrointenstinal tract of pelagic and demersal fish from the English Channel. Mar. Pollut. Bull. 67. https://doi.org/10.1016/j.marpolbul.2012.11.028.

MacAulay, S., Masud, N., Cable, J., 2024. Analysis of bamboo fibres and their associated dye on a freshwater fish host-parasite system. Environ. Sci. Pollut. Control Ser. 31. https://doi.org/10.1007/s11356-024-34626-7.

Malis, D., Jeršek, B., Tomšič, B., Štular, D., Golja, B., Kapun, G., Simončič, B., 2019. Antibacterial activity and biodegradation of cellulose fiber blends with incorporated ZnO. Materials 12 (3399). https://doi.org/10.3390/ma12203399.

McCoy, K.A., Hodgson, D.J., Clark, P.F., Morritt, D., 2020. The effects of wet wipe pollution on the Asian clam, Corbicula fluminea (Mollusca: bivalvia) in the River Thames, London. Environ. Pollut. 264. https://doi.org/10.1016/j.envpol.2020.114577.

McGoran, A.R., Clark, P.F., Morritt, D., 2017. Presence of microplastic in the digestive tracts of European flounder, platicthys flesus, and European smelt, Osmerus eperlanus, from the River Thames. Environ. Pollut. 220, 744–751. https://doi.org/10.1016/j.envpol.2016.09.078.

Met Office, 2025. Climate station: cardiff, bute park. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/location-specific-long-term-averages/gcjszmp44. Accessed: 27/03/25.

Metcalf, R., Fellows, R., White, H.L., Quilliam, R.S., 2024. Persistence of 'wet wipes' in beach sand: an unrecognised reservoir for localised E. coli contamination. Mar. Pollut. Bull. 201. https://doi.org/10.1016/j.marpolbul.2024.116175.

Mulholland, P.J., et al., 2001. Inter-biome comparison of factors controllings stream metabolism. Freshw. Biol. 46. https://doi.org/10.1046/j.1365-2427.2001.00773.x.

Nagamine, R., Kobayashi, K., Kusumi, R., Wada, M., 2022. Cellulose fiber biodegradation in natural waters: river water, brackish water, and seawater. Cellulose. https://doi.org/10.1007/s10570-021-04349-w.

Ó Briain, O., Mendes, A.R.M., McCarron, S., Healy, M.G., 2020. The role of wet wipes and sanitary towels as a source of white microplastic fibres in the marine environment. Water Res. 182. https://doi.org/10.1016/j.watres.2020.116021.

ONS, 2024. Estimated population of Wales in 2023, by unitary authority. Statista: Available at: https://www-statista-com.cardiff.idm.oclc.org/statistics/971295/regional-population-wales/. Accessed: 27/03/25.

Patil, I., 2021. Visualizations with statistical details: the 'ggstatsplot' approach. J. Open Source Softw. 6 (61). https://doi.org/10.21105/joss.03167.

Paul, M.J., Meyer, J.L., 2001. Streams in the urban landscape. Annu. Rev. Ecol. Evol. Syst. 32. https://doi.org/10.1146/annurev.ecolsys.32.081501.114040.

Pedersen, A.F., Vasquez, A.A., Thorsby, J.S., Gorrell, M., Petriv, A.-M.V., Miller, C.J., Baker, T.R., 2022. Sewage transport volumes and physical degradation rates of personal care wipes. J. Am. Water Resour. Assoc. 58 (6). https://doi.org/10.1111/1752-1688.13046.

Pedersen, T.L., 2024. Patchwork: the composer of plots. Available at: https://patchwork.data-imaginist.com. Accessed: 12/12/24.

Pingram, M.A., Clapcott, J.E., Hamer, M.P., Atalah, J., Ozkundakci, D., 2020. Exploring temporal and spatial variation in cotton tensile-strength loss to assess the ecosystem health of non-wadeable rivers. Ecol. Indic. 108. https://doi.org/10.1016/j.ecolind.2019.105773.

Polman, E.M.N., Gruter, G.-J.M., Parsons, J.R., Tietema, A., 2021. Comparison of the aerobic biodegradation of biopolymers and the corresponding bioplastics: a review. Sci. Total Environ. 753. https://doi.org/10.1016/j.scitotenv.2020.141953.

Robinson, D., Hayes, A., Couch, S., 2025. Broom: Convert statistical objects into tidy tibbles. Available at: https://CRAN.R-project.org/package=broom. Accessed: 21/11/24.

Sait, S.T.L., Sørensen, L., Kubowicz, S., Vike-Jonas, K., Gonzalez, S.V., Asimakopoulos, A.G., Booth, A.M., 2021. Microplastic fibres from synthetic textiles: environmental degradation and additive chemical content. Environ. Pollut. 268. https://doi.org/10.1016/j.envpol.2020.115745.

Salama, P., Gliksberg, A., Cohen, M., Tzafrir, I., Ziklo, N., 2021. Why are wet wipes so difficult to preserve? Understanding the intrinsic causes. Cosmetics 8 (73), 1–11. https://doi.org/10.3390/cosmetics8030073.

Smith, M.M., Zambrano, M., Ankeny, M., Daystar, J., Pires, S., Pawlak, J., Venditti, R.A., 2024. Aquatic aerobic biodegradation of commonly flushed materials in aerobic wastewater treatment plant solids, seawater, and lakewater. Bioresources 19. https://doi.org/10.15376/biores.19.1.1150-1164.

Solargis, 2025. Global solar Atlas. Available at: https://globalsolaratlas.info/map?c=51,486193,-3.210411,12&m=site&a=-3.267561,51.448762,-3.267561,51.539833,-3.124172,51.539833,-3.124172,51.448762,-3.267561,51.448762 [Accessed: 27/03/25.

Steinman, A.D., Lamberti, G.A., Leavitt, P.R., 2007. Biomass and pigments of benthic algae. In: Hauer, F.R., Lamberti, G.A. (Eds.), Methods in Stream Ecology, Second Edition. Elsevier, Academic Press, pp. 357–379.

Thames21, 2019. 23 thousand wet wipes discovered on stretch of Thames river bank. Available at: https://www.thames21.org.uk/2019/04/23-thousand-wet-wipes-discovered-stretch-thames-river-bank/. Accessed: 1/04/2022.

Tiegs, S.D., et al., 2024. Human activities shape global patterns of decomposition rates in rivers. Science 384 (6701). https://doi-org.cardiff.idm.oclc.org/10.1126/science.adn1262.

Tiegs, S.D., Clapcott, J.E., Griffiths, N.A., Boulton, A.J., 2013. A standardized cotton-strip assay for measuring organic-matter decomposition in streams. Ecol. Indic. 32, 131–139. https://doi.org/10.1016/j.ecolind.2013.03.013.

Tiegs, S.D., et al., 2019. Global patterns and drivers of ecosystem functioning in rivers and riparian zones. Sci. Adv. 5. https://doi.org/10.1126/sciadv.aav0486.

Tiegs, S.D., Langhans, S.D., Tockner, K., Gessner, M.O., 2007. Cotton strips as a leaf surrogate to measure decomposition in river floodplain habitats. The North American Benthological Society 26, 70–77. https://doi.org/10.1899/0887-3593(2007)26[70:CSAALS]2.0.CO, 2.

Tipper, M.J., 2016. Flushability of Nonwoven Wet Wipes. University of Leeds. PhD Thesis.

Tserendorj, D., et al., 2024. Microfiber emission from a municipal wastewater treatment plant in Hungary. Sci. Rep. 14. https://doi.org/10.1038/s41598-024-62817-2, 12041.

UKWIR, 2022. 22/WW/06/22 - Plastics Recieved by the Water Industry & How Best to Tackle Them Through Source Control. Prepared by Eunomia Research & Consulting Ltd. Available at: https://ukwir.org/eng/plastics-received-by-the-water-industry-and-how-best-to-tackle-them-through-source-control-0. Accessed: 1/09/2023.

Wickham, H., et al., 2019. Welcome to the tidyverse. J. Open Source Softw. 4 (43). https://doi.org/10.21105/joss.01686.

Windler, L., Height, M., Nowack, B., 2013. Comparative evaluation of antimicrobials for textile applications. Environ. Int. 53. https://doi.org/10.1016/j.envint.2012.12.010.

Windsor, F.M., Tilley, R.M., Tyler, C.R., Ormerod, S.J., 2019. Microplastic ingestion by riverine macroinvertebrates. Sci. Total Environ. 646, 68–74. https://doi.org/10.1016/j.scitotenv.2018.07.271.

Winterbourn, M.J., Hildrew, A.G., Orton, S., 1992. Nutrients, algae and grazers in some British streams of contrasting pH. Freshw. Biol. 28 (2). https://doi.org/10.1111/j.1365-2427.1992.tb00574.x.

Wood, S.N., 2011. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. J. Roy. Stat. Soc. 73. https://doi.org/10.1111/j.1467-9868.2010.00749.x.

Woodward, G., et al., 2012. Continental-Scale effects of nutrient pollution on stream ecosystem functioning. Science 336 (6087). https://doi.org/10.1126/science.1219534.

Zambrano, M.C., Pawlak, J.J., Daystar, J., Ankeny, M., Goller, C.C., Venditti, R.A., 2020. Aerobic biodegradation in freshwater and marine environments of textile microfibers generated in clothes laundering: effects of cellulose and polyester-based microfibers on the microbiome. Mar. Pollut. Bull. 151. https://doi.org/10.1016/j.marpolbul.2019.110826.

Zancarini, A., Echenique-Subiabre, I., Debroas, D., Taib, N., Quiblier, C., Humbert, J.-F., 2017. Deciphering biodiversity and interactions between bacteria and microeukaryotes within epilithic biofilms from the Louie river, France. Sci. Rep. 7 (4344). https://doi.org/10.1038/s41598-017-04016-w.

Ziklo, N., Yuli, I., Bibi, M., Salama, P., 2024. The influence of physical characteristics of wet wipe fabrics on the microbial biomass accumulation. Cosmetics 11 (4). https://doi.org/10.3390/cosmetics11040106.

Zimmerman, L., Dombrowski, A., Volker, C., Wagner, M., 2020. Are bioplastics and plant-based materials safer than conventional plastics? In vitro toxicity and chemical composition. Environ. Int. 145. https://doi.org/10.1016/j.envint.2020.106066.