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Matthew Dunn
University of Rhode Island

Nicholas Noons

Simon Vojta
University of Rhode Island

Jitka Becanova

Heidi Pickard
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Unregulated Active and Closed Textile Mills Represent a Significant Vector of PFAS Contamination into Coastal Rivers

Authors

Matthew Dunn, Nicholas Noons, Simon Vojta, Jitka Becanova, Heidi Pickard, Elsie M. Sunderland, and Rainer Lohmann

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1 **Unregulated Active and Closed Textile Mills Represent a Significant Vector of PFAS**
2 **Contamination into Coastal Rivers**

3

4 Matthew Dunn¹, Nicholas Noons², Simon Vojta¹, Jitka Becanova¹, Heidi Pickard³, Elsie M.
5 Sunderland³, Rainer Lohmann^{1,*}

6

7 1. Graduate School of Oceanography, University of Rhode Island, Narragansett, RI, 02882 USA

8 2. Rhode Island Department of Environmental Management, Providence, RI, 02980 USA

9 3. Harvard University John A. Paulson School of Engineering and Applied Sciences, Cambridge,
10 MA, 02138 USA

11 *. Corresponding author: rlohmann@uri.edu

12

13 **Abstract**

14 Despite concerns over the ubiquity of per- and polyfluoroalkyl substances (PFAS), little is
15 known about the diversity of input sources to surface waters and their seasonal dynamics.

16 Frequent use of PFAS in textiles means both active and closed textile mills require evaluation as
17 PFAS sources. We deployed passive samplers at seven sites in an urban river and estuary

18 adjacent to textile mills in Southern Rhode Island (USA) over 12 months. We estimated monthly
19 mass flows (g month^{-1}) of perfluorohexanoic acid (PFHxA: 45 ± 56), and perfluorooctanoic acid

20 (PFOA: 30 ± 45) from the upstream river influenced by an active mill. Average mass flows were
21 73-155% higher downstream, where historical textile waste lagoons contributed long chain

22 perfluoroalkyl acids (PFAA). Mass flows of PFNA increased from 7.5 to 21 g month^{-1} between
23 the upstream and downstream portions of the rivers. Distinct grouping of the two main PFAS

24 sources, active textile mills and historical waste lagoons, were identified using principal
25 components analysis. Neither suspect screening nor extractable organofluorine analysis revealed
26 measurable PFAS were missing beyond the targeted compounds. This research demonstrates that
27 both closed and active textile mills are important ongoing PFAS sources to freshwater and
28 marine regions and should be further evaluated as a source category.

29

30 **Synopsis:** Active and closed textile mills have distinct PFAS fingerprints and are important
31 sources to freshwater and marine regions in the Northeastern U.S.

32

33 **Key words:** PFAS, textile mills, waste lagoons, PFCA, passive sampling

34

35

36 **Introduction**

37 Per- and polyfluoro alkyl substances (PFAS) are ubiquitous in the environment, and
38 many PFAS are persistent, and bioaccumulative.¹⁻⁷ With their myriad of negative health impacts
39 on affected communities, this makes identifying PFAS fate and transport in our environment a
40 vital challenge for scientists and policy makers.^{2,8,9} Across the US there are an estimated 57,000
41 contamination sites based on likely sources from aqueous film forming foams, industrial
42 facilities, or PFAS containing waste.¹⁰ Confirming and quantifying the magnitude of PFAS
43 concentrations in surface and ground water at these sites is paramount to the efforts of managing
44 public health in relation to PFAS exposure.

45 Much of our understanding of aquatic PFAS dynamics across different sources, seasonal
46 trends, and environments is based on grab (active) samples.^{6,11-17} These results are limited in the

47 scope, due to their discrete nature, as they only reflect one time, place, and set of conditions that
48 is often then extrapolated across time.¹⁸⁻²⁰ Instead, integrative passive sampling of PFAS can be
49 deployed to provide time weighted average concentrations, which reflect changes across
50 climactic events, tidal changes, or irregular discharges of contamination.^{18,19,21-23}

51 Previous studies on riverine and estuarine dynamics have focused on the impacts of
52 environmental parameters on PFAS such as particulate organic carbon and salinity gradients, as
53 opposed to studying strong temporal trends.^{15,24,25} Few studies with limited temporal aspects
54 demonstrated that PFAS concentrations were quickly diluted when transitioning from river to
55 tidal and estuarine environments, or during rain events/seasons.^{11,12} This contrasts other work on
56 rivers which show some enhancement of PFAS during flooding events suggesting that the
57 specific sources of PFAS to each system are important to understanding the temporal
58 dynamics.^{26,27}

59 To establish a better understanding of PFAS dynamics in the environment, a river and
60 estuary impacted by textile activity was studied over the course of a year. Textile mill
61 contribution to PFAS contamination in North America is a generally understudied field despite
62 the known use of PFAS within the textile industry.^{1,28-30} Furthermore, many studies from China
63 and Italy highlight the textile industry's role in introducing PFAS to their environment.^{29,31-35}
64 According to the US Environmental Protection Agency (EPA) toxic release inventory, there are
65 93 textile mills that report to the toxic reporting inventory, with the vast majority centered
66 around the US East Coast's northern and southern regions.³⁶ Recently, the EPA announced their
67 Current Effluent Guidelines Plan, which highlights plans to study the use of PFAS in the textile
68 sector and their discharges.³⁷

69 Long chain (>C8) perfluorocarboxylic acids (PFCAs) are likely present in the waste
70 lagoons, long after textile activity has ceased at some of these mills.^{1,28,38} The likely source of
71 these long chain PFCA compounds is the degradation of side chain fluorinated polymers or
72 fluorotelomers historically used in the textile industry.¹ For example, perfluorooctanesulfonyl
73 fluoride (POSF)-based urethane and acrylate polymers, were produced by 3M in large quantities.
74 These side-chain fluorinated polymers (SCFPs) were produced and used within the same time
75 frame that these mills were active (1960-2019).^{39,40} If fluoropolymer mixtures were used at these
76 historical mills instead, perfluorooctanoic acid (PFOA) or perfluorononanoic acid (PFNA)
77 compounds used as processing aids alongside long chain fluorotelomer alcohol compounds
78 (FTOHs) were likely released as impurities in waste from polymers during the textile treatment
79 process.³⁹ While specific data on the production of PFAS tied to textile industries is limited, 3M
80 reported that in the range of 4.5-227 tonnes of POSF-derivative and n:2 fluorotelomer PFAS
81 were produced in the US in 2001, and 11-45 tonnes per year between 2012-2015.⁴³ With SCFPs
82 heavily used in textiles, it is likely a large portion of this production ended up at textile mills and
83 their waste lagoons.

84 Our study site, the Pawcatuck River (Rhode Island, USA) has previously been identified
85 in literature as contaminated, with elevated levels of PFAS compounds.¹⁶ Thanks to both active
86 and closed mills, the varied PFAS sources and temperate climate make this a potentially
87 representative environment for studying the PFAS dynamics of textile impacted surface waters in
88 the United States.^{10,28,41-43} Sources of PFAS to this river will come from on-going releases and
89 precipitation mobilizing water and solids from the waste lagoons associated with closed textile
90 mills based on previously observed desorption from tributary sediments.⁴⁴ Additionally, active
91 textile mills near the start and end of the river provide additional pathways for PFAS to enter the

92 river. In particular, the active textile mill located upstream has a permitted direct discharge to the
93 Pawcatuck River for treated industrial wastewater. This could be an especially significant source
94 of shorter chain (<C8) PFCAs or byproducts of replacement compounds used in today's textile
95 industry, which has previously been investigated in the textile hub of Dhaka, Bangladesh where
96 water PFAS profiles were dominated by shorter chain PFCA⁴⁵. Shorter chain FTOHs or
97 perfluorobutanesulfonyl fluoride (PBSF)-based side chain fluorinated polymers are likely
98 reaching surface waters through atmospheric deposition or wastewater releases where they can
99 transform into PFCAs.^{28,31,38,39,46,47}

100 The objectives of this study were hence to i) assess sources of PFAS along the Pawcatuck
101 River and estuary during a yearlong study, ii) combine target analysis of PFAS with suspect
102 screening and total extractable organic fluorine analysis to constrain the presence of non-target
103 PFAS; and iii) identify seasonal trends in PFAS surface water concentrations across changing
104 environmental conditions.

105

106 **Materials & Methods**

107 **Chemicals and Reagents.** Liquid chromatography-grade methanol (LC-MeOH), and
108 water (LC-water) were purchased from Fisher Scientific (New Hampshire, USA) along with
109 ammonium hydroxide (NH₄OH), ammonium acetate (C₂H₇NO₂), ACS-grade ethanol (EtOH) and
110 ACS-grade methanol (MeOH). Analytical standards were used to create native compound
111 standards from the Wellington PFAC-30PAR mix plus an additional four analytical compounds
112 (Table S1). Mass labeled surrogate solutions were derived from Wellington Laboratories'
113 MPFAC-24ES plus an additional three mass labeled compounds purchased from Wellington
114 Laboratories (Canada) (Table S1).

115 **Field Deployments & Sampling.** Microporous polyethylene tube passive samplers with
116 hydrophilic lipophilic balance sorbent were assembled and prepared in accordance with
117 previously described methods.^{21,22,48} Passive samplers were deployed for 24-36 days
118 consecutively between April 2021 and February 2022 across 7 locations spanning the fresh-salt
119 water continuum of the Pawcatuck River in Westerly, Rhode Island (Fig S1, Table S2). These
120 passive samplers are integrative in nature, using sampling rates to convert accumulated mass to
121 ambient water concentrations; currently no performance reference compounds have been
122 validated for their use.²¹ Sites 1-5 are fresh water, riverine sites while sites 6 and 7 are tidal,
123 saline environments at the end of the river before it opens into the Atlantic Ocean. Discrete
124 salinity was not measured as part of this study. Grab samples were also collected at two
125 historical waste lagoons that served as retention ponds for textile mills that are connected to the
126 river during high flow events. These waste lagoons will be hence referred to as WL1 (Bradford
127 Waste Lagoon) and WL2 (Griswold Waste Lagoon). WL1 spans over a mile of river length,
128 while WL2 spans only a half mile portion of the river. For more details on sites, see map Fig S1
129 and table S2.

130 **Environmental Conditions.** Three USGS gauge locations along the Pawcatuck River
131 provide additional understanding of how environmental conditions are changing seasonally in
132 the river (Fig S1). USGS 01117430 (G1) was located downstream of site 1 but preceded site 2
133 near an active textile mill by 0.82 miles. USGS 01117500 (G2) was located between sites 2 and
134 3, approximately 2.2 miles downstream from site 2. USGS 01118500 (G3) is located further
135 down the river, just upstream of site 6 by 0.62 miles, near an active textile mill (Fig S1).
136 Discharge was reported for all three sites and averaged on a monthly basis (Table S3). For
137 additional details on water temperature and precipitation data, see SI.⁴⁹

138 **Passive Sampler Extraction.** Passive samplers were spiked with surrogate internal
139 surrogates and extracted with LC-methanol following previously published methods with slight
140 modifications, including the addition of a third 24-hour extraction sequence, to yield a final
141 extract volume of 18 mL that was then concentrated to 0.5 mL under a gentle stream of nitrogen
142 at 40 °C.²¹

143 **Grab Sample Solid Phase Extraction.** Grab water samples (250-500 mL river water and
144 50 mL waste lagoon water) were mixed, weighed, and spiked with 4 ng of isotopically labeled
145 internal standard followed by offline Oasis weak anion exchange (WAX) solid phase extraction
146 (SPE) using 500 mg cartridges, in accordance with previously published methods based on EPA
147 Method 533.^{21,50} A matrix spike experiment was performed to assess the recovery of a native
148 PFAS spike in tap water and the results can be seen in table S4, with recoveries ranging from 84-
149 110%.

150 **Instrumental Analysis.** The instrumental analysis was performed using a SCIEX
151 ExionLC AC UHPLC system coupled to a SCIEX X500R quadrupole time-of-flight tandem
152 mass spectrometer (QTOF MSMS). The concentrated methanol extract was reconstituted with
153 10mM ammonium acetate in water to reach final ratio of 40/60 (methanol/water). A Phenomenex
154 Gemini 3µm C18 110Å 50x2mm LC analytical column preceded with a Phenomenex
155 SecurityGuard cartridge was used for the analyte separation. Another Phenomenex Gemini 5µm
156 C18 110 Å 50x4.6mm LC analytical column was used to delay the PFASs instrumental
157 contributions. The aqueous mobile phase (MPA) was 10mM ammonium acetate in water and the
158 organic mobile phase (MPB) was 10mM ammonium acetate in methanol. LC parameters were
159 set as follows: flow 0.3mL/min, injection 20 µL, column oven 45°C. Solvent gradient of MPB

160 gradually increased from 40% to 80% (1 to 5.5 min), 80% to 100% (5.5 to 7 min), then held for
161 one minute and finally drop to 40% (8 to 8.5 min) and held for another 6.5 minutes.

162 For the quantification of the target analytes, a HRMS/MS (MRM HR) method was used.
163 The negative ESI with the following parameters was used: curtain Gas at 30 psi, ion source gas 1
164 at 40 psi, ion source gas 2 at 60 psi, temperature 450°C (see Table S1 for compound dependent
165 parameters). The MS data for the suspect screening analysis were collected using Sequential
166 Window Acquisition of All Theoretical Mass Spectra (SWATH) mode with the IS operated
167 under the same conditions.

168 **Suspect Screening Analysis.** The identification of PFAS using the suspect screening
169 approach was performed using the SCIEX OS software and was based on precursor mass,
170 isotope pattern, retention time, exact mass accuracy, and MS/MS fragmentation matching
171 (SCIEX Fluorochemical HR-MS/MS Spectral Library 2.0 Activ). Following the conventions
172 from Schymanski et al. (2014) and Charbonnet et al. (2022), all the identified compounds were
173 assigned the confidence level of identification 2a (MS/MS spectral library match) or 5a (suspect
174 screening list exact mass match).^{51,52}

175 **Extractable Organofluorine Extraction & Analysis.** Since targeted analysis only
176 quantifies a small suite of PFAS with available analytical standards and suspect screening is a
177 qualitative method to identify suspect PFAS, a subset of passive samplers were extracted and
178 analyzed for extractable organofluorine (EOF) to quantify organofluorine, a proxy for total
179 PFAS and other organofluorine compounds. Passive samplers for EOF were co-deployed with
180 passive samplers for targeted PFAS analysis in April of 2022 at sites 2 and 7. These sites were
181 chosen because they represent either active textile activity (Site 2) or the final downstream signal
182 being exported to the ocean (Site 7). Passive samplers were soaked for 24 hours in 6 mL a 0.01%

183 ammonium hydroxide in water (v/v) to remove inorganic fluoride prior to extraction, as detailed
184 for SPE cartridge analysis in previous publications.^{42,53} Passive sampler extraction was then
185 performed with slight alterations to the method described above, by neglecting the mass labeled
186 surrogate spike and ENVI-Carb cleanup steps. For additional details on instrumentations see SI.

187 **Quality Assurance and Quality Control.** For the targeted analysis data, method
188 detection limits (MDLs) were calculated from laboratory and field blanks collected for both
189 passive sampler and solid phase extraction procedures. The median and three times standard
190 deviation of the blank concentrations were summed to determine MDLs. MDL for passive
191 samplers and grab samplers can be found in Table S5. Method efficiency for SPE and passive
192 sampler extraction was evaluated using mass labeled surrogates purchased from Wellington,
193 including their MPFAC24 mixture and 3 additional single compounds (Table S6). Recoveries of
194 mass labeled surrogates ranging from 60-140% were accepted, with an exception made for
195 Perfluoroundecanoic acid (PFUnDA) which had an average recovery of 55% and is reported here
196 with less confidence than that of the other compounds. For additional details on these quality
197 assurance steps and further EOF quality assurance and control methods, see SI.

198 **Deriving Time Weighted Average Concentrations.** Sampling rates for PFAS from
199 Gardiner et al. 2022 were chosen and adjusted for temperature changes using an approach
200 outlined in Dunn et al. 2022 and applied to calculate time weighted average (TWA)
201 concentration from passive sampler uptake in this study (Fig S2, S7).^{21,22} To adjust sampling
202 rates for temperature, monthly deployments were grouped into three temperature categories; 5,
203 15, and 25 °C. Winter months (January, February, March) were assigned to 5 °C, spring and fall
204 months (September, October, November, March, April, May) were assigned to 15 °C and

205 summer months (June, July, August) were assigned to 25 °C, all in accordance with reported
206 water temperatures (Table S2). Two compounds, 7:3 FTCA and PFUnDA, did not have sampling
207 rates published thus sampling rates for compounds with similar chemistry were used to calculate
208 concentrations (PFOA and PFDA respectively). The sampling rates from Gardiner et al. 2022
209 have an average percent standard deviation of 34%, which could be considered the mean
210 uncertainty in each TWA concentration reported.²² For additional details on the determination of
211 sampling rates and TWA, see SI and Fig S2.

212 **Statistical Analysis.** Principal component analysis (PCA) was performed using the
213 R software (Version 2022.07.0+548) and the prcomp package, which normalizes the data for
214 analysis based on composition, not concentration. All zeros, non-detects, or samples with values
215 below reported method detection limits (MDL) were replaced by either the MDL/sqrt(2) for grab
216 samples or using the MDL/sqrt(2) as the mass for passive samplers for analytes with $\geq 70\%$
217 detection. All other compounds, specifically the shorter than 8 chain (<C8) perfluoroalkyl
218 sulfonic acids, were not included in the analysis. A non-parametric Kendall Rank Correlation test
219 was performed in R for investigating the relationship between rain and PFAS concentrations. A
220 non-parametric test was chosen for this non-normally distributed, dynamic environmental data
221 set to see if broad correlation could be found between monthly sum rain, and monthly average
222 PFAS concentrations. Kruskal-Wallis and Pairwise Wilcox tests were used to evaluate statistical
223 differences among the seven sites with passive-sampler derived sum PFAS concentrations.

224 **Results and Discussion**

225 **Active Sampling Showed PFAS Are Ubiquitous in the Pawcatuck River.** In
226 the active grab samples, PFCAs were most prevalent: C₄-C₁₁ PFCAs were detected at all sites,

227 with PFBA, PFPeA, PFHxA, PFHpA, and PFOA detected in almost every sample collected
228 (Table S8). Concentrations in discrete grab samples ranged from 0.51 to 27 ng L⁻¹ (Table S8).
229 PFPeA and PFOA were the only compounds detected in every single grab sample analyzed.
230 Other regularly detected compounds include PFBA, PFHxA, PFNA, and PFDA, present in 67%,
231 82%, 64%, and 77% of grab samples respectively (Table S7). In grab samples the highest
232 average concentration was 11 ng L⁻¹ for PFHpA, but this compound was only detected in 25% of
233 samples above method detection limits (Table S8). The second highest average concentration
234 throughout the river was PFHxA, at 7.0 ng L⁻¹, which was also detected at more sites than
235 PFHpA. PFASs were less prevalent; PFHpS was not detected once above detection limits,
236 PFHxS ranged from 0.46-1.6 ng L⁻¹, and PFOS ranged from 2.2-7.8 ng L⁻¹ in grab samples
237 (Table S8). There were no statistical differences between sum PFAS concentrations across all
238 seven river sites (Pair Wilcox rank sum test, p <0.05, Table S9).

239 **Passive Samplers Suggest PFAS Are Ubiquitous Across Long Temporal Scales.**

240 Time weighed average concentrations of PFAS deduced from passive samplers ranged from
241 0.0023 ng L⁻¹ to 18 ng L⁻¹ (Table S10). The lower end of this range displays the advantage of the
242 passive sampler over water grab samples; pre-concentration in the passive sampler allows for
243 lower reported TWA concentrations in water. 0.0023 ng L⁻¹ was back calculated from a mass of
244 PFAS in the tube passive sampler that was above the sampler MDL, despite producing a
245 calculation below the reported grab sample MDL. PFCAs were most prevalent in the passive
246 sampler results, with PFBA, PFPeA, PFHxA, PFHpA, and PFOA detected in every sampler
247 deployed. PFASs such as PFHxS, PFHpS, and PFOS were detected less frequently in passive
248 samplers, with TWA concentrations ranging from 0.18 – 11 ng L⁻¹. PFOS was the dominant

249 PFASs in passive samplers, ranging from 0.05-11 ng L⁻¹, but was only detected in 31% of
250 samples (Table S8).

251 From this point forward, TWA concentration results from passive samplers will be
252 used for most of the discussion and interpretation of environmental dynamics of PFAS within the
253 Pawcatuck River. PFAS TWA concentrations across month long passive sampler deployments
254 were calculated for each of the 7 field sites along the Pawcatuck River and estuary using
255 previously published sampling rates, adjusted for temperature using a model described in
256 literature.^{21,22} Due to the longer duration deployments in this study, it is difficult to verify that all
257 PFAS compounds are truly present in the river every day. However, even if the passive samplers
258 were only capturing plumes as they irregularly pass by, it is clear that enough PFAS are traveling
259 down this river on a monthly basis to be detected. It is worth noting that discrete active samples
260 (reflecting the beginning or end of passive sampler deployments), may not perfectly match TWA
261 concentrations in this complex and dynamic river and estuary system. Passive samplers were
262 well separated from grab samples during principal component analysis, and this result suggests
263 that using passive samplers in lieu of traditional grab sampling will result in different findings
264 (Fig S3). This highlights the potential benefits of using passive samplers as valuable tools for

265 long term monitoring of PFAS as they may provide a more representative picture of time
266 weighted contamination.

267 PFAS concentration profiles for rivers of this size are rare, as most literature has focused
268 on larger rivers in the region such as the Hudson and Delaware Rivers.^{54,55} The Hudson river has
269 been identified as having a median concentration of 20 ng L⁻¹ PFOA, not far from the most
270 contaminated sites, site 5 and the waste lagoons in this study.⁵⁴ The Delaware River has had a

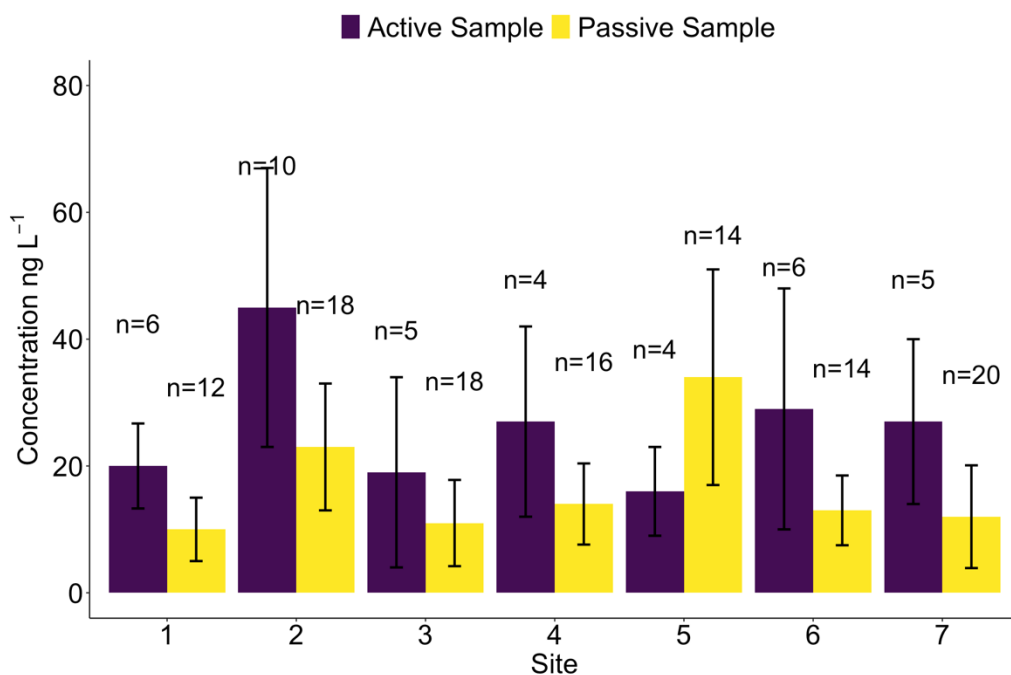


Figure 1. Spatial trends in average sum PFAS concentrations of 12 compounds across 7

sites. Both active and passive samples identified site 2 as capturing a large source of PFAS to the upstream portion of the river. Both sampling approaches show relatively consistent profiles of sum PFAS concentrations across the river. However, the passive samplers identified site 5 as having a spike in concentration from intermittent waste lagoon discharges that is not captured by discrete active sampling.

271 much larger range of reported contamination, with individual concentrations ranging from 1-
272 1000 ng L⁻¹ for PFCA and PFSA compounds also investigated in this study.⁵⁵ However, the
273 mean concentration of all compounds across reported study sites was 30 ng L⁻¹, which is again
274 similar to the higher level sites in this study⁵⁵. Another recent study employed the same passive

275 samplers and same sampling rates to report very similar concentration ranges to this study (8-30
276 ng L⁻¹) in the Danube River, highlighting the broad applicability of these methods.⁵⁶

277 **Organofluorine mass balance** Nine water grab samples and 10 passive samplers from
278 May 2021 were screened for semi-quantitative analysis of suspect compounds. In both matrices,
279 no additional suspect compounds were identified. Water grabs were analyzed at each site,
280 including the waste lagoons, while passive samplers were only analyzed for sites 1, 3, 4, 5, and
281 7.

282 To better investigate if these analytical approaches with passive samplers captured the
283 majority of organofluorine present in the river, passive samplers recovered in April 2022 from
284 sites 2 and 7 were measured for extractable organic fluorine (EOF). In both cases EOF
285 measurements were below the method detection limits of 45-66 ng F sampler⁻¹. Concentrations
286 of target PFAS from co-deployed passive samplers were converted to F equivalents: site 2 had an
287 average of 17 (1.6) ng F sampler⁻¹ EOF and 7 had an average of 4.6 (3.9) ng F sampler⁻¹
288 EOF. However, because highly elevated levels above method detection limits are not displayed
289 in the EOF analysis of these passives, it can be concluded that the target and suspect screening
290 analyses are not missing a large portion of the organofluorine contamination in the Pawcatuck
291 River.

292 There are limitations to these results, including that suspect screening is only limited to
293 compounds found on a predetermined suspect list within the instrumental software. No true non-
294 target analysis was performed, additionally some compounds including non-polar PFAS such as
295 FTOHs or ultra short chain PFAS could not be detected by the LC MS/QTOF used in this study.
296 Some additional unexplained organofluorine might exist in this river, however it did not exceed

297 the EOF detection limits. For reference, EOF concentrations in a contaminated river with similar
298 levels of targeted PFAS but additional contamination from an AFFF source in the region ranged
299 from 280-500 ng F L⁻¹.⁵³

300 **Active Mills Might Show Distinct Fingerprint.** Separate from this discussion of
301 precipitation and transport dynamics, site 2 provided an important aspect of PFAS source
302 dynamics from an active textile mill. Generally, there was no apparent trend in PFAS TWA
303 concentrations at this site across the year, or with changes in precipitation (Fig S4). This, paired
304 with the detection and magnitude of 7:3 Fluorotelomer carboxylic acid (FTCA), a compound
305 rarely found in the rest of the river, suggests the air emissions or wastewater discharges of the
306 active mill may be a constant, consistent source of PFAS. Site 2's profile clearly separated out
307 from the other samples during principal component analysis, suggesting that the presence of
308 increased 7:3 FTCA concentrations may serve as a good tracer for active mill fingerprinting (Fig
309 2).

310 The compound 7:3 FTCA, a common product of the degradation of volatile
311 fluorotelomer alcohol (FTOH) compounds into terminal PFCA compounds, was detected at 6 of
312 the 7 sites in this study (Table S9).^{47,57} TWA concentrations ranged from 0.12 ng L⁻¹ to 3.3 ng L⁻¹,
313 with the highest concentrations being detected at site 2, located downstream from an active
314 textile mill (Table S8). Additionally, there was no detection of 7:3 FTCA at site 1, the source
315 pond for the river that sits upstream of all textile activity (Table S10). The presence of this
316 compound downstream of the mill suggests use and emissions of FTOH-chemistry, which is then
317 degraded to terminal PFCAs that are detected in elevated concentrations at this site in the river,
318 such as PFHxA (Fig 3).^{46,47,58} Due to the temperate nature of this region, there is monthly rain,

319 which may aid in the consistent deposition of airborne PFAS from air to surface water, without
320 the discrete seasonal changes in source dynamics seen downstream of waste lagoons. Beyond

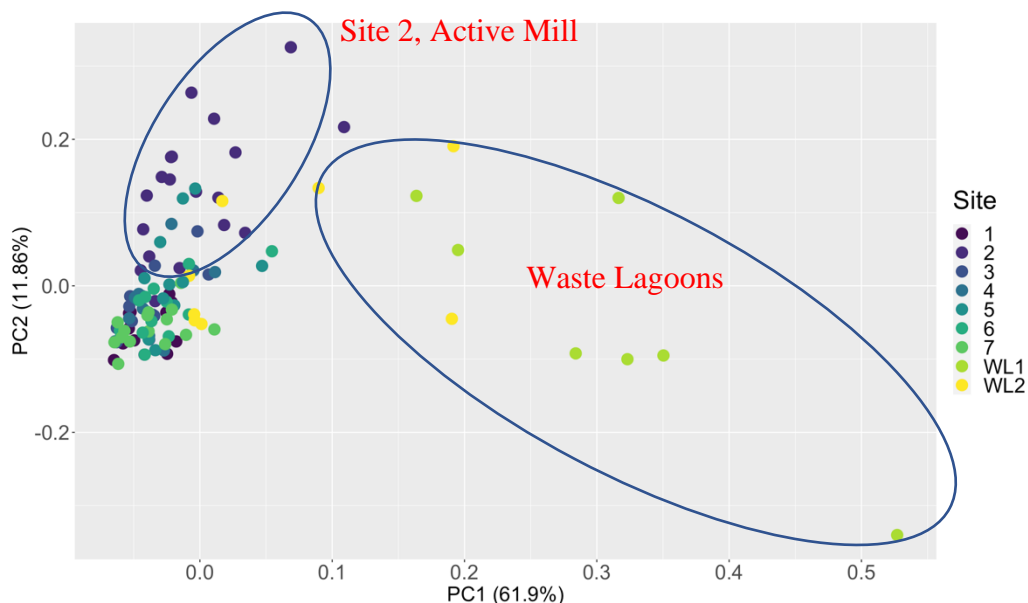


Figure 2. Principal Component Shows Separation of Active Mill and Historical Waste Lagoon Fingerprint. Site 2 is located downstream of an active textile mill and exhibited increased concentrations in a PFCA precursor, 7:3 FTCA, as well as elevated shorter chain PFCAs. Sites WL1 and WL2 were historical waste lagoons with high concentrations of long chain PFCAs. Ovals were added to guide the eye and represent specific sites as labeled.

321 fluorotelomer-based chemistry (explaining the presence of 7:3 FTCA), the large quantities of
322 short chain PFCAs in the upstream section of the river suggests the potential use of shorter side
323 chain fluorinated polymers that degrade into four and six chain PFCAs.⁴¹ There are limited
324 studies that discuss FTCA compounds, with most focused on biotransformation or
325 bioaccumulation instead.^{47,59,60} In contrast, most studies of rivers and coastal systems have not
326 included FTCAs in their methods and results.⁵⁴⁻⁵⁶

327 **Spatial Trends In Dissolved PFAS concentrations.** Sites 2 and 5 consistently
328 showed elevated concentrations of PFAS compared to their neighboring sites (Fig 3), which was
329 confirmed to be statistically significant by the Pairwise Wilcox rank sum exact test ($p < 0.05$)
330 (Table S11). Overall, a Kruskal-Wallis rank sum test showed significant differences amongst all

331 sites in this study ($p < 0.05$). Interestingly, the passive sampler data showed statistically
332 significant differences between sites, while the grab sample data did not (Table S8, S10).
333 Arguably, passive sampler derived concentrations reveal long-term concentrations that are site-
334 specific.

335 Sum of 12 PFAS concentrations ($\Sigma_{12}\text{PFAS}$) at site 2 ranged from 13 ng L^{-1} to 43 ng L^{-1} . Short
336 chain compounds PFPeA and PFHxA, as well as legacy compound PFOA had the highest
337 average concentrations and detection frequency, with 2.9 ng L^{-1} , 7.4 ng L^{-1} , and 3.6 ng L^{-1}
338 respectively across the year at site 2 (Table S10). Active and passive sampler results are
339 generally in good agreement on spatial trends in concentration, except for at site 5, which is
340 influenced the most by intermittent discharges from waste lagoons, suggesting the need for
341 passive samplers to monitor these dynamic systems (Fig 1).

342 At site 5, average Σ_{12} PFAS ranged from 17-67 ng L⁻¹, with PFHxA, PFHpA, and
 343 PFOA dominating (8.8, 3.8, and 7.2 ng L⁻¹ mean TWA concentrations, respectively). There was
 344 also a noticeable increase in concentrations of long chain (<C8) PFCA, PFNA (2.6 ng L⁻¹),
 345 PFDA (2.1 ng L⁻¹), and PFUnDA (1.6 ng L⁻¹) at site 5 (at other sites, these compounds never
 346 exceeded TWA concentrations of 1 ng L⁻¹ each). Historical use of these long chain compounds
 347 was suspected at the closed textile mills and their presence in the river has been previously
 348 reported suggesting that nearby waste lagoons may be a source of C8+ chemistry PFAS (see

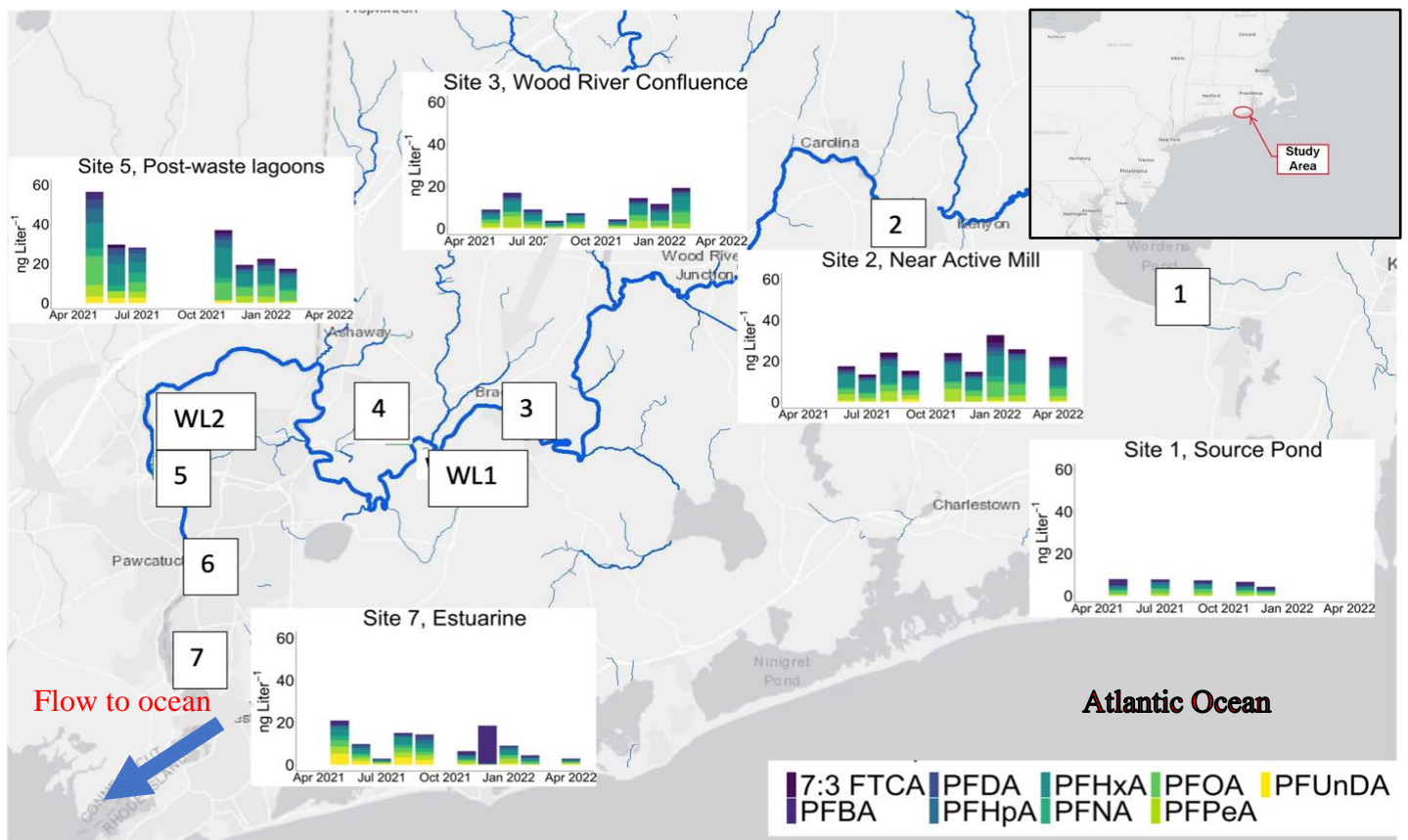


Figure 3. Spatial trends in monthly time weighted average concentration across the Pawcatuck River.

349 below).¹⁶ Overall, the PFCA and telomer signals are strongly observed in this river both
 350 temporally and spatially, while the sulfonic acid signatures are not.

351 **Source dynamics of PFAS in the Pawcatuck River.** We first investigated known
352 PFAS sources to the river including two waste lagoons (WL1 and WL2) associated with
353 historical textile mills that have since closed. Passive samplers were not used in the waste
354 lagoons due to their stagnant nature, which complicates modeling of PFAS uptake.²¹ Grab
355 samples showed that WL1 consistently exhibited higher concentrations than all river sites and
356 the WL2 lagoon, with concentrations of individual PFCA exceeding 10 ng L⁻¹, and some long
357 chain compounds such as PFOA, PFDA, and PFUnDA ranging from 13-63 ng L⁻¹ (Table S7).
358 These longer chain compounds are known for a strong binding affinity to organic matter/
359 sediments, increasing their residence time in these stagnant waste lagoons.⁶¹ Additionally, these
360 waste lagoon sites were well separated from the river samples by principal component analysis,
361 suggesting a unique signature that is generally diluted in the river sites (Fig 2). WL2
362 concentrations were generally lower, with the same long chain compounds ranging from 0.5 – 26
363 ng L⁻¹. PFOS concentrations were highest in WL1, consistently an order of magnitude above any
364 active samples from the river (Table S7). This unique PFOS signature and the presence of long
365 chain PFCAs larger than the C9 PFNA suggests that either POSF-based SCFPs or longer telomer
366 compounds not identified by suspect screening were used at these closed textile mills.^{39,40}
367 However, it is important to note that a fluorotelomer-based signature would result in increased
368 PFOA and PFNA concentrations, which are also observed in these waste lagoons, and it is
369 possible due to the age of these lagoons that all precursor compounds have since been
370 transformed, resulting in the lack of longer chain FTCA compounds detected in this study.
371 Overall, this suggests that possibly both POSF-based and FT-based mixtures were used at these
372 closed mills.³⁹

373 These PFAS-contaminated waste lagoons represent a potential long-term source of
374 PFAS to the Pawcatuck River, especially for the longer chain PFCAs: PFNA, PFDA, and
375 PFUnDA TWA concentrations increased by 280-340% between river sites 4 and 5. This sharp
376 spike demonstrates the likeliness of these waste lagoons as point sources (Fig 4-5). The presence
377 of the longer chain PFCAs is especially concerning as they are known to strongly bioaccumulate
378 in biota in temperate estuaries.^{3,62} During and following precipitation events, spillover of waste
379 lagoon water flowed into the river, where PFAS can be taken up by downstream benthic
380 invertebrates and larvae in commercially and ecologically important estuarine nurseries.

381 Other, likely minor sources of PFAS to this system might exist. The river's basin
382 has generally unconfined interactions between groundwater and surface water, meaning PFAS
383 could be drawn down from the surface sources into groundwater at various points throughout the
384 river.⁶³ Likewise, there could be some contribution from private wells during groundwater-river
385 recharge.⁶³ However the majority of the river's freshwater section is a very rural area that is not
386 likely to compete with the large mass flows from the textile industry.⁶³ Additionally, there is a
387 waste water treatment plant (WWTP) located between sites 6 and 7 on the tidal portion of the
388 river. This is an obvious source of PFAS. However, because it is confined to the tidal zone, there
389 is very little chance of this PFAS signal mixing upwards out of the tidal zone into the freshwater
390 portion of the river. For these reasons, we generally assume there are no other major sources of
391 PFAS aside from textile activity on this rural river.

392 **Influence of precipitation on PFAS concentrations.** We found pronounced
393 season patterns in PFAS concentrations using passive sampler data. Generally, when
394 precipitation was lowest in the winter months of November 2021- January 2022, sites 4 and 5

395 exhibited TWA concentrations below the annual mean for PFAS (Fig 3, Fig S5) (TWA
396 concentrations were corrected for temperature). Furthermore, the highest TWA concentrations
397 were observed in the spring and early summer of sites site 4 and site 5, at the same time the
398 highest concentrations of PFAS were found in the waste lagoons upstream of these sites (Fig 4).

399 We did not find a statistically significant correlation (Kendall Rank Correlation test
400 for non-normative data at $p < 0.05$) between sum of monthly rainfall and individual monthly
401 PFAS concentrations in the river. The statistically strongest correlations between PFAS
402 concentration and rain were observed at site 1 for only PFHxA and PFOA ($r = 0.73$ and 0.86
403 respectively), suggesting the site's surface waters may be contaminated by rain-driven runoff.
404 Increases in PFAS concentrations at sites 4 and 5, both downstream of waste lagoons, were not
405 significantly related (Kendall Rank Correlation, $p > 0.05$) to increased rainfall during spring and
406 early fall; however visible flooding of the river with waste lagoon discharge following rain
407 events has been observed by the authors.

408 Principal component analysis showed that rain may be providing only limited
409 directionality to the spread of data (Fig S4). These results suggest that the 28+ day deployment
410 length of these passive samplers obscures the pulse from individual precipitation events on PFAS
411 concentrations. Despite waste lagoon intrusion into the river being primarily driven by rain and
412 flooding, there was no clear statistical example of this within the collected data set (Fig S3). A
413 high-resolution active sampling campaign would be needed to prove this with 7 or 14-day
414 deployment lengths for passive samplers or water grab sampling during rain events. Both rain
415 and desorption of PFAS mobilized from tributary sediment have been identified as important

416 sources in some surface water studies, and remain the most likely explanation, albeit on shorter
417 time scales, for PFAS intrusion into the Pawcatuck River.^{26,44}

418 **Can River Discharge Explain PFAS Dynamics?** Despite higher concentrations in
419 waste lagoon 1, the adjacent site 4 does not exhibit a higher concentration profile across the year
420 than the previous site 3, which has no impact from these waste lagoons. A large increase in
421 concentration is not seen until site 5, for which the adjacent waste lagoon 2 is less contaminated
422 than waste lagoon 1. This is likely reflective of the small-scale changes in river discharge and
423 volume, and how they can play a large role in concentration dynamics. Additionally, sorption to
424 settling organic carbon particles has been highlighted as an important parameter for PFAS fate
425 and transport, but the passive samplers we deployed only measure dissolved PFAS
426 concentrations, which might be why the signal at WL 1 was muted.^{15,24} This suggests there might
427 be a lag period which we hypothesize is due to OC-bound PFAS traveling downstream from
428 WL1 at site 4 and partitioning out into less contaminated river surface waters. The higher
429 concentrations of less polar long chain compounds with high K_{oc} in WL1 compared to WL2 may
430 explain their preferential binding to OC at this site compared to others.⁶¹

431 **Mass Flow of PFAS to the Atlantic Ocean and Long Island Sound.** Estimates of
 432 PFAS loading from the Pawcatuck into the Atlantic Ocean and nearby Long Island Sound were
 433 calculated from TWA concentrations and average discharge from the river at USGS gauge
 434 01118500, located approximately 6 miles from the Atlantic Ocean (Table S3, Figure S4).
 435 Monthly discharge at this site (0.62 miles upstream) of the tidal site 6 ranged from 7.3×10^3 to
 436 3.1×10^4 Liters second^{-1} , highlighting the dynamics of the river's water flow in this temperate
 437 region (Table S3). TWA dissolved PFAS concentrations were used to calculate mass flows.
 438 Monthly flows of PFAS to the Little Narragansett Bay estuary, which feeds into the Atlantic
 439 Ocean were consistent from June 2021 through December 2021, before a sharp increase
 440 associated with increases in river flow discharge in early 2022 (Fig 5, Table S3). PFBA, PFPeA,
 441 PFHxA, and PFOA were the dominant compounds discharged into the ocean from the
 442 Pawcatuck with mean monthly mass flows of 110, 56, 87, and 31 g month^{-1} respectively. The

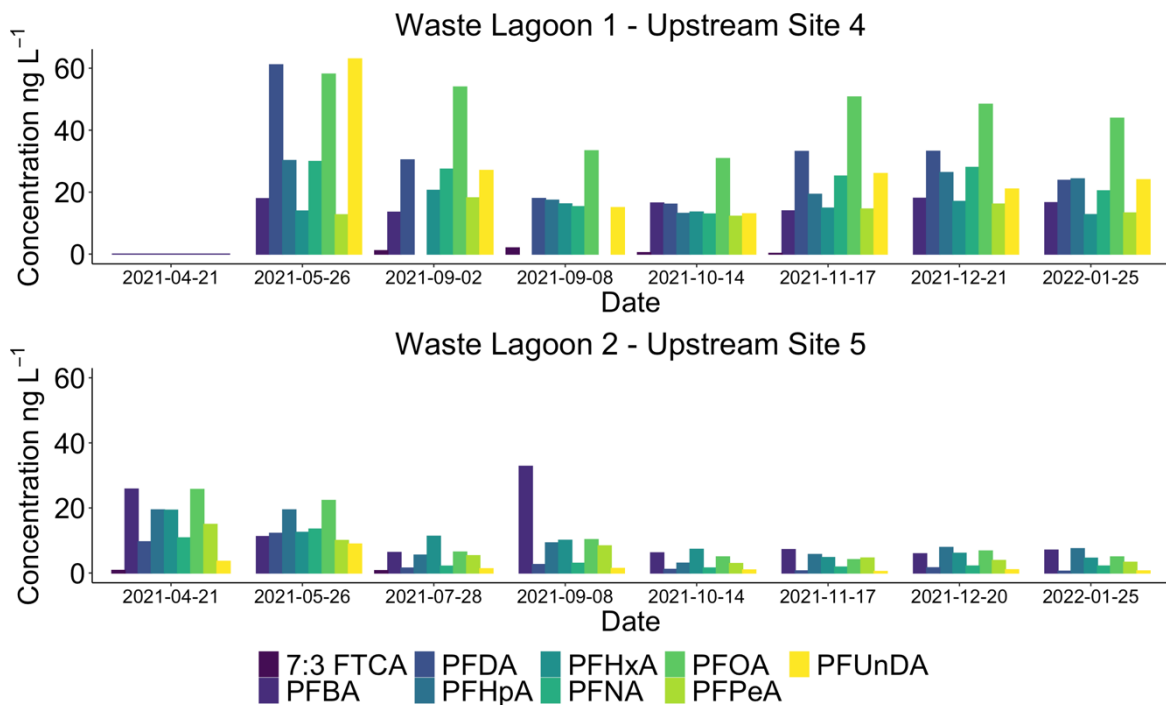


Figure 4. Active sampling of waste lagoons shows elevated PFAS concentrations.

Concentrations are calculated from solid phase extraction of discrete grab samples. No samples taken at Waste lagoon 1 in April 2021.

443 scale of the mass flow is large, with monthly discharges on the order of tens of grams of
444 individual PFAS from this relatively small river into the Atlantic Ocean (Fig 5). This is one of
445 many potential sources to the Atlantic Ocean, though a study performed nearby this region has
446 displayed elevated levels of PFHxA, PFOA, PFNA, PFDA, and PFOS in the coastal ocean that
447 could be partially sourced from the Pawcatuck River.⁶⁴ Additionally prior large scale modeling
448 efforts show that rivers are the likely dominant source of PFAS detected in Atlantic seawater
449 samples, which emphasizes the need for source attribution and mitigation in coastal rivers.⁶⁵

450 The large increase in PFBA concentrations at site 6 from December 2021 to
451 January 2022 was also exhibited downstream at site 7 during the same deployment time (Table
452 S8, Figure 5). While the exact cause is unknown, it could be a seasonal input from a nearby
453 wastewater treatment plant (WWTP). Floor stripping and waxing is known to predominantly
454 release PFBA, and this could have happened in public buildings during the winter break, and
455 reached the river through the municipal plumbing and eventually the WWTP outfall.⁶⁶ Perhaps
456 an equally likely explanation could be the limited time use of quantities of PFBA or PFBA-
457 precursors at the active textile mill along the river, less than a mile upstream of site 6.

458 The PFHpA/PFOA ratio has previously been identified as a method for examining
459 the importance of atmospheric deposition as a source of PFAS to a given area.⁴⁶ The ratio of
460 PFHpA/PFOA was basically the same downriver (0.69 ± 0.22 downstream versus 0.65 ± 0.32
461 upstream). This ratio being <1.0 is likely explained by the large inputs of longer chain
462 compounds from the waste lagoons in the downstream portion.

463 **Contribution of Waste Lagoons to Pawcatuck River Mass Flow.** The waste
464 lagoons are two large pools of PFAS, providing a multitude of pathways for their contaminants

465 to enter the Atlantic Ocean. Mass flows of PFAS were calculated at three different sites along the
466 river using the local USGS discharge data and TWA concentrations continuously increase in
467 PFAS mass flows was observed downriver, suggesting limited losses (Table S12). There was an
468 average increase of 16-fold, or 41 grams, in monthly discharge of each PFAS compound between
469 the sites upstream and downstream of all textile mills. In addition, there was an average increase
470 of 7-fold, or 28 grams, in monthly discharge of each PFAS compound between the mid-river site
471 and the site downstream of the waste lagoons (Fig 5, Table S12). The largest increases are seen
472 in the long chain PFCA compounds PFNA, PFDA, and PFUnDA, which all increase in mass
473 flow by over 10-fold (Table S12). Additionally, no compounds displayed an average decrease in
474 mass flow after the waste lagoon, with the lowest increase reported for a compound equating to
475 2.3-fold for 7:3 FTCA, which specifically has likely upstream sources (Table S12). These mass
476 flows are underestimated to some degree, as we did not capture and quantify the particle phase
477 bound PFAS in waste lagoons or rivers with the use of passive samplers. These passive samplers
478 and overall mass flows better represent the bioavailable fraction of dissolved PFAS. However,
479 PFAS Log K_{oc} values range from 1.3-5.6, increasing with chain length and hydrophobicity. If we
480 assume a typical coastal fOC of 0.02, and a total suspended solid (TSS) value of 30 mg L⁻¹
481 previously reported for the river, the vast majority of the PFAS will remain in the dissolved
482 phase, with potential underestimation of only the longest chain compounds such as PFDA and
483 PFUnDA as fOC increases (Table S13).^{67,68}

484 Remediating these waste lagoons is an obvious step that can be taken towards
485 mitigating these ongoing PFAS releases to the river and its estuary. Prior to the waste lagoon
486 intrusion into the river, the suspected source from the active mill upstream contributes average
487 mass flow of 3.6-45 g month⁻¹ of each of the 9 PFAS discussed here (Fig 5, Table S11). These

488 flows are dominated by shorter chain PFPeA and PFHxA compounds in addition to PFOA
 489 (Table S11). This signature is still seen in the downstream flow, but with large increases in
 490 PFBA, PFHpA, and the previously discussed long chain compounds, suggesting that the
 491 upstream sources are still significant drivers of PFAS pollution to the Pawcatuck as well.

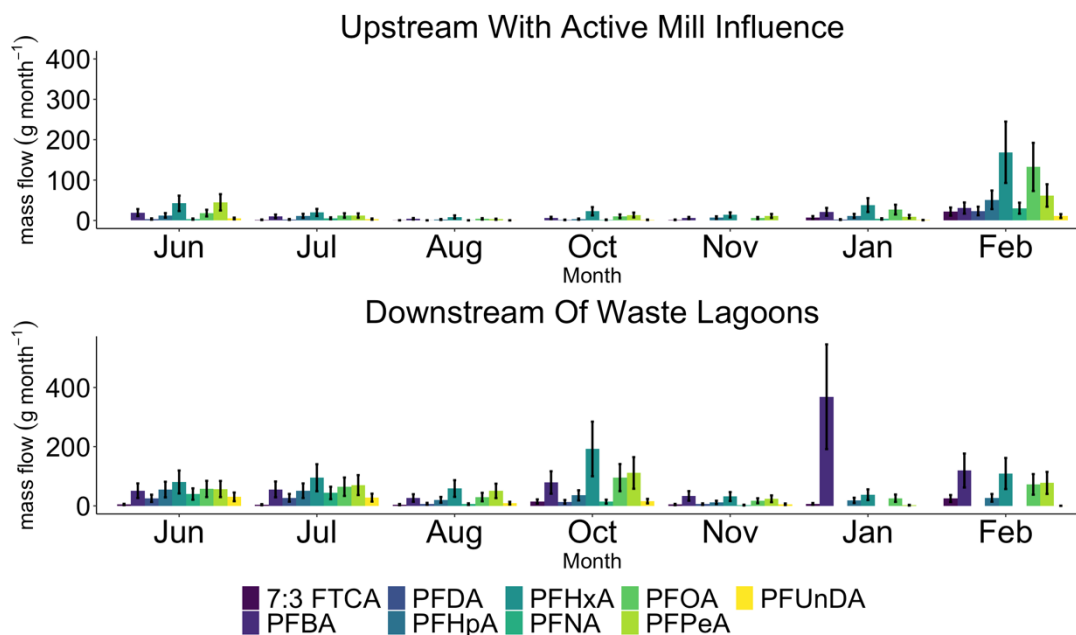


Figure 5. Comparison of the Mass Flux of PFAS Up and Downstream of the Waste

Lagoons. Values of mass flux in g month⁻¹ are calculated using time weighted average concentrations from passive samplers and discharge results from nearby USGS gauges. Error bars are propagated uncertainty derived from mean standard deviation of time weighted average water concentrations and mean standard deviation of monthly discharge data.

492

493 **Conclusions.** The Pawcatuck River and estuary are highly dynamic systems that are not
 494 well captured by discrete grab samples, the traditional method from which the scientific
 495 community has drawn conclusions on PFAS fate and transport. Changes in PFAS concentrations
 496 can be seen both spatially and temporally across the river, underlining the need for broad, TWA
 497 data sets for stakeholders and management of this contaminated water body. Many factors,
 498 including precipitation, river discharge and volume, and the presence of salinity and organic

499 carbon must be accounted for in future studies of PFAS dynamics. Using shorter passive sampler
500 deployment durations, such as 7-14 days instead of the 28+ day timescale of this study might be
501 able to better capture local PFAS responses to climate factors. These environmental
502 considerations play a sizeable role in controlling the Pawcatuck River's PFAS load delivered to
503 the coastal ocean and Long Island Sound estuary from these textile-influenced point sources. In
504 this riverine system and estuary, precipitation demonstrates the potential to mobilize PFAS from
505 sources, and increase dissolved concentrations in the river.

506 With a long history of early industrialization, it is likely that several rivers in this region
507 of the US have similar PFAS contamination, and indeed it appears much larger rivers have
508 similar concentrations.^{54,55} The Environmental Protection Agency's Toxics Release Inventory
509 lists 93 active textile mills in the United States, with unknown use of PFAS, or PFAS emission.³⁶
510 Beyond these active mills, there are likely an even greater number of former textile mills around
511 with archaic waste lagoon systems, such as those investigated in this study. Results from the
512 Pawcatuck River could serve as an example of the scale and fingerprint of PFAS contamination
513 for other textile industrialized rivers.

514 With tens to hundreds of tonnes of PFAS and side chain fluorinated polymers produced
515 yearly, the potential scope of environmental pollution is quite large. These side chain fluorinated
516 polymers were heavily used in textiles, and it is likely a large portion of this production ended up
517 at textile mills and their waste lagoons. Scaling the emissions documented from the Pawcatuck
518 River to other sites would yield large annual emissions of individual PFAS reaching important
519 biological nurseries and the Atlantic Ocean. With the continued optimization and deployment of
520 passive samplers, additional sources, and transport mechanics of PFAS in coastal systems can be

521 investigated, and ideally guide how and where to best use funds for remediation and
522 management of textile mills and their PFAS contamination.

523 **Supporting Information**

524 The Supporting Information is available free of charge on the ACS Publications website at DOI:
525 It contains additional analytical details, methods, standards, quality control and assurance,
526 LC/MS conditions, details on field deployment sites and conditions, sampling rates for all PFAS
527 compounds of interest, and further details on calculations of mass flows and time weighted
528 average concentrations.

529

530 **Author Information**

531 Corresponding Author: Rainer Lohmann – Graduate School of Oceanography, University of
532 Rhode Island, Narragansett Rhode Island. 02882, United States. [Orcid.org/0000-0001-8796-](https://orcid.org/0000-0001-8796-3229)
533 [3229](https://orcid.org/0000-0001-8796-3229). Phone: (401) 874-6612; email: rlohmann@uri.edu

534 Matthew Dunn – Graduate School of Oceanography, University of Rhode island, Narragansett,
535 Rhode Island 02882, United States; orcid.org/000-0002-8902-8434

536 Nicholas Noons – Office of Land Revitalization and Sustainable Materials Management, Rhode
537 Island Department of Environmental Management, Providence, Rhode Island 02908, United
538 States

539 Simon Vojta - Graduate School of Oceanography, University of Rhode Island, Narragansett,
540 Rhode Island 02882, United States;

541 Jitka Becanova – Graduate School of Oceanography, University of Rhode Island, Narragansett,
542 Rhode Island 02882, United States; orcid.org/0000-0002-3091-1054

543 Heidi Pickard – John A. Paulson School of Engineering and Applied Sciences, Harvard,
544 Cambridge, Massachusetts 02138, United States; orcid.org/0000-0001-8312-7522

545 Elsie M. Sunderland – John A. Paulson School of Engineering and Applied Sciences, Harvard,
546 Cambridge, Massachusetts 02138, United States; orcid.org/0000-0003-03869548

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555 **References**

- 556 (1) Buck, R. C.; Franklin, J.; Berger, U.; Conder, J. M.; Cousins, I. T.; Voogt, P. D.; Jensen, A. A.;
557 Kannan, K.; Mabury, S. A.; van Leeuwen, S. P. J. Perfluoroalkyl and Polyfluoroalkyl
558 Substances in the Environment: Terminology, Classification, and Origins. *Integr. Environ.*
559 *Assess. Manag.* **2011**, *7* (4), 513–541. <https://doi.org/10.1002/ieam.258>.
- 560 (2) Sunderland, E. M.; Hu, X. C.; Dassuncao, C.; Tokranov, A. K.; Wagner, C. C.; Allen, J. G. A
561 Review of the Pathways of Human Exposure to Poly- and Perfluoroalkyl Substances
562 (PFASs) and Present Understanding of Health Effects. *J. Expo. Sci. Environ. Epidemiol.* **2019**,
563 *29* (2), 131–147. <https://doi.org/10.1038/s41370-018-0094-1>.
- 564 (3) Munoz, G.; Mercier, L.; Duy, S. V.; Liu, J.; Sauvé, S.; Houde, M. Bioaccumulation and
565 Trophic Magnification of Emerging and Legacy Per- and Polyfluoroalkyl Substances (PFAS)

- 566 in a St. Lawrence River Food Web. *Environ. Pollut.* **2022**, 309.
567 <https://doi.org/10.1016/j.envpol.2022.119739>.
- 568 (4) Robuck, A. R.; Cantwell, M. G.; McCord, J. P.; Addison, L. M.; Pfohl, M.; Strynar, M. J.;
569 McKinney, R.; Katz, D. R.; Wiley, D. N.; Lohmann, R. Legacy and Novel Per- and
570 Polyfluoroalkyl Substances in Juvenile Seabirds from the U.S. Atlantic Coast. *Environ. Sci.*
571 *Technol.* **2020**, 54 (20), 12938–12948. <https://doi.org/10.1021/acs.est.0c01951>.
- 572 (5) De Silva, A. O.; Armitage, J. M.; Bruton, T. A.; Dassuncao, C.; Heiger-Bernays, W.; Hu, X. C.;
573 Kärrman, A.; Kelly, B.; Ng, C.; Robuck, A.; Sun, M.; Webster, T. F.; Sunderland, E. M. PFAS
574 Exposure Pathways for Humans and Wildlife: A Synthesis of Current Knowledge and Key
575 Gaps in Understanding. *Environ. Toxicol. Chem.* **2021**, 40 (3), 631–657.
576 <https://doi.org/10.1002/etc.4935>.
- 577 (6) Yamazaki, E.; Taniyasu, S.; Wang, X.; Yamashita, N. Per- and Polyfluoroalkyl Substances in
578 Surface Water, Gas and Particle in Open Ocean and Coastal Environment. *Chemosphere*
579 **2021**, 272, 129869. <https://doi.org/10.1016/j.chemosphere.2021.129869>.
- 580 (7) Taylor, M. D.; Johnson, D. D. Preliminary Investigation of Perfluoroalkyl Substances in
581 Exploited Fishes of Two Contaminated Estuaries. *Mar. Pollut. Bull.* **2016**, 111 (1–2), 509–
582 513. <https://doi.org/10.1016/j.marpolbul.2016.06.023>.
- 583 (8) Ojo, A. F.; Peng, C.; Ng, J. C. Assessing the Human Health Risks of Per- and Polyfluoroalkyl
584 Substances: A Need for Greater Focus on Their Interactions as Mixtures. *J. Hazard. Mater.*
585 **2021**, 407 (August 2020), 124863. <https://doi.org/10.1016/j.jhazmat.2020.124863>.
- 586 (9) Pelch, K. E.; Reade, A.; Wolffe, T. A. M.; Kwiatkowski, C. F. PFAS Health Effects Database:
587 Protocol for a Systematic Evidence Map. *Environ. Int.* **2019**, 130 (July).
588 <https://doi.org/10.1016/j.envint.2019.05.045>.
- 589 (10) Salvatore, D.; Mok, K.; Garrett, K. K.; Poudrier, G.; Brown, P.; Birnbaum, L. S.; Goldenman,
590 G.; Miller, M. F.; Patton, S.; Poehlein, M.; Varshavsky, J.; Corder, A. Contamination Based
591 on Likely Sources. **2022**. <https://doi.org/10.1021/acs.estlett.2c00502>.
- 592 (11) Katz, D. R.; Sullivan, J. C.; Rosa, K.; Gardiner, C. L.; Robuck, A. R.; Lohmann, R.; Kincaid, C.;
593 Cantwell, M. G. Transport and Fate of Aqueous Film Forming Foam in an Urban Estuary.
594 *Environ. Pollut.* **2022**, 300 (October 2021), 118963.
595 <https://doi.org/10.1016/j.envpol.2022.118963>.
- 596 (12) Cai, Y.; Wang, X.; Wu, Y.; Zhao, S.; Li, Y.; Ma, L.; Chen, C.; Huang, J.; Yu, G. Temporal Trends
597 and Transport of Perfluoroalkyl Substances (PFASs) in a Subtropical Estuary: Jiulong River
598 Estuary, Fujian, China. *Sci. Total Environ.* **2018**, 639, 263–270.
599 <https://doi.org/10.1016/j.scitotenv.2018.05.042>.
- 600 (13) Munoz, G.; Budzinski, H.; Babut, M.; Lobry, J.; Selleslagh, J.; Tapie, N.; Labadie, P. Temporal
601 Variations of Perfluoroalkyl Substances Partitioning between Surface Water, Suspended
602 Sediment, and Biota in a Macrotidal Estuary. *Chemosphere* **2019**, 233, 319–326.
603 <https://doi.org/10.1016/j.chemosphere.2019.05.281>.
- 604 (14) Munoz, G.; Labadie, P.; Botta, F.; Lestremau, F.; Lopez, B.; Geneste, E.; Pardon, P.; Dévier,
605 M. H.; Budzinski, H. Occurrence Survey and Spatial Distribution of Perfluoroalkyl and
606 Polyfluoroalkyl Surfactants in Groundwater, Surface Water, and Sediments from Tropical
607 Environments. *Sci. Total Environ.* **2017**, 607–608, 243–252.
608 <https://doi.org/10.1016/j.scitotenv.2017.06.146>.

- 609 (15) Munoz, G.; Budzinski, H.; Labadie, P. Influence of Environmental Factors on the Fate of
610 Legacy and Emerging Per- and Polyfluoroalkyl Substances along the Salinity/Turbidity
611 Gradient of a Macrotidal Estuary. *Environ. Sci. Technol.* **2017**, *51* (21), 12347–12357.
612 <https://doi.org/10.1021/acs.est.7b03626>.
- 613 (16) Zhang, X.; Lohmann, R.; Dassuncao, C.; Hu, X. C.; Weber, A. K.; Vecitis, C. D.; Sunderland, E.
614 M. Source Attribution of Poly- and Perfluoroalkyl Substances (PFASs) in Surface Waters
615 from Rhode Island and the New York Metropolitan Area. *Environ. Sci. Technol. Lett.* **2016**,
616 *3* (9), 316–321. <https://doi.org/10.1021/acs.estlett.6b00255>.
- 617 (17) Allinson, M.; Yamashita, N.; Taniyasu, S.; Yamazaki, E.; Allinson, G. Occurrence of
618 Perfluoroalkyl Substances in Selected Victorian Rivers and Estuaries: An Historical
619 Snapshot. *Heliyon* **2019**, *5* (9), e02472. <https://doi.org/10.1016/j.heliyon.2019.e02472>.
- 620 (18) Godlewska, K.; Stepnowski, P.; Paszkiewicz, M. *Pollutant Analysis Using Passive Samplers:
621 Principles, Sorbents, Calibration and Applications. A Review*; Springer International
622 Publishing, 2021; Vol. 19. <https://doi.org/10.1007/s10311-020-01079-6>.
- 623 (19) Li, Ying., Kaserzon, S.L., Baduel, C., Godlonton, C., & Mueller, J. F. A Passive Sampling Tool
624 for Time Integrative Monitoring Of Perfluoroalkyl Substances In Aqueous Environments.
625 [36th International Symposium on Halogenated Persistent Organic Pollutants \(Dioxin
626 2016\)](https://espace.library.uq.edu.au/view/UQ:696061) **2016**, 1–4. <https://espace.library.uq.edu.au/view/UQ:696061>
- 627 (20) Vrana, B.; Allan, I. J.; Greenwood, R.; Mills, G. A.; Dominiak, E.; Svensson, K.; Knutsson, J.;
628 Morrison, G. Passive Sampling Techniques for Monitoring Pollutants in Water. *TrAC -
629 Trends Anal. Chem.* **2005**, *24* (10), 845–868. <https://doi.org/10.1016/j.trac.2005.06.006>.
- 630 (21) Dunn, M.; Becanova, J.; Snook, J.; Ruyle, B.; Lohmann, R. Calibration of Perfluorinated Alkyl
631 Acid Uptake Rates by a Tube Passive Sampler in Water. *ACS EST Water* **2023**,
632 *acsestwater.2c00384*. <https://doi.org/10.1021/acsestwater.2c00384>.
- 633 (22) Gardiner, C.; Robuck, A.; Becanova, J.; Cantwell, M.; Kaserzon, S.; Katz, D.; Mueller, J.;
634 Lohmann, R. Field Validation of a Novel Passive Sampler for Dissolved PFAS in Surface
635 Waters. *Environ. Toxicol. Chem.* **2022**, *41* (10), 2375–2385.
636 <https://doi.org/10.1002/etc.5431>.
- 637 (23) Kaserzon, S. L.; Vijayasarathy, S.; Bräunig, J.; Mueller, L.; Hawker, D. W.; Thomas, K. V.;
638 Mueller, J. F. Calibration and Validation of a Novel Passive Sampling Device for the Time
639 Integrative Monitoring of Per- and Polyfluoroalkyl Substances (PFASs) and Precursors in
640 Contaminated Groundwater. *J. Hazard. Mater.* **2019**, *366* (September 2018), 423–431.
641 <https://doi.org/10.1016/j.jhazmat.2018.12.010>.
- 642 (24) Navarro, D. A.; Oliver, D. P.; Simpson, S. L.; Kookana, R. S. Organic Carbon and Salinity
643 Affect Desorption of PFAS from Estuarine Sediments. *J. Soils Sediments* **2022**, *22* (4), 1302–
644 1314. <https://doi.org/10.1007/s11368-022-03172-5>.
- 645 (25) Li, C.; Zhang, C.; Gibbes, B.; Wang, T.; Lockington, D. Coupling Effects of Tide and Salting-
646 out on Perfluorooctane Sulfonate (PFOS) Transport and Adsorption in a Coastal Aquifer.
647 *Adv. Water Resour.* **2022**, *166* (July 2021), 104240.
648 <https://doi.org/10.1016/j.advwatres.2022.104240>.
- 649 (26) Nguyen, M. A.; Norström, K.; Wiberg, K.; Gustavsson, J.; Josefsson, S.; Ahrens, L. Seasonal
650 Trends of Per- and Polyfluoroalkyl Substances in River Water Affected by Fire Training Sites
651 and Wastewater Treatment Plants. *Chemosphere* **2022**, *308*, 136467.
652 <https://doi.org/10.1016/j.chemosphere.2022.136467>.

- 653 (27) Wang, S.; Ma, L.; Chen, C.; Li, Y.; Wu, Y.; Liu, Y.; Dou, Z.; Yamazaki, E.; Yamashita, N.; Lin, B.
654 L.; Wang, X. Occurrence and Partitioning Behavior of Per- and Polyfluoroalkyl Substances
655 (PFASs) in Water and Sediment from the Jiulong Estuary-Xiamen Bay, China. *Chemosphere*
656 **2020**, *238*, 124578. <https://doi.org/10.1016/j.chemosphere.2019.124578>.
- 657 (28) Glüge, J.; Scheringer, M.; Cousins, I. T.; Dewitt, J. C.; Goldenman, G.; Herzke, D.; Lohmann,
658 R.; Ng, C. A.; Trier, X.; Wang, Z. An Overview of the Uses of Per- And Polyfluoroalkyl
659 Substances (PFAS). *Environ. Sci. Process. Impacts* **2020**, *22* (12), 2345–2373.
660 <https://doi.org/10.1039/d0em00291g>.
- 661 (29) Hill, P. J.; Taylor, M.; Goswami, P.; Blackburn, R. S. Substitution of PFAS Chemistry in
662 Outdoor Apparel and the Impact on Repellency Performance. *Chemosphere* **2017**, *181*,
663 500–507. <https://doi.org/10.1016/j.chemosphere.2017.04.122>.
- 664 (30) Peaslee, G. F.; Wilkinson, J. T.; McGuinness, S. R.; Tighe, M.; Caterisano, N.; Lee, S.;
665 Gonzales, A.; Roddy, M.; Mills, S.; Mitchell, K. Another Pathway for Firefighter Exposure to
666 Per- and Polyfluoroalkyl Substances: Firefighter Textiles. *Environ. Sci. Technol. Lett.* **2020**, *7*
667 (8), 594–599. <https://doi.org/10.1021/acs.estlett.0c00410>.
- 668 (31) Heydebreck, F.; Tang, J.; Xie, Z.; Ebinghaus, R. Emissions of Per- and Polyfluoroalkyl
669 Substances in a Textile Manufacturing Plant in China and Their Relevance for Workers'
670 Exposure. *Environ. Sci. Technol.* **2016**, *50* (19), 10386–10396.
671 <https://doi.org/10.1021/acs.est.6b03213>.
- 672 (32) Liu, S.; Jin, B.; Arp, H. P. H.; Chen, W.; Liu, Y.; Zhang, G. The Fate and Transport of
673 Chlorinated Polyfluorinated Ether Sulfonates and Other PFAS through Industrial
674 Wastewater Treatment Facilities in China. *Environ. Sci. Technol.* **2022**, *56* (5), 3002–3010.
675 <https://doi.org/10.1021/acs.est.1c04276>.
- 676 (33) Mumtaz, M.; Bao, Y.; Li, W.; Kong, L.; Huang, J.; Yu, G. Screening of Textile Finishing Agents
677 Available on the Chinese Market: An Important Source of per- and Polyfluoroalkyl
678 Substances to the Environment. *Front. Environ. Sci. Eng.* **2019**, *13* (5), 67.
679 <https://doi.org/10.1007/s11783-019-1145-0>.
- 680 (34) Ma, D.; Zhong, H.; Lv, J.; Wang, Y.; Jiang, G. Levels, Distributions, and Sources of Legacy
681 and Novel per- and Perfluoroalkyl Substances (PFAS) in the Topsoil of Tianjin, China. *J.*
682 *Environ. Sci.* **2022**, *112*, 71–81. <https://doi.org/10.1016/j.jes.2021.04.029>.
- 683 (35) Valsecchi, S.; Rusconi, M.; Mazzoni, M.; Viviano, G.; Pagnotta, R.; Zaghi, C.; Serrini, G.;
684 Polesello, S. Occurrence and Sources of Perfluoroalkyl Acids in Italian River Basins.
685 *Chemosphere* **2015**, *129*, 126–134. <https://doi.org/10.1016/j.chemosphere.2014.07.044>.
- 686 (36) EPA, USA. *Toxics Release Inventory*.
687 <https://edap.epa.gov/public/extensions/TRIToxicsTracker/TRIToxicsTracker.html#continue>
688 (accessed on 2022-11-04).
- 689 (37) EPA, USA. *Current Effluent Guidelines Program Plan*. Current Effluent Guidelines Program
690 Plan. <https://www.epa.gov/eg/current-effluent-guidelines-program-plan> (accessed 2023-
691 05-15).
- 692 (38) Robel, A. E.; Marshall, K.; Dickinson, M.; Lunderberg, D.; Butt, C.; Peaslee, G.; Stapleton, H.
693 M.; Field, J. A. Closing the Mass Balance on Fluorine on Papers and Textiles. *Environ. Sci.*
694 *Technol.* **2017**, *51* (16), 9022–9032. <https://doi.org/10.1021/acs.est.7b02080>.

- 695 (39) Lohmann, R.; Letcher, R. J. The Universe of Fluorinated Polymers and Polymeric
696 Substances and Potential Environmental Impacts and Concerns. *Curr. Opin. Green Sustain.*
697 *Chem.* **2023**, 100795. <https://doi.org/10.1016/j.cogsc.2023.100795>.
- 698 (40) OECD (2022), Synthesis Report on Understanding Side-Chain Fluorinated Polymers and
699 Their Life Cycle, OECD Series on Risk Management, No. 73, Environment, Health and
700 Safety, Environment Directorate, OECD.
- 701 (41) Schellenberger, S.; Liagkouridis, I.; Awad, R.; Khan, S.; Plassmann, M.; Peters, G.; Benskin,
702 J. P.; Cousins, I. T. An Outdoor Aging Study to Investigate the Release of Per- and
703 Polyfluoroalkyl Substances (PFAS) from Functional Textiles. *Environ. Sci. Technol.* **2022**, *56*
704 (6), 3471–3479. <https://doi.org/10.1021/acs.est.1c06812>.
- 705 (42) Ruyle, B. J.; Thackray, C. P.; McCord, J. P.; Strynar, M. J.; Mauge-Lewis, K. A.; Fenton, S. E.;
706 Sunderland, E. M. Reconstructing the Composition of Per- And Polyfluoroalkyl Substances
707 in Contemporary Aqueous Film-Forming Foams. *Environ. Sci. Technol. Lett.* **2020**.
708 <https://doi.org/10.1021/acs.estlett.0c00798>.
- 709 (43) Weber, A. K.; Barber, L. B.; LeBlanc, D. R.; Sunderland, E. M.; Vecitis, C. D. Geochemical and
710 Hydrologic Factors Controlling Subsurface Transport of Poly- and Perfluoroalkyl
711 Substances, Cape Cod, Massachusetts. *Environ. Sci. Technol.* **2017**, *51* (8), 4269–4279.
712 <https://doi.org/10.1021/acs.est.6b05573>.
- 713 (44) Balgooyen, S.; Remucal, C. K. Tributary Loading and Sediment Desorption as Sources of
714 PFAS to Receiving Waters. *ACS EST Water* **2022**, *2* (3), 436–445.
715 <https://doi.org/10.1021/acsestwater.1c00348>.
- 716 (45) Morales-McDevitt, M. E.; Dunn, M.; Habib, A.; Vojta, S.; Becanova, J.; Lohmann, R. Poly-
717 and Perfluorinated Alkyl Substances in Air and Water from Dhaka, Bangladesh. *Environ.*
718 *Toxicol. Chem.* **2022**, *41* (2), 334–342. <https://doi.org/10.1002/etc.5255>.
- 719 (46) Simcik, M. F.; Dorweiler, K. J. Ratio of Perfluorochemical Concentrations as a Tracer of
720 Atmospheric Deposition to Surface Waters. *Environ. Sci. Technol.* **2005**, *39* (22), 8678–
721 8683. <https://doi.org/10.1021/es0511218>.
- 722 (47) Nilsson, H.; Kärrman, A.; Rotander, A.; van Bavel, B.; Lindström, G.; Westberg, H.
723 Biotransformation of Fluorotelomer Compound to Perfluorocarboxylates in Humans.
724 *Environ. Int.* **2013**, *51* (2013), 8–12. <https://doi.org/10.1016/j.envint.2012.09.001>.
- 725 (48) Fauvelle, V.; Montero, N.; Mueller, J. F.; Banks, A.; Mazzella, N.; Kaserzon, S. L. Glyphosate
726 and AMPA Passive Sampling in Freshwater Using a Microporous Polyethylene Diffusion
727 Sampler. *Chemosphere* **2017**, *188*, 241–248.
728 <https://doi.org/10.1016/j.chemosphere.2017.08.013>.
- 729 (49) Weather Spark. Climate and Average Weather Year Round In Westerly.
730 [https://weatherspark.com/y/26174/Average-Weather-in-Westerly-Rhode-Island-United-](https://weatherspark.com/y/26174/Average-Weather-in-Westerly-Rhode-Island-United-States-Year-Round#Sections-Precipitation)
731 [States-Year-Round#Sections-Precipitation](https://weatherspark.com/y/26174/Average-Weather-in-Westerly-Rhode-Island-United-States-Year-Round#Sections-Precipitation) (accessed 2022-06-12).
- 732 (50) Becanova, J.; Saleeba, Z. S. S. L.; Stone, A.; Robuck, A. R.; Hurt, R. H.; Lohmann, R. A
733 Graphene-Based Hydrogel Monolith with Tailored Surface Chemistry for PFAS Passive
734 Sampling. *Environ. Sci. Nano* **2021**, *8* (10), 2894–2907.
735 <https://doi.org/10.1039/d1en00517k>.
- 736 (51) Schymanski, E. L.; Jeon, J.; Gulde, R.; Fenner, K.; Ruff, M.; Singer, H. P.; Hollender, J.
737 Identifying Small Molecules via High Resolution Mass Spectrometry: Communicating

- 738 Confidence. *Environ. Sci. Technol.* **2014**, *48* (4), 2097–2098.
739 <https://doi.org/10.1021/es5002105>.
- 740 (52) Charbonnet, J. A.; McDonough, C. A.; Xiao, F.; Schwichtenberg, T.; Cao, D.; Kaserzon, S.;
741 Thomas, K. V.; Dewapriya, P.; Place, B. J.; Schymanski, E. L.; Field, J. A.; Helbling, D. E.;
742 Higgins, C. P. Communicating Confidence of Per- and Polyfluoroalkyl Substance
743 Identification via High-Resolution Mass Spectrometry. *Environ. Sci. Technol. Lett.* **2022**, *9*
744 (6), 473–481. <https://doi.org/10.1021/acs.estlett.2c00206>.
- 745 (53) Ruyle, B. J.; Pickard, H. M.; LeBlanc, D. R.; Tokranov, A. K.; Thackray, C. P.; Hu, X. C.; Vecitis,
746 C. D.; Sunderland, E. M. Isolating the AFFF Signature in Coastal Watersheds Using
747 Oxidizable PFAS Precursors and Unexplained Organofluorine. *Environ. Sci. Technol.* **2021**,
748 *55* (6), 3686–3695. <https://doi.org/10.1021/acs.est.0c07296>.
- 749 (54) Carpenter, C. M. G.; Helbling, D. E. Widespread Micropollutant Monitoring in the Hudson
750 River Estuary Reveals Spatiotemporal Micropollutant Clusters and Their Sources. *Environ.*
751 *Sci. Technol.* **2018**, *52* (11), 6187–6196. <https://doi.org/10.1021/acs.est.8b00945>.
- 752 (55) MacGillivray, A. R. Temporal Trends of Per- and Polyfluoroalkyl Substances in Delaware
753 River Fish, USA. *Integr. Environ. Assess. Manag.* **2021**, *17* (2), 411–421.
754 <https://doi.org/10.1002/ieam.4342>.
- 755 (56) Beggs, C.; Mackie, R.; Vrana, B.; Prokeš, R.; Gorji, S. G.; Schulze, B.; Thomas, K. V.; Mueller,
756 J. F.; Kaserzon, S. L. Estimation of Per- and Poly-Fluoroalkyl Substances Mass Loads in the
757 Danube River Using Passive Sampling. *Sci. Total Environ.* **2023**, *892*, 164458.
758 <https://doi.org/10.1016/j.scitotenv.2023.164458>.
- 759 (57) Zhao, J.; Yang, L.; Yang, X.; Zhao, X.; Li, M.; Zhao, S.; Zhu, L.; Zhan, J. Degradation of 8:2
760 Fluorotelomer Carboxylic Acid (8:2 FTCA) by Plants and Their Co-Existing Microorganisms.
761 *J. Hazard. Mater.* **2023**, *451*, 131129. <https://doi.org/10.1016/j.jhazmat.2023.131129>.
- 762 (58) Morales-McDevitt, M. E.; Becanova, J.; Blum, A.; Bruton, T. A.; Vojta, S.; Woodward, M.;
763 Lohmann, R. The Air That We Breathe: Neutral and Volatile PFAS in Indoor Air. *Environ. Sci.*
764 *Technol. Lett.* **2021**. <https://doi.org/10.1021/acs.estlett.1c00481>.
- 765 (59) Chen, D.; Zhao, Y.; Xu, W.; Pan, Y.; Wei, Q.; Xie, S. Biotransformation and Tissue
766 Bioaccumulation of 8:2 Fluorotelomer Alcohol in Broiler by Oral Exposure. *Environ. Pollut.*
767 **2020**, *267*, 115611. <https://doi.org/10.1016/j.envpol.2020.115611>.
- 768 (60) George, S. E.; Baker, T. R.; Baker, B. B. Nonlethal Detection of PFAS Bioaccumulation and
769 Biomagnification within Fishes in an Urban- and Wastewater-Dominant Great Lakes
770 Watershed. *Environ. Pollut.* **2023**, *321*, 121123.
771 <https://doi.org/10.1016/j.envpol.2023.121123>.
- 772 (61) Campos Pereira, H.; Ullberg, M.; Kleja, D. B.; Gustafsson, J. P.; Ahrens, L. Sorption of
773 Perfluoroalkyl Substances (PFASs) to an Organic Soil Horizon – Effect of Cation
774 Composition and pH. *Chemosphere* **2018**, *207*, 183–191.
775 <https://doi.org/10.1016/j.chemosphere.2018.05.012>.
- 776 (62) Munoz, G.; Budzinski, H.; Babut, M.; Drouineau, H.; Lauzent, M.; Menach, K. L.; Lobry, J.;
777 Selleslagh, J.; Simonnet-Laprade, C.; Labadie, P. Evidence for the Trophic Transfer of
778 Perfluoroalkylated Substances in a Temperate Macrotidal Estuary. *Environ. Sci. Technol.*
779 **2017**, *51* (15), 8450–8459. <https://doi.org/10.1021/acs.est.7b02399>.
- 780 (63) Bent, G.C., Zarriello, P.J., Granato, G.E., Masterson, J.P., Walter, D.A., Waite, A.M., and
781 Church, P.E., 2011, Simulated effects of water withdrawals and land-use changes on

782 streamflows and groundwater levels in the Pawcatuck River Basin, southwestern Rhode
783 Island and southeastern Connecticut: U.S. Geological Survey Scientific Investigations
784 Report 2009–5127, 254 p., available at <http://pubs.usgs.gov/sir/2009/5127>.

785 (64) Benskin, J. P.; Muir, D. C. G.; Scott, B. F.; Spencer, C.; De Silva, A. O.; Kylin, H.; Martin, J. W.;
786 Morris, A.; Lohmann, R.; Tomy, G.; Rosenberg, B.; Taniyasu, S.; Yamashita, N.
787 Perfluoroalkyl Acids in the Atlantic and Canadian Arctic Oceans. *Environ. Sci. Technol.*
788 **2012**, *46* (11), 5815–5823. <https://doi.org/10.1021/es300578x>.

789 (65) Zhang, X.; Lohmann, R.; Sunderland, E. M. Poly- and Perfluoroalkyl Substances in Seawater
790 and Plankton from the Northwestern Atlantic Margin. *Environ. Sci. Technol.* **2019**, *53* (21),
791 12348–12356. <https://doi.org/10.1021/acs.est.9b03230>.

792 (66) Zhou, J.; Baumann, K.; Chang, N.; Morrison, G.; Bodnar, W.; Zhang, Z.; Atkin, J. M.; Surratt,
793 J. D.; Turpin, B. J. Per- and Polyfluoroalkyl Substances (PFASs) in Airborne Particulate
794 Matter (PM_{2.0}) Emitted during Floor Waxing: A Pilot Study. *Atmos. Environ.* **2022**, *268*,
795 118845. <https://doi.org/10.1016/j.atmosenv.2021.118845>.

796 (67) Lupo, C. D.; McCutcheon, C. M.; Kenner, S. J. *Pawcatuck River Watershed Final HSPF*
797 *Modeling Report*; Connecticut Department of Energy and Environmental Protection, 2022;
798 p 119. [https://portal.ct.gov/-/media/DEEP/water/tmdl/Pawcatuck-Nutrient-TMDL/Final-](https://portal.ct.gov/-/media/DEEP/water/tmdl/Pawcatuck-Nutrient-TMDL/Final-Pawcatuck-River-Watershed-HSPF-Modeling-Report.pdf)
799 [Pawcatuck-River-Watershed-HSPF-Modeling-Report.pdf](https://portal.ct.gov/-/media/DEEP/water/tmdl/Pawcatuck-Nutrient-TMDL/Final-Pawcatuck-River-Watershed-HSPF-Modeling-Report.pdf) (accessed on 2023/01/23).

800 (68) Schwarzenbach, R.; Gschwend, P.; Imboden, D. *Environmental Organic Chemistry*, 3rd
801 Edition.; John Wiley & Sons, Inc., 2017.
802