1 **Constraints and Drivers of Dissolved Fluxes of Pyrogenic Carbon in Soil and Freshwater Systems: a**
2 **Global Review and Meta-analysis**

3

4

5 **Authors: R. B. Abney[1], M. E. Barnes[2], A. Moss[1], and F. Santos[3]**

6

7 [1]Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA.
8 [2]Biological Sciences Division, Pacific Northwest National Laboratory, Richland, WA.
9 [3]Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN.

10

11

12

13 Corresponding author: Rebecca Abney (Rebecca.Abney@uga.edu)

14

15

16

17 **Key Points:**

18 - Burn intensity of pyrogenic carbon (PyC) controls its dissolution and aqueous transport.

19 - From headwater streams to oceans, dissolved PyC increases in radiocarbon age, implying
20   environmental persistence.

21 - Environmental controls, including climate, topography, biota, parent material, and time
22   impact the pool size and flux of dissolved PyC.

23

24

Abstract

Pyrogenic carbon (PyC) is a significant component of the global soil carbon pool due to its longer environmental persistence than other soil organic matter components. Despite PyC's persistence in soil, recent work has indicated that it is susceptible to loss processes such as mineralization and leaching, with the significance and magnitude of these largely unknown at the hillslope and watershed scale. We present a review of the work concerning dissolved PyC transport in soil and freshwater. Our analysis found the primary environmental controls on dissolved PyC (dPyC) transport are the formation conditions and quality of the PyC itself, with longer and higher temperature charring conditions leading to less transport of dPyC. While correlations between dPyC and dissolved organic carbon in rivers and other pools are frequently reported, the slope of these correlations was pool-dependent (i.e., soil-water, precipitation, lakes, streams, rivers), suggesting site-specific environmental controls. However, the lack of consistency in analytical techniques and sample preparation remains a major challenge to quantifying environmental controls on dPyC fluxes. We propose that future research should focus on the following: (1) consistency in methodological approaches, (2) more quantitative measures of dPyC in pools and fluxes from soils to streams, (3) turnover times of dPyC in soils and aquatic systems, and (4) improved understanding of how mechanisms controlling the fate of dPyC in dynamic post-fire landscapes interact. With more refined quantitative information about the controls on dPyC transport at the hillslope and landscape scale, we can increase the accuracy and utility of global carbon models.

1. Introduction

1.1 Significance of dissolved pyrogenic carbon in the global C cycle

Wildfire is a a major ecosystem disturbance that has been progressively getting more severe and intense in many areas of the world (Descals et al., 2022; Feurdean et al., 2020; Turco et al., 2018; Westerling et al., 2006). Additionally, significant increases in the use of prescribed fire have been observed across the US in the past few decades, especially in the southeast (Kolden, 2019). Natural and anthropogenic fires transform carbon (C) into a range of materials across the burn continuum including lightly charred material to charcoal, soot, and ash, collectively known as pyrogenic carbon, or PyC (Bird et al., 2015; Masiello, 2004). This PyC is initially deposited on the soil surface and emitted to the atmosphere. Pyrogenic C has received considerable research attention over the past decades due to its generally longer persistence times in soil relative to other soil organic matter components, which has implications for C storage (Abney and Berhe, 2018; Bird et al., 2015; Santín et al., 2016).

There has been limited research synthesizing the production, pools, and fluxes of PyC across large spatial scales (Coppola et al., 2022; Wagner et al., 2018) to better constrain the role of PyC in the global C cycle. It has been estimated that approximately 2.7% of the global land area is burned annually (Giglio et al., 2013), resulting in 50-270 Tg y^{-1} of PyC produced (Kuhlbusch and Crutzen, 1995). At a global scale, PyC can account for a significant portion (~10%) of dissolved organic carbon (DOC) fluxes in rivers (Jaffé et al., 2013; Wagner et al., 2018). The global annual transport of particulate PyC from rivers into oceans ranges from 19-80 Tg/yr (Druffel, 2004; Elmquist et al., 2008; Jaffé et al., 2013), although this is most frequently reported as 26-27 Tg/yr (Elmquist et al., 2008; Jaffé et al., 2013). However, current estimates of dPyC

69   export from terrestrial to inland aquatic systems range over several orders of magnitude, from
70   ~0.4-18 Tg/yr (Bowring et al., 2022; Jones et al., 2020), likely reflecting both the variability in
71   processes (e.g., decomposition, photooxidation, water availability, etc.) influencing PyC
72   dynamics at the terrestrial- aquatic interface and insufficient PyC measurements.

74   While parameterization of global PyC models typically includes annual production, stocks,
75   large-scale fluxes, and some smaller-scale controls (i.e., decomposition (Bird et al., 2015;
76   Reisser et al., 2016; Santín et al., 2016), several critical knowledge gaps remain regarding what
77   controls those fluxes, especially at smaller scales. In particular, we need a more detailed
78   understanding of how PyC is transported from source burned material into the soil system, across
79   hillslopes and watersheds, and into aquatic environments (Myers-Pigg et al., 2015; Wagner et al.,
80   2019c; Wagner et al., 2015b). Some recent research has demonstrated inconsistencies between
81   the sources and ages of dPyC flowing from rivers and those from PyC measured in the ocean
82   (Wagner et al., 2019b), which indicates limitations in our mechanistic understanding of (1) the
83   cycling of dPyC within terrestrial systems, (2) cycling within freshwater aquatic systems, and (3)
84   connections between terrestrial and aquatic systems. This paper addresses this limitation by
85   synthesizing our current knowledge of dPyC fluxes and the processes that control its
86   transformation within and between terrestrial and aquatic ecosystems to inform research
87   communities involved in management, modeling, and description of PyC.

1.2 Aims

91   The aim of this paper is two-fold. The first aim is to conduct a meta-analysis of available primary
92   literature on dPyC to synthesize the state of knowledge of pools and fluxes in soils and
93   freshwater systems. We use the meta-analysis to quantify methodological differences in
94   measurements of dPyC and examine whether they confound our understanding of reported dPyC
95   fluxes. We also assess the interactive effects of precursor material, burn conditions, and
96   environmental factors on the magnitude and quality of reported dPyC fluxes.

98   The second aim is to synthesize known and unknown processes that affect the aqueous transport
99   and composition of dPyC across the terrestrial to aquatic interface. We conclude this paper by
100   identifying gaps in our understanding of processes that drive dPyC retention and mobility in soils
101   and freshwater. By providing a more quantitative understanding of dPyC in soil and freshwater
102   carbon cycles, this work should provide useful information for researchers, land managers, and
103   for land models that handle fire disturbances used to predict future climates.

1.3 Defining dPyC

106   Pyrogenic C is a continuum of materials that range from lightly charred organic matter to highly
107   charred and condensed organic matter (Bird et al., 2015; Masiello, 2004; Santín et al., 2016).
108   Therefore, dPyC also exists on such a continuum (Wagner et al., 2017). In a review paper on the
109   PyC cycle, Bird et al. (2015) define dPyC as the "thermochemically altered carbon present in the
110   environment as a component of dissolved organic carbon in the <0.45 µm fraction of natural
111   waters." However, current methodological approaches restrict measurement of PyC or dPyC to
112   only subsections of the continuum (Wiedemeier et al., 2015). Thus, by selecting a single method,
113   most studies are operationally defining PyC or dPyC. Previous studies have measured dPyC
114   using filter sizes ranging from 0.1 um to 0.7 um (see section **3.2.2**, **Figure S2**), and while much

115 of environmental research considers materials <0.45 µm to be dissolved, these materials may in
116 fact be very small colloidal organic matter. Benzene, which makes up the backbone of many PyC
117 molecules, is only slightly soluble in water, but it is small enough that it would easily pass
118 through a 0.45 µm filter (diameter of benzene ring is 2.49 A or approximately 0.000249 µm). Of
119 course, most PyC does not exist as single benzene rings but is comprised of large, complex
120 macromolecules with some soluble functional groups (Bird et al., 2015), and much of what is
121 often described as dPyC are chemical fragments and colloids (Wagner et al., 2017). In this work,
122 we are using "dPyC" to indicate both fully dissolved and colloidal PyC within an aqueous matrix
123 due to inconsistencies in analytical approaches used for quantifying PyC pools and fluxes.
124

2. Methods

2.1 Data acquisition

128 Publications were selected for the meta-analysis following a modification of the PSALSAR
129 (research protocol and reporting results) method adapted by Mengist et al. (2020) from the
130 SALSA (search, appraisal, synthesis, and analysis) method (Grant and Booth, 2009), which
131 outlines organized methods for systematic literature reviews. Complete search and selection
132 criteria can be found in **Table S1**. The scope of the meta-analysis was defined as original
133 research papers that either quantified dPyC or characterized its chemical composition within
134 terrestrial or freshwater systems, including in-lab simulations and extractions. The search for
135 publications was initiated on August 23, 2021, via Web of Science and returned 12,776
136 publications, which we narrowed to 175 final publications via three successive rounds of filtering
137 (**Figure S1).**

2.2 Data processing and analysis

2.2.1 Extraction of dPyC values and other relevant information

142 Data were extracted from the 175 meta-analysis publications (**Figure 1** and **Table S1**) and
143 organized in Microsoft Excel as study- and observation-level information, where the study-level
144 information was applicable to the entire publication (e.g., publication year) and observation-level
145 information was specific to individual data points (e.g., pyrolysis temperature for a single PyC
146 type). Observations were included when they were reported concentrations of dPyC or
147 concentrations of DOC when only PyC (charcoal, biochar) was extracted in the laboratory. Data
148 which were presented only in figures (e.g., a bar graph of dPyC concentration means) were
149 extracted using Web Plot Digitizer (Rohatgi, 2022).
150

2.2.2 Assignment of location and climate regions to dPyC field observations

153 Latitude and longitude were assigned to each field study observation using provided coordinates,
154 map figures (using Web Plot Digitizer as described earlier), or via Google Earth using
155 descriptions in the manuscript text. Climate assignments were made by entering coordinates into
156 ClimateCharts.net (Laura Zepner, 2020), a web-based application which generates climate
157 classifications based on the Geiger-Köppen system. These assignments may contain some error
158 at locations where regional or local scale climate variation exists at too fine a resolution to be

159 captured by the raster cell size from ClimateCharts.net or within the Geiger-Köppen
160 classification system itself. We further binned the climate assignments into broader classes, such
161 as temperate, tropical, polar, etc.

162

163 The map figure was generated in R version 4.2.1 using the geom_map() function in the ggplot2
164 package. To reduce noise in the map, observations were binned by river basin using ArcMap
165 from ArcGIS Desktop 10.8.1(2020) with the World Wildlife Fund hydroSHEDS level 5 basins
166 dataset (Lehner and Grill, 2013). Detailed workflows for processing of spatial data are provided
167 in the Supplemental Materials.

168

169 **2.2.4 Statistical analyses**
170 We subset the data to analyze field and laboratory studies separately, and for field studies we
171 only used dPyC observations measured by the benzene polycarboxylic acid (BPCA) method to
172 maintain comparability across studies, and because this method comprised ~71% of the field
173 observations. Data were found not normal using the Shapiro-Wilk test and were analyzed using
174 non-parametric tests: the Kruskal-Wallis rank sum test and pairwise comparisons using the
175 Wilcoxon rank sum test. For field studies dissolved pyrogenic carbon was the dependent variable
176 and sample type (Flowing Water, Still Water, Precipitation, Soil Field Extract, and Throughflow
177 and Stemflow), methodology, filter size, and climate bin (Cold, Polar, Temperate, and Tropical)
178 were independent variables. For laboratory studies, we used observations across all
179 methodologies to increase our sample size and because no method was employed to produce a
180 majority of the observations. For some continuous variables in the laboratory observations
181 (charring temperature, time, extraction ratios), we used linear regression to describe their
182 relationship with dPyC. Significance was determined at the $p < 0.05$ level.

183

184 **3. Results**
185 **3.1 Overview of dataset**
186 Here we provide an overview of the meta-analysis based on 175 studies on dPyC, which yielded
187 2084 individual observations of dPyC (**Table 1**). The majority of observations (66%) were from
188 the field setting, and they were reported across five different non-convertible units. We analyzed
189 the data in light of methodological differences (Section 3.2), field measurements (Section 3.3),
190 and laboratory-based measurements (Section 3.4) to quantify the relative magnitude of different
191 dPyC pools and fluxes across the terrestrial to aquatic interface and to identify important
192 mechanisms driving these fluxes.

193

194 The most frequently published journals were Science of the Total Environment (n = 20);
195 Environmental Science and Technology (n = 17); Chemosphere (n = 13); Journal of Hazardous
196 Materials (n = 9); Organic Geochemistry (n = 8); Global Biogeochemical Cycles (n = 6);
197 Biogeochemistry (n = 5); Frontiers in Earth Science (n = 5); and Water Research (n = 5); The
198 remainder had fewer than five studies included in this meta-analysis. This wide range in scope of
199 journal topics reflects our broad search aims to connect terrestrial and aquatic research on dPyC.

200

201 Of the studies selected for this meta-analysis, only 12 measured soil, sediment, or other PyC
202 source concentration in combination with dPyC. Many of these reported positive correlations

between dPyC with DOC and source PyC concentration (Bi et al., 2018; Güereña et al., 2015; Jones et al., 2019). Contrastingly, several studies reported a reduction in DOC export after biochar or charcoal amendment (e.g.(Abdelrahman et al., 2018; Braun et al., 2020; Zand and Grathwohl, 2016), suggesting initial sorption of OM to the added PyC surface. Furthermore, many of these papers noted or measured variability in time post-fire or post-addition of charcoal as factors in leaching of PyC (e.g.(Bi et al., 2018; Jones et al., 2019; Santos et al., 2017), highlighting the need for investigation on the controls on the timing of post-fire dPyC fluxes.

Table 1. Number of dPyC measurements (n) across all methods that were extracted in this meta-analysis. A measurement of dPyC was considered whenever the observation was associated with a quantitative measurement of dPyC concentration. Dissolved organic C was considered a measurement of dPyC when charcoal only was extracted in the laboratory setting. Proportion of dPyC in DOC is denoted as %.

Sample Type	Individual Observations (n)	dPyC mg L^{-1} (n)	dPyC mg kg^{-1} (n)	DOC mg L^{-1} (n)	DOC mg kg^{-1} (n)	dPyC % (n)
Field	1389	1300	64	NA	NA	25
Lab	695	67	121	302	159	46
Total	2084	1367	185	302	159	71

3.2. The influence of methodological approaches on dPyC measurements
3.2.1 Detection and measurement of dissolved PyC

A major obstacle to closing the gaps in the global PyC cycle is associated with the methodological difficulty in measuring PyC, because there are a broad suite of methods used to quantify PyC and dPyC, and each method only captures a small portion of the PyC continuum (Bird et al., 2015; Masiello, 2004; Wiedemeier et al., 2015). As such, a suite of analytical methods was used by the studies selected for our meta-analysis. Many studies utilized qualitative methods for identification of the chemical composition of PyC in solution, including ultrahigh resolution mass spectrometry, Nuclear Magnetic Resonance spectroscopy (NMR), X-ray photoelectron spectroscopy (XPS), and X-ray absorption spectroscopy (XAS). For additional details on how these methods can be utilized for dPyC, see (Chen et al., 2022; Hameed et al., 2023; Wagner et al., 2018). Quantitative methods provide a measurement of PyC concentration in solution, which we further investigated in our meta-analysis, and included molecular markers (Benzene Polycarboxylic Acid or BPCA, levoglucosan, and polycyclic aromatic hydrocarbons or PAHs), chemo-thermal oxidation, as well as analytical and hydrogen pyrolysis, all of which have been extensively discussed elsewhere (Ascough et al., 2009; Gustafsson et al., 2001; Hameed et al., 2023; Hammes et al., 2007; Schmidt et al., 2001; Wagner et al., 2018). In 3.2.2, we examined the most common methods applied for the chemical characterization and quantification of dPyC in field and laboratory studies.

3.2.2 The impact of methodological approach on measured dPyC concentration

240　　To assess how the use of different methodological approaches may influence dPyC values, we
241　　examined concentrations (in mg L^{-1} or mg kg^{-1}) across field- and lab-based observations (**Figure**
242　　**2**). Field measurements of dPyC ranged over six orders of magnitude, and the lowest reported
243　　concentrations were in some of the methods that describe the narrowest section of the PyC
244　　continuum, namely GC-MS for PAHs and levoglucosan. BPCA is a widely used method across
245　　many sample types and this is reflected in the concentrations reported from this method.
246　　Approximately 50% of all data points were derived from the BPCA method, the majority of
247　　which were from field-based studies. The majority of field observations were obtained from the
248　　BPCA method (71%), while the majority of laboratory observations were made by quantifying
249　　total DOC from PyC (as charcoal, biochar, etc. extractions), followed by GC-MS techniques.
250　　The GC-MS techniques were also common in field studies (16% of observations). The vast
251　　majority (>80%) of laboratory studies quantified dPyC on char extracts, and the remainder
252　　quantified dPyC on soil extracts.
253
254　　Most of the samples were filtered prior to analysis, with approximately a third of samples (n =
255　　742) filtered using a 0.7 μm filter size, followed by 664 samples filtered with a 0.45 μm filter.
256　　Laboratory studies most frequently used the 0.45 μm filter (n = 412), whereas field-based studies
257　　most frequently used the 0.7 μm filter size (n = 670).
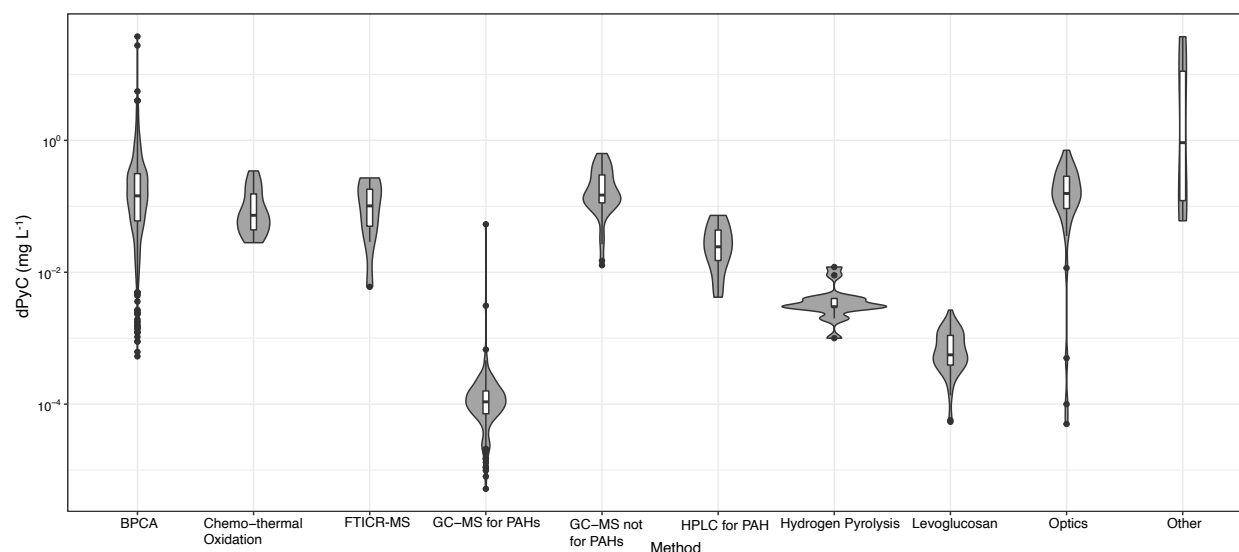258

259
260　　**Figure 2.** Field measurements of dPyC across different methodologies (n.b. Y axis is in log
261　　scale). Methods included in this figure are: BPCA (benzene polycarboxylic acid), Chemo-thermo
262　　oxidation, FTICR-MS (Fourier-transform ion cyclotron resonance mass spectrometry), GC-MS
263　　(gas chromatography mass spectrometry) for PAHs (polycyclic aromatic hydrocarbons), HPLC
264　　(High-performance liquid chromatography) for PAHs, hydrogen pyrolysis, levoglucosan, optics
265　　(includes fluorescence excitation-emission matrix-parallel factor analysis and other optical
266　　techniques), and other (includes liquid-chromatography for organic carbon detection and
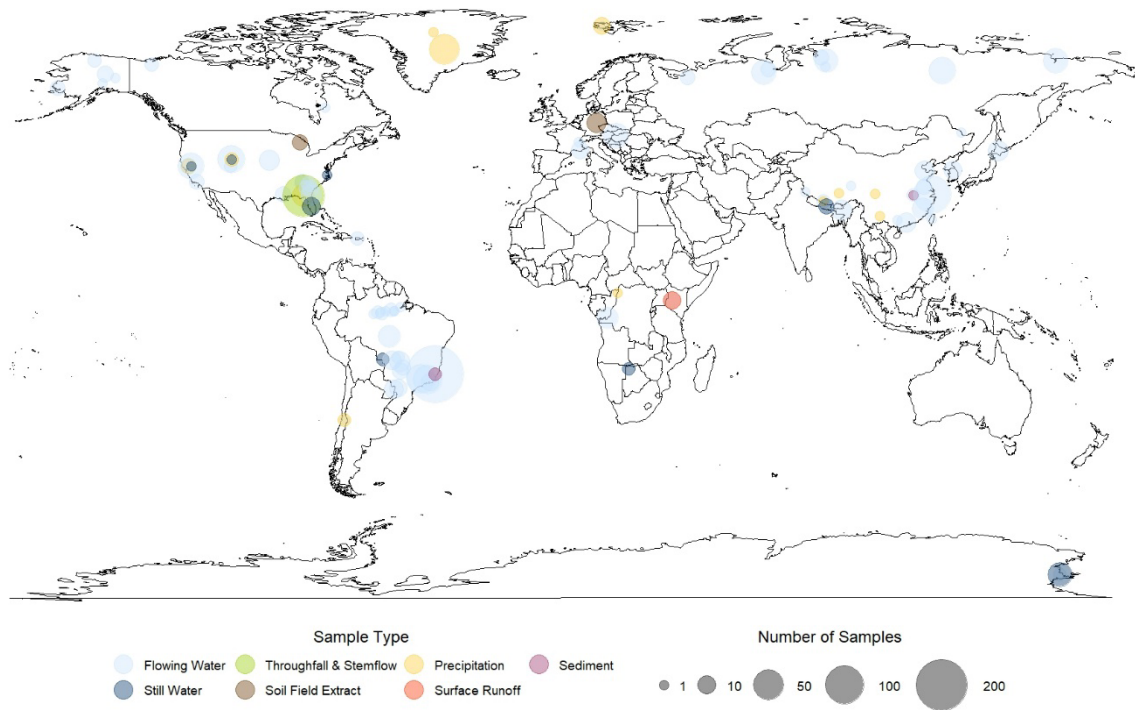267　　dichromate sulfuric acid oxidation).
268
269
270
271

Figure 3. Field-based measurements of dPyC across all methods with color indicating sample type and size of point denoting the number of observations.

3.3 Field-based measurements

Field-based observations of dPyC were distributed globally, including six of seven continents (excluding Australia). We extracted dPyC observations from several locations, including central and western Asia, eastern Europe, central America, and northern Africa (**Figure 3**). The majority of field-based observations of dPyC using the BPCA method were made in flowing water (n= 734; 78%), with only a few that were soil related (n = 5; 0.5%). There were significant differences in dPyC concentration across different sample types (p < 0.05; $\chi2$ = 109.42, df = 4; Kruskal-Wallis, **Figure 4A**). A pairwise Wilcoxon rank sum test indicated that each sample type was significantly different from each other (p < 0.05), except for soil field extract and throughflow and stemflow. The highest concentrations of dPyC in the field were found in water that had more time to interact with PyC (e.g., still water, soil pore water, throughfall and stemflow vs. precipitation and flowing water (**Figure 4A**). Additionally, there were significant differences in dPyC concentration across different climate zones (p < 0.05; $\chi2$ = 78.434, df = 3; Kruskal-Wallis; **Figure 4B**). This likely reflects differences in burn season length, vegetation available for burning, and mean air temperature and availability of water for dissolution.
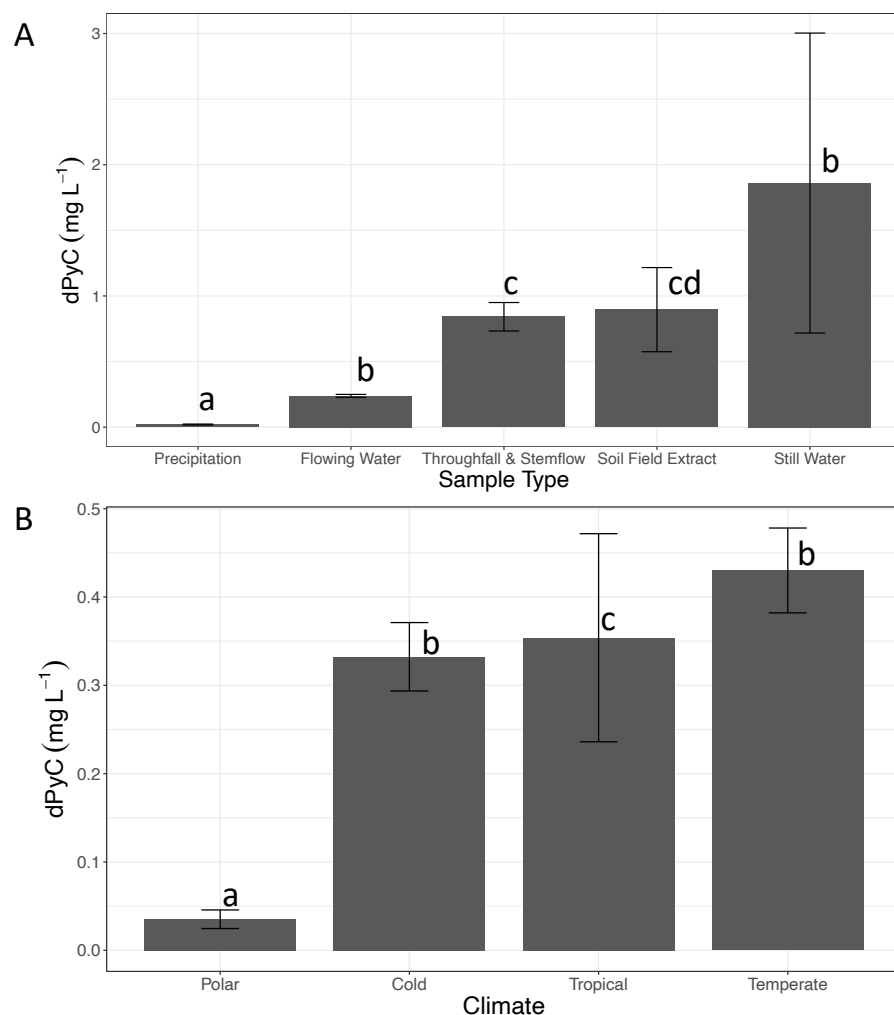
Figure 4A and B. Mean concentration with standard error of dPyC across different sample types (A) and climate zones (B) observed in the field derived from the BPCA method. Letters denote significant differences based on Wilcox rank sum test.

Correlation of dPyC with DOC
In field-based studies across all sample types, dPyC comprised approximately 5% of DOC. In sample types where water has more contact time with PyC (i.e., soil water, still water, see **Figure 4A**), dPyC makes up a greater proportion of DOC. In many of the flowing water studies the dPyC was correlated with discharge rates ([Dittmar et al., 2012](#); [Drake et al., 2020](#)), as is DOC, but some found a dilution effect of higher precipitation with lower dPyC concentrations under certain hydrologic conditions ([Bao et al., 2019](#); [Jones et al., 2019](#)). Together, this suggests that the mobility of dPyC is controlled by many of the same processes as DOC; hence the positive correlations between dPyC and DOC (**Figure 5**). However, studies that incorporate measurements of urban and anthropogenic sources of dPyC have noted a decoupling of dPyC from DOC where there are higher combustion activities ([Bi et al., 2018](#)).
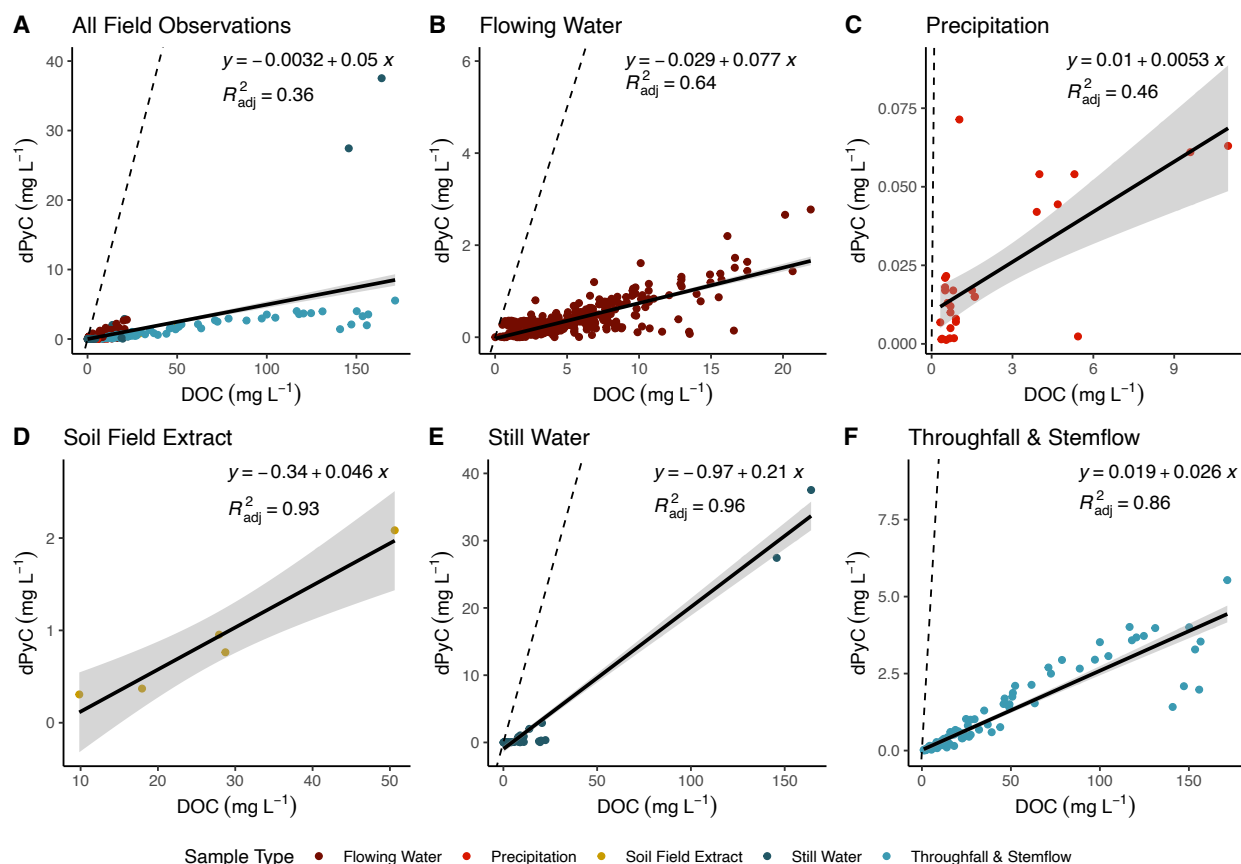
Figure 5. Dissolved PyC as measured by the BPCA method was correlated with DOC in field data from all sample types (panel A) and separated by different sample types (panels B-F). Solid black lines indicate linear fit with grey shaded area indicating 95% CI. The dashed line represents a 1:1 ratio of dPyC to DOC.

3.4 Laboratory-based measurements

For laboratory studies, we considered measurements of DOC from char-extracts as a measure of dPyC since the source material would be classified as PyC. This was the most frequently used method in laboratory studies that measured dPyC, followed by the BPCA method. The highest concentrations of dPyC measured in the lab were with the DOC method, which generated values approximately 1-2 orders of magnitude higher than the next highest concentrations, which were determined via the BPCA method. We found two categories of sample types in laboratory studies – soil extracts and charcoal extracts (n = 35 and 86, respectively). The soil extract had a not significantly lower concentration of dPyC compared with charcoal extracts (in units dPyC mg/kg; p = 0.11; $\chi 2$ = 2.5347; df = 1).

There were significant differences in dPyC concentrations across different filter sizes (**Figure S2**), but these did not increase with filter pore size, therefore these differences were more likely due to differing methodological approaches and/or sample characteristics. For laboratory studies, the BPCA method was most frequently used (n = 69 observations), followed by GC-MS for PAHs (n = 46). There were significant differences in the dPyC concentration based on the analytical method used to quantify dPyC (p < 0.05; $\chi 2$ = 40.268; df = 3), with BPCA having

336 significantly higher concentrations than GC-MS and GC-MS specifically applied for PAHs
337 (**Figure S3**).
338
339 **3.4.1 Inferences from char extracts**
340
341 We also examined drivers of dPyC by analyzing data from charcoal extractions, with feedstocks
342 categorized as woody or herbaceous. There were no significant differences in dPyC
343 concentration based upon feedstock type (woody vs herbaceous) alone ($\chi2 = 0.011081$; df = 1; p
344 > 0.05) or on extraction time ($\chi2 = 5.1529$, df = 5; p > 0.05). Extraction by shaking released
345 significantly more dPyC than other methods ($\chi2 = 22.512$; df = 1; p < 0.05). There was a
346 significant negative correlation between the solid to solution ratio (p<0.05; F=5.648; df = 65),
347 which reflects more PyC available for dissolution, although this only explained a small portion
348 of the variability in this data (Adjusted $R^2 = 0.068$). For extraction solvents, organic solvents
349 resulted in significantly lower concentrations of dPyC than water or other solvents ($\chi2 = 33.883$;
350 df = 2; p<0.05). We did find significant negative relationships between dPyC and burn time (F =
351 10; df = 18; p < 0.05) and maximum charring temperature (F = 8.57; df = 40; p < 0.05). This is
352 consistent with freshly produced charcoal at high charring intensities being some of the least
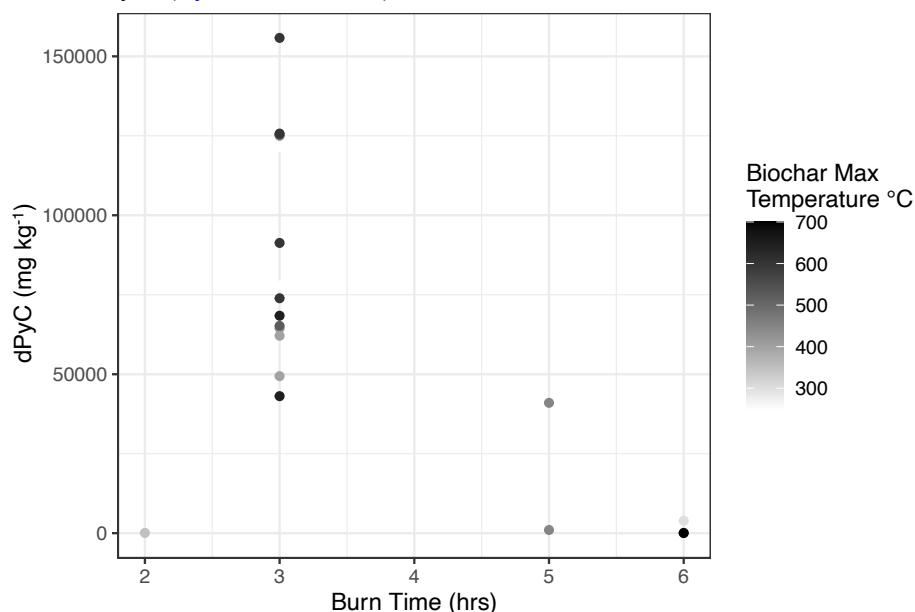353 soluble PyC ([Pyle et al., 2015](#)).

354
355 **Figure 6**. Concentration of dPyC across a range of burn time (hours) from charcoal extractions.
356

357 **4. Synthesis: factors affecting the fate of dPyC in terrestrial ecosystems**
358
359 Through this meta-analysis we have identified some key factors controlling the observed
360 magnitudes of dPyC fluxes reported by previous studies, including formation properties of PyC
361 and the method used to quantify dPyC. We found that methods that quantify a smaller
362 component of the dPyC continuum report lower concentrations of dPyC, and we found that
363 higher solid to solution ratios resulted in higher dPyC concentrations. We found that higher
364 charring time and temperature produced PyC that yielded lower amounts of dPyC (**Figure 6**).

365 These results make intuitive sense in terms of mechanistic controls on decomposition and
366 already described properties of the solubility of PyC.
367
368 In the following sections we synthesize the controls on pools and fluxes of dPyC along the
369 terrestrial to aquatic continuum. We will explore the main chemical and physical properties of
370 dPyC that impact its transport, discuss the role of fire in changing soil and landscape properties
371 that guide dPyC movement through terrestrial to aquatic systems, and finally describe how the
372 soil forming factors can be applied as a model for understanding the relevant environmental
373 controls on dPyC formation, transformation, and loss in watersheds.
374

4.1 Properties of PyC and dPyC that impact persistence and cycling in soil

376
377 Once PyC is deposited on the soil surface after a fire, its physical and chemical properties dictate
378 how it may be transformed by the many processes that control its cycling and long-term fate.
379 Earlier work on PyC suggested that it persists in soil on the millennial time scale; however recent
380 work indicates that it has a centennial mean residence time in soil (Bird et al., 2015; Dittmar,
381 2015; Wagner et al., 2015a). The persistence of PyC in soil is typically longer when it is formed
382 at higher temperatures and/or longer burning durations, and there is evidence that when formed
383 at lower temperatures (<250 ºC), it may breakdown over relatively shorter time scales (6 months;
384 (Abney et al., 2019a; Whitman et al., 2015). Santos et al. (2012) showed that this early
385 breakdown can occur even when PyC is formed in the laboratory at relatively higher controlled
386 temperatures (450 ºC). This more rapid breakdown has been mainly attributed to decomposition;
387 however, some loss is likely due to leaching (Hockaday et al., 2006; Hockaday et al., 2007;
388 Major et al., 2010) and erosion (Abney et al., 2019a; Abney and Berhe, 2018; Abney et al.,
389 2019b; Abney et al., 2017). In this section we discuss how chemical and physical properties of
390 PyC and leached dPyC impact its transport in soils. We propose the following function can be
391 used to understand the controls on the fate and properties of PyC and dPyC throughout post-fire
392 landscapes (**Figure 7**).
393
394 *f = Burn conditions + Climate + Topography + Organisms + Parent material + Time + post-*
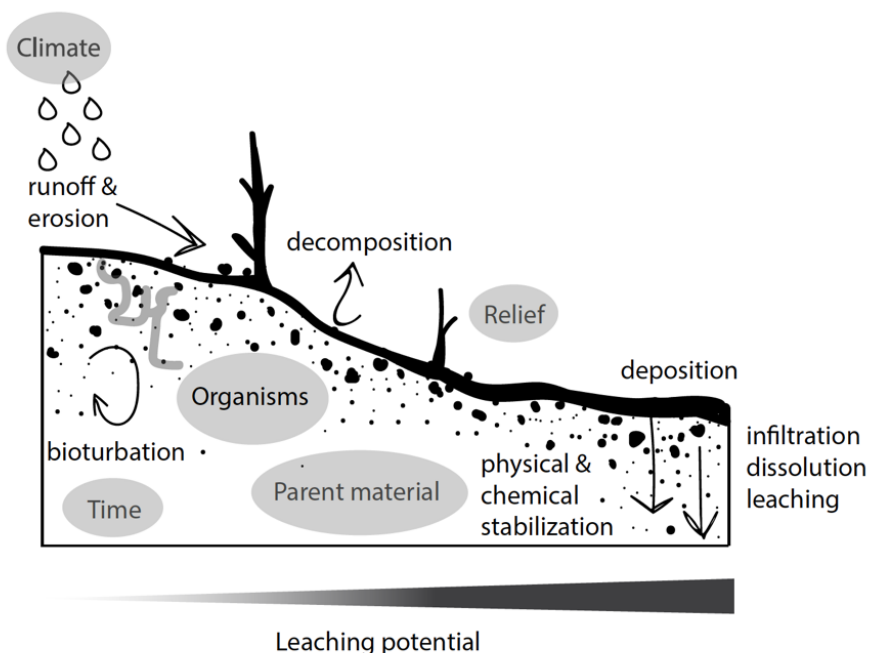395 *fire soil properties*

Figure 7. The interaction of soil forming factors and burn conditions and severity likely dictate the fate and transport of dissolved pyrogenic carbon across dynamic landscapes. By studying post-fire landscapes through these two lenses, we can build predictive models of post-fire PyC dynamics.

4.1.2 Chemical composition

The chemical composition of solid PyC before dissolution influences the quantity and quality of dPyC. At lower burn temperatures, the resulting PyC formed is high in nutrients and oxygenated functional groups, while higher burn temperatures tend to produce more condensed aromatic PyC with low H:C and O:C ratios (Baldock and Smernik, 2002; Hammes et al., 2006; Schneider et al., 2010). Bostick et al. (2018) investigated dPyC from PyC formed from oak wood and grass and reported that the leachates were mostly chemically composed of a complex mixture of C groups such as carbohydrates, aliphatics, and low molecular mass compounds, suggesting that the DOM resulting from PyC is not solely comprised of condensed aromatics. Wozniak et al. (2020) demonstrated the chemical heterogeneity of dPyC derived from low (250 ºC) heating temperatures exhibits a wide range of O/C values and the presence of highly unsaturated aliphatics, polyphenolic formulas, and condensed aromatic C compounds, whereas compounds with low O/C ratios dominated the composition of dPyC from higher heating temperatures (400 ºC). Taken together, these recent findings suggest that DOM leached from solid PyC is heterogeneous and exists also along a chemical continuum (Wagner et al., 2021).

4.1.3 The influence of burn conditions

The physical structure of PyC has been described by porosity, density, and particle size, each of which can impact the stability of PyC and its potential for dissolution. For many biochars, increasing charring temperature increases skeletal density and porosity, which may alter the availability of the char for microbial decomposition or leaching (Brewer et al., 2014). We found

that the formation conditions (burn time and temperature) of dPyC were most critical for predicting the concentration of dPyC (**Figure 6**). Together, burn temperature and duration represent burn intensity, which controls solid PyC properties, such that increasing burn temperature and duration generate PyC that is increasingly aromatic and condensed, contributing to lower solubility. Solid PyC can physically break down via comminution that leads to its dissolved transport, and there is evidence that weaker portions of the solid PyC structure are responsible for the production of dPyC (Spokas et al., 2014). Some of the variability in strength in char structure reflects the precursor plant material structure, which is often retained in PyC (Brewer et al., 2014; Hockaday et al., 2007), although we found no effect of precursor material on resulting dPyC concentration. Additionally, size of the solid PyC controls the distance it can be transported via erosion (Bellè et al., 2021), which may result in the PyC being eroded into a landform position with more favorable leaching conditions. The physical properties of charcoal have been linked to its persistence in soil, however more rsearch is needed on how and if these properties dictate quantity and quality of leached dPyC.

4.2 Environmental and soil factors controlling transport and fate of dPyC along a terrestrial to aquatic gradient

In addition to the properties of PyC, there are a number of environmental and soil factors that play a role in the transport of dPyC across landscapes. Below we discuss the relative importance of site factors, such as climate, topography, biotic factors, parent material, time, and burn conditions, in dPyC formation and transport. Additionally, these factors are frequently interacting across dynamics landscapes and are often difficult to disentangle in natural systems.

4.2.1 Climate: precipitation and temperature
The climate of an area regulates vegetation type, biomass (see **section 4.2.3**) and the availability and quantity of water required for leaching, erosion, and microbial functioning. We found dPyC concentration varied with climate (**Figure 4B**). Precipitation can affect erosion of PyC and transport of dPyC. However, we found no significant relationship between MAP and field-reported concentration of dPyC ($p > 0.05$). This is likely because climate describes average expected weather for an area, whereas leaching is largely driven by individual precipitation events. Climate does dictate the amount of time throughout the year that precipitation-driven erosion events can occur (i.e., rainfall vs snowfall), which can drive overall sediment transport (Nearing et al., 2004). Importantly, extreme precipitation events can be major drivers of erosion in post-fire landscapes, and previous research has described how rain-on-snow events, can lead significant erosional losses of PyC (Abney et al., 2019b; Carroll et al., 2007), and increase runoff (see **Section 4.2.2**).

From this meta-analysis, studies that measured dPyC concentrations in precipitation samples found some of the lowest ratios of dPyC to DOC and the weakest correlation between DOC and dPyC (**Figure 5**). The variability in dPyC in precipitation is likely due to the movement of air masses containing particulate PyC (either from fire ash and soot or fossil fuel burning) and variability in timing of precipitation events. The timing of the initial interaction of PyC and water is critical for its fate in the post-fire landscape (Masiello and Berhe, 2020).

Environmental temperatures likely impact the potential for PyC leaching via a few key mechanisms, mainly through the form of precipitation (snow vs rain fall), the speed of

468 decomposition, and interactively through controlling the vegetation available to serve as fuel for
469 a fire (formation quantity and quality of PyC). Slow rates of snow melting or rainfall could
470 possibly drive more downwards migration of PyC through leaching than the rapid runoff of an
471 intense rainfall event. The role of precipitation intensity as a driver of leaching is also moderated
472 by soil type (Bellè et al., 2021).

474 Temperature is a significant driver of the rate of decomposition. Due to the higher temperature
475 sensitivity of PyC and other more persistent SOM components to rising temperatures (Conant et
476 al., 2011; Davidson and Janssens, 2006; von Lützow and Kögel-Knabner, 2009), these materials
477 may have increased decomposition that leads to greater dissolution of PyC (Abiven et al., 2011;
478 Hockaday et al., 2006). Temperature also impacts the solubility of DOM, where considerably
479 higher temperatures may lead to greater solubility of PyC. We did not find a significant
480 relationship between mean annual temperature and field dPyC concentrations ($p > 0.05$), but we
481 expect that this is due to confounding site-specific environmental and methodological factors.

4.2.2 Topography and Erosion

484 Topographic landform positions are likely significant controls on dPyC and SOM dynamics, as
485 previous research has reported an increase in SOC in depositional landform positions, which
486 receive eroded material from upslope landform positions (Berhe et al., 2018; Berhe et al., 2007).
487 Additionally, the preferential erosion of PyC likely indicates that it can accumulate in these
488 depositional landform positions (Abney et al., 2019a; Abney and Berhe, 2018; Rumpel et al.,
489 2009; Rumpel et al., 2006). An unanswered question, however, is whether PyC in depositional vs
490 eroding landform positions could be more susceptible to leaching when there are higher nutrient
491 concentrations and higher water content. The impacts of erosion on the long-term fate of PyC in
492 soil are complex and depend on the landscape erosion and deposition dynamics along with the
493 soil water conditions and precipitation events (Abney and Berhe, 2018; Masiello and Berhe,
494 2020).

496 Elevated rates of post-fire erosion are widely reported across a range of different ecosystem
497 types (Carroll et al., 2007; de Dios Benavides-Solorio and MacDonald, 2005; Johansen et al.,
498 2001; Robichaud et al., 2007). The magnitude of post-fire erosion is dependent on a number of
499 soil and precipitation factors, which also control the (1) amount of PyC available for leaching
500 and (2) the soil-water conditions to allow for leaching to occur. Furthermore, there is a direct
501 relationship between burn severity, SOM chemical composition, and erosion (Abney and Berhe,
502 2018; Abney et al., 2019b), such that areas that experience higher burn severities have SOM that
503 is more aromatic and more erosion prone. This suggests that conversely, areas that experience
504 moderate- to low-severity burning may have SOM that is more prone to leaching due to both its
505 chemical composition and higher soil infiltration rates (de Dios Benavides-Solorio and
506 MacDonald, 2005; Morris and Moses, 1987; Robichaud, 2000; Torreano and Morris, 1998).
507 Several studies have indicated that PyC is preferentially transported post-fire (Abney et al.,
508 2019b; Rumpel et al., 2009; Rumpel et al., 2006). However, since most SOM that is transported
509 via erosion is deposited within the same or adjacent watershed (Stallard, 1998), this transported
510 PyC may still be available for and susceptible to leaching, depending on where in the landscape
511 it is deposited and if it is buried after erosion.

4.2.3 Organisms

Vegetation can play a role in controlling the fate of dPyC through several pathways including acting as fuel for PyC formation and regulating burn severity, moderating microbial decomposition/cycling of PyC, and dictating post-fire erosion via soil stabilization and ground cover (Cerdà and Doerr, 2005; Kim et al., 2021). These factors will vary with functional type, diversity, and structure of vegetation, along with pre- and post-fire growth, and likely will drive shifts in SOM composition and stability (Hart et al., 2005; Viedma et al., 2020).

In addition to vegetation growing in soil, soil fauna can impact the fate of PyC and dPyC. Soil macro- and meso-fauna can fragment PyC into smaller particle sizes and remobilize it (Domene, 2016; Domene et al., 2015; Elmer et al., 2015), likely increasing the availability of PyC to be microbially degraded and/or to enter the dissolved phase. However, the specific role of bioturbation in controlling dPyC production has, to the authors' knowledge, not yet be demonstrated. The addition of PyC to soil also has been demonstrated to increase microbial biomass and cause shifts in microbial community composition (Dove et al., 2021; Lehmann et al., 2011). While in some cases bacteria can be more sensitive to fire relative to fungi (Brown et al., 2019), emerging general trends suggest that fungi are more sensitive to fire than bacteria (Beals et al., 2022; Pressler et al., 2019; Qin and Liu, 2021), and may (Dove and Hart, 2017) or may not (Miyamoto et al., 2021) recover to pre-fire levels over a decade post burn. These microbial and other organismal community changes can alter soil functioning, possibly leading to shifts in dPyC formation and leaching.

4.2.4 Parent material and mineralogy

A soil's parent material will influence the soil texture, structure, and reactivity, among other soil characteristics. Coarse, sandy soils formed under granodiorite parent material may have a low cation exchange capacity (CEC) and little aggregation. These soils can have high infiltration and hydraulic conductivity, with potential for rapid leaching of PyC; however, clay-rich soils may have lower hydraulic conductivity, with lower potential for leaching (Barnes et al., 2014). The parent material also influences the hydrologic properties of a soil, including depth to ground water, hydrologic flow paths, and connectivity to nearby aquatic systems. In soils where surface water is connected to groundwater, post-fire leaching can transport dPyC to groundwater, with some concerns regarding contamination (i.e., with PAHs) of aquifers (Mansilha et al., 2014).

Mineral-PyC associations have been demonstrated by several studies (Brodowski et al., 2005; Glaser et al., 2000; Hockaday et al., 2007), but it is not yet known if these form from the interaction of dPyC or solid PyC with soil minerals. Cusack et al. (2012) found positive correlations between PyC, SOC, and short range order minerals. They did not find selective preservation of PyC over non-pyrogenic SOC, but they indicate that physical protection is likely an important stabilization mechanism for PyC. A study by Brodowski et al. (2006) found that PyC was preferentially stored in the occluded particulate matter fraction. Some labeling studies have also found PyC associated with heavy fractions or aggregate-protected fractions in less than a year post-application (Singh et al., 2014; Soong and Cotrufo, 2015). In subsurface soils, increasing mineral surface area and soil aggregates (Lehmann and Kleber, 2015) may control dPyC mobility or stabilization via sorption. Mineral stabilization of PyC needs more

558 investigation to fully understand how mineral-PyC associations are similar to other mineral-OM
559 interactions, how important the quality and quantity of dPyC is for the formation of these
560 associations, and how these interactions impact the long-term fate of PyC in soil.
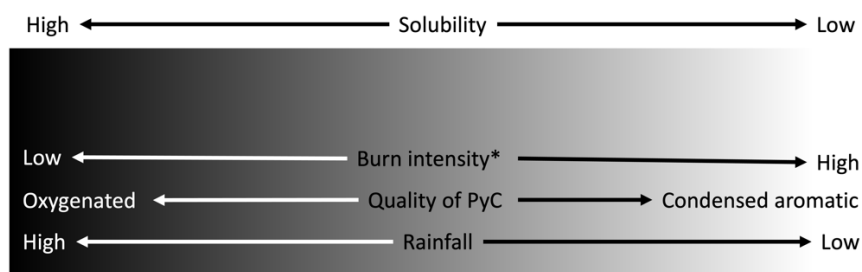561
562 **4.2.5 Time**
563 Time is a critical factor in controlling the availability of PyC for forming dPyC. Some research
564 has shown that dPyC increases over time, as the processes of chemical, physical, and biological
565 breakdown of PyC progress, leading to more soluble DOC and dPyC with increasing time since
566 fire (Bostick et al., 2021; Wagner et al., 2017; Wagner et al., 2018). This increase in solubility is
567 due to introduction of oxygen and nitrogen containing functional groups (Hilscher and Knicker,
568 2011). There are also lags in between the formation and deposition of PyC and the transport of
569 dPyC into adjacent and further downstream aquatic systems, as evidenced by increasing ages of
570 PyC and dPyC (see **Table S2, Figure 9**) along terrestrial to aquatic gradients to the ocean
571 (Wagner et al., 2019a; Wagner et al., 2018). Contrastingly, some evidence indicates that PyC is
572 more bioavailable immediately after production than previously considered, in part because PyC
573 also contains C compounds that are readily available for microorganisms and leaching (Cheng et
574 al., 2008; Schiedung et al., 2020). In addition, the amount of soil PyC available for forming
575 dPyC is highest in the time immediately post fire. There are many environmental factors that
576 may control dPyC concentrations in soils and streams post-fire, such as soil PyC concentrations,
577 hydrological connectivity of a watershed, potential for decomposition of PyC, etc. At least one
578 study has found no significant relationship between time since fire or between fire return interval
579 and dPyC concentration in stream water (Ding et al., 2013), which may be reflective of these
580 complexities. Wagner et al. (2015a) show different timing of particulate PyC compared with
581 dPyC peak fluxes in a fire-impacted watershed in Colorado, USA. With this variability in timing
582 of export of dPyC from fire-impacted watersheds, we need more mechanistic understanding of
583 the hillslope-scale processes that are driving this export.

584 **4.3 Fire-driven changes in soil properties that impact mobilization of PyC**
585
586 Fire has long been described as a major disturbance and driver of changes in soil properties,
587 including pH, hydrophobicity, texture, nutrient availability, organic matter, and vegetation
588 dynamics, among others (DeBano, 1991; Gonzalez-Perez et al., 2004; MacDonald and Huffman,
589 2004; Mataix-Solera et al., 2011). Increases in pH may shift the solubility of PyC, as shown by
590 Rebollo et al. (2008). Increases in hydrophobicity post-fire have frequently been reported in low
591 to moderate burn severity fires (MacDonald and Huffman, 2004; Mataix-Solera and Doerr,
592 2004), and increases in hydrophobicity may at least temporarily reduce the leaching of PyC into
593 the soil matrix, as there is reduced infiltration (with potential for increased runoff of dPyC).
594 Reduced infiltration and increased runoff post-fire due to loss of vegetation also can lead to
595 elevated rates of erosion, at least until vegetation cover has returned (de Dios Benavides-Solorio
596 and MacDonald, 2005). As vegetation regrows post-fire and the hydrophobic layer begins to
597 breakdown, it is not yet clear what timescales are relevant for processes of dissolution, within
598 soil redistribution, and hydrologic export of dPyC or how these processes are moderated by burn
599 severity to impact the transport of dPyC.
600
601 Results from this meta-analysis indicate that burn intensity (energy x duration, **Figure 6**) is a
602 significant driver of formation and quality of dPyC in soils. Other works have suggested that
603 quality of PyC is important for formation of dPyC (more aliphatic, from either lower temperature

604 burns or more decomposed/oxidized). For example, as PyC ages, its surfaces may be broken into
605 more digestible aliphatic functional groups, which are more soluble, but as these are consumed,
606 this can leave behind a more aromatic PyC (Hockaday et al., 2007) that is less soluble and more
607 persistent in soil. Rainfall can also drive fluxes of dPyC, as multiple papers monitoring dPyC in
608 rivers noted positive correlations with higher discharge (Jaffé et al., 2013; Wagner et al., 2019c;
609 Wagner et al., 2015a). Bao et al. (2019) argue that different pools of DOC are accessed at
610 different flow rates through soil, so we often find higher DOC and dPyC with higher flows. The
611 actual solubility (or potential for leaching) of PyC across landscapes is controlled by the
612 interaction of several environmental factors, including the burn intensity, chemical and physical
613 properties of the PyC, and amount and timing of rainfall (**Figure 8**).

614

*Burn intensity = temperature + duration of burning

615
616 **Figure 8.** Controls on the solubility (leaching potential) of PyC in soil. There are other factors
617 controlling the leaching of PyC, including decomposition, sorption, fire return intervals, aging of
618 PyC etc. However, these are complex and likely do not follow linear relationships.

619 **5. Ecosystem implications**

620

621 **5.1 Fluxes and internal cycling of dPyC between ecosystem components**
622 If we assume that the main sources of dPyC for soil water include precipitation and dissolution
623 of *in situ* charcoal or PyC, then our data currently suggest that *in situ* dissolution is much more
624 important than precipitation, which has a dPyC concentration three orders of magnitude lower
625 than reported soil extract dPyC concentrations (**Figure 6**). The radiocarbon age of dPyC in
626 precipitation was, however, older than reported in some tributaries and rivers (**Figure 9, Table
627 S2**), which likely suggests that there is some contribution of very old pyrogenic material (i.e.,
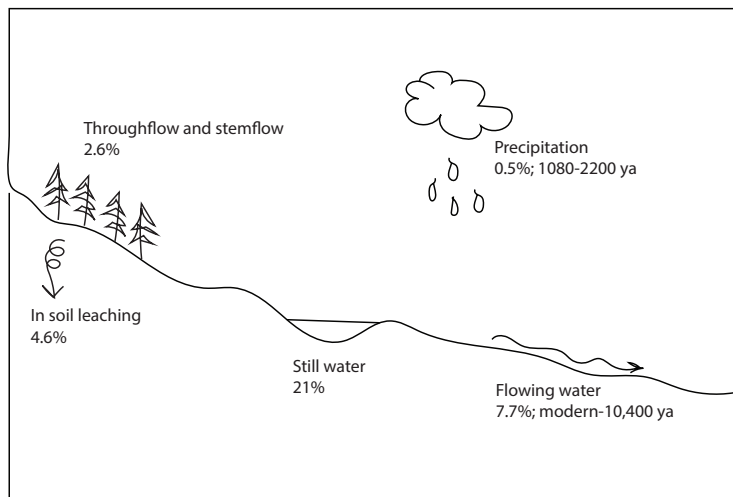628 fossil fuel combustion) with more modern fuel burning to this dPyC pool.

629
630 **Figure 9**. Transport of dPyC across different global pools is presented as percent of dissolved
631 organic carbon. The range of radiocarbon ages of the dPyC in each pool is reported in years (see
632 **Table S2** for more details).

633
634 In terrestrial waters, there is an increase in dPyC age reported with increasingly larger and more
635 mixed pools (rivers and oceans compared with tributaries). The age of dPyC in tributaries was
636 also highly variable, which is likely reflective of the significance of individual watershed
637 properties on controlling DOM release and quality. The proportion of PyC that makes it from
638 deposition on soil surface post-fire to ocean dPyC is likely infinitesimally small. Bostick et al.
639 (2018) report that between 1-14% of dPyC in soil is leached to river dPyC, so the vast majority is
640 internally recycled. However, the controls on the variability and rates of transport of soil dPyC to
641 riverine dPyC need further investigation.

642
643 **5.2 Future research directions**
644 There are numerous needs for future research on dPyC in soil and across the terrestrial-aquatic
645 interface. Arguably the most urgent of these is for more quantitative measurements of dPyC in
646 soil and in fluxes between soil and aquatic systems. For example, two previous studies showed
647 that PyC can be redistributed through leaching at rates of 1-17% of soil PyC (Boot et al., 2015;
648 Major et al., 2010), however considerable further quantification is required before we can predict
649 dPyC leaching across different soil and precipitation regimes. There is also a need for better
650 intercomparison between quantitative approaches, or perhaps wider adoption of a smaller
651 number of methodologies. In field studies, the BPCA method has been widely used and has the
652 advantage of being able to measure PyC in both solid and liquid samples, as well as provide
653 information about the condensation of PyC. However, BPCA is time consuming, somewhat
654 expensive, and is quantitative only for higher burn intensity dPyC. Combining BPCA with
655 methods that are more sensitive to the lower burn intensity portion of the dPyC continuum will
656 allow for a more complete understanding of transport of dPyC in the natural environment. BPCA
657 can also be used to train models that enable researchers to obtain more quantiative measurements
658 from less costly, high-throughput methods such as mid-infrared spectroscopy(Cotrufo et al.,
659 2016; Sanderman et al., 2020).

660

661 A second methodological need is for studies designed to provide infromation within relevant
662 timescales for dPyC pools and fluxes post fire. Many studies have reported some "aging"
663 required for PyC to become dissolved (Abiven et al., 2011; Hockaday et al., 2007), but these do
664 not frequently have more than two or three timepoints for comparing dissolution. Additionally,
665 we know that some of this aging is related to photooxidation, microbial processing, and physical
666 comminution (Abney et al., 2019a; Spokas et al., 2014; Ward et al., 2014), but these processes
667 likely operate on different timescales and interact with one another to dynamically impact the
668 fate of PyC in soil. Some post-fire soil changes have been well-described, such as increases in
669 pH, weaking of soil aggregation, and decreases in infiltration due to increases in hydrophobicity,
670 and these post-fire changes can be short (<1 year) or medium term (>10 years), depending on
671 burn severity and post-fire vegetation recovery (Abney et al., 2017; Gonzalez-Perez et al., 2004;
672 Mataix-Solera and Doerr, 2004). Longer-term experiments and field studies which encompass
673 repeated burn events are needed to capture the time component of dPyC transformation.
674
675 There is also a need for a better understanding of dPyC cycling within soils and watersheds – in
676 particular, the fate of PyC once it becomes dissolved in soil water. We hypothesize that PyC can
677 also form mineral-organic interactions and become stabilized, but there is only limited evidence
678 for this process (Brodowski et al., 2006). We also hypothesize that dPyC is more susceptible to
679 decomposition than solid PyC, but this needs further evidence. We also need a better
680 understanding of the transport and connectivity of DOC and dPyC between different aquatic
681 reservoirs (i.e., soil water to stream to lake, etc.). There is considerable research attention on
682 DOC at the terrestrial-aquatic interface (Rasilo et al., 2015; Tank et al., 2018), but this has yet to
683 focus on the transport of dPyC, which may be a significant persistent source of OC in this pool.
684 Future research should also focus on the interaction between disturbances and loss processes
685 impacting PyC and dPyC, such as erosion, leaching, decomposition, land management activities,
686 and climate change.
687
6. Conclusions
689 The concentration and flux of dPyC from soils represents a critical soil C loss mechanism that
690 has received a limited amount of research attention. In rivers and the ocean, dPyC has been well
691 characterized, but more investigation is needed to understand the connections between terrestrial
692 and aquatic dPyC. One major challenge to integrating current research on dPyC is the range of
693 different methodological approaches used to quantitatively and qualitatively describe dPyC. In
694 this meta-analysis we found over ten different methodologies used to quantify dPyC, but in field
695 studies BPCA was the most frequently used method. We found that methods that quantify a
696 smaller component of the dPyC continuum reported smaller concentrations of dPyC, which
697 indicates that these may be underestimating dPyC more than other methods.
698
699 Due to the variability in the data in this meta-analysis, we found only a few significant trends in
700 controlling factors for dPyC. In field studies, sample types where PyC was in contact with water
701 for longer periods had higher concentrations of dPyC (i.e., still water vs flowing water), and
702 samples from areas with more fire-prone climates had higher dPyC concentrations (i.e.,
703 temperate vs polar). Across sample types we also found high correlations between DOC and
704 dPyC, which suggests that these are controlled by many of the same environmental processes,
705 although these processes need further elucidation. From laboratory studies, we found that burn
706 temperature and burn time were negatively correlated with freshly extracted dPyC, which is

707   consistent with chemical controls of dPyC solubility. We have also identified a significant need
708   for (1) more mechanistic understanding of the environmental controls of movement of dPyC
709   from soil to aquatic systems, (2) more soil-water measurements of dPyC, and (3) better
710   understanding of the temporal scales relevant to environmental fluxes of dPyC. Future research
711   on dPyC that provides quantitative measures of the fate of dPyC in soils will be useful for
712   models of soil C and can potentially provide critical information to land managers seeking to
713   increase soil C storage.
714

715

729

730 **Open Research**

731 The data extracted from the meta-analysis are available in a table uploaded in the supplemental

732 information of this manuscript as a .xslx file.

733
734

References

2020. ArcGIS Desktop 10.8.1 Esri Inc.

Abdelrahman, H. et al., 2018. Historical charcoal additions alter water extractable, particulate and bulk soil C composition and stabilization. Journal of Plant Nutrition and Soil Science, 181, 809-817.

Abiven, S., Hengartner, P., Schneider, M.P.W., Singh, N., Schmidt, M.W.I., 2011. Pyrogenic carbon soluble fraction is larger and more aromatic in aged charcoal than in fresh charcoal. Soil Biology & Biochemistry, 43, 1615-1617.

Abney, R., Jin, L., Berhe, A., 2019a. Soil properties and combustion temperature: Controls on the decomposition rate of pyrogenic organic matter. Catena, 182, 104127.

Abney, R.B., Berhe, A.A., 2018. Pyrogenic carbon erosion: implications for stock and persistence of pyrogenic carbon in soil. Frontiers in Earth Science, 6, 26.

Abney, R.B. et al., 2019b. Pyrogenic carbon erosion after the Rim Fire, Yosemite National Park: the role of burn severity and slope. Journal of Geophysical Research: Biogeosciences, 124, 432-449.

Abney, R.B., Sanderman, J., Johnson, D., Fogel, M.L., Berhe, A.A., 2017. Post-wildfire erosion in mountainous terrain leads to rapid and major redistribution of soil organic carbon. Frontiers in Earth Science, 5, 99.

Ascough, P. et al., 2009. Hydropyrolysis as a new tool for radiocarbon pre-treatment and the quantification of black carbon. Quaternary Geochronology, 4, 140-147.

Baldock, J.A., Smernik, R.J., 2002. Chemical composition and bioavailability of thermally altered Pinus resinosa (Red pine) wood. Organic Geochemistry, 33, 1093-1109.

Bao, H., Niggemann, J., Huang, D., Dittmar, T., Kao, S.J., 2019. Different responses of dissolved black carbon and dissolved lignin to seasonal hydrological changes and an extreme rain event. Journal of Geophysical Research: Biogeosciences, 124, 479-493.

Barnes, R.T., Gallagher, M.E., Masiello, C.A., Liu, Z., Dugan, B., 2014. Biochar-Induced Changes in Soil Hydraulic Conductivity and Dissolved Nutrient Fluxes Constrained by Laboratory Experiments. Plos One, 9.

Beals, K.K., Scearce, A.E., Swystun, A.T., Schweitzer, J.A., 2022. Belowground mechanisms for oak regeneration: Interactions among fire, soil microbes, and plant community alter oak seedling growth. Forest Ecology and Management, 503, 119774.

Bellè, S.-L. et al., 2021. Key drivers of pyrogenic carbon redistribution during a simulated rainfall event. Biogeosciences, 18, 1105-1126.

Berhe, A.A., Barnes, R.T., Six, J., Marín-Spiotta, E., 2018. Role of soil erosion in biogeochemical cycling of essential elements: carbon, nitrogen, and phosphorus. Annual Review of Earth and Planetary Sciences, 46, 521-548.

Berhe, A.A., Harte, J., Harden, J.W., Torn, M.S., 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience, 57, 337-346.

Bi, C., Wang, X., Jia, J., Chen, Z., 2018. Spatial variation and sources of polycyclic aromatic hydrocarbons influenced by intensive land use in an urbanized river network of East China. Science of the Total Environment, 627, 671-680.

Bird, M.I., Wynn, J.G., Saiz, G., Wurster, C.M., McBeath, A., 2015. The pyrogenic carbon cycle. Annual Review of Earth and Planetary Sciences, 43, 273-298.

Boot, C., Haddix, M., Paustian, K., Cotrufo, M., 2015. Distribution of black carbon in ponderosa pine forest floor and soils following the High Park wildfire. Biogeosciences, 12, 3029-3039.

Bostick, K.W. et al., 2021. Biolability of Fresh and Photodegraded Pyrogenic Dissolved Organic Matter From Laboratory-Prepared Chars. Journal of Geophysical Research: Biogeosciences, 126, e2020JG005981.

Bostick, K.W., Zimmerman, A.R., Wozniak, A.S., Mitra, S., Hatcher, P.G., 2018. Production and Composition of Pyrogenic Dissolved Organic Matter From a Logical Series of Laboratory-Generated Chars. Frontiers in Earth Science, 6.

Bowring, S.P., Jones, M.W., Ciais, P., Guenet, B., Abiven, S., 2022. Pyrogenic carbon decomposition critical to resolving fire's role in the Earth system. Nature Geoscience, 15, 135-142.

Braun, M., Kappenberg, A., Sandhage-Hofmann, A., Lehndorff, E., 2020. Leachable soil black carbon after biochar application. Organic Geochemistry, 143, 103996.

Brewer, C.E. et al., 2014. New approaches to measuring biochar density and porosity. Biomass and bioenergy, 66, 176-185.

Brodowski, S., Amelung, W., Haumaier, L., Abetz, C., Zech, W., 2005. Morphological and chemical properties of black carbon in physical soil fractions as revealed by scanning electron microscopy and energy-dispersive X-ray spectroscopy. Geoderma, 128, 116-129.

Brodowski, S., John, B., Flessa, H., Amelung, W., 2006. Aggregate-occluded black carbon in soil. European journal of soil science, 57, 539-546.

Brown, S.P. et al., 2019. Context dependent fungal and bacterial soil community shifts in response to recent wildfires in the Southern Appalachian Mountains. Forest Ecology and Management, 451, 117520.

Carroll, E.M. et al., 2007. Spatial analysis of a large magnitude erosion event following a Sierran wildfire. Journal of environmental quality, 36, 1105-1111.

Cerdà, A., Doerr, S.H., 2005. Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. International Journal of Wildland Fire, 14, 423-437.

Chen, Y. et al., 2022. Analytical methods, molecular structures and biogeochemical behaviors of dissolved black carbon. Carbon Research, 1, 23.

Cheng, C.H., Lehmann, J., Thies, J.E., Burton, S.D., 2008. Stability of black carbon in soils across a climatic gradient. Journal of Geophysical Research: Biogeosciences, 113.

Conant, R.T. et al., 2011. Temperature and soil organic matter decomposition rates–synthesis of current knowledge and a way forward. Global change biology, 17, 3392-3404.

Coppola, A.I. et al., 2022. The black carbon cycle and its role in the Earth system. Nature Reviews Earth & Environment, 3, 516-532.

Cotrufo, M.F. et al., 2016. Quantification of pyrogenic carbon in the environment: An integration of analytical approaches. Organic Geochemistry, 100, 42-50.

Cusack, D.F., Chadwick, O.A., Hockaday, W.C., Vitousek, P.M., 2012. Mineralogical controls on soil black carbon preservation. Global Biogeochemical Cycles, 26.

Davidson, E.A., Janssens, I.A., 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature, 440, 165-173.

de Dios Benavides-Solorio, J., MacDonald, L.H., 2005. Measurement and prediction of post-fire erosion at the hillslope scale, Colorado Front Range. International Journal of Wildland Fire, 14, 457-474.

DeBano, L.F., 1991. The effect of fire on soil properties, Proceedings management and productivity of western-Montane. Forest Soils, pp. 151-155.

Descals, A. et al., 2022. Unprecedented fire activity above the Arctic Circle linked to rising temperatures. Science, 378, 532-537.

Ding, Y., Yamashita, Y., Dodds, W.K., Jaffé, R., 2013. Dissolved black carbon in grassland streams: Is there an effect of recent fire history? Chemosphere, 90, 2557-2562.

Dittmar, T., 2015. Reasons Behind the Long-Term Stability of Dissolved Organic Matter.

Dittmar, T. et al., 2012. Continuous flux of dissolved black carbon from a vanished tropical forest biome. Nature Geoscience, 5, 618-622.

Domene, X., 2016. A critical analysis of meso-and macrofauna effects following biochar supplementation, Biochar Application. Elsevier, pp. 268-292.

Domene, X., Hanley, K., Enders, A., Lehmann, J., 2015. Short-term mesofauna responses to soil additions of corn stover biochar and the role of microbial biomass. Applied Soil Ecology, 89, 10-17.

Dove, N.C., Hart, S.C., 2017. Fire reduces fungal species richness and in situ mycorrhizal colonization: a meta-analysis. Fire Ecology, 13, 37-65.

Dove, N.C., Klingeman, D.M., Carrell, A.A., Cregger, M.A., Schadt, C.W., 2021. Fire alters plant microbiome assembly patterns: integrating the plant and soil microbial response to disturbance. New Phytologist, 230, 2433-2446.

Drake, T.W. et al., 2020. Du feu à l'eau: source and flux of dissolved black carbon from the Congo River. Global Biogeochemical Cycles, 34, e2020GB006560.

Druffel, E.R., 2004. Comments on the importance of black carbon in the global carbon cycle. Marine Chemistry, 92, 197-200.

Elmer, W., Lattao, C., Pignatello, J., 2015. Active removal of biochar by earthworms (Lumbricus terrestris). Pedobiologia, 58, 1-6.

Elmquist, M., Semiletov, I., Guo, L., Gustafsson, Ö., 2008. Pan-Arctic patterns in black carbon sources and fluvial discharges deduced from radiocarbon and PAH source apportionment markers in estuarine surface sediments. Global Biogeochemical Cycles, 22.

Feurdean, A. et al., 2020. Recent fire regime in the southern boreal forests of western Siberia is unprecedented in the last five millennia. Quaternary Science Reviews, 244, 106495.

Giglio, L., Randerson, J.T., Van Der Werf, G.R., 2013. Analysis of daily, monthly, and annual burned area using the fourth-generation global fire emissions database (GFED4). Journal of Geophysical Research: Biogeosciences, 118, 317-328.

Glaser, B., Balashov, E., Haumaier, L., Guggenberger, G., Zech, W., 2000. Black carbon in density fractions of anthropogenic soils of the Brazilian Amazon region. Organic geochemistry, 31, 669-678.

Gonzalez-Perez, J.A., Gonzalez-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fire on soil organic matter - a review. Environment International, 30, 855-870.

Grant, M.J., Booth, A., 2009. A typology of reviews: an analysis of 14 review types and associated methodologies. Health information & libraries journal, 26, 91-108.

Güereña, D.T. et al., 2015. Terrestrial pyrogenic carbon export to fluvial ecosystems: Lessons learned from the White Nile watershed of East Africa. Global Biogeochemical Cycles, 29, 1911-1928.

Gustafsson, Ö. et al., 2001. Evaluation of a protocol for the quantification of black carbon in sediments. Global Biogeochemical Cycles, 15, 881-890.

Hameed, R. et al., 2023. Structural characteristics of dissolved black carbon and its interactions with organic and inorganic contaminants: A critical review. Science of The Total Environment, 162210.

Hammes, K. et al., 2007. Comparison of quantification methods to measure fire-derived (black/elemental) carbon in soils and sediments using reference materials from soil, water, sediment and the atmosphere. Global Biogeochemical Cycles, 21.

Hammes, K. et al., 2006. Synthesis and characterisation of laboratory-charred grass straw (Oryza sativa) and chestnut wood (Castanea sativa) as reference materials for black carbon quantification. Organic Geochemistry, 37, 1629-1633.

Hart, S.C., DeLuca, T.H., Newman, G.S., MacKenzie, M.D., Boyle, S.I., 2005. Post-fire vegetative dynamics as drivers of microbial community structure and function in forest soils. Forest Ecology and Management, 220, 166-184.

Hilscher, A., Knicker, H., 2011. Degradation of grass-derived pyrogenic organic material, transport of the residues within a soil column and distribution in soil organic matter fractions during a 28 month microcosm experiment. Organic Geochemistry, 42, 42-54.

Hockaday, W.C., Grannas, A.M., Kim, S., Hatcher, P.G., 2006. Direct molecular evidence for the degradation and mobility of black carbon in soils from ultrahigh-resolution mass spectral analysis of dissolved organic matter from a fire-impacted forest soil. Organic Geochemistry, 37, 501-510.

Hockaday, W.C., Grannas, A.M., Kim, S., Hatcher, P.G., 2007. The transformation and mobility of charcoal in a fire-impacted watershed. Geochimica et Cosmochimica Acta, 71, 3432-3445.

Jaffé, R. et al., 2013. Global charcoal mobilization from soils via dissolution and riverine transport to the oceans. Science, 340, 345-347.

Johansen, M.P., Hakonson, T.E., Breshears, D.D., 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. Hydrological processes, 15, 2953-2965.

Jones, M.W. et al., 2020. Fires prime terrestrial organic carbon for riverine export to the global oceans. Nature communications, 11, 2791.

Jones, M.W. et al., 2019. Environmental controls on the riverine export of dissolved black carbon. Global Biogeochemical Cycles, 33, 849-874.

Kim, Y., Kim, C.-G., Lee, K.S., Choung, Y., 2021. Effects of post-fire vegetation recovery on soil erosion in vulnerable montane regions in a monsoon climate: a decade of monitoring. Journal of Plant Biology, 64, 123-133.

Kolden, C.A., 2019. We're not doing enough prescribed fire in the Western United States to mitigate wildfire risk. Fire, 2, 30.

Kuhlbusch, T., Crutzen, P., 1995. Toward a global estimate of black carbon in residues of vegetation fires representing a sink of atmospheric CO2 and a source of O2. Global Biogeochemical Cycles, 9, 491-501.

Laura Zepner, P.K., Felix Wiemann & Lars Bernard 2020. ClimateCharts.net – an interactive climate analysis web platform. International Journal of Digital Earth.

Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. Nature, 528, 60-68.

Lehmann, J. et al., 2011. Biochar effects on soil biota–a review. Soil biology and biochemistry, 43, 1812-1836.

915   Lehner, B., Grill, G., 2013. Global river hydrography and network routing: baseline data and
916   new approaches to study the world's large river systems. Hydrological Processes, 27,
917   2171-2186.
918   MacDonald, L.H., Huffman, E.L., 2004. Post-fire soil water repellency: persistence and soil
919   moisture thresholds. Soil Science Society of America Journal, 68, 1729-1734.
920   Major, J., Lehmann, J., Rondon, M., Goodale, C., 2010. Fate of soil-applied black carbon:
921   downward migration, leaching and soil respiration. Global Change Biology, 16, 1366-
922   1379.
923   Mansilha, C., Carvalho, A., Guimarães, P., Espinha Marques, J., 2014. Water quality concerns
924   due to forest fires: Polycyclic aromatic hydrocarbons (PAH) contamination of
925   groundwater from mountain areas. Journal of Toxicology and Environmental Health, Part
926   A, 77, 806-815.
927   Masiello, C.A., 2004. New directions in black carbon organic geochemistry. Marine Chemistry,
928   92, 201-213.
929   Masiello, C.A., Berhe, A.A., 2020. First interactions with the hydrologic cycle determine
930   pyrogenic carbon's fate in the Earth system. Earth Surface Processes and Landforms, 45,
931   2394-2398.
932   Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., Zavala, L., 2011. Fire effects on soil
933   aggregation: a review. Earth-Science Reviews, 109, 44-60.
934   Mataix-Solera, J., Doerr, S.H., 2004. Hydrophobicity and aggregate stability in calcareous
935   topsoils from fire-affected pine forests in southeastern Spain. Geoderma, 118, 77-88.
936   Mengist, W., Soromessa, T., Legese, G., 2020. Method for conducting systematic literature
937   review and meta-analysis for environmental science research. MethodsX, 7, 100777.
938   Miyamoto, Y., Danilov, A.V., Bryanin, S.V., 2021. The dominance of Suillus species in
939   ectomycorrhizal fungal communities on Larix gmelinii in a post-fire forest in the Russian
940   Far East. Mycorrhiza, 31, 55-66.
941   Morris, S.E., Moses, T.A., 1987. Forest fire and the natural soil erosion regime in the Colorado
942   Front Range. Annals of the association of American geographers, 77, 245-254.
943   Myers-Pigg, A.N. et al., 2015. Labile pyrogenic dissolved organic carbon in major Siberian
944   Arctic rivers: Implications for wildfire-stream metabolic linkages. Geophysical Research
945   Letters, 42, 377-385.
946   Nearing, M., Pruski, F., O'neal, M., 2004. Expected climate change impacts on soil erosion rates:
947   a review. Journal of soil and water conservation, 59, 43-50.
948   Pressler, Y., Moore, J.C., Cotrufo, M.F., 2019. Belowground community responses to fire: meta-
949   analysis reveals contrasting responses of soil microorganisms and mesofauna. Oikos,
950   128, 309-327.
951   Pyle, L.A. et al., 2015. Chemical and isotopic thresholds in charring: implications for the
952   interpretation of charcoal mass and isotopic data. Environmental Science & Technology,
953   49, 14057-14064.
954   Qin, Q., Liu, Y., 2021. Changes in microbial communities at different soil depths through the
955   first rainy season following severe wildfire in North China artificial Pinus tabulaeformis
956   forest. Journal of Environmental Management, 280, 111865.
957   Rasilo, T., Ojala, A., Huotari, J., Starr, M., Pumpanen, J., 2015. Concentrations and quality of
958   DOC along the terrestrial–aquatic continuum in a boreal forested catchment. Freshwater
959   Science, 34, 440-455.

Rebollo, N. et al., 2008. Structural characterization of charcoal exposed to high and low pH: implications for 14C sample preparation and charcoal preservation. Radiocarbon, 50, 289-307.

Reisser, M., Purves, R.S., Schmidt, M.W.I., Abiven, S., 2016. Pyrogenic Carbon in Soils: A Literature-Based Inventory and a Global Estimation of Its Content in Soil Organic Carbon and Stocks. Frontiers in Earth Science, 4.

Robichaud, P., Elliot, W., Pierson, F., Hall, D., Moffet, C., 2007. Predicting postfire erosion and mitigation effectiveness with a web-based probabilistic erosion model. Catena, 71, 229-241.

Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. Journal of hydrology, 231, 220-229.

Rohatgi, A., 2022. WebPlotDigitizer.

Rumpel, C., Ba, A., Darboux, F., Chaplot, V., Planchon, O., 2009. Erosion budget and process selectivity of black carbon at meter scale. Geoderma, 154, 131-137.

Rumpel, C. et al., 2006. Preferential erosion of black carbon on steep slopes with slash and burn agriculture. Catena, 65, 30-40.

Sanderman, J., Savage, K., Dangal, S.R., 2020. Mid-infrared spectroscopy for prediction of soil health indicators in the United States. Soil Science Society of America Journal, 84, 251-261.

Santín, C. et al., 2016. Towards a global assessment of pyrogenic carbon from vegetation fires. Global Change Biology, 22, 76-91.

Santos, F., Torn, M.S., Bird, J.A., 2012. Biological degradation of pyrogenic organic matter in temperate forest soils. Soil Biology and Biochemistry, 51, 115-124.

Santos, F., Wagner, S., Rothstein, D., Jaffe, R., Miesel, J.R., 2017. Impact of a historical fire event on pyrogenic carbon stocks and dissolved pyrogenic carbon in spodosols in northern Michigan. Frontiers in Earth Science, 5, 80.

Schiedung, M., Bellè, S.-L., Sigmund, G., Kalbitz, K., Abiven, S., 2020. Vertical mobility of pyrogenic organic matter in soils: a column experiment. Biogeosciences, 17, 6457-6474.

Schmidt, M.W. et al., 2001. Comparative analysis of black carbon in soils. Global biogeochemical cycles, 15, 163-167.

Schneider, M.P., Hilf, M., Vogt, U.F., Schmidt, M.W., 2010. The benzene polycarboxylic acid (BPCA) pattern of wood pyrolyzed between 200 C and 1000 C. Organic Geochemistry, 41, 1082-1088.

Singh, N. et al., 2014. Transformation and stabilization of pyrogenic organic matter in a temperate forest field experiment. Global Change Biology, 20, 1629-1642.

Soong, J.L., Cotrufo, M.F., 2015. Annual burning of a tallgrass prairie inhibits C and N cycling in soil, increasing recalcitrant pyrogenic organic matter storage while reducing N availability. Global change biology, 21, 2321-2333.

Spokas, K. et al., 2014. Physical disintegration of biochar: an overlooked process. Environmental Science & Technology Letters, 1, 326-332.

Stallard, R.F., 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. Global biogeochemical cycles, 12, 231-257.

Tank, S.E., Fellman, J.B., Hood, E., Kritzberg, E.S., 2018. Beyond respiration: Controls on lateral carbon fluxes across the terrestrial-aquatic interface. Limnology and Oceanography Letters, 3, 76-88.

1005   Torreano, S.J., Morris, L.A., 1998. Loblolly pine root growth and distribution under water stress.
1006   Soil Science Society of America Journal, 62, 818-827.
1007   Turco, M. et al., 2018. Exacerbated fires in Mediterranean Europe due to anthropogenic warming
1008   projected with non-stationary climate-fire models. Nature communications, 9, 1-9.
1009   Viedma, O. et al., 2020. Disentangling the role of prefire vegetation vs. burning conditions on
1010   fire severity in a large forest fire in SE Spain. Remote Sensing of Environment, 247,
1011   111891.
1012   von Lützow, M., Kögel-Knabner, I., 2009. Temperature sensitivity of soil organic matter
1013   decomposition—what do we know? Biology and Fertility of soils, 46, 1-15.
1014   Wagner, S. et al., 2019a. Isotopic composition of oceanic dissolved black carbon reveals non-
1015   riverine source. Nature communications, 10, 1-8.
1016   Wagner, S. et al., 2019b. Isotopic composition of oceanic dissolved black carbon reveals non-
1017   riverine source. Nature Communications, 10.
1018   Wagner, S. et al., 2019c. Dissolved black carbon in throughfall and stemflow in a fire-managed
1019   longleaf pine woodland. Biogeochemistry, 146, 191-207.
1020   Wagner, S., Cawley, K.M., Rosario-Ortiz, F.L., Jaffe, R., 2015a. In-stream sources and links
1021   between particulate and dissolved black carbon following a wildfire. Biogeochemistry,
1022   124, 145-161.
1023   Wagner, S. et al., 2021. Questions remain about the biolability of dissolved black carbon along
1024   the combustion continuum. Nature communications, 12, 4281.
1025   Wagner, S., Ding, Y., Jaffé, R., 2017. A new perspective on the apparent solubility of dissolved
1026   black carbon. Frontiers in Earth Science, 5, 75.
1027   Wagner, S., Dittmar, T., Jaffe, R., 2015b. Molecular characterization of dissolved black nitrogen
1028   via electrospray ionization Fourier transform ion cyclotron resonance mass spectrometry.
1029   Organic Geochemistry, 79, 21-30.
1030   Wagner, S., Jaffe, R., Stubbins, A., 2018. Dissolved black carbon in aquatic ecosystems.
1031   Limnology and Oceanography Letters, 3, 168-185.
1032   Ward, C.P., Sleighter, R.L., Hatcher, P.G., Cory, R.M., 2014. Insights into the complete and
1033   partial photooxidation of black carbon in surface waters. Environmental Science:
1034   Processes & Impacts, 16, 721-731.
1035   Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier
1036   spring increase western US forest wildfire activity. science, 313, 940-943.
1037   Whitman, T., Singh, B.P., Zimmerman, A.R., Lehmann, J., Joseph, S., 2015. Priming effects in
1038   biochar-amended soils: implications of biochar-soil organic matter interactions for carbon
1039   storage. Biochar for environmental management: Science, technology and
1040   implementation, 2, 455-488.
1041   Wiedemeier, D.B. et al., 2015. Aromaticity and degree of aromatic condensation of char.
1042   Organic Geochemistry, 78, 135-143.
1043   Wozniak, A.S. et al., 2020. Molecular heterogeneity in pyrogenic dissolved organic matter from
1044   a thermal series of oak and grass chars. Organic Geochemistry, 148, 104065.
1045   Zand, A.D., Grathwohl, P., 2016. Enhanced Immobilization of Polycyclic Aromatic
1046   Hydrocarbons in Contaminated Soil Using Forest Wood-Derived Biochar and Activated
1047   Carbon under Saturated Conditions, and the Importance of Biochar Particle Size. Polish
1048   Journal of Environmental Studies, 25.
1049