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1 **Title:** Replacing time with space: Using laboratory fires to explore the effects of repeated
2 burning on black carbon degradation

3
4 **Suggested Running Head:** Repeated burning on black carbon residence times

5
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21
22 **Abstract**

23 Soil organic matter plays a key role in the global carbon cycle, representing three to four times
24 the total carbon stored in plant or atmospheric pools. Although fires convert a portion of the
25 faster cycling organic matter to slower cycling black carbon (BC), abiotic and biotic degradation
26 processes can significantly shorten BC residence times. Repeated fires may also reduce residence
27 times, but this mechanism has received less attention. Here we show that BC exposed to repeated
28 experimental burns is exponentially reduced through four subsequent fires, by 37.0, 82.5, 98.6,
29 and 99.0% respectively. Repeated burning can thus be a significant BC degradation mechanism,
30 particularly in ecosystems where burning rates are high, relative to BC soil incorporation rates.
31 We further consider loss rates in the context of simulated BC budgets, where 0-100% of BC is

32 protected from subsequent fires, implicitly representing ecosystems with varying fire regimes
33 and BC transport and incorporation rates. After five burns, BC storage was as much as 338%
34 lower than predicted if degradation from burning was ignored. These results illustrate the
35 importance of accounting for BC loss from repeated burning, further highlighting the potential
36 conflict between managing forests for increasing soil carbon storage vs. maintaining historic fire
37 regimes.

38

39 **Keywords:** *carbon storage, ecosystems, fire regimes, soil incorporation, CTO-375*

40

41 **Table of Contents Summary:**

42 We highlight via a laboratory experiment the potential of repeated burns to reduce black carbon
43 residence times. Our results indicate that for black carbon that remains in-situ to be most
44 effective as a net carbon sink, it must be incorporated deep into the organic layer or into the
45 mineral soil matrix prior to subsequent burning.

46

47

48 **Introduction**

49 Globally, soil carbon represents a large and persistent carbon sink, partially offsetting
50 increasing concentrations of atmospheric carbon dioxide (Schimel *et al.* 1994; Forbes *et al.*
51 2006). Biomass burning converts a portion of the faster cycling organic matter to slower cycling
52 black carbon (Lehmann *et al.* 2008) that has been shown to be resistant to biologic degradation
53 due to changes in its chemical structure (Schmidt and Noack 2000; Masiello 2004). Increased
54 fire activity is predicted for many ecosystems in response to climate change and will likely lead
55 to significant changes in the global carbon cycle (Chapin III *et al.* 2000; Westerling *et al.* 2006;
56 Moritz *et al.* 2012; Flannigan *et al.* 2013; IPCC 2013). Given the potential for long-term carbon
57 storage, this process has received significant attention in recent decades (Goldberg 1985; Preston
58 and Schmidt 2006), especially given the growing interest in black carbon's contribution to the
59 soil organic carbon (SOC) pool which can exceed 50% in some ecosystems (Lehmann *et al.*
60 2008; Schmidt *et al.* 2011; Santín *et al.* 2015a).

61 Radiocarbon dating of black carbon in soils have shown residence times on the order of
62 100's~1000's of years (Schmidt *et al.* 2002, 2011; Preston *et al.* 2006; Kuzyakov *et al.* 2009;
63 Zimmerman 2010). However, abiotic and biotic degradation processes can significantly shorten
64 black carbon residence times (Czimczik and Masiello, 2007; Zimmerman, 2010). Repeated
65 biomass burning has been postulated to also reduce residence times of black carbon (Ohlson and
66 Tryterud, 2000; Rovira *et al.* 2009), but this mechanism has received less attention (Santín *et al.*
67 2013). However, in forested ecosystems that have a high propensity to produce black carbon
68 through combustion of woody fuels (Hurteau and Brooks, 2011), the typical time between fires is
69 of the order of decades to centuries. This temporal scale limits the ability to observe and quantify
70 this process in a field setting, thus emphasizing the need for an experimental approach.

71

72 **Black Carbon Cycling**

73 Fire altered carbon from biomass burning exists on a recalcitrance gradient, from
74 partially charred vegetation to soot (Masiello 2004; Smith and Hudak 2005; Preston and Schmidt
75 2006; Keiluweit *et al.* 2010). Given research into black carbon covers soil science,
76 biogeochemistry, and fire science; differing terminology occurs and can lead to some confusion.
77 As described in Preston and Schmidt (2006), the term pyrogenically altered carbon is used to

78 describe the entire spectrum of carbon that has been thermally altered by the fire, whereas black
79 carbon (BC) often only refers only to a narrow range of pyrogenic carbon produced through
80 thermal exposure. Specifically, BC is defined here as the highly recalcitrant material that exhibits
81 significantly lowered H:C and O:C ratios (Kuhlbusch *et al.* 1996; Hatten and Zabowski, 2009;
82 Brewer *et al.* 2013). Pyrogenic organic matter (PyOM) is another widely used term that has been
83 used to describe black carbon (Santin *et al.* 2015a). A source of confusion may arise from the
84 term pyrogenic carbon, which is used to describe black carbon (Santin *et al.* 2013) due to its
85 closeness to the Preston and Schmidt (2006) terminology. In this study, we do not advocate the
86 usage of a specific set of terminology but rather elect to follow the terminology associated with
87 the analytical BC calculation method used in this paper (Hatten and Zabowski, 2009).

88 The coupling of BC production, soil incorporation rates, and BC loss via combustion
89 creates a complex relationship between the frequency of burning in an ecosystem and total BC
90 storage. The molecular composition alone cannot singularly predict carbon's persistence in soils;
91 rather, ecosystem processes and soil matrix interactions play important roles in its longevity
92 (Czimczik and Masiello 2007; Jenkins *et al.* 2014). Specifically, the physical protection of BC,
93 for example via soil incorporation or off-site transport, is required for it to be preserved over
94 geologic time scales (DeLuca and Aplet 2008; Fang *et al.* 2014). With each new fire BC is
95 generated, but the existing BC from prior fires may also be consumed if it remains near the soil
96 surface and exposed to the fire (Ohlson and Tyrterus 2000; Czimczik *et al.* 2003; Rovira *et al.*
97 2011). Two recent field two studies measured consumption of pre-existing BC (charcoal) by fire
98 in contrasting environments. Santin *et al.* 2013 found median mass losses <15% of BC samples
99 placed within the surface of the organic layer consumed in a boreal forest fire and Saiz *et al.*
100 (2014) reported average mass losses <8% of BC and on the soil surface in a prescribed fire in
101 open savannah woodland. Schmidt and Noack 2000) suggest that thermal degradation not
102 usually achieved at mineral soil depths > 30 mm. However, research quantifying soil
103 incorporation rates of pyrogenic carbon are limited (Nocentini *et al.* 2010). Preliminary
104 projections suggest that it takes decades to centuries for fire residues to be adequately
105 incorporated into a soil matrix to ensure protection from fires (Eckmeier *et al.* 2007; Lehmann *et*
106 *al.* 2008), considerably longer than fire return intervals in most dry temperate forests (Littel *et al.*
107 2009). Given fire frequencies in many ecosystems are expected to change due to climate

108 (Flannigan *et al.* 2013), it is important to address how repeated burning may influence BC pools
109 (Czimczik and Masiello 2007; Schmidt *et al.* 2011; Santin *et al.* 2015b). For the first time, we
110 present results from a controlled laboratory experiment to quantify how exposure to repeated fire
111 events degrades BC pools. We hypothesize that neglecting the BC losses associated with
112 repeated burning may lead to significant overestimates of soil BC storage. In turn, accounting for
113 BC losses as a function of repeated burning will lead to significantly less BC available for
114 incorporation into the passive SOC pool, and these losses will likely have significant impacts on
115 net soil BC estimates.

116

117 **Methods**

118 ***Sample collection and construction of fuel beds***

119 To test our hypothesis we selected woody surface fuels that had been controlled for their
120 particle size and moisture content, allowing for a controlled laboratory combustion experiment.
121 The constructed fuel beds represented a so-called masticated fuel matrix within a western North
122 American temperate conifer forest, dominated by western white pine (*Pinus monticola*),
123 Douglas-fir (*Pseudotsuga menziesii*), and lodgepole pine (*Pinus contorta*). Full details on sample
124 collection and fuel bed construction are outlined in a prior study (Brewer *et al.* 2013). Fuels were
125 collected from an 8 ha stand within the Clearwater National Forest (latitude: 46.80N, longitude:
126 119.47 W) that included white pine (*Pinus monticola*), Douglas Fir (*Pseudotsuga menziesii*), and
127 lodgepole pine (*Pinus contorta*). During mastication the woody particles were chipped into
128 predominately small-diameter particles (<7.6 cm). Fuels were collected following the sampling
129 protocols in Hood and Wu (2006). Fifteen fuel beds were constructed representing a typical
130 woody surface fuel loading (5835 g m⁻²) observed in mesic mixed conifer systems of the
131 northwestern United States (Kreye *et al.* 2014; Sparks *et al.* in review).

132

133 ***Initial burn methodology***

134 Fire experiments were conducted at the Idaho Fire Initiative for Research and Education
135 (IFIRE) laboratory located in a climatically controlled environment, shielded from weather
136 effects (Brewer *et al.* 2013; Smith *et al.* 2013). The experimental burn and residue collection and
137 analysis methodologies followed the procedures detailed by Brewer *et al.* (2013). Burns were

138 considered to be complete when mass loss had ceased, as measured with a Sartorius EB Series
139 scale (precision: 1 g, range: 0.0005-65.0000 kg, Goettingen, Germany). Following combustion,
140 post-fire residues were sieved into > 6 mm, 1-6 mm, and < 1 mm size classes and weighed using
141 a Sartorius scale (precision: 0.1 g, range: 0.1-2,000.0 g), with two ~1 g sub-samples collected for
142 BC proportion analysis. BC was quantified using thermo-chemical methods adapted from
143 CTO375 protocols, which isolates the biologically resistant portion of the pyrogenic carbon
144 (Hatten and Zabowski 2009; Sánchez-García *et al.* 2012). Following elemental analysis of BC
145 proportions ($CTO375_{BC(\%)}$), the BC mass (BC_{mass} , g) was calculated by (Equation 1; Hatten and
146 Zabowski 2009):

$$147 \quad 148 \quad 149 \quad BC_{mass} = CTO375_{BC(\%)} * [pre_{mass} - post_{mass}] \quad [1]$$

150 Where, pre_{mass} (g) and $post_{mass}$ (g) are the original fuel loading and mass of post-fire residues
151 respectively.

152

153 ***Repeated burns methodology***

154 To quantify how the $post_{mass}$ residues from the initial burns persisted under subsequent
155 burns, each burnt fuel bed (n=15) was subsequently exposed to four consecutive burn trials. We
156 acknowledge that ideally we should observe BC incorporation and losses in a field setting (e.g.,
157 Santin *et al.* 2013), however we contend that this experimental approach allows us to simulate
158 long fire return intervals that otherwise could only be inferred from modeling studies. In each of
159 the subsequent burn trials a consistent litter fall was included as a layer of solely pine needles
160 consisting proportionally (by mass) of lodgepole pine and ponderosa pine (*Pinus ponderosa*),
161 which are common early seral species in the study region. The pine needles were added to the
162 top of the fuel bed and selected in lieu of other litter components due to ease of replication, with
163 the mass of needles increased to account to other missing components (e.g. leaves, twigs). These
164 pine needle fuel beds were constructed to resemble an upper limit of litter fuel loading (1,700 g
165 m^{-2}) in temperate conifer forests throughout western U.S. and Canada (Law *et al.* 2003, Hyde *et al.*
166 2011). Fuel moisture was controlled by placing prepared fuel beds in a drying oven prior to
167 combustion (Table 1a; Brewer *et al.* 2013).

168 In each repeated burn after the initial characterization, only the residues > 6 mm in size
169 underwent additional elemental analysis for assessing BC proportions and were carried forward
170 into the subsequent burn trials where these particles were mixed throughout the pine needle fuel
171 beds. Residues < 6 mm were not carried forward, as these were indistinguishable from the newly
172 burnt pine needles. To calculate percent BC remaining [Equation 2] in subsequent burns, BC
173 masses were standardized against BC produced in the initial burn, by:

$$BC_{Ri} = \frac{BC_1 - BC_i}{BC_1} \quad [2]$$

176
177 Where, BC_{Ri} is the normalized remaining BC after burn number i , BC_1 is the BC
178 produced from burn number 1, BC_i is the remaining BC after burn number i , and i represents the
179 burn number from 2 to 5. Since grass fires likely produce charcoal residues smaller than 6 mm,
180 the conclusions of this experiment are limited to ecosystems with woody vegetation (i.e. trees
181 and shrubs), which represents a continuum from savannah to mesic forest.

182
183
184

185 ***Statistical analysis and modeling***

186 A repeated measures ANOVA was used to test for differences in post-fire residue and BC
187 masses for each burn trial. When Mauchly's sphericity assumption was not met, the Greenhouse-
188 Geisser statistic was used. A Bonferroni post-hoc test was used to compare the main effect of
189 burn number. To generalize our results and construct a range of partially protected BC budgets,
190 we fit a negative exponential model, robust to outliers, to the individual mass-loss percentages of
191 the exposed > 6 mm residues: $y = a^{bx}$, where y is the predicted remaining BC mass (% of the
192 original mass), and x is the burn number (where $x = 1$ for the initial burn). When $a = 259.0$
193 (239.5 - 278.6, 95% prediction bounds) and $b = -0.9476$ (-1.008 - -0.8877) this model explained
194 98% of the variability in our observed data. The observed BC mass loss rate was used to develop
195 a BC budget through the five burns. Further, a series of hypothetical BC budgets, representing
196 varying degrees (0-100%) of protection of the total BC produced in the initial burn are produced

197 and discussed. Statistical analyses were conducted using IBM SPSS predictive analytics software
198 (version 19), and Matlab technical computing software (version 7.11.1). For this modeling the
199 BC produced in the combustion of the added litter layer is not included, as it is indistinguishable
200 from the fine woody BC.

201

202 **Results and Discussion**

203 Following the initial burns, ~61, 31, and 8% of the BC produced were in the < 1 mm, 1-6
204 mm, and > 6 mm residue size classes respectively. The > 6 mm BC from the initial burns
205 declined significantly across the four subsequent burns (Table 1b, Figure 1). Following Burn 5,
206 only 1% of the exposed BC produced in the initial burn (Burn 1) remained. Integrating the
207 charred residues throughout the litter layer probably made the charred particles more susceptible
208 to thermal degradation. However, in a field setting charred residues could remain at the interface
209 of the O-horizon and mineral soil or the material could be mixed throughout the O-horizon
210 depending on the pedoturbation intensity, eventually the char will become incorporated into
211 mineral soil (Gavin 2003). Consequently, our estimates are likely to represent an upper bound of
212 losses associated with repeated burning, however more research is needed on pedoturbation
213 within the O-horizon. Moving the charred particles to the bottom of the litter horizon or placing
214 them below the soil surface to simulate soil incorporation could have resulted in lower loss rates
215 (Santín *et al.* 2013). Regardless, our results support the prevailing hypothesis that in forest types
216 with high-frequency burning and little to no soil incorporation, repeated burning can be a
217 significant mechanism for BC loss (Ohlson and Tryterud 2000; Preston and Schmidt 2006;
218 Rovira *et al.* 2009). Our results also indicate that in as few as two repeated burns, the majority
219 (~80%, Table 1b, Figure 1) of the exposed BC produced in an initial fire can be lost. Given that
220 the best estimates of BC loss rates through biotic and abiotic (non-pyric) mechanisms range from
221 < 1% to 37% over 100 years (Zimmerman 2010), our results highlight repeated burning as a
222 potentially significant mechanism of carbon loss.

223 To estimate the compounding effects of BC loss through repeated burning, we used the
224 experimental loss rates of the exposed carbon (Figure 1) to construct a BC budget spanning five
225 burns (Figure 2; Ohlson and Tryterud 2000; Czimczik and Masiello 2007; Zimmerman 2010).
226 This budget assumes all material < 1 mm is protected from future burning, implicitly

227 representing immediate off-site wind transport like is common in grassland and savannah
228 ecosystems or incorporation of fine particles into the soil and litter matrix (Rumpel *et al.* 2009).
229 Protecting the < 1 mm BC left 39% of the BC produced in an initial burn available for further
230 thermal degradation. This scenario also assumes that an equivalent masticated fuel loading was
231 reached between each burn, representing a best case estimate for dry forest types, were net BC
232 increases at a near-linear rate of approximately 70% per burn (Figure 2). By the fifth burn, BC
233 storage is 381% of that created in the initial burn, whereas an estimate ignoring BC loss from
234 repeated burning would predict a value of 500%. The “missing” 119% represents the tradeoff
235 between generating new BC while consuming existing BC with each successive burn. We
236 acknowledge that a full accounting of the residual size categories would provide more accurate
237 estimates of BC loss rates, however given analytical limitations of separating these fine char
238 fractions from pine needle residues in the repeated burns, this was not feasible.

239 The degree to which repeated burning impacts a BC budget inherently reflects
240 assumptions on BC protection from future burning (e.g., via soil incorporation or off-site
241 transport; Rumpel *et al.* 2009; Dittmar *et al.* 2012; Santín *et al.* 2013). To generalize our results
242 and explore this sensitivity, we used the fitted model in Figure 1 to calculate net BC storage
243 under five different scenarios, where 0%, 25%, 50%, 75%, and 100% of the BC produced in the
244 initial burns is protected from future burning (Figure 3). As expected, BC budgets are highly
245 sensitive to protection rates: at 0% protection, net BC asymptotes around 160% after four burns,
246 implying an upper limit to BC storage. Under the scenario with 75% BC protected, values fail to
247 asymptote and reach 416% following the fifth burn (Figure 3).

248 Implicitly, these scenarios represent a series of generalized environmental conditions that
249 dictate both BC protection rate (% yr⁻¹) and the rate of burning (fire yr⁻¹) in a given ecosystem.
250 For example, while savanna ecosystems are frequented by fires every 3-5 years, mesic forests
251 can exhibit fire return intervals of 200+ years. Although less well known, soil incorporation and
252 off-site wind transfer rates of BC in these systems likewise vary (Rumpel *et al.* 2009; Nocentini
253 *et al.* 2010; Kasin and Ohlson 2013), principally due to differences in soil exposure, topography,
254 precipitation, temperature, and wind regimes. If burning rates are faster than the protection rate
255 required to safeguard BC, then most BC produced in a burn will still be exposed during
256 subsequent burns. This may represent a scenario of 0% or 25% BC protection, which may be

257 comparable to the short fire return interval found in savannah systems if off-site transport
258 through wind and water erosion is minimal; a scenario that may not be commonplace given these
259 particles are often considered very susceptible to offsite transport by wind and water processes in
260 grasslands and savannahs. In contrast, the longer fire return intervals of boreal forests may allow
261 time for BC to be incorporated far enough into the litter and duff or mineral soil to be protected
262 during subsequent burns; this ecosystem is more likely represented by a scenario of 75% or
263 100% BC protection (Santín *et al.* 2015).

264 The ratio of these two processes (fires yr^{-1}), i.e., the rate of burning (fire yr^{-1}) and rate of
265 BC protection ($\% \text{ yr}^{-1}$), will determine the sensitivity of soil BC budgets to repeated burning.
266 Assessment of this ratio will enable studies to determine if and when net BC storage reaches an
267 asymptote, but will require understanding the protection and production rates of residues of
268 different sizes in different ecosystems. If frequent fire consumes black carbon faster than it can
269 be protected by soils or off-site transport, then those forested systems with short fire return
270 intervals would have lower amounts of black carbon compared to forested systems with long fire
271 return intervals. While unexplored in their study, this hypothesis is generally supported by Jaus
272 *et al.* (2015) whose data show that the black carbon content of the forested soils has a significant
273 relationship with fire return interval ($r^2=0.452$). However, the per-fire production rate of black
274 carbon is a function of other factors such as fire severity, fire return interval, and net primary
275 productivity which could account for some of the unexplained variability in this relationship.

276 Ultimately, these processes likewise dictate the feasibility of using fuel treatments and
277 fire hazard management (prescribed, wildland fire use fires, etc.) as tools to increase soil BC
278 storage (Deluca and Aplet 2008; Santín *et al.* 2015b). Given the tradeoff between BC production
279 and consumption in subsequent burns, maintaining “natural” fire regimes in ecosystems
280 historically characterized by high-frequency fires (e.g., burning once every several years to
281 decades), may be at odds with maximizing soil BC storage.

282

283 **Conclusion**

284 This study highlights the potential importance of physical degradation of BC through
285 repeated burning, adding combustion as a key mechanism to previous work demonstrating BC
286 loss through biological and physical degradation (Preston and Schmidt 2006; Schmidt *et al.*

287 2011). For BC that remains in-situ to be most effective as a net carbon sink, it must be
288 incorporated deep into the organic layer or into the mineral soil matrix prior to subsequent
289 burning. Our work is a first step towards quantifying BC loss rates from repeated burning to
290 more accurately model long-term BC storage in soil organic pools, but projecting the long-term
291 impacts on carbon budgets requires more precise estimates of BC protection rates (Santin et al.
292 2015b). Biogeochemical models that track BC should be sensitive to the combined effects of
293 burning, soil incorporation, and off-site transport rates, as exemplified by the ratio of these
294 processes.

295

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301

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425 **Table 1.** (A) Mean (SD) fuel bed characteristics and burn conditions for the five experimental
 426 burns (n = 15 replicates for each burn number). (B) Median (SD) production and loss rates
 427 associated with the post-fire residues and BC (n=15). Burn 1 represents the initial fire, which
 428 produced all of the > 6 mm residues used in the subsequent burns represented as burns 2-5. All
 429 p-values <0.001 reported from repeated measures ANOVA. Homogenous subsets as identified
 430 by Bonferonni post-hoc analysis are identified as a, b, c, d, and e.
 431

A)	Burn Number	Bulk Density (kg m ⁻³)	Fuel Loading (g m ⁻²)	Consumption (%)	Fuel Moisture (%)	Temperature (°C)	Relative Humidity (%)
	1	102.1 (9.3)	5829.7 (211.7)	90.6 (2.6)	10.0 (3.5)	17.8 (7.3)	38.4 (12.3)
	2	58.7 (6.3)	2107.2 (242.6)	45.5 (13.1)	9.7 (3.9)	16.7 (2.7)	36.1 (11.0)
	3	48.8 (8.4)	1771.9 (201.7)	57.4 (18.1)	11.0 (3.3)	21.5 (2.9)	34.3 (5.2)
	4	45.9 (4.4)	1752.5 (97.7)	57.3 (15.4)	9.5 (4.6)	25.4 (5.2)	30.2 (6.9)
	5	52.5 (16.9)	1763.3 (128.4)	61.8 (18.1)	10.2 (4.7)	21.5 (2.3)	33.6 (6.4)

B)	Burn Number	Residues (g m ⁻²)	Residue Remaining (%)	BC (g m ⁻²)	BC Remaining (%)
	1	198.1 (77.3) a	100.0 (0.0)	0.0650 (0.0290) a	100.0 (0.0)
	2	113.7 (50.9) b	55.6 (17.1)	0.0400 (0.0180) b	63.0 (23.0)
	3	78.8 (46.6) c	38.5 (15.0)	0.0140 (0.0100) c	17.5 (13.4)
	4	54.8 (34.9) d	24.2 (13.3)	0.0010 (0.0006) d	1.4 (0.8)
	5	39.5 (30.7) e	15.7 (13.3)	0.0007 (0.0004) d	1.0 (0.6)

433 **Figure 1.** BC loss with repeated burning. Burn number 1 represents total carbon produced after
434 the initial burn (100%). For each repeated burn (Burn number 2-5), 15 replicates are shown as
435 observations. The solid black line represents a robust fit of the model $y = a e^{bx}$, and the dashed
436 lines represent 95% prediction intervals. The fitted model explains 98% of the variability in the
437 observations ($r^2_{\text{adj}} = 0.98$), when $a = 259.0$ and $b = -0.9476$.

438
439 **Figure 2.** BC budget based on experimental observations. The net BC contributed to the budget
440 of each burn number is tracked via varying shades of grey. Each burn contributes 100% of the
441 BC generated in the initial burn, and 61% (39%) of this BC is protected from (exposed to)
442 degradation in subsequent burns.

443
444 **Figure 3.** BC budgets based on varying modeled scenarios. Using the fitted model in Figure 1 to
445 calculate loss rates, each budget assumes that a varying level of BC is protected from subsequent
446 burning (from 0-100%). Each scenario implicitly represents varying soil incorporation rates and
447 fire frequencies across a range of fire-prone ecosystems.

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