

Quantifying the Total and Bioavailable Polycyclic Aromatic Hydrocarbons and Dioxins in Biochars

Sarah E. Hale,^{†,*} Johannes Lehmann,[‡] David Rutherford,[§] Andrew R. Zimmerman,^{||}
Robert T. Bachmann,[†] Victor Shitumbanuma,[#] Adam O'Toole,[▽] Kristina L. Sundqvist,[○]
Hans Peter H. Arp,[†] and Gerard Cornelissen,^{‡,◆,¶}

[†]Department of Environmental Engineering, Norwegian Geotechnical Institute NGI, P.O. Box 3930 Ullevål Stadion, N-0806, Oslo, Norway

[‡]Department of Crop and Soil Sciences, 908 Bradfield Hall, Cornell University, Ithaca, New York 14853, United States

[§]U.S. Geological Survey, MS 408, PO Box 25046, Building 95, Denver Federal Center, Denver, Colorado 80225, United States

^{||}Department of Geological Sciences, University of Florida, 241 Williamson Hall, P.O. Box 112120, Gainesville, Florida 32611-2120, United States

[†]Malaysian Institute of Chemical and Bioengineering Technology (MICET), Universiti Kuala Lumpur, Lot 1988, Taboh Naning, 78000 Alor Gajah, Malaysia

[#]Department of Soil Science, University of Zambia, PO Box 32379, Lusaka, Zambia

[▽]Soil and Environment Department, Bioforsk, Frederik A Dahls vei 20, NO-1432 Ås, Norway

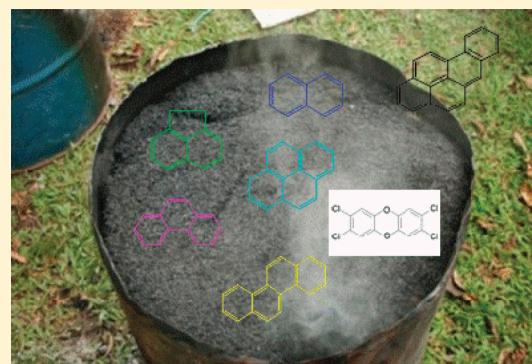
[○]Department of Chemistry, Umeå University, SE-901 87 Umeå, Sweden

[◆]Department of Applied Environmental Sciences, Stockholm University, 10691 Stockholm, Sweden

[¶]Department of Plant and Environmental Sciences, University of Life Sciences, 5003 Ås, Norway

Supporting Information

ABSTRACT: Biochar soil amendment is advocated to mitigate climate change and improve soil fertility. A concern though, is that during biochar preparation PAHs and dioxins are likely formed. These contaminants can possibly be present in the biochar matrix and even bioavailable to exposed organisms. Here we quantify total and bioavailable PAHs and dioxins in a suite of over 50 biochars produced via slow pyrolysis between 250 and 900 °C, using various methods and biomass from tropical, boreal, and temperate areas. These slow pyrolysis biochars, which can be produced locally on farms with minimum resources, are also compared to biochar produced using the industrial methods of fast pyrolysis and gasification. Total concentrations were measured with a Soxhlet extraction and bioavailable concentrations were measured with polyoxymethylene passive samplers. Total PAH concentrations ranged from 0.07 $\mu\text{g g}^{-1}$ to 3.27 $\mu\text{g g}^{-1}$ for the slow pyrolysis biochars and were dependent on biomass source, pyrolysis temperature, and time. With increasing pyrolysis time and temperature, PAH concentrations generally decreased. These total concentrations were below existing environmental quality standards for concentrations of PAHs in soils. Total PAH concentrations in the fast pyrolysis and gasification biochar were 0.3 $\mu\text{g g}^{-1}$ and 45 $\mu\text{g g}^{-1}$, respectively, with maximum levels exceeding some quality standards. Concentrations of bioavailable PAHs in slow pyrolysis biochars ranged from 0.17 ng L^{-1} to 10.0 ng L^{-1} which is lower than concentrations reported for relatively clean urban sediments. The gasification produced biochar sample had the highest bioavailable concentration ($162 \pm 71 \text{ ng L}^{-1}$). Total dioxin concentrations were low (up to 92 pg g^{-1}) and bioavailable concentrations were below the analytical limit of detection. No clear pattern of how strongly PAHs were bound to different biochars was found based on the biochars' physicochemical properties.



INTRODUCTION

Biochar, the carbonaceous material produced during the pyrolysis of biomass can contribute positively to the mitigation of climate change, in waste management strategies, to energy production, and for soil improvement.¹ When added to soil, biochar can increase pH, increase cation exchange capacity,² improve physical properties (such as aggregation, porosity,

aeration and water holding capacity), and adsorb hydrophobic organic pollutants,³ and therefore ultimately increase soil

Received: November 8, 2011

Revised: January 27, 2012

Accepted: February 9, 2012

Published: February 9, 2012

productivity.⁴ Biochar is a deliberately generated black carbonaceous material and is not to be confused with atmospheric black carbon particles that have complex effects on the climate,^{5,6} (although such particles can be a small byproduct of biochar production). There are also potential complications regarding biochar implementation that remain under investigation. Because of their sorption affinity for organic compounds, biochars can immobilize herbicides in unpredictable ways, which may yield unexpected outcomes when following standard agricultural practices.^{3,7,8}

In addition, the inherent nature of the biochar production process makes the formation of polycyclic aromatic hydrocarbons (PAHs)^{9–11} likely and the formation of dioxins and furans (PCDD/Fs)¹² possible. During pyrolysis, organic compounds contained in the biomass are partly cracked into smaller and unstable fragments. These fragments are composed of highly reactive free radicals that combine and lead to more stable PAHs through recombination reactions. Dioxins are mostly formed on solid surfaces when the pyrolysis temperature is 200–400 °C and the pyrolysis times are seconds.¹³ Often the presence of chlorine from, for example, salt (NaCl) or polyvinylchloride enhances the observed levels of dioxins.¹⁴ To date there have been no studies to address this important issue in depth.

Both from a regulatory and environmental perspective, using a soil amendment material that contains PAHs and dioxins that pose a threat to the soil is unacceptable. From a regulatory perspective, guideline values of toxin concentrations that should not be exceeded in order to protect the environment exist in many developed countries. For example, benzo(a)pyrene, a common toxic PAH has a maximum acceptable soil concentration of 0.6, 3, 2–10, 0.5, 0.15, and 0.1–7.5 µg g⁻¹ in Canada,¹⁵ Denmark,¹⁶ Germany,¹⁷ Norway,¹⁸ the UK,¹⁹ and other European Countries.²⁰ A full list of soil environmental quality standards for PAHs is given in Table S1 in the Supporting Information (SI). A Swedish guideline value for dioxin contamination in nonsensitive land use has been set at 250 µg g⁻¹ soil toxic equivalents (TEQ).²¹ Another important legislative point relates to regulation surrounding the reuse of wood ash as a fertilizer as this could be analogous to using some biochars for agricultural improvement. For example, the Finnish Ministry of Agriculture and Forestry stipulates that the highest total PAH content that is permitted in an ash product intended to be spread on forest land is 2 µg g⁻¹. In many Nordic countries large amounts of wood ash is generated annually in forest industry and energy plants and some of this is used as a forest fertilizer. Many of the currently advocated biochars are not from wood but from agricultural waste, therefore it is unclear to what extent these guidelines apply to biochars from wood feedstock.

As useful as these guideline values are, they only give a very rough approximation of actual environmental quality, as they are based on the total pollutant concentration in the soil, and not on the bioavailable concentrations, which are more closely related to risk.^{22,23} It is common practice when assessing environmental risk to use the total pollutant concentration as a proxy for the level of contamination, but evidence has been presented to show that bioavailable pollutant concentrations are more relevant.^{22,26,27} A bioavailable compound has been defined as one that is freely available to cross an organism's cellular membrane from the medium at a given time.²⁴ Once this transfer has occurred, storage, assimilation and degradation processes can take place within the organism. In addition, these

bioavailable compounds are more labile and able to leach to the surrounding areas. Currently only one upper acceptable total PAH concentration in biochar of 16 µg g⁻¹ has been recommended if the biochar is to be sold and marketed as having desirable properties.²⁵

There are very few studies that have evaluated the total and bioavailable concentration of PAHs and dioxins in biochars. Previous research has focused on PAHs (most often the 16 priority PAHs defined by the U.S. Environmental Protection Agency (USEPA)) in the syn-gas^{28–32} and bio-oil,^{33–36} products. Those studies that have chemically extracted a limited number of biochars^{19,37–43} have reported total PAH concentrations (Σ 16 USEPA PAHs) from negligible⁴¹ up to 45 µg g⁻¹.³⁸ The results of these studies are shown in Table 1. The biomass feedstock, pyrolysis temperature^{37,44} and the biochar production method^{37,39} were important in determining concentrations and distribution of PAHs by ring size. However, the results from these studies were not always clear and consistent. Brown et al.,³⁷ observed that low-temperature chars contained higher relative concentrations of low-molecular weight, high-vapor pressure PAHs. However, the opposite was observed by Nakajima et al.,⁴⁴ as higher pyrolysis temperatures resulted in a dominance of these PAHs. There has only been one study that attempted to quantify the concentration of bioavailable PAHs.⁴⁵ A wood chip biochar produced at 450 °C was extracted with a supercritical water method but did not result in any detectable PAHs. Studies have quantified dioxins in biochars produced via gasification and combustion,^{46,47} but not via pyrolysis.

To address this important knowledge gap, we measured total and bioavailable PAHs and dioxins in a vast array of biochars. The 59 biochars were produced from 23 different feedstocks (both tropical, temperate and boreal), through slow pyrolysis using various laboratory and in situ equipment (household stoves, traditional kilns, muffle furnaces, double drum, modern slow pyrolysis units) and via fast pyrolysis and gasification.

MATERIALS AND METHODS

Biochars. Biochars were produced from digested dairy manure (DDM), food waste (FW), paper mill waste (PMW), corn stover (CS), wheat straw (WS), rubberwood sawdust (RWSD *Hevea brasiliensis*), loblolly pine (LPP), pine wood (PW *Pinus ponderosa*), switch grass (SG *Panicum virgatum*), laurel oak (*O. Quercus laurifolia*), loblolly pine (P *Pinus taeda*), eastern gamma grass (G *Tripsacum dactyloides*), hardwood (HW), heartland pine (HP), secondary mixed wood, corn cob, rice husk, maize residues, corn stover, sawdust, empty fruit bunches (EFB), and coconut shell (CCS). Biochars were produced at 250–900 °C, commercially, in the laboratory or in the field via slow pyrolysis, fast pyrolysis and gasification. Gasification is a partial oxidation process that aims to optimize the syngas yield in comparison to pyrolysis which is carried out in the almost complete absence of oxygen. The char used here had a residence time in the fluidized bed and thermal cracker (which comprise gasification) of 12.5 s. Slow and fast pyrolysis can be distinguished based on the length of time of the pyrolysis where fast pyrolysis encompasses those processes of only a few seconds. Biochars were ground and sieved to 300 µm. Biochar physicochemical properties and production methods are given in SI Table S2. SI Table S3 shows the time and conditions the biochars were stored under prior to use. This is important to consider because PAHs sorbed to biochar can show variable stabilities and may degrade even in the absence of

Table 1. Reported Concentrations of USEPA PAHs in Biochars from Different Source Materials, Produced under Different Conditions and Quantified Using Different Methods

biochar source material	biochar production method	PAH extraction method	PAH concentration ($\mu\text{g g}^{-1}$)	reference
wood chips (<i>Eucalyptus camaldulensis</i>)	wood chips and pea straw were heated at 450 °C for 1 h in a muffle furnace.	extraction for 1 h in an ultrasonic bath with 70 mL heptane. Extracts were filtered through a 0.5 μm poresize PTFE millipore filter	0.07 0.12 <0.01 (PHEN to B(ghi)P)	39
pea straw (<i>Pisum sativum</i>)	vegetation fire residues were collected 4 days after the controlled burning of native vegetation along the boundary of a nature reserve in Australia	soxhlet extraction with toluene:ethanol 3:7 for 16 h	0.08 – 0.11 0.06 – 0.12 0.08 – 0.15 (all PAHs except NAP)	44
vegetation fire residues (predominantly leaves and branches of eucalyptus trees)	furnace with carbonization temperatures of 400, 600, 800, and 1000 °C held at the final temperature for 1 h	analysis carried out by the National Institute of Standards and Technology	1.01 – 3.68	37
Japanese Hinoki cypress (<i>Chamaecyparis obtuse</i>)	pyrolysis at 450 – 1000 °C at various ramp rates and held at the final temperature for 30 min. Natural charring was via prescribed burning	toluene extraction with sample preparation according to Amtliche Sammlung § 35 LMBG Method L07/00–40	0.005 0.013	40
Japanese chestnut (<i>Castanea crenata</i>)	slow pyrolysis under a heating rate of 1.5 – 2 °C/minute to a final temperature of 600 °C	soxhlet extraction with dichloromethane for 12 h	Negligible	41
Moso bamboo (<i>Phyllostachys pubescens</i>)	data not given	soxhlet extraction with 70 mL of toluene for 16 h	45	38
xylem from a freshly cut pitch pine log (<i>Pinus rigida</i>) and naturally charred branches of pitch pine	slow pyrolysis at 400 or 550 °C with and without steam activation. Chars were pyrolyzed at 5 – 10 °C/minute heating rate and a 40 min residence time	soxhlet extraction with 70 mL of toluene for 16 h	45	38
birch				
pine				
<i>Eucalyptus saligna</i> wood, <i>Eucalyptus saligna</i> leaves, paper sludge, cow manure and poultry litter				
charcoal from the “combustion of bark” particles <150 μm				

light.⁴⁸ It is outside the scope of the present paper to investigate this fully, and we chose to deploy biochars as they would be used in practice.

Total PAH and Dioxin Concentrations in Biochar. Total PAH and dioxin concentrations were measured via triplicate Soxhlet extractions (0.5 g biochar, 90 mL of toluene, 6 h, 160 °C). Toluene was selected as the extraction solvent as it has previously been identified to give optimal recovery of PAHs from a biochar produced from the combustion of bark.³⁸ A small scale test was carried out before undertaking the extractions where toluene and toluene:methanol were compared. The highest PAH concentration (1.6 $\mu\text{g g}^{-1}$ for toluene and 1.0 $\mu\text{g g}^{-1}$ for toluene:methanol) and largest recovery of deuterated standard (average of 73% compared to 65% for toluene:methanol) was obtained when toluene was used as the extraction solvent. Prior to extraction, a mixture of deuterated PAHs (d_{10} phenanthrene, d_{10} pyrene, d_{12} benzo(*a*)anthracene, d_{12} benzo(*a*)pyrene and d_{12} benzo(*ghi*)perylene) were added to the extraction thimble to monitor the recovery of PAHs or 17 ¹³C-labeled dioxins and furans were added prior to clean up to monitor the recovery of dioxins. The recovery of PAHs ranged from 56 to 79% across 178 extractions. Individual PAH compounds were not corrected based on the recovery of just five standards. In some cases recovery was the lowest for the biochars produced at the highest temperatures, although clear trends across all of the biochar types were difficult to observe. The recovery of dioxins ranged from 80 to 97%. Following the Soxhlet extraction, the toluene was reduced to 1 mL, cleaned up with dimethylformamide:water 95:5⁴⁹ and then eluted with 10 mL heptane over a precleaned (6 mL heptane) silica gel column (3 cm, 10% deactivated silica gel) topped with sodium sulfate. The solvent was reduced to less than 1 mL and each sample was spiked with polychlorinated biphenyl 77 as an internal standard before gas chromatography mass spectroscopy (GCMS) analysis.⁵⁰ Total dioxins were quantified via GCMS following a multilayer silica column cleaning and fractionation on carbon/Celite and a clean up on a miniaturized multilayer silica column.⁵¹

Bioavailable PAH and Dioxin Concentration in Biochar. Polyoxymethylene (POM) passive samplers were precleaned with methanol (1 day), heptane (1 day), Millipore water (1 day) and rinsed with Millipore water prior to use. POM (0.4 g), either 55 μm thick (PAHs) or 17 μm thick (dioxins) were exposed to a mixture of biochar (0.2 g) and water (40 mL spiked at 1% volume NaN₃ from a 20 g/L stock) for 2 (PAHs) and 6 (dioxins) months, respectively (all carried out in triplicate). During this time, an equilibrium between the sampler and the water is reached and the bioavailable pollutant concentration can be obtained.⁵² Equilibrium is achieved more slowly for dioxins^{53,54} (4 months for POM-17) than PAHs⁵⁵ (10 days for POM-55), hence a thinner membrane was used. After this time, POM samplers were removed, cleaned, extracted with a 20:80 mixture of acetone:heptane for 2 days and the solvent cleaned up using a silica gel column before GCMS analysis. Prior to extraction, the deuterated PAHs were spiked to the solvent to monitor process recovery and the same 17 ¹³C-labeled dioxins and furans were used prior to clean up. Extraction recovery was within an acceptable 70–130% range for PAHs and from 87 to 107% for dioxins and furans. In addition, sample blanks were treated in the same way as biochar and POM samples and gave comparable recovery standard results. Bioavailable concentrations were calculated using predetermined POM-water partitioning coefficients specific to each

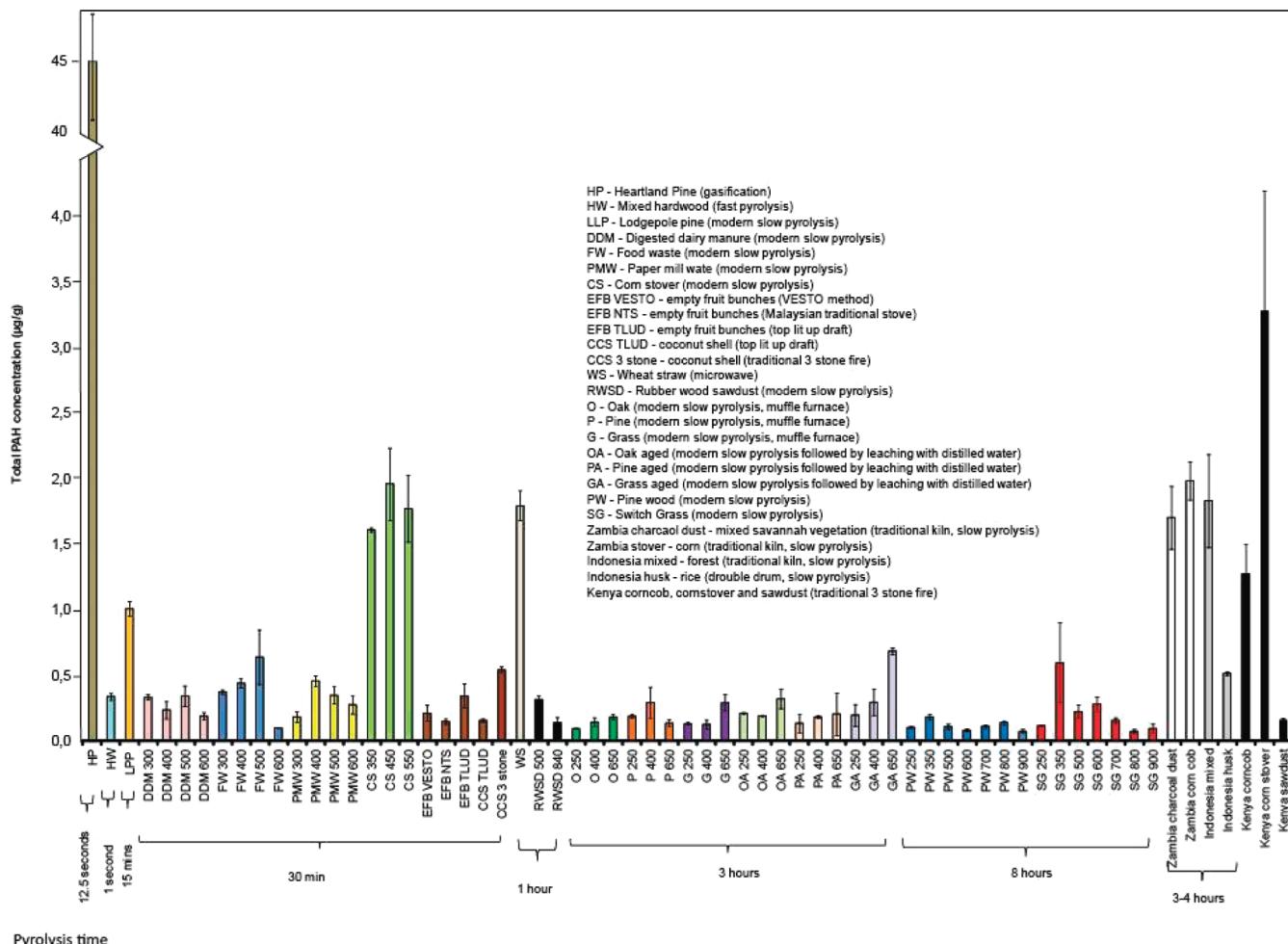


Figure 1. Total PAH concentration ($\mu\text{g g}^{-1}$) in biochars. Bars are grouped by color according to the biochar source material and country of production.

PAH⁵⁵ and dioxin⁵⁴ compound and the measured POM concentration according to $C_W = K_{POM}/C_{POM}$, where C_W ($\mu\text{g mL}^{-1}$) is the aqueous phase concentration (and is the pollutant concentration that is bioavailable), K_{POM} (mL g^{-1}) is the POM-water partitioning coefficient and C_{POM} ($\mu\text{g g}^{-1}$) is the measured POM concentration.

RESULTS AND DISCUSSION

Total PAH Concentrations. Figure 1 shows the total PAH concentration (16 USEPA compounds) in the biochars tested, with the commercial and lab samples grouped together and the field samples grouped together. SI Table S4 contains the concentration of individual PAHs in all of the biochars extracted. For the slow pyrolysis biochars, total PAH concentrations ranged from $0.07 \mu\text{g g}^{-1}$ (switch grass 800 and pine wood 900) to $3.27 \mu\text{g g}^{-1}$ (Kenya corn stover) and varied with biomass source. These total concentrations we measured fall within the range of concentrations in other biochars reported previously,^{19,37–41,43} with the exception of a much higher concentration in a charcoal obtained from the “combustion of bark”³⁸ (however details are not provided regarding the production and process conditions of this material). Adding biochar at 0.5 to 135 t/ha has been shown to produce positive soil enhancement responses.⁴ Even at doses greater than this, with the dose equaling the mass of soil, the levels of introduced PAHs and dioxins from these biochars

would not pose a concern. In general, the biochars with the highest PAH concentrations were those produced in uncontrolled field conditions (Zambia, Indonesia, and Kenya) in traditional kiln, drum, and stove setups. However, Corn Stover 350, 450, and 550 produced under more controlled high-tech pyrolysis conditions contained $1.5\text{--}2 \mu\text{g g}^{-1}$ total PAHs.

PAH concentrations were affected by pyrolysis time and temperature. Across the suite of biochars produced in more controlled conditions, PAH concentrations are generally higher in biochars pyrolyzed for shorter times. Concentrations decreased in the order: lodge pole pine > digested dairy manure ≈ food waste ≈ paper mill waste ≈ VESTO ≈ TLUD ≈ CCS ≈ rubber wood saw dust > oak ≈ pine ≈ grass > pine wood ≈ switch grass, as the pyrolysis times increased from 30 min to 1, 3, and 8 h. With respect to pyrolysis temperature, in general, the higher pyrolysis temperatures resulted in the lowest total PAH concentrations. The PAH concentration in pine wood 900 was statistically significantly lower (t test, $P = 0.05$) than the concentration at all other production temperatures (except 600 °C) and the PAH concentration in switch grass 900 was significantly lower than concentrations at all production temperatures except 800 °C. The greatest PAH concentrations were generally observed among biochars produced in the 350–550 °C range. This observation is in contrast to what has been observed previously (as PAH formation has been observed to increase at higher temperatures⁵⁷).

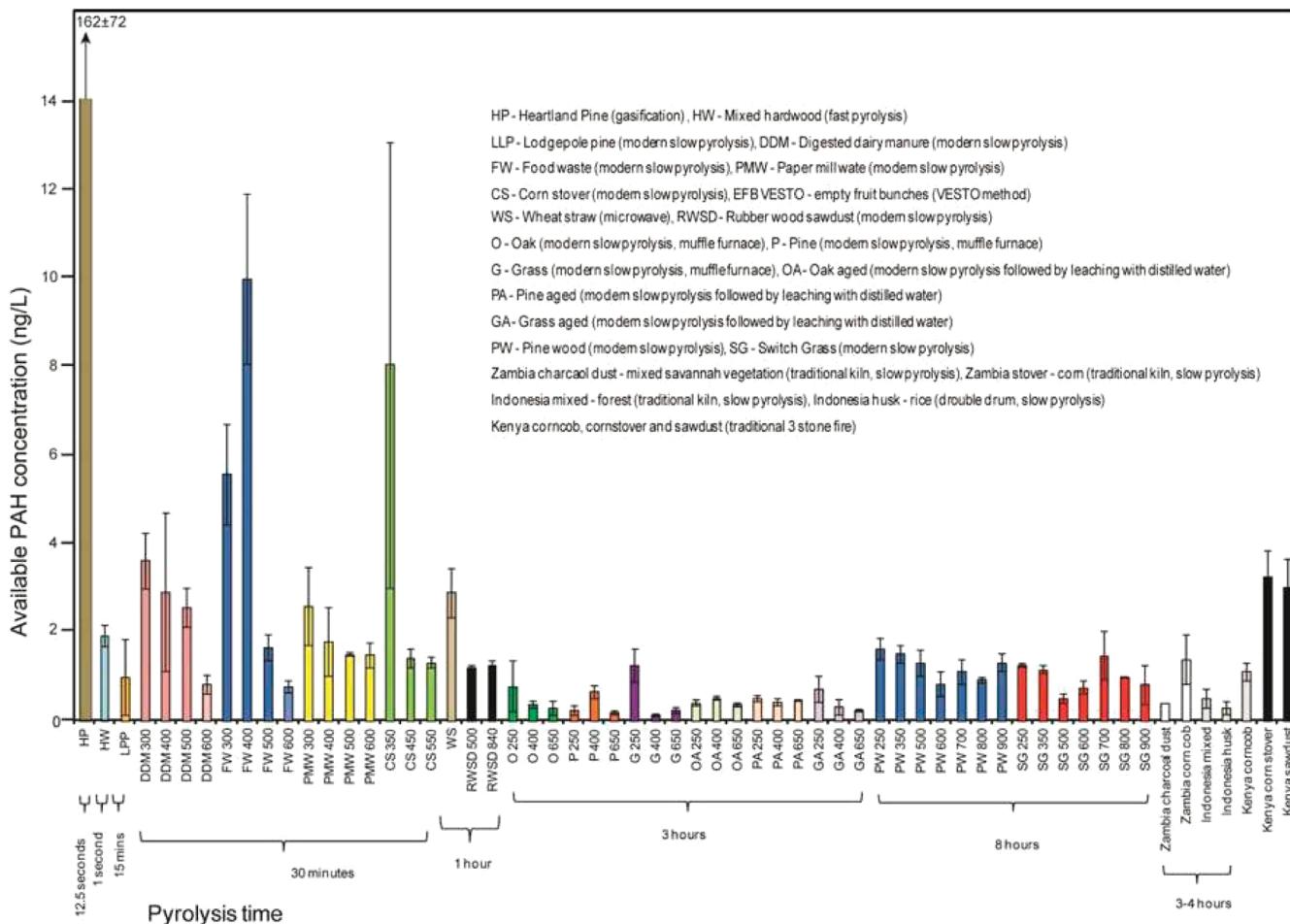


Figure 2. Concentration of bioavailable PAHs (ng L^{-1}) in biochars. Bars are grouped by color according to the biochar source material and country of production.

Artificially aged biochars obtained by successive batch leaching of oak, pine and grass with distilled deionized, UV oxidized water with DOC completely removed, for a period of 100 days in the lab (therefore not exposed to the atmosphere) were also examined. The concentrations of PAHs were greater in the aged materials than the unaged materials (except pine age 250 and pine age 400), and were statistically significant (t test, $P = 0.1$) for oak 250 and oak age 250, oak 650 and oak age 650 and grass 400 and grass age 400. Leaching likely removed the hydrophilic portion of the biochar source material and possibly ash minerals, but not the PAHs, leaving a PAH-enriched hydrophobic portion. The chemical changes that take place with such aging is the subject of a publication in preparation by Zimmerman et al.

The biochars produced via fast pyrolysis (hard wood) and via gasification (heartland pine) gave different results than for slow pyrolysis. The slow pyrolysis sample had a low total PAH concentration ($0.3 \mu\text{g g}^{-1}$), while the gasification sample had the highest total PAH concentration of any sample, at $45 \mu\text{g g}^{-1}$. We speculate that slow pyrolysis results in any PAHs that are produced during the process escaping to the gaseous phase while during gasification and fast pyrolysis, any PAHs that are produced may condense on the biochar material itself. This observation also supports the notion that with greater access to oxygen (gasification is carried out in the presence of some oxygen), the concentration of PAHs is higher.⁵⁸ It could also be possible that fast pyrolysis simply allows a higher frequency of

PAH forming reactions. This area requires further investigation, especially if a gasification biochar is being considered in a remediation context, as the level of $45 \mu\text{g g}^{-1}$ measured here exceeds soil quality guidelines. Clear trends with regards to temperature affecting the concentration of PAHs by ring size were not observed, however over the temperature range of $250\text{--}900^\circ\text{C}$ a comparison of grass (grass and switch grass) and wood (pine wood, oak and pine) biomass sources demonstrated that grass biochars contained five and six ring PAHs around the production temperature range of $350\text{--}650^\circ\text{C}$ while wood biochars were mostly devoid of these larger PAHs (see Figure S1 and discussion in the SI).

To give some additional context to these total PAH concentrations they are compared to the concentration of PAHs in other soil amendments. For example cattle manure slurries can contain $87\text{--}309 \mu\text{g kg}^{-1}$, pig slurries $66\text{--}339 \mu\text{g kg}^{-1}$, sewage sludge $1.7\text{--}126 \mu\text{g g}^{-1}$, and compost $0.8\text{--}2.7 \mu\text{g g}^{-1}$.^{59\text{--}61} The presently studied biochars contain lower total PAH concentrations than these materials.

Bioavailable PAHs. Concentration of bioavailable PAHs ranged from $0.17 \pm 0.04 \text{ ng L}^{-1}$ for Pine 650 to $10.0 \pm 1.1 \text{ ng L}^{-1}$ for Food Waste 400 (Figure 2 and full list in Table S5). The concentration of bioavailable PAHs in the gasification biochar was $162 \pm 71 \text{ ng L}^{-1}$ with the large standard deviation likely being the result of variable naphthalene concentrations. Concentrations were dependent on the biochar source material, pyrolysis production temperature and pyrolysis time.

Concentrations decreased as production temperature increased. In concurrence with total concentrations, bioavailable concentrations generally decreased with increasing pyrolysis time, with the exception of the pine wood and the switch grass biochars. The concentrations for all of the biochars except the one produced via gasification are in line with the previously mentioned report for wood chip biochar.⁴⁵ To add some context to these results they can be compared to the concentration of bioavailable PAHs found in various sediment pore waters.⁶² Urban sediments have concentrations of 0.08–342 µg L⁻¹ and those impacted by aluminum smelters and manufactured gas plants have concentrations of 0.02–401 µg L⁻¹ and 0.1–10867 µg L⁻¹ respectively (when considering all 34 EPA PAHs). As biochars have substantially lower concentrations than the lowest of such values, biochar bioavailable concentration can be considered background or trace. One reason why PAHs have a very low bioavailability lies in their mechanism of binding to biochar. Biochar consists of condensed, aromatic sheets that show highly favorable π – π interactions with the planar, aromatic PAHs. In addition, part of the PAHs are formed simultaneously with the biochar, and can become occluded and completely locked up within the biochar structure, rendering them completely unavailable. Some of the occluded PAHs maybe nonextractable, even with the exhaustive solvent extraction, and therefore total PAH could be somewhat higher than analyzed.⁵⁶

The correlation of bioavailable concentrations with pyrolysis temperature resulted in significantly negative slopes for the digested dairy manure ($P = 0.05$), paper mill waste ($P = 0.1$), pine wood ($P = 0.2$), and the correlation of all chars was significant at the 0.1 confidence level. A full statistical analysis for the bioavailable concentration verses biochar production temperature is shown in SI Table S6. The concentration of bioavailable PAHs were affected by artificial aging in the opposite direction to the total concentrations, that is, concentrations decrease following aging (statistically significant differences observed following a t test $P = 0.05$ for pine 250 compared to pine age 250 and pine 650 compared to pine age 650). There was no clear correlation between PAH distribution by ring size with pyrolysis temperature, pyrolysis time or artificial aging.

Biochar-Water Partitioning Coefficients. Biochar-water partitioning coefficients ($\log K_{\text{biochar-water}}$ mL g⁻¹) were calculated for each PAH by dividing the average biochar PAH concentration for a specific biochar (C_{biochar} µg g⁻¹) by its average concentration of bioavailable PAHs (C_{water} µg mL⁻¹). It should be noted the method used to obtain these values is subject to errors; in $K_{\text{POM-water}}$ values (maximally 0.2 log units for chrysene), total PAH concentrations (likely 30–40% given the 56–79% recovery) and the GCMS analyses (around 30% given the 70–130% recovery). In total, the errors in $K_{\text{biochar-water}}$ values can amount to up to around 0.3 log units (factor of 2). However, this error is encountered in the vast majority of $K_{\text{passive sampler-water}}$ concentrations published to date.⁶³

Figure 3 shows the range of values for each PAH compound (Figure 3a) and the relationship between the $\log K_{\text{biochar-water}}$ values and the H:C and O:C atomic ratios for the biochars (Figure 3b and 3c grouped according to biochar type). The $K_{\text{biochar-water}}$ values are high which indicates strong PAH sorption to the biochar matrix and thus a low bioavailable concentration. The large range of $\log K_{\text{biochar-water}}$ values for each PAH, covering 3 orders of magnitude, is indicative of the range of biochar sorption affinities due to the different biomass sources

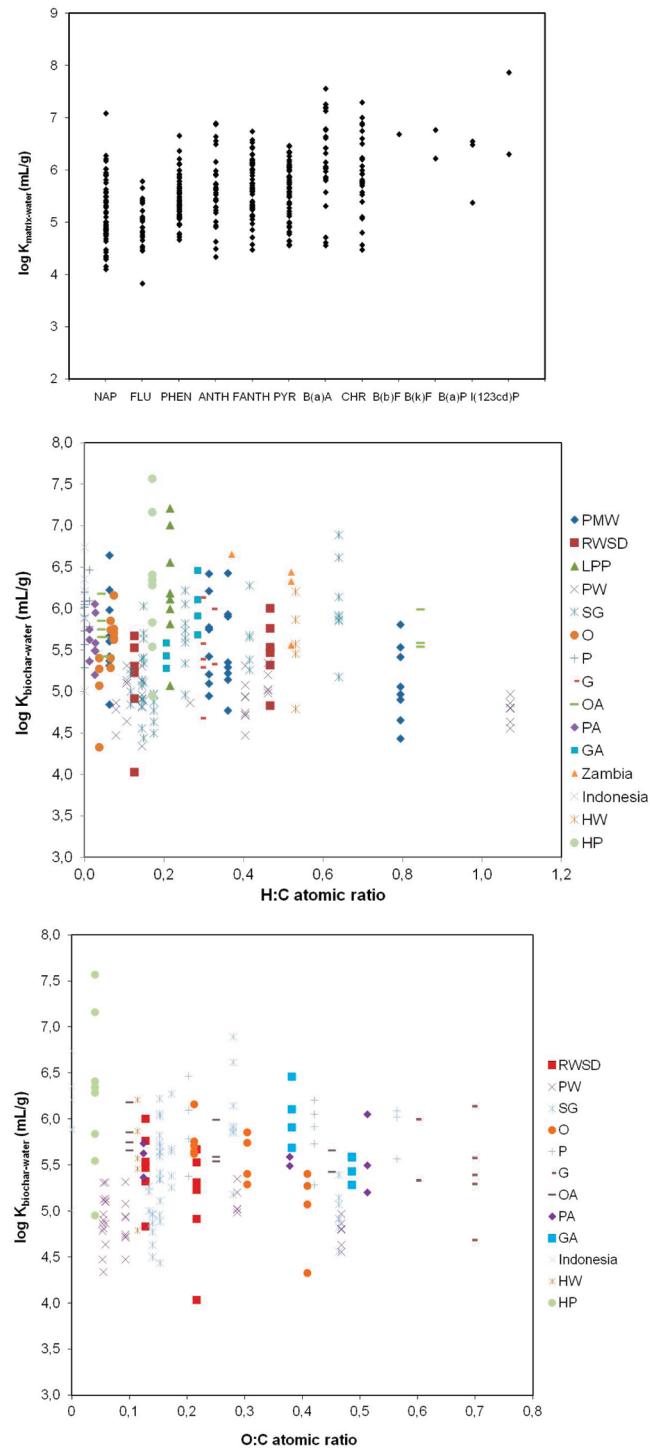


Figure 3. correlation between experimental $K_{\text{biochar-water}}$ (mL g⁻¹) values (a) for each PAH, (b) with biochar H:C ratio and (c) with biochar O:C ratio.

and production methods. The sorption properties of individual biochars are likely heterogeneous, and different levels of PAH occlusion in the biochar matrix likely also contributed to this large scatter. No meaningful mechanism or parameter could be identified that reduced this distribution.

Extrapolation of these partitioning coefficients to other biochars should therefore be done with great caution. This point is also illustrated in previous studies where $K_{\text{biochar-water}}$ values for naphthalene and phenanthrene ranged by up to

10 orders of magnitude (e.g., naphthalene log $K_{\text{biochar-water}}$ was between 1.0^{64} and 10.4^{65} mL g $^{-1}$ calculated at an aqueous concentration of 1 ng L $^{-1}$). SI Table S7 shows the median and range of experimental log $K_{\text{biochar-water}}$ values with a comparison to literature data. The log $K_{\text{biochar-water}}$ values are larger than log K_d values for sediment 62,66 and soil 67 but smaller than log $K_{\text{AC-water}}$ values 68 implying that for biochars the ratio of bioavailable PAHs to total PAHs is lower than for sediments but higher than for AC. The smallest log $K_{\text{biochar-water}}$ value measured was 4 (i.e., 0.01% of total PAHs was bioavailable) and the largest was 7.6 (i.e., 0.000001% was bioavailable), with most values being around 5.5. In line with Rey-Salgueiro et al. 58 and Jonker and Hawthorne, 69 who carried out long-term leaching studies of black carbon materials, these results indicate the biochar could be a long-term stable PAH reservoir. However, this cannot be said with certainty as no long-term or harsh leaching/desorption studies were performed.

Figure 3b and c also illustrate the variability of $K_{\text{biochar-water}}$ values when related to biochar physicochemical properties (H:C and O:C biochar atomic ratio). These H:C and O:C ratios provide an indication of the chemical differences between the biochars, with higher H:C and O:C indicating a more condensed residue. 70 The sorption capacity of the biochars does not clearly increase for the PAH compounds when H:C and O:C ratios decrease as may be expected as the number of strong sorption sites are expected to increase with increased carbon content. 56

Dioxin Concentrations. The total dioxin concentrations in 14 biochars are shown in Table 2. These values represent the

Table 2. Dioxin Concentrations in Biochars

biochar	total tetra-octa chlorinated dioxin concentration (pg/g) ^a	total 2,3,7,8-substituted dioxin concentration (pg/g)	toxic dioxin concentration (pg/g TEQ)
FW 400	92.0	12.2	0.15
FW 600	84.0	7.5	0.16
DDM 600	85.8	8.0	0.13
PW 900	91.5	10.7	0.15
LPP	86.0	10.5	0.18
FW 300		13.3	1.20
FW 500		0.39	0.008
O 650		1.5	0.02
G 650		2.4	0.02
PW 800		0.5	0.005
SG 800		0.8	0.008
SG 900		2.2	0.22
PMW 600		0.6	0.06

^aTotal of 130 toxic and nontoxic congeners.

analysis of only the 2,3,7,8-substituted “toxic” congeners (17 congeners) for eight biochars and a complete analysis of a total of 130 nontoxic and toxic congeners (2,3,7,8 substituted congeners and the 4–8 substituted congeners) for five biochars. SI Table S8 shows the concentration of toxic and nontoxic congeners found in the biochars. In all cases the concentrations of toxic congeners are lower than the Swedish guideline values for dioxin contamination for nonsensitive land use (250 pg g $^{-1}$ TEQ). Food waste, which often has a high salt content, has been shown to contain significant amounts of dioxins. 71 The food waste biochars used here generally contained higher levels of dioxins than the other biochars possibly because the chlorine content was higher (food waste 300 contained 2.9% chlorine,

food waste 600 had 3.4% chlorine and switch grass 800 had just 0.44%).

The concentration of bioavailable dioxins measured with POM passive samplers were below the analytical limit of detection for all of the biochars analyzed, even though the state-of-the-art passive sampling method is capable of measuring dioxin concentrations down to 1 pg m $^{-3}$ 53 and the analysis of the 130 congeners was carried out at a leading dioxin laboratory. SI Table S9 shows the analytical LOD for the 130 congeners analyzed and SI Table S10 reports minimum biochar-water distribution ratios.

Further Use of Biochar in Environmental Management.

Potential negative side effects associated with the use of biochar in agricultural improvement must be addressed before such a practice will become acceptable from a social, environmental and legislative standpoint. Based on the presently reported screening of a large number of mainly slow pyrolysis biochars, some recommendations can be made regarding the production of a biochar that minimizes the risk of exposure to organic toxins contained within the biochar. Biochar source, production temperature, pyrolysis time and aging all affected the level of toxins in the resulting biochars. Producing biochars from woody feedstocks, using higher production temperatures (between 500 and 600 °C) and longer pyrolysis times resulted in lower toxin concentrations. This is encouraging in those cases where a slow pyrolysis method of making biochar can be incorporated into traditional farming practices in order to produce biochar for soil amendment. All of the total and bioavailable PAHs and dioxins in the slow-pyrolysis biochars were below environmental guidelines for the levels of toxins in soils; however the concentrations for the gasification biochar were above these safe levels. Further investigation is needed to determine whether generalizations can be made regarding the production of a safe gasification biochar and slow pyrolysis biochars using feedstocks and production conditions not tested here.

ASSOCIATED CONTENT

S Supporting Information

Additional information as noted in the text. This material is available free of charge via the Internet at <http://pubs.acs.org>.

AUTHOR INFORMATION

Corresponding Author

*E-mail: Sarah.Hale@NGI.NO.

Notes

The authors declare no competing financial interest.

ACKNOWLEDGMENTS

The following are thanked for the production of various biochars; Jonah Levine (former Biochar Engineering Company and now Biochar Solutions), Colleen Rostad (USGS), Peter Aagaard and Gibson Simusokwe (CFU Zambia), Alex Heikens (UNDP regional office Bangkok), Dorisel Torres and Kelly Hanley (Cornell University), and Daniel Rasse (Bioforsk). Horst Rottler (Oekometric GmbH) is thanked for the analysis of toxic dioxins in eight biochar samples. Aina Winther (NGI) is thanked for laboratory assistance, Vanja Alling (NGI) is thanked for help with producing figures and Gijs Breedveld (NGI) and Jan Mulder (University of Life Sciences) are thanked for constructive discussions.

REFERENCES

- (1) Lehmann, J. A handful of carbon. *Nature* **2007**, *447* (7141), 143–144.

- (2) Liang, B.; Lehmann, J.; Solomon, D.; Kinyangi, J.; Grossman, J.; O'Neill, B.; Skjemstad, J. O.; Thies, J.; Luizao, F. J.; Petersen, J.; Neves, E. G. Black carbon increases cation exchange capacity in soils. *Soil Sci. Soc. Am. J.* **2006**, *70* (5), 1719–1730.
- (3) Yu, X. Y.; Ying, G. G.; Kookana, R. S. Reduced plant uptake of pesticides with biochar additions to soil. *Chemosphere* **2009**, *76* (5), 665–671.
- (4) Glaser, B.; Lehmann, J.; Zech, W. Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal - a review. *Biol. Fertil. Soils* **2002**, *35* (4), 219–230.
- (5) Ramanathan, V.; Crutzen, P. J.; Kiehl, J. T.; Rosenfeld, D. Atmosphere - Aerosols, climate, and the hydrological cycle. *Science* **2001**, *294* (5549), 2119–2124.
- (6) Clarke, A. D.; Noone, K. J. Soot in the Arctic snowpack—A cause for perturbations in radiative-transfer. *Atmos. Environ.* **1985**, *19* (12), 2045–2053.
- (7) Yang, Y. N.; Sheng, G. Y. Enhanced pesticide sorption by soils containing particulate matter from crop residue burns. *Environ. Sci. Technol.* **2003**, *37* (16), 3635–3639.
- (8) Yang, Y. N.; Sheng, G. Y.; Huang, M. S. Bioavailability of diuron in soil containing wheat-straw-derived char. *Sci. Total Environ.* **2006**, *354* (2–3), 170–178.
- (9) Wornat, M. J.; Sarofim, A. F.; Longwell, J. P. Changes in the degree of substitution of polycyclic aromatic-compounds from pyrolysis of a high-volatile bituminous coal. *Energy Fuels* **1987**, *1* (5), 431–437.
- (10) Hajaligol, M.; Waymack, B.; Kellogg, D. Low temperature formation of aromatic hydrocarbon from pyrolysis of cellulosic materials. *Fuel* **2001**, *80* (12), 1799–1807.
- (11) Mastral, A. M.; Callen, M. S. A review an polycyclic aromatic hydrocarbon (PAH) emissions from energy generation. *Environ. Sci. Technol.* **2000**, *34* (15), 3051–3057.
- (12) Garcia-Perez, M. *The Formation of Polycyclic Hydrocarbons and Dioxins During Pyrolysis: A Review of the Literature with Descriptions of Biomass Composition, Fast Pyrolysis Technologies and Thermochemical Reactions*; Washington State University: Pullman, Washington, 2008.
- (13) Stanmore, B. R. The formation of dioxins in combustion systems. *Combust. Flame* **2004**, *136* (3), 398–427.
- (14) Wikstrom, E.; Lofvenius, G.; Rappe, C.; Marklund, S. Influence of level and form of chlorine on the formation of chlorinated dioxins, dibenzofurans, and benzenes during combustion of an artificial fuel in a laboratory reactor. *Environ. Sci. Technol.* **1996**, *30* (5), 1637–1644.
- (15) Canadian Council of Ministers of the Environment. *Canadian Soil Quality Guidelines, Carcinogenic and other polycyclic aromatic hydrocarbons (PAHS)*; Canadian Council of Ministers of the Environment, 2008.
- (16) Miljøstyrelsen. *Liste over kvalitetskriterier i relation til forurenset jord og kvalitetskriterier for drikkevand*, 2009.
- (17) GmbH Ein Service des Bundesministeriums der Justiz in Zusammenarbeit mit der juris. *Bundes-Bodenschutz- und Altlastenverordnung (BBodSchV)*, 1999.
- (18) *Tilstandsklasser for Forurenset Grunn*; Norwegian Pollution Control Authority, 2009.
- (19) *Using Science to Create a Better Place*; Environment Agency, 2008.
- (20) J. R. C., Ispra Derivation Methods of Soil Screening Values in Europe. A Review and Evaluation of National Procedures Towards Harmonization; European Commission, 2007.
- (21) Elert, M.; Jones, C.; Norman, F. *Development of Generic Guideline Values: Model and Data Used for Generic Guideline Values for Contaminated Soils in Sweden*; Swedish Environment Agency, 1997.
- (22) Kraaij, R.; Mayer, P.; Busser, F. J. M.; Bolscher, M. V.; Seinen, W.; Tolls, J. Measured pore-water concentrations make equilibrium partitioning work—A data analysis. *Environ. Sci. Technol.* **2003**, *37* (2), 268–274.
- (23) Hawthorne, S. B.; Azzolina, N. A.; Neuhauser, E. F.; Kreitinger, J. P. Predicting Bioavailability of sediment polycyclic aromatic hydrocarbons to *Hyalella azteca* using equilibrium partitioning, Supercritical fluid extraction, and pore water concentrations. *Environ. Sci. Technol.* **2007**, *41* (17), 6297–6304.
- (24) Semple, K. T.; Doick, K. J.; Jones, K. C.; Burauel, P.; Craven, A.; Harms, H. Defining bioavailability and bioaccessibility of contaminated soil and sediment is complicated. *Environ. Sci. Technol.* **2004**, *38* (12), 228A–231A.
- (25) IBI. *International Biochar Initiative: Biochar Product Definition and Standard DRAFT VERSION*, 2011.
- (26) Alexander, M. How toxic are toxic-chemicals in soil. *Environ. Sci. Technol.* **1995**, *29* (11), 2713–2717.
- (27) Vinturella, A. E.; Burgess, R. M.; Coull, B. A.; Thompson, K. M.; Shine, J. P. Use of passive samplers to mimic uptake of polycyclic aromatic hydrocarbons by benthic polychaetes. *Environ. Sci. Technol.* **2004**, *38* (4), 1154–1160.
- (28) Hedberg, E.; Kristensson, A.; Ohlsson, M.; Johansson, C.; Johansson, P. A.; Swietlicki, E.; Vesely, V.; Wideqvist, U.; Westerholm, R. Chemical and physical characterization of emissions from birch wood combustion in a wood stove. *Atmos. Environ.* **2002**, *36* (30), 4823–4837.
- (29) Chao, C. Y. H.; Kwong, P. C. W.; Wang, J. H.; Cheung, C. W.; Kendall, G. Co-firing coal with rice husk and bamboo and the impact on particulate matters and associated polycyclic aromatic hydrocarbon emissions. *Bioresour. Technol.* **2008**, *99* (1), 83–93.
- (30) Conde, F. J.; Ayala, J. H.; Afonso, A. M.; Gonzalez, V. Emissions of polycyclic aromatic hydrocarbons from combustion of agricultural and silvicultural debris. *Atmos. Environ.* **2005**, *39* (35), 6654–6663.
- (31) Schauer, J. J.; Kleeman, M. J.; Cass, G. R.; Simoneit, B. R. T. Measurement of emissions from air pollution sources. 3. C-1-C-29 organic compounds from fireplace combustion of wood. *Environ. Sci. Technol.* **2001**, *35* (9), 1716–1728.
- (32) Jenkins, B. M.; Jones, A. D.; Turn, S. Q.; Williams, R. B. Emission factors for polycyclic aromatic hydrocarbons from biomass burning. *Environ. Sci. Technol.* **1996**, *30* (8), 2462–2469.
- (33) Williams, P. T.; Horne, P. A. Analysis of aromatic-hydrocarbons in pyrolytic oil derived from biomass. *J. Anal. Appl. Pyrolysis.* **1995**, *31*, 15–37.
- (34) Williams, P. T.; Nugranad, N. Comparison of products from the pyrolysis and catalytic pyrolysis of rice husks. *Energy* **2000**, *25* (6), 493–513.
- (35) Fabbri, D.; Adamiano, A.; Torri, C. GC-MS determination of polycyclic aromatic hydrocarbons evolved from pyrolysis of biomass. *Anal. Bioanal. Chem.* **2010**, *397* (1), 309–317.
- (36) Horne, P. A.; Williams, P. T. Influence of temperature on the products from the flash pyrolysis of biomass. *Fuel* **1996**, *75* (9), 1051–1059.
- (37) Brown, R. A.; Kercher, A. K.; Nguyen, T. H.; Nagle, D. C.; Ball, W. P. Production and characterization of synthetic wood chars for use as surrogates for natural sorbents. *Org. Geochem.* **2006**, *37* (3), 321–333.
- (38) Jonker, M. T. O.; Koelmans, A. A. Extraction of polycyclic aromatic hydrocarbons from soot and sediment: Solvent evaluation and implications for sorption mechanism. *Environ. Sci. Technol.* **2002**, *36* (19), 4107–4113.
- (39) Fernandes, M. B.; Brooks, P. Characterization of carbonaceous combustion residues: II. Nonpolar organic compounds. *Chemosphere* **2003**, *53* (5), 447–458.
- (40) Zhurinsh, A.; Zandersons, J.; Dobe, G. Slow pyrolysis studies for utilization of impregnated waste timber materials. *J. Anal. Appl. Pyrolysis.* **2005**, *74* (1–2), 439–444.
- (41) Singh, B.; Singh, B. P.; Cowie, A. L. Characterisation and evaluation of biochars for their application as a soil amendment. *Aust. J. Soil Res.* **2010**, *48* (6–7), 516–525.
- (42) Wornat, M. J.; Ledesma, E. B.; Sandrowitz, A. K.; Roth, M. J.; Dawsey, S. M.; Qiao, Y. L.; Chen, W. Polycyclic aromatic hydrocarbons identified in soot extracts from domestic coal burning stoves of Henan Province, China. *Environ. Sci. Technol.* **2001**, *35* (10), 1943–1952.
- (43) Sharma, R. K.; Hajaligol, M. R. Effect of pyrolysis conditions on the formation of polycyclic aromatic hydrocarbons (PAHs) from polyphenolic compounds. *J. Anal. Appl. Pyrolysis* **2003**, *66* (1–2), 123–144.
- (44) Nakajima, D.; Nagame, S.; Kuramochi, H.; Sugita, K.; Kageyama, S.; Shiozaki, T.; Takemura, T.; Shiraishi, F.; Goto, S.

- Polycyclic aromatic hydrocarbon generation behavior in the process of carbonization of wood. *Bull. Environ. Contam. Toxicol.* **2007**, *79* (2), 221–225.
- (45) Freddo, A.; Hughes, S.; Raven, J.; Reid, B. J. Presence and significance of priority pollutants in biochar matrices. *SETAC Europe: 20th Annual Meeting 23–27 May 2010, Seville, Spain*. 2010.
- (46) Kulkarni, P. S.; Crespo, J. G.; Afonso, C. A. M. Dioxins sources and current remediation technologies—A review. *Environ. Int.* **2008**, *34* (1), 139–153.
- (47) Gordon, M. Dioxin characterisation, formation and minimisation during municipal solid waste (MSW) incineration: Review. *Chem. Eng. J.* **2002**, *86* (3), 343–368.
- (48) Korfomacher, W. A.; Natusch, D. F. S.; Taylor, D. R.; Mamantov, G.; Wehry, E. L. Oxidative transformations of polycyclic aromatic-hydrocarbons adsorbed on coal fly-asH. *Science* **1980**, *207* (4432), 763–765.
- (49) Mandalakis, M.; Zebuhr, Y.; Gustafsson, O. Efficient isolation of polycyclic aromatic fraction from aliphatic compounds in complex extracts using dimethylformamide-pentane partitionings. *J. Chromatogr., A* **2004**, *1041* (1–2), 111–117.
- (50) Brandli, R. C.; Hartnik, T.; Henriksen, T.; Cornelissen, G. Sorption of native polycyclic aromatic hydrocarbons (PAH) to black carbon and amended activated carbon in soil. *Chemosphere* **2008**, *73* (11), 1805–1810.
- (51) Sundqvist, K.; Tysklind, M.; Cato, I.; Bignert, A.; Wiberg, K. Levels and homologue profiles of PCDD/Fs in sediments along the Swedish coast of the Baltic Sea. *Environ. Sci. Pollut. Res.* **2009**, *16* (4), 396–409.
- (52) Jonker, M. T. O.; Koelmans, A. A. Polyoxyethylene solid phase extraction as a partitioning method for hydrophobic organic chemicals in sediment and soot. *Environ. Sci. Technol.* **2001**, *35* (18), 3742–3748.
- (53) Cornelissen, G.; Wiberg, K.; Broman, D.; Arp, H. P. H.; Persson, Y.; Sundqvist, K.; Jonsson, P. Freely dissolved concentrations and sediment-water activity ratios of PCDD/Fs and PCBs in the open Baltic Sea. *Environ. Sci. Technol.* **2008**, *42* (23), 8733–8739.
- (54) Cornelissen, G.; Cousins, I. T.; Wiberg, K.; Tysklind, M.; Holmström, H.; Broman, D. Black carbon-dominated PCDD/Fs sorption to soils at a former wood impregnation site. *Chemosphere* **2008**, *72* (10), 1455–1461.
- (55) Cornelissen, G.; Pfttersen, A.; Broman, D.; Mayer, P.; Breedveld, G. D. Field testing of equilibrium passive samplers to determine freely dissolved native polycyclic aromatic hydrocarbon concentrations. *Environ. Toxicol. Chem.* **2008**, *27* (3), 499–508.
- (56) Cornelissen, G.; Gustafsson, O.; Bucheli, T. D.; Jonker, M. T. O.; Koelmans, A. A.; Van Noort, P. C. M. Extensive sorption of organic compounds to black carbon, coal, and kerogen in sediments and soils: Mechanisms and consequences for distribution, bioaccumulation, and biodegradation. *Environ. Sci. Technol.* **2005**, *39* (18), 6881–6895.
- (57) Baek, S. O.; Field, R. A.; Goldstone, M. E.; Kirk, P. W.; Lester, J. N.; Perry, R. A review of atmospheric polycyclic aromatic-hydrocarbons - sources, fate and behavIOR. *Water, Air, Soil Pollut.* **1991**, *60* (3–4), 279–300.
- (58) Rey-Salgueiro, L.; Garcia-Falcon, M. S.; Soto-Gonzalez, B.; Simal-Gandara, J. Procedure to measure the level of polycyclic aromatic hydrocarbons in wood ashes used as fertilizer in agroforestry soils and their transfer from ashes to water. *J. Agric. Food. Chem.* **2004**, *52* (12), 3900–3904.
- (59) Wild, S. R.; McGrath, S. P.; Jones, K. C. The polynuclear aromatic hydrocarbon (pah) content of archived sewage sludges. *Chemosphere* **1990**, *20* (6), 703–716.
- (60) Wild, S. R.; Jones, K. C. Polynuclear aromatic-hydrocarbons in the united-kingdom environment—A preliminary source inventory and budget. *Environ. Pollut.* **1995**, *88* (1), 91–108.
- (61) Berset, J. D.; Holzer, R. Organic micropollutants in swiss agriculture: Distribution of polynuclear aromatic-hydrocarbons (PAH) and polychlorinated-biphenyls (PCB) in soil, liquid manure, sewage-sludge and compost samples; A comparative-study. *Int. J. Environ. Anal. Chem.* **1995**, *59* (2–4), 145–165.
- (62) Arp, H. P. H.; Azzolina, N. A.; Cornelissen, G.; Hawthorne, S. B. Predicting pore water EPA-34 PAH concentrations and toxicity in pyrogenic-impacted sediments using pyrene content. *Environ. Sci. Technol.* **2011**, *45* (12), 5139–5146.
- (63) Lohmann, R. Critical review of low-density polyethylene's partitioning and diffusion coefficients for trace organic contaminants and implications for its use as a passive sampler. *Environ. Sci. Technol.* **2011**, *46*, 606–618.
- (64) Wang, X. L.; Xing, B. S. Sorption of organic contaminants by biopolymer-derived chars. *Environ. Sci. Technol.* **2007**, *41* (24), 8342–8348.
- (65) Chen, B. L.; Yuan, M. X. Enhanced sorption of polycyclic aromatic hydrocarbons by soil amended with biochar. *J. Soils Sed.* **2011**, *11* (1), 62–71.
- (66) Arp, H. P. H.; Breedveld, G. D.; Cornelissen, G. Estimating the in situ sediment-porewater distribution of PAHs and chlorinated aromatic hydrocarbons in anthropogenic impacted sediments. *Environ. Sci. Technol.* **2009**, *43* (15), 5576–5585.
- (67) Nguyen, T. H.; Goss, K. U.; Ball, W. P. Polyparameter linear free energy relationships for estimating the equilibrium partition of organic compounds between water and the natural organic matter in soils and sediments. *Environ. Sci. Technol.* **2005**, *39* (4), 913–924.
- (68) Hale, S. E.; Werner, D. Modeling the mass transfer of hydrophobic organic pollutants in briefly and continuously mixed sediment after amendment with activated carbon. *Environ. Sci. Technol.* **2010**, *44* (9), 3381–3387.
- (69) Jonker, M. T. O.; Hawthorne, S. B.; Koelmans, A. A. Extremely slowly desorbing polycyclic aromatic hydrocarbons from soot and soot-like materials: Evidence by supercritical fluid extraction. *Environ. Sci. Technol.* **2005**, *39* (20), 7889–7895.
- (70) Hammes, K.; Schmidt, M. W. I.; Smernik, R. J.; Currie, L. A.; Ball, W. P.; Nguyen, T. H.; Loucheurn, P.; Houel, S.; Gustafsson, O.; Elmquist, M.; Cornelissen, G.; Skjemstad, J. O.; Masiello, C. A.; Song, J.; Peng, P.; Mitra, S.; Dunn, J. C.; Hatcher, P. G.; Hockaday, W. C.; Smith, D. M.; Hartkopf-Froeder, C.; Boehmer, A.; Luer, B.; Huebert, B. J.; Amelung, W.; Brodowski, S.; Huang, L.; Zhang, W.; Gschwend, P. M.; Flores-Cervantes, D. X.; Largeau, C.; Rouzaud, J. N.; Rumpel, C.; Guggenberger, G.; Kaiser, K.; Rodionov, A.; Gonzalez-Vila, F. J.; Gonzalez-Perez, J. A.; de la Rosa, J. M.; Manning, D. A. C.; Lopez-Capel, E.; Ding, L. 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 Biogeochem. Cycles.* **2007**, *21* (3).
- (71) Srogi, K. Levels and congener distributions of PCDDs, PCDFs and dioxin-like PCBs in environmental and human samples: A review. *Environ. Chem. Lett.* **2008**, *6* (1), 1–28.