

Trace Element Deposition on Mount Everest

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abstract

The objective of this study of the North Ridge of Everest was to examine trace element concentrations and altitudinal trends in soil and snow. Mount Everest was selected because its remote location and extreme elevation isolates it from localized pollution sources. Soil samples were collected on the Rongbuk glacier of Mount Everest (Qomolangma) from 5334 to 6553 m, and fresh surface snow samples (0–10 cm) were collected along the climbing route of the northeast ridge from 6858 m to 7752 m. The samples were analyzed for Pb, Zn, Cd, Ni, Cr, Co, Cu, As, Mn, Hg, and V using inductively coupled plasma spectroscopy. Results show that As and Cd are both above USEPA drinking water guidelines in all snow samples, and arsenic is above the USEPA soil screening guidelines in all soil samples. There was a clear trend in element variation in the soil samples, with the highest concentrations found at 5944 m. There was no clear trend detected in the snow samples, possibly due to vertical mixing of surface snow. Anthropogenic sources are suspected to have contributed to the elevated concentrations of both cadmium and arsenic.



Due to its rapid increase in industrialization, Asia is currently the greatest contributor of atmospheric anthropogenic pollutants in the world (Pacyna and Pacyna, 2001). The combustion of fossil fuels (Cr, Ni, Sb) and gasoline (Pb) and the production of nonferrous metals (Cu, Zn, As, Sd) contribute the bulk of trace metals to the atmosphere as Asia increases its gross domestic product (Pacyna et al., 2007; Pacyna and Pacyna, 2001). The scope and range of this deposition is not precisely known, but it is believed that trace elements in the troposphere can be transported more than 2500 km (Marx et al., 2004). Mount Everest's (Qomolangma) extension into the remote troposphere, above the boundary layer (the demarcation between lower elevations influenced by localized pollution and the unaffected region (atmosphere) above) positions it to receive deposition from these anthropogenic sources and provides researchers the ideal setting to examine the quantity and type of trace element deposition.

Nearly 20% of the Earth's surface is comprised of mountains that provide a valuable surface water storage reservoir and vital resource for millions of humans, including the Tibetans and Nepalis living below Everest (USEPA, 2008). It is estimated that approximately one-tenth of the world's population relies heavily on fresh water from snowpack in alpine regions, reinforcing the importance of a deeper understanding of elemental concentrations in atmospheric deposition at high elevations (USEPA, 2008). Mount Everest's (27°59'N, 86°55'E, 8844 m) remote setting far from industry, extension into the troposphere, and intermediate location between northern China and India provide an exceptional laboratory for the study of atmospheric deposition and chemistry. Trace metal concentrations in fresh snow on the north (Lee et al., 2008; Kang et al., 2007, 2004, 2002) and south slopes of Everest (Marinoni et al., 2001) have been evaluated, as well as on adjacent Himalayan mountains (Balerna et al., 2003; Shrestha et al., 1997; Mayewski and Lyons, 1983) and at remote sites in Alaska, Antarctica, Norway, Japan, Europe, and New Zealand (Douglas and Sturm, 2004; Ikegawa et al., 1999; Norwegian Institute for Air Research, 2008; Hou et al., 2005; Walker et al., 2003; Marx et al., 2004). Some studies have found that primary anthropogenic (Cd, Zn, Pb) trace metal concentrations in fresh snow on Mount Everest are lower than detected in large cities, such as Hong Kong, and that other trace metals in Everest surface snow (Zn, Mn, Cu, As, V, and Cr) have low concentrations that closely mirror remote sites such as Antarctica (Kang et al., 2007). This suggests that Everest is not affected by anthropogenic pollution (Kang et al., 2007; Marinoni et al., 2001). However, a recent study of fresh snow samples from the North Ridge of Everest contradicts these findings, with trace element concentrations that are three and four magnitudes higher than Antarctica (Lee et al., 2008), suggesting that snow on Everest has been polluted by anthropogenic sources.

The determination of the provenance of trace elements is a very complex process because deposition is often a composite of multiple sources (Marx et al., 2004). Snow chemistry on the northern slopes of Everest is influenced by desert dust from central Asia (Lee et al., 2008; Kang et al., 2007; Balerna et al., 2003), while the southern slopes are relatively free from this deposition (Marinoni et al., 2001). Research has suggested that the bulk of trace metals in surface snow on Mount Everest are derived from crustal aerosols from adjacent rocks in addition to dust storms (Kang et al., 2007), but recent research utilized a crustal



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Images from the expedition, including a sunset from the top of the world and the author, Bill Yeo, in full gear.

enrichment factor (Wedepohl, 1995; Qureshi, 2001) to determine the contribution of crustal aerosols and found they were insignificant compared to the amount of deposition from anthropogenic sources (Lee et al., 2008). Studies have suggested that precipitation rates in alpine environments are strongly correlated with trace metal deposition levels and could be the primary source of trace metal fallout in mountain environments (Zechmeister, 2004). During the pre-monsoon season Everest receives very little precipitation because of its location beside the arid Tibetan Plateau, and the bulk of deposition arrives from long-range western sources (Lee et al., 2008). Concentrations of trace elements are higher in the pre-monsoon season than in the post-monsoon season (Lee et al., 2008; Kang et al., 2007).

Little is known about the potential for trace elements to become more concentrated at specific altitudes. Some studies have found no obvious trends in trace element accumulation with altitude and believe it is unlikely due to mixing of surface snow by relentless winds and potential inputs from crustal aerosols. This determination was made through the examination of 14 snow samples collected from the North Ridge of Everest in which no clear altitudinal trend for elemental variation was found (Kang et al., 2007). Surface snow samples from an adjacent mountain, Cho Oyu, also didn't exhibit any elemental trends with altitude, but only three sample locations were used (Balerna et al., 2003).

Knowledge of trace metal contents of soils and snow is important because in labile forms there is potential for migration into the ecosystem, including food and water sources (Navas and Lindhorfer, 2005). The negative impact of trace metal toxicity to humans is widespread and well understood. Trace elements in urban and suburban soils have been studied extensively (Langley-Turnbaugh and Belanger, 2007; Langley-Turnbaugh and Evans, 2001; Steinnes et al., 1997), and numerous studies have focused on mountain environments (Wang et al., 2007; Navas and Lindhorfer, 2005; Evans et al., 2004; Walker et al., 2003; Moysse and Fernandez, 1987; Reiners et al., 1975), but there has been limited study of the Aridisols of Mount Everest. Trace metal analysis of soils on northeastern United States mountains in New Hampshire were conducted (Reiners et al., 1975), and elevated levels of Pb and Zn were said to be in the range of roadside soils. This study suggested there was an increase in deposition as altitude increased, as was found in other alpine soil trace metal studies in the European Alps (Zechmeister, 2004), on the Tibetan

Plateau (Wang et al., 2007), and on the Rongbuk glacier at the foot of Mount Everest (Zhang et al., 2006).

The objective of this study of the North Ridge of Everest was to examine trace element concentrations and altitudinal trends in soil and snow. This vertical analysis will provide a more complete understanding of the immediate and cumulative trace element concentrations above the boundary layer and provide insight to the scope and range of pollution.

Site Description

Mount Qomolangma (Everest) is located on the border of Nepal and China (Fig. 1). This study was conducted on the northern slope of Qomolangma, northeast of the town of Zhaxizong in Tingri county Tibet. The Tibetan plateau consists of arid alpine grassland vegetation, average elevation 4500 m, with an average annual precipitation of 3.2 cm (Zhang et al., 2006). Shrubs are randomly dispersed throughout the study site and are the dominate vegetation below approximately 5800 m (the tree line).

Soil samples were collected every 305 m between 5334 and 6553 m in the 8-km² glacial moraine of the Rongbuk glacier, and replicate samples were collected every 305 m above through the east Rongbuk glacier, a deep valley flanked by massive walls of dark crushed glacial till that winds in and out of penitentes (Fig. 2). The soil in this region is generally undisturbed, and no form of agriculture existed anywhere near the sample collection locations. Surface soil on the Tibetan plateau is routinely redistributed by winds with an annual mean speed of 4.4 m/s (Zhang et al., 2006) and on the Rongbuk glacier during the pre-monsoon season can average 7.5 m/s (Kang et al., 2007).

The snow samples were collected every 305 m between 6858 and 7772 m, on a steep exposed face that is interspersed with crevasses and hanging seracs and extends upward to the North Col. This face was predominantly ice on the lower slopes until the last week of May (2006) when the sun intensified. Samples were collected on the exposed ridge (7144 m) of the North Col, a site that leads to the upper reaches of Qomolangma (7448 m) to the northeast and the summit of Changtse (7752 m) to the west. The potential of vertical mixing of snow exists at these altitudes, but the distance between samples should be far enough to yield comparable data. The snow samples were collected from different aspects of the mountain because collection sites were limited to

the direction of the ridge; thus the deposition could be influenced by prevailing winds. Samples collected above the North Col could yield very different results because this location is not blocked by the imposing North Col wall or anything else in the world.

Methods

Replicate soil samples were collected in sterile Whirl-Pak (Nasco, Fort Atkinson, WI) containers on May 13, 2006, using standard methods (Soil Survey Division Staff, 1993). Surface snow samples (0–10 cm) were collected on May 8, 10, and 11, 2006 in sterile, acid-cleaned, high density polyethylene HDPE (Nalgene, Thermo Fisher Scientific, Waltham, MA) containers. During sample collection extreme care was taken to minimize contamination.



Fig. 1. Location map of Mount Everest.



Fig. 2. Satellite image of Everest and North Col Ridge route. Samples of snow and soil were collected on this route.

Soil samples were dried, ground, and sieved, and snow samples were stabilized after collection with trace metal grade nitric acid. All samples were analyzed for Cd, Ni, Zn, Cr, Co, Cu, As, Mg, Hg, V, and Pb. Soil samples were digested using USEPA Method 3051b, a nitric acid microwave digestion method (USEPA, 1995), and all samples were quantified using ICP-AES following USEPA Method 6010 (USEPA, 1996). The limit of detection for most elements was 10 µg/L. A standard reference material (NIST Soil Standard SRM 2711, Montana Soil) was inserted into the sampling stream carried through the digestions, extractions, and analysis as part of the quality assurance protocol. Duplicates and reagent blanks were also used to ensure accuracy and precision in the analysis. Results were analyzed for patterns in trace element concentration with elevation.

Results Snow

The average concentrations of As at all elevations exceeded the USEPA's maximum contaminant level (MCL) of 10 ng/g for drinking water (Table 1, Fig. 3). The lowest concentration of As, 10.9 ng/g, was detected at 7144 m, and the highest concentration of 23.6 ng/g was detected at 7448 m (Fig. 3). Cadmium also exceeded the USEPA's MCL of 5 ng/g for drinking water, with the lowest average concentration of 5.2 ng/g at 6858 m and the highest concentration of 6.8 ng/g at 7752 m. Cadmium and Zn concentrations increased with increasing altitude. Lead, Co, and Hg were all below detection limits.

Soil

The highest concentrations of Ni, Cr, Co, Mn, and V in Everest soil were at 5944 m (Table 1, Fig. 4). Copper and As levels contradicted this trend, with the highest concentrations at 5639 and 6553 m, respectively. Arsenic concentrations in soil were above critical limits (USEPA) at all altitudes, with the lowest concentration of 12.2 mg/kg at 5639 m and the highest, 49.3 mg/kg, at 6553 m. Cadmium concentrations were below detection limits.

Discussion Snow

Elevated concentrations of Cd and As at 7144 m indicate that between May 5 and 7, 2006 a deposition event took place through a significant snowstorm. Elevated Cd and As concentrations detected in fresh snow on May 10 and 11, 2006 at 7752 m, 7448 m, and 6858 m were from another snow event accompanied by heavy winds that struck Everest on the May 9, 2006. Because climbers consume approximately 6 to 8 L of water a day for an average of 6 d while camping at the North Col and above, there is potential for Cd and As to accumulate in climbers after ingesting melted snow (Pomroy et al., 1980).

Table 1. USEPA levels for soil and water.

	USEPA water	Everest	USEPA soil	Everest
	guidelines†	snow	guidelines‡	soil
	ng/g		mg/kg	
As	10	12.3–23.6	10	12.4–49.9
Cd	5	5.2–6.8	27	ND§
Co	NA¶	ND	NA	2.7–11.5
Cr	100	3.0–4.4	950	6.8–24.8
Cu	1300	3.4–8.7	650	9.1–33.5
Hg	2	ND	60	ND
Mn	NA	3.1–8.75	NA	109.1–339.8
Ni	NA	0.6–3.3	3800	4.5–18.8
Pb	15	0.1	375	1.4–29.9
Zn	5000	35.9–76.2	1500	0–95.9

† USEPA maximum contaminant level for drinking water (USEPA, 2006a).

‡ Remedial action guidelines for contaminated soils (Maine Department of Environmental Protection, 1996; USEPA, 2006b).

§ ND, no detection.

¶ NA, not available.

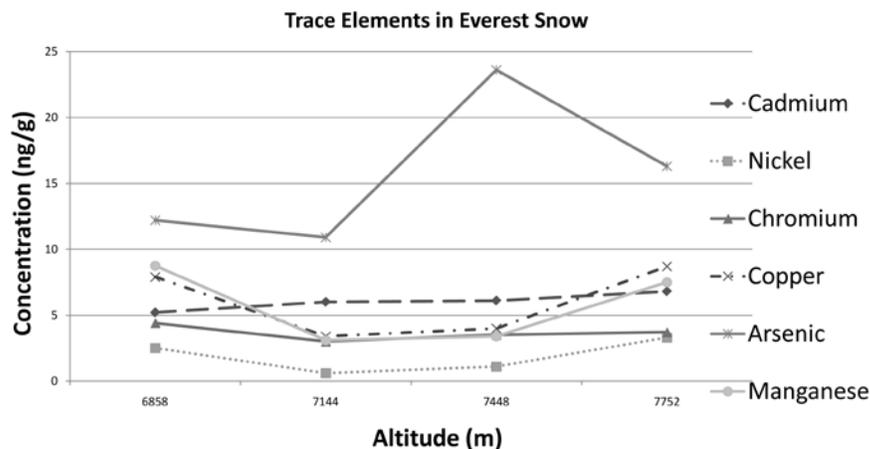


Fig. 3. Changes in trace element concentration with elevation in Everest snow.

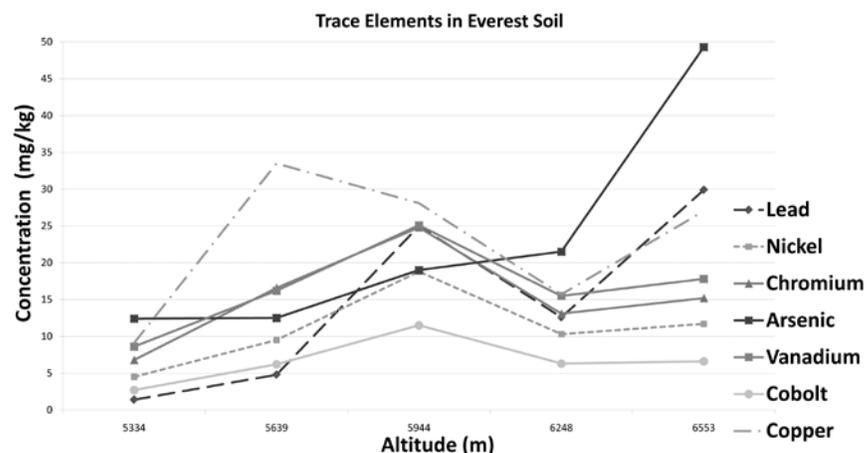


Fig. 4. Changes in trace element concentration with elevation in Everest soil.

Arsenic has long been associated with bladder, kidney, and skin cancers, with a biological half-life of about 4 d (Buchet et al., 1981) to 8 d (Pomroy et al., 1980) in the human body and up to a month in skin, esophagus, tongue, stomach and oral cavity (Vahter et al., 1982) before excretion by urine. Cadmium is classified as a probable human carcinogen (USEPA, 1992) associated with lung and prostate cancer through



inhalation and ingestion of food and water (Armstrong and Kazantzis, 1983; Sorahan & Waterhouse, 1983). Arsenic and Cd deposition in Asia is the result of nonferrous metal production and the combustion of fossil fuels (Pacyna and Pacyna, 2001). In Tibet, samples from the Dasuopu Glacier (6700 m) on Mount Xixiabangama, adjacent to Everest, indicate that low temperature combustion of coal, particularly in the heating season, contribute to Himalayan deposition levels (Wang et al., 2008), and studies have shown that the combustion of coal introduces As to the atmosphere (Contreras et al., 2009). India combusts 335 million tons of coal annually to supply one-third of their energy requirements (International Energy Association, 2002). Arsenic and Cd deposition in remote areas have been associated with anthropogenic processes (Ikegawa et al., 1999; Cunningham and Zoller, 1981).

The trace element concentrations from this study are higher than those found in a study on the North Ridge of Everest in May 2005, which reported Cd below detection limits and As at 183 pg/g (Kang et al., 2007). Another study conducted in the autumn of 2004 to spring 2005 of an excavated snow pit at 6575 m below the North Col on the North Ridge of Everest reported average concentrations of 5 pg/g for Cd and 83 pg/g for As (Lee et al., 2008), likely from anthropogenic sources since the concentrations exceeded potential rock and soil dust contributions. Because there are multiple factors that influence the provenance of deposition, including prevailing winds, precipitation patterns, and point source emissions, each deposition event is different and can originate from various sources located in the western region of central Asia (Lee et al., 2008) or south from India (Wang et al., 2008). Since sources vary, so should deposition levels in fresh snow samples collected from Everest. In addition, samples collected above the North Col could yield very different results because it is not blocked by the imposing North Col wall or anything else in the world.

It has also been shown that the pre-monsoon (May–July) conditions experienced during this study are more likely to yield elevated concentrations of trace elements compared to monsoon conditions (Lee et al., 2008; Kang et al., 2007). Studies have shown that deposition in the Himalayas during the pre-monsoon season originates from anthropogenic sources in India (Valsecchi et al., 1999) or from western sources in central Asia (Lee et al., 2008). It has been shown that anthropogenic sources directly influence deposition levels at remote sites such as Antarctica (Scarponi et al., 1997), and this finding was reinforced by several studies that found a rapid decrease in Pb immediately after its removal from gas-line (Evans et al., 2004).

In this study concentrations of Cd increased with altitude, from 5.2 to 6.8 $\mu\text{g/g}$ (Fig. 3). In high-altitude environments there is potential for wind to redistribute snow vertically and interfere with the development of demarcation zones between altitudes, although the 305-m difference between sample plots was enough for Cd to exhibit an increasing trend. It is unknown why this trend exists on Everest and reinforces the need for further study in the Himalayas and other remote high-altitude locations. Another comparable study (Kang et al., 2007) discovered that no trends existed in snow sampled on the North Ridge of Everest and concluded that influences from crustal aerosols or strong winds eliminated altitudinal stratification.

A comprehensive study of annual precipitation amounts on Everest would help determine potential input of trace elements because studies have shown that precipitation rates are directly related to deposition quantities (Zechmeister, 2004). Snyder-Conn et al. (1997) suggested that

snowflake size and the associated slow fall rate of snow compared to rain have potential to carry more air pollutants. Deposition levels on Everest have been compared to remote sites such as Antarctica, which has an annual average snow accumulation rate of less than 3 cm (Antarctic Connection, 2010), but precipitation rates need to be factored to obtain comparable results because of Everest's elevated snow accumulation rate.

Soil

There has been one other comprehensive study completed on the soils of the northern slope of Everest (Zhang et al., 2006). Comparable soil studies have been conducted on smaller mountains (Navas and Lindhorfer, 2005; Evans et al., 2004; Lin et al., 1996; Zechmeister, 2004; Moyses and Fernandez, 1987), but they are all within the range of influence of the boundary layer, which acts as a barrier, holding localized pollution closer to the ground. Studies conducted above the boundary layer are able to detect deposition that has been transported longer distances in the atmosphere and that is not influenced by localized sources. The elevated soil concentrations of As detected at all elevations compared to USEPA acceptable levels (USEPA, 2006b) are the result of deposition that has accumulated over time. Yet, soils on the north side of Everest contain very little organic matter (Wang et al., 2007) and thus have low potential to bind with metals (Moyse and Fernandez, 1987; Tyler and McBride, 1982). Therefore, soil As could be introduced to water sources through wind and snow melt, posing a health risk to mountain communities on and below Everest. Wind-blown dust could also potentially carry As and be inhaled, swallowed, or consumed in food frequently cooked in open-air settings. Humans inhale approximately 20 m³/d of air, with an average daily value of 10 µg/m³ of dust (American Lung Association of Maine, 2006), suggesting that 0.2 mg of dust could be inhaled each day. People climbing Everest and in many Tibetan villages below are routinely exposed to persistent winds carrying dust. From base camp (5334 m) to advanced base camp (6553 m) winds carrying dust and dirt periodically scour eyes and skin, reducing visibility and leaving dust deposits in the mouth. Fine particles driven by wind often penetrate the seams of tents, depositing a visible layer of dust on clothing, equipment, and food.

The most evident increases in soil concentrations with altitude were for Mn, As, and Co. A distinct altitudinal trend existed, with peak trace element concentrations of Mn, Co, Cr, Ni, Pb, and V at 5944 m. Some studies on other mountains have also found increases in trace element concentrations with increasing altitude in soil (Zhang et al., 2006; Wang et al., 2007), lichen (Bergamaschi et al., 2002), and moss (Zechmeister, 2004), but other studies have not reported any trends (Kang et al., 2007). This trend on Everest might exist because winds are blocked from the east and west, and wind from the north can travel directly up the valley from base camp. Another hypothesis is that 5944 m might be an altitude that is ideally positioned to receive deposition from outside sources. It is also possible that winds from the south above 5944 m redistribute deposition and dust to this lower protected valley, where it accumulates.

Soil As concentrations were highest at 6553 m, which suggests potential crustal enrichment in this area associated with East Rongbuk glacial moraine upheaval or sources above advanced base camp, such as the unstable rock wall of Changtse (7543 m), although the gradual increase in deposition with altitude is not supported by this theory. In 2006 more than 200 climbers slept at this elevation for an average of 14 nights, with many expeditions burning fossil fuels to run generators, although this use may not be related to elevated As concentrations due to the lower concentrations of the other elements that are typically asso-

ciated with As in fuels used. Advanced base camp (6553 m) could have elevated levels of As due to input from the more than 1400 m of elevation above the site that could potentially intercept deposition that is eventually transferred to lower elevations. Base camp (5334 m) has more human traffic and more fossil fuel combustion due to cooking (propane, white gas, butane, kerosene, yak dung), generators, and vehicle access. It consists of an 8-km² area that is not flanked by high walls that might trap pollution or potentially introduce crustal material as suspected at 6553 m. In the future crustal enrichment factors may be used to compare the influence of natural background levels of rocks and Tibetan plateau dust to soil levels. Another potential source of Cd and As could be from volcanic activity (Gabrielli et al., 2005), which can be transported more than 2500 km in dust (Marx et al., 2004).

Conclusions

Two specific snowstorm events on Everest introduced elevated concentrations of As and Cd to fresh surface snow on the upper North Col. Soil samples collected from base camp (5334 m) to advanced base camp (6553 m) contained elevated concentrations of As that gradually increased with altitude. These findings indicate that further research needs to be done to determine the provenance of these metals and suggests that individuals who are routinely exposed to this environment might consider filtering water and wearing a particle mask.

Trace element analysis at ultra-high elevations is a very difficult field of study due to the physical and technical factors restricting the number of samples collected; therefore, limited research is available for comparison. Future research needs to be conducted in other remote high-altitude settings around the world to develop a comprehensive database to assess the scope and range of trace element transport and to determine if they are derived from natural or anthropogenic sources. Ultra-high elevation sites could prove to be the "catcher's mitt" for deposition and provide the ideal laboratory to assess impact.

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Black Carbon in Soils: Relevance, Analysis, Distribution

Thomas Nehls and Richard K. Shaw

abstract

Black carbon (BC), a graphitic, particulate form of carbon that results from the incomplete combustion of organic material, is a significant part of the soil organic carbon (SOC) in soils worldwide. This article gives an overview on the sources and the behavior of soil BC, discusses the different analytical approaches for BC, and reviews literature on its distribution in soils. Black carbon is released by biomass and fossil fuel burning in the form of soot and char and charcoal. It is ubiquitous in the atmosphere and in soils. In soils, BC is more persistent and less reactive than other SOC constituents. It has turnover periods ranging from some decades to some thousands of years. Its stability, together with chemical and physical features such as high surface area and porosity, has important implications for nutrient storage, contaminant adsorption, and bulk SOC stability, as well as for global climate change mitigation. Biochar BC production and storage in soils are discussed as very promising carbon sequestration strategies. Because BC includes a continuum of materials, a wide array of analytical approaches has been developed for its investigation, including optical, chemical, and thermal methods. This makes it difficult to compare different studies and to draw a big picture of BC distribution. Black carbon assessment in soil survey would be valuable from a use and management perspective, might help identify the factors and processes affecting BC in soils, and possibly provide an important clue in understanding the global carbon cycle.

Black carbon, also called pyrogenic C (C_{pyr}), is a particulate, graphitic form of carbon produced during the incomplete combustion of biomass or fossil fuels or is a residue of it (Novakov, 1984; Goldberg, 1985). Char and charcoal, now called *biochar*, still show some morphological and chemical features of the charred material, which can be identified by scanning electron microscopy (SEM) (Skjemstad et al., 1996). In contrast, soot forms by condensation of burning products during combustion. Therefore, the term *black carbon* describes a continuum of substances with decreasing O/C ratio, ranging from char and charcoal (O/C = 0.4) to soot (O/C = 0.2) and graphitic C (Hedges et al., 2000; Hammes et al., 2007) (see Fig. 1). Aromaticity is a key structural feature of BC, as studied by solid state ^{13}C nuclear magnetic resonance (NMR) (Skjemstad et al., 1996; Schmidt and Noack, 2000).

The majority of BC can be found in the clay- and silt-sized light fraction of soil organic matter (SOM < 1.6 to 2.0 Mg m⁻³) (Glaser and Amelung, 2003; Rodionov et al., 2006).

The relevance of pyrogenic carbon arises from its ubiquitous distribution in ecosystems and soils, its persistence in soils and sediments, and its function as an adsorber for nutrients and contaminants, as well as its role as a historical marker wherever we find it. Black carbon is released in high amounts. Crutzen and Andreae (1990) estimated the global annual formation rate of vegetation fire residues (coarse grain BC) to be

200 to 600 Tg, while Kuhlbusch and Crutzen (1995) calculated 50 to 216 Tg/year but projected that, depending on the formation temperature, it could be double. These coarse fire residues (charcoal pieces) are mostly not transported from the fire site but incorporated in the soil and contribute to the BC pool in soils. The soil BC pool has still not been assessed globally. It is interesting to note that fossil fuel burning and vegetation fires contribute in more or less equal amounts to BC stored dynamically in the atmosphere, which is approximately 1.2 Tg. The emission rates for aerosol BC from vegetation fires and fossil fuel burning are 5 to 6 and 6 to 9 Tg/yr (Kuhlbusch, 1998), with oceans and soils as the terminal storages for BC.

Persistence in Soils

Black carbon is very stable in soils, but it is not inert. It has frequently been found in the very old fractions of SOM, for example, up to 7000 yr old for an Amazonian Anthrosol (Liang et al., 2008) and up to 2000 yr old for Mediterranean soils (Thinin, 1978). The turnover times for BC have been estimated to be some thousands of years (Preston and Schmidt, 2006). Contrary to these findings, BC was degraded within tens and hundreds of years under laboratory conditions (Shneour, 1966; Hamer et al., 2004; Hammes et al., 2008). In well-aerated soils from the subtropics BC shows half-life times in the same order (Bird et al., 1999). That conforms to our understanding of microbial decomposers and their ability to decompose everything organic at least co-metabolically, as long as temperature, humidity, and aeration are comfortable and primary food is available. It is widely accepted that BC must be degraded to some extent (Schmidt and Noack, 2000; Schmidt, 2004). Otherwise we would find the corresponding ancient BC in thick layers in our soils.

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