Soils and vulnerability to climate change

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Organic matter decay and carbon storage

Soils represent an immense reservoir of carbon, and moisture and temperature regimes at a global scale play important roles in determining soil carbon stocks (Post et al. 1982; Dixon 1994). Within forested landscapes, soil carbon stocks generally increase with site productivity (Homann et al. 2005). In the Sub-Boreal Spruce (SBS) moist cold subzone near Smithers, for example, soil carbon stocks on richer sites were almost double that of poorer sites (from 64 Mg C/ha on xeric, poor Lodgepole pine - Cladonia sites to 116 Mg C/ha on sub-hygric, rich Hybrid spruce - Oak fern sites); in these forests, soil carbon represents almost 30% of total ecosystem carbon (120 to 725 Mg C/ha, depending on site type; Kranabetter 2009). Less certain are the trends in soil carbon storage with stand age, but old-growth forests may continue to accumulate carbon in forest floors and mineral horizons in the absence of fire (Entry and Emmingham 1998, Giesen et al. 2008, Diochon and Kellman 2009).

The influence of shifting climatic regimes on the decay of soil organic matter (and, consequently, the sequestration of carbon) has been widely studied but there remains considerable disagreement about which soil pools will be most strongly affected and for how long. A multi-year litter decomposition study conducted across Canada demonstrated that temperature and precipitation have a strong influence on litter decay (Moore et al. 1999; Trofymow et al. 2002); in general, litter decomposition is stimulated by warmer and wetter conditions and suppressed by cooler and drier conditions. Soil respiration rates, which are a function of heterotrophic respiration by roots and soil biota, and the decay of roots, soil organisms and soil organic matter, are also typically stimulated by warmer temperatures under experimental conditions; however, this effect is usually only temporary (i.e., one to three years) and is followed by an acclimation period, during which respiration rates return to pre-warming levels (Rustad et al. 2001; Knorr et al. 2005). Recent research suggests that this acclimation effect is caused by thermal adaptation of the soil microbial community, a reduction in the size of the microbial biomass, and a depletion of the small pool of readily decomposed soil organic matter (Bradford et al. 2008). Although the larger, more decay-resistant pool of soil carbon has intrinsic turnover times of up to several thousands of years and, therefore, responds more slowly to a warmer climate, this pool may actually be more sensitive to increased temperatures than readily decomposable carbon forms; consequently, there may be a long-term positive feedback loop between global temperatures and soil organic matter decay (e.g., Fang et al. 2005; Knorr et al. 2005; Hartley and Ineson 2008; Conant et al. 2008).

Nutrient availability and site productivity

Altered rates of organic matter decay will also influence the rate at which nutrients bound in soil organic matter and litterfall are made available for re-uptake by plants and soil biota. This is a critical consideration because increased plant productivity can only be sustained if sufficient nutrients are
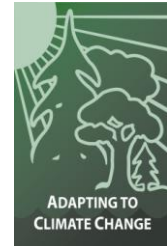
available. Fertilization experiments have demonstrated that the productivity of many BC forests is constrained by nitrogen availability although other nutrients, e.g., boron and sulphur, may also limit tree growth under some conditions (Brockley and Simpson 2004; BCMOFR 2007). In general, warmer temperatures have been reported to stimulate net nitrogen mineralization rates, suggesting greater nutrient availability for plants and soil microbes, but this effect is not always observed and is frequently short-lived (Rustad et al. 2001; Saxe et al. 2001; Pendall et al. 2004; Reich et al. 2006). There is also some concern that nutrients could be lost from forest and grassland soils via leaching or denitrification if warmer temperatures stimulate a more rapid release of nutrients than can be taken up by the plant and microbial biomass (Saxe et al. 2001). However, the influence of warmer temperatures on nutrient availability will depend strongly on many other factors, such as soil physical properties, the degree of warming and soil moisture. For example, in an Alaskan black spruce stand with restricted drainage (due to underlying permafrost), two years of soil warming stimulated the decomposition of the forest floor, increased forest floor concentrations of mineral N and P and enhanced foliar concentrations of N, P and K (Van Cleve et al. 1990). In contrast, three years of soil warming in an Alaskan black spruce stand that was not underlain by permafrost caused soil moisture to decline, thereby suppressing organic matter decay (Allison and Treseder 2008). Thus the influence of climate change on ecosystem productivity and soil carbon storage will be difficult to accurately predict because of the numerous interacting factors that affect the direction and rate of change in soil properties.

Shifts in species composition and migration of soil biota

The activity and structure of the soil biotic community is strongly influenced by the composition of the plant community (e.g., Saetre and Bååth 2000; Staddon et al. 2002; Hannam et al. 2006). Therefore, shifts in the presence and abundance of individual plant species are expected to have cascading effects on the structure and function of the belowground biological community. For example, deciduous tree species are expected to be more resilient to the anticipated effects of climate change than many coniferous tree species (Hamann and Wang 2006) and may be favoured by an increase in disturbance frequency (e.g. wildfire in the boreal forest; Johnstone 2006). Soils in deciduous stands are typically more biologically active than those in coniferous stands, likely because of differences in root and litter chemistry and microclimate effects (Prescott et al. 2000; Lindo and Visser 2003; Hannam et al. 2006). Large-scale stand conversions over relatively short time spans could give rise to novel soil biotic communities, depending on migration rates and habitat ranges of individual species, with potential losses in functional attributes associated with less disturbed forests. At the same time, microbial and invertebrate communities under rapid climate change may be under significant dispersal pressure, especially specialist species that have more defined niches and limited distributions (Clavel et al. 2011). As a result, cosmopolitan species such as *Cenoccocum geophilum* (likely the most abundant ectomycorrhizal fungi in the world; Douhan and Rizzo 2005) may play a larger role in sustaining ecosystem processes during periods of flux.

We are only beginning to appreciate how species interact belowground, and recent discoveries have demonstrated that these relationships are both critical and complex. For example, the presence of saprophytic fungi, such as *Hypholoma fasciculare*, appears to interfere with the infection of roots by the pathogenic fungus *Armillaria ostoyae* (Chapman et al. 2004); mycorrhizal linkages with mature trees

may facilitate seedling establishment under dry conditions by alleviating moisture stress (Teste and Simard 2008); many achlorophyllous plants are completely reliant on mycorrhizal fungi associated with nearby trees (Bidartondo and Bruns 2001). Rapid dispersal and migration of plants, animals and soil biota may uncouple such complex relationships. At a finer scale, a degree of heritability exists between trees and mycorrhizal fungal communities that reflects coevolutionary selection pressures of the local abiotic environment (Hoeksema and Thompson 2010). In the case of assisted migration, where tree populations are moved to match anticipated future climates, ectomycorrhizal community composition was altered with introduced host trees, which may cause a degree of maladaptation for the host-fungi mutualism (Kranabetter et al. 2012). Both spatial variability and heritability in ectomycorrhizal communities are therefore relevant in quantifying the vulnerability of forests to rapid climate change.

Special cases: earthworms and permafrost

Rapid climate change may facilitate the movement and establishment of weedy and invasive species, many of which can have a strong, direct influence on the ecology and productivity of the soil. For example, most of the earthworms in British Columbia are believed to be recent introductions from Europe (Addison 2008); under a warming climate, exotic earthworms are expected to expand their ranges further north (Bohlen et al. 2004; Addison 2008). Invasion by exotic earthworms in trembling aspen and lodgepole pine stands in the Kananaskis Valley and temperate forests in the north eastern U.S. have altered or eliminated the forest floor, changed the distribution and availability of soil carbon, nitrogen and phosphorus, caused shifts in the size and composition of the soil biotic community and reduced the diversity of the understory vegetation (Bohlen et al. 2004; Addison 2008). In grasslands, earthworms may disrupt soil crusts (Haeussler 2007), thereby increasing surface erosion and reducing infiltration (Belnap 2006), which is of particular concern if rainfall intensity increases (Nearing et al. 2004).

The influence of global warming patterns on permafrost has received considerable attention in the media because northern regions are warming faster than the global average (IPCC 2007) and because, until now, low temperatures have suppressed organic matter decay and emission of methane from these soils (Trumbore and Czimczik 2008). Permafrost is not widespread in British Columbia. It can be found in the fringe of the discontinuous permafrost zone, which is located in the northeastern corner of the province, where permafrost occurs as small isolated patches in peat bogs and on north-facing slopes in heavily shaded areas. South of the discontinuous permafrost zone, alpine permafrost can occur in mountainous areas, typically on high, north-facing slopes that retain little or no snow cover. Soils with alpine permafrost are often mineral, rather than organic, soils (Valentine and Lavkulich 1978) and can, therefore, be expected to release smaller quantities of carbon when warmed compared with permafrost in more northern regions. Thus, the impact of thawing permafrost on the decay of soil carbon will probably be more of an issue farther north. However, an increase in landslides in northern BC over the last 30 years may, in part, be caused by the warming of alpine permafrost (Geertesema et al. 2006). Therefore, the degradation of permafrost in alpine areas and its influence on slope stability may be a greater concern in British Columbia, although such effects will be localised (Pike et al. 2008).

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