Litter and Carbon Accumulation in Soils after Forest Restoration: the Australian Experience after Bauxite Mining

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Abstract

Soil is the primary store of terrestrial carbon, and is seriously disturbed by mining activities. Soil carbon exists in various forms that are functionally different and have contrasting residence times as part of the soil organic matter store. Here I explore the nature of soil carbon, from surface litter stocks to humified fractions, as measured from various rehabilitated (reforested) bauxite mined land across Australia. Litter in rehabilitated forests tends to accumulate to higher masses than in the surrounding native forests. This may simply be a function of extra litterfall during forest regrowth and higher stem densities than in the natural forests, or of lower decomposition rates. The higher litter stocks in restored forests are sometimes reflected in a higher carbon concentration in the mineral soil. However the type of carbon that accumulates in the mineral soils after bauxite mining may be primarily of particulate organic form that is not stable in the long-term and may readily mineralize to CO_2. Further research is required to establish the stability of carbon in the soils of rehabilitated forests and woodlands compared to their natural systems and the implication for carbon accounting and climate change.

Key Words
Carbon, litter, organic matter, carbon sequestration, mining, restoration, forests.

Introduction
Bauxite mining in Australia is a surficial, strip-mining operation that modifies large areas of land compared to most other types of mining. A typical bauxite mine in Australia will disturb around 500 hectares of *Eucalyptus* woodland each year. The preferred type of rehabilitation practice is integrated with mining operations and allows top soils and sub-soils to be utilised with little or no storage where possible (Koch 2007, Tibbett 2010). Despite this practice, organic carbon is lost from the soils between an undisturbed soil and a new rehabilitated profile (Banning *et al*. 2008; Sawada 1999; Schwenke *et al*. 2000a; Spain *et al*. 2005). This is likely due to mixing during stripping, stockpiling, replacement and ripping practices, as well as greater oxidation (to CO_2) through microbial activity as new surfaces are exposed in the soil through disturbance.

Interest in soil carbon is increasing as concern mounts about the steadily rising concentrations of CO_2 in the atmosphere; which has risen by 31% since 1750 from fossil fuel combustion and land use change (IPCC 2001, Lal 2004). There is a hope that soil carbon may have a role to play in sequestering greater amounts of atmospheric carbon, with a particular expectation for the restoration of degraded landscapes in this respect (Lal 2004; Lal *et al*. 2007). Carbon accounting is likely to become commonplace (e.g. Waterworth and Richards 2008), and tracking terrestrial carbon stores, their transformations and net balance with respect to the atmosphere may become routine for mining companies. The component of terrestrial carbon that is most difficult to assess in a manner that limits risk for markets (likely to trade in carbon) is the carbon held belowground. In this paper I will give a brief overview of soil carbon, its types and measurement and report on the current data on soil carbon after bauxite mining in Australia. Although plant roots play an important role in belowground carbon they will not be considered in detail here.

The landscapes mined for bauxite in Australia
Australia has three areas in which bauxite mining is active. These include Weipa and Gove around the gulf of Carpenatria (both in the far North of Queensland and the Northern Territory respectively) and Boddington, Huntley and Willowdale in the coastal south west of Western Australia (See Tibbett 2010). Former mines in the south west were also at Del Park and Jarrahdale (Figure 1). The natural vegetation is an open *Eucalyptus tetradonta* woodland, in the northern sites and Jarrah (*E. marginata*) forest in the Western sites. The soils are ancient highly weathered, lateritic and broadly classed as Kandosols or Oxisols. The

formation of bauxite (Al) ore has accumulated a few metres from the current land surface after millennia of eluviation from the upper soil horizons.

Figure 1. Location of bauxite mines in Australia where forest restoration has been implemented.

Carbon in post-mining soils

The primary source of soil organic carbon is from plant litter and this is related to the productivity of the vegetation biomass. In developing reforested systems, litterfall tends to be higher than in native stable systems. In bauxite mine rehabilitation, the density of seedlings that emerge and overall stem populations and tree basal area tend to also be higher than in the adjacent native forests. While belowground deposition is relatively difficult to estimate, aboveground litter stocks can be easily measured as a litter layer above the mineral soil; although the litter layer can sometimes be discriminated into upper and lower layers, with the lower layer having undergone greater comminution (Tibbett 2010). Stocks of litter on the forest floor are typically greater in restored sites than native, unmined forests sites. Litter accumulation has been reported as being almost four times greater (after 15 years of rehabilitation in Del Park Mine in Western Australia) than the surrounding Jarrah forest (Ward and Koch 1996). Litter masses that accumulated at two contrasting bauxite mines, one at Gove in the Northern Territory and the other at Boddington in Western Australia, are both greater in the rehabilitated than the native forests after around fifteen years (Figure 2). At Boddington, litter accumulation that matched the native systems took only ten years to occur while at Gove this took a little longer at 14 years. At the now closed Jarrahdale minesite (Western Australia) the amount of carbon held in the litter layer increased by 50% between 8 year old and 15 year old rehabilitated sites (Sawada 1999) This timeframe is in keeping with the natural senescence of many of the re-seeding, short-lived *Acacia* species that tend to dominate the canopy of the rehabilitated forests until this time. At Gove, for example, the *Acacia* tree basal area and the phyllode component of the litter tends to decrease from around 14 years (Tibbett 2010). From this, it seems likely that a rapid increase in litter stocks occurs as part of the natural successional cycle as (primarily legumous) re-seeders are replaced by the standard re-sprouting trees that make up the over-story in a mature forest canopy. Fire, an important aspect of litter mass balance in natural systems, has not been introduced into many of the systems reported here. This may have a profound effect on litter dynamics and the formation of inert organic matter; but will not be discussed further in this paper.

Litter is the feedstock for soil organic carbon and perhaps not surprisingly this affects the depth distribution of carbon in soils. Even within the top 10 cm of a natural (unmined) forest soil at Gove, carbon content declines steeply from the top centimetre of the mineral soil (ca. 2.6%) to the five to ten centimetre zone (ca. 1.2%) (Figure 3). This type of depth profile for carbon has been commonly reported for other soils in Australia (Spain *et al*. 1983). In the rehabilitated soils there is no depth discrimination of carbon in the earliest year and it takes five years to develop a measurable difference in the profile. As the age of rehabilitation progresses so does the discrimination of carbon percentage with depth. By year 14 this reflected the natural soils. After 14 years soil carbon concentrations increase to approximately double the values found in the natural soils. In such cases this represents a significant amount of extra carbon sequestered. In the sites that were measured between 20 and 26 years there is a notable decrease in the concentration of carbon in the soil and this may represent a stabilisation of organic carbon dynamics towards the native quotients. If this is so the long-term prospects for sequestering carbon in the soil may be no greater than the surrounding natural forests. Soil carbon, however, has a very complex chemistry and the total carbon measurements (based on wet oxidation or dry combustion) mask these details. Molecular analysis is possible using advanced analytical tools such as solid state nuclear magnetic resonance and pyrolysis followed by gas chromatography – mass spectrometry, that can be applied to characterise soil organic matter

compounds in great detail. However, a useful intermediate technique is to characterise the organic carbon by particle size fractionations (Cambardella and Elliott 1992). In this method organic particle sizes are separated by wet sieving, and the size fractions are known to be related to the stability and recalcitrance. This in turn is related to the residence time of the organic carbon in the soil: the smaller the size fraction the longer the residence time.
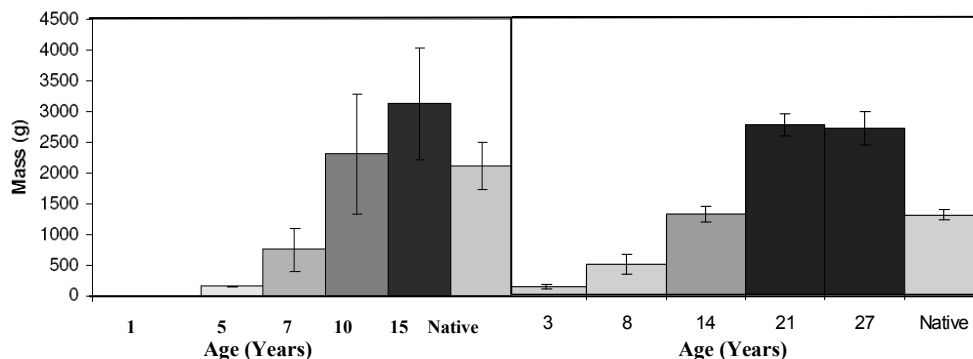

Figure 2. Surface litter masses (g/m^2) of land rehabilitated with *E. marginata* (Jarrah) forest at Boddington (left panel) and *Eucalyptus tetradonta* forest at Gove (right panel) and comparative litter masses on unmined (native) forest soil. Bars = SE. (After Tibbett 2010).

Various computer models attempt to determine the fate of these different fractions as soil carbon "pools". These models attempt to predict changes in soil carbon pools through time under different conditions. Their accuracy is limited but they provide a useful tool to anticipate the effects of changes in land use, management and climate. Their precision is partly governed by how well they can represent processes in soil and their calibration to real soils and conditions.

Some of these models have been commonly used in Australia and include RothC, CENTURY and Socrates (Kirschbaum *et al*. 2001). The century model tries to predict the active, slow and passive pools and has been used to model organic matter at the Weipa mine in Queensland (Schwenke *et al*. 2000b). The results of this modelling (for 100 years) showed that total soil organic carbon would increase steadily and that both active and slow pools would increase with time. The passive soil organic carbon, however, remained constant over this period.

Conclusions
The capacity of reforestation to accumulate carbon in soil, and sequester it there for the long-term, may be more limited than superficial measurements of carbon reveal. In studies outside the mining industry this has been shown to be the case (Arai *et al*. 2007; Jackson *et al*. 2002; Richter *et al*. 1999; Schlesinger and Lichter 2001) but the time frames and mechanisms needed to sequester recalcitrant carbon in the soil remain elusive. On this basis, there is clearly a need for more detailed studies on the soil carbon dynamics of post-mining reforestation schemes, both in terms of measurements of stocks, fractions and fluxes as well as predictive modeling.
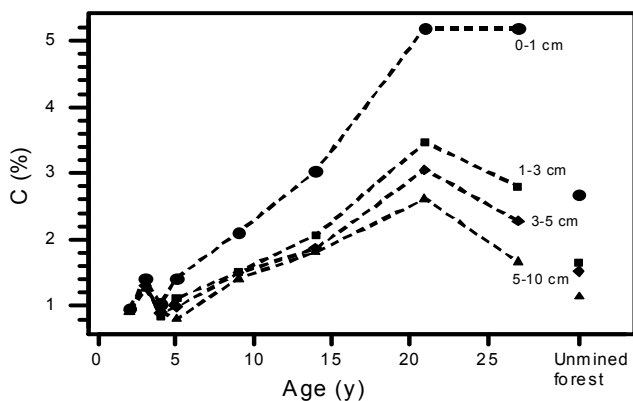

Figure 3. Carbon concentration at a range of depths in the upper profile in the soils of a 26 year chronosequence of rehabilitated sites (and in adjacent unmined native forest sites) at the Gove mine site (after Spain *et al*. 2005)

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