Biochar assessment

The term “biochar” encompasses a wide variety of materials that vary in their chemical and physical composition and in their effects on soil biology and productivity. The degree of variability is affected by the type of feedstock used as well as by the type of biochar production. A fundamental prerequisite of this project was a comprehensive biochemical and physical assessment of biochars produced from different feedstocks and under different conditions in order to determine the degree of variability between and within biochars derived from different feedstocks.

Biochar types: nutrients, carbon, pH and CEC

Biochar varies widely in its chemical characteristics. Figure 1 provides an example of the variation in some key chemical characteristics measured across the biochar database and evaluates the spread of data as a function of feedstock. The data reflect the inherent diverse nature of biochars and as such enable the targeted application of biochars for specific purposes. For example, carbon (C) content varies from very high (>90%) to comparatively low (<30%), while the pH range spans from acidic (<5) to strongly alkaline (12).

Biochar and soil interaction

Subsequent to the findings on the variability of biochar, results from several experiments of this project have shown that the effects of one type of biochar can vary depending on which soil type, which rate and under which climatic conditions it is applied. Similar interactions are known to occur with fertilisers and composts and as such, a similar degree of specification is needed for biochars to ensure the best possible outcomes.

Biochar stability in agricultural soils of Australia

To understand the C sequestration potential of biochars in soils, a long-term (~2-year) laboratory incubation study was undertaken. Two wood biochars (produced at 450 °C and 550 °C) were incubated at the rate of 2% w/w with four agricultural soils of contrasting soil properties (WA tenosol, NSW ferrosol, SA calcarsol, QLD vertosol) at 20, 40 and 60 °C. A carbon isotopic approach was used to quantify the stability of the biochars in the soils.

The results showed that between 0.7 and 8.9% of the applied biochar carbon was released (mineralised) in the form of CO₂ during the first 18 months of the incubation (Figure 2). The higher temperature (550 °C) biochar was more stable than the lower temperature (450 °C) biochar in all soils. Biochar carbon stability decreased with increasing incubation temperature in all four soil types. The effect of soil properties on biochar carbon stability was evident only at higher incubation temperatures (60 °C), with significantly lower mineralisation of the 450 °C biochar in the ferrosol (NSW) compared with the other soils at 60 °C (Figure 2).
Biochar and mineral associations

The NSW ferrosol contains a significant proportion of iron and aluminium oxides, which were intimately associated with the biochar (Figure 3).

The close association of minerals and higher native organic C content in the NSW ferrosol possibly contributed to the biochar stability in this soil. In the tensof from WA, the electron map of Si indicates that a few quartz particles filled the biochar pore space (indicated by the arrows in Figure 3B) and even fewer kaolinite particles were associated with the biochar particles (as indicated by electron maps of Si and Al in Figure 3 C&D). The minor association of kaolinite with biochar in the WA soil, due to its low (<1.3%) clay content is likely to provide little or no protection of the biochar-C against mineralisation.

Biochar and GHG mitigation

One of the most potent greenhouse (GHG) gases released in the agricultural soil environment is nitrous oxide (N₂O), with a 280 times greater GHG potential compared with CO₂. While previous studies have reported that biochar may have a mitigating effect on N₂O emissions from soil, the data were inconsistent. As a result, this project invested into several laboratory and field studies to determine under what soil, and climatic conditions a mitigating effect is possible.

Biochar and N₂O: field experiments

Rainfall events during summer when no crop is grown contribute up to 50% of annual N₂O emissions from semi-arid soils (Barton et al. 2008). The capacity for biochar to mitigate N₂O emissions from these soils was investigated in Western Australia using poultry litter biochar (Figure 4). Results indicate that biochar did not alter N₂O emissions after summer rainfall events or during the winter period (Figure 5).

Concurrent field trials were undertaken in northern NSW using similar poultry litter biochar over the summer crop 2011-2012. Figure 6 shows differences in emissions between treatments and Figure 7 shows the automated chamber set-up.
Table 1. Total N₂O emissions from the Wollongbar field site (auto-chambers)

<table>
<thead>
<tr>
<th>N₂O-N (kg ha⁻¹)</th>
<th>N₂O Emission Factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urea</td>
<td>1.5</td>
</tr>
<tr>
<td>Biochar plus urea</td>
<td>1.4</td>
</tr>
<tr>
<td>Poultry litter</td>
<td>4.9</td>
</tr>
<tr>
<td>Litter</td>
<td>8.4</td>
</tr>
</tbody>
</table>

Figure 7: The N₂O field experiment on ferrosol in northern NSW, using automated chambers for greenhouse gas measurements.

While no differences were detected in emissions between urea and biochar plus urea, statistically significant reductions in emissions of N₂O were detected when comparing raw poultry litter with the pyrolysed form (i.e. poultry litter biochar) (Table 1). Effects on corn yield and N uptake were similar between poultry litter and its biochar; urea alone resulted in lower yield and lower N-use efficiency than biochar or biochar plus urea.

Biochar and N₂O: laboratory experiments

Laboratory incubation studies using large soil cores have demonstrated a range of effects of biochar on GHG emissions in different soil types (Figure 8). Up to 70% reduction in emissions from a WA tenosol was possible when this soil was maintained in a very wet condition, conducive to denitrification (Table 2). The most effective biochars for GHG reduction were oil mallee and wheat straw biochars, while poultry litter biochar was less effective. The incubation studies revealed that some soil types (e.g. the vertosol) did not produce detectable emissions of N₂O when collected from the field, but when synthetic root exudate was applied (a microbial C source), significant emissions occurred. It was also shown that biochar could help to reduce the loss of Nitrogen due to leaching from the soil cores.

Table 2: Emissions (in kg N₂O-N ha⁻¹) from 8 months of analyses.

<table>
<thead>
<tr>
<th>Soil Type</th>
<th>Nil biochar</th>
<th>Oil Mallee biochar</th>
<th>Wheat straw biochar</th>
<th>Poultry litter biochar</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calciosol</td>
<td>0.36</td>
<td>0.06a</td>
<td>0.10</td>
<td>0.12</td>
</tr>
<tr>
<td>Vertosol</td>
<td>0.01</td>
<td>0.01</td>
<td>0.01</td>
<td>0.02</td>
</tr>
<tr>
<td>Ferrosol</td>
<td>0.05</td>
<td>0.05</td>
<td>0.09</td>
<td>0.18</td>
</tr>
<tr>
<td>Ferrosol plus synthetic</td>
<td>0.20</td>
<td>0.38</td>
<td>0.46b</td>
<td>0.18</td>
</tr>
<tr>
<td>root exudate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tenosol</td>
<td>1.95</td>
<td>0.58a</td>
<td>0.95a</td>
<td>1.14</td>
</tr>
<tr>
<td>Tenosol plus synthetic</td>
<td>2.13</td>
<td>0.63a</td>
<td>0.72a</td>
<td>1.58</td>
</tr>
<tr>
<td>root exudate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Factors 'a' = significant (p<0.05) decrease in emission compared with control

A range of mechanisms for these impacts on N₂O production were studied. It was shown that biochars can increase soil porosity and soil air content, thus reducing effects of water-logging. The biochar treatments also altered the soil microbial communities associated with production of N₂O. Work is continuing to optimise the benefits of applying biochar to reduce the production of N₂O.

Biochar and life cycle assessment

Greenhouse gas impact of different biochar systems

Life cycle assessment (LCA) was used to systematically quantify the net GHG emissions and removals across the biochar life cycle. The assessment included emissions from the production, transport and processing of biomass, the construction and operation of the pyrolysis plant, the transport and application of biochar, and quantification of avoided emissions from stabilisation of biomass, displacement of fossil energy sources, and reduced emissions of N₂O from soil. For each scenario, the impact on GHG emissions was assessed by comparing with the applicable reference system, which represents the conventional use of biomass, source of electricity and fertiliser.

Most biochar scenarios examined led to substantial reduction in GHG emissions. The greatest reduction, 3.2 kg CO₂-e per kg biochar, was estimated for poultry litter biochar applied to maize growing in a ferrosol on the NSW north coast. Greater benefits were seen from application of biochar to maize than to wheat, due to:

• assumed greater yield response of maize to biochar
• greater GHG emissions from conventional maize production compared with wheat (thus greater potential to reduce emissions with biochar application)
• higher N₂O emissions from soil in the wetter climate in which maize is grown conventionally (hence greater potential for emissions reduction when biochar is applied).

Benefits were greater for higher than lower temperature biochars, due to their greater stability (Figure 9). The results are highly sensitive to the assumptions employed.

Figure 8: Examples of incubation chambers with the four soil types.

Figure 9: Net GHG emissions from use of 1 kg biochar in a maize or wheat cropping system. Labels indicate the target crop (maize or wheat), the biochar feedstock (GW: greenwaste; PL: poultry litter; WS: wheat straw) and pyrolysis temperature (450 °C or 550 °C).
Biochar does not always reduce emissions compared with the reference system: disposing of biomass in a landfill facility with methane capture and electricity generation may give greater GHG mitigation than using the biomass for biochar.

As the result is highly situation-specific, the optimal use of biomass should be evaluated for each case in which biochar use is proposed, considering also other environmental and production objectives. An example of a C-negative (i.e. storing C) case is shown in Figure 10.

**Impact of biochar on herbicide efficacy**

Biochars have been reported to affect pesticide behaviour (degradation and bioavailability) in soil to the extent that their efficacy may be compromised. This project evaluated 450 °C-wheat straw biochar for its ability to influence the efficacy of two commonly used herbicides with different modes of action (atrazine and trifluralin) in controlling ryegrass in the NSW ferrosol and the SA Calcarosol.

The results from this study showed that biochar did affect the weed control of ryegrass with the two herbicides. However, the effect depended on the herbicide and to some extent on soil type. The effect of biochar on atrazine efficacy was so prominent that in soil receiving 10 t ha\(^{-1}\) biochar, the herbicide rate had to be increased 2-3 times (Figure 12). The effect was much smaller in the case of trifluralin efficacy and only a slight increase in application rate was needed for weed control. Atrazine is more mobile than trifluralin and therefore was in greater contact with biochar particles, allowing a more efficient deactivation by biochar compared with the more stationary trifluralin (Martin et al., in press).

For both herbicides, the adjustment needed was higher in the calcarosol than in the ferrosol. These results suggest that the effect of biochar in reducing herbicide efficacy would vary with herbicide and soil type. However, these results from laboratory studies need to be verified under field conditions and over longer time frames as it is possible that the detrimental effect of biochar on herbicide efficacy may disappear rapidly as the biochar surface undergoes weathering.

**Recommendations for biochar use**

It is important for practical application to consider the limitations of analysing pure biochar materials: We found strong dependencies between biochar and soil types, affecting how well a particular biochar function depends on the herbicide and to some extent on soil type. The effect of biochar on atrazine efficacy was so prominent that in soil receiving 10 t ha\(^{-1}\) biochar, the herbicide rate had to be increased 2-3 times (Figure 12). The effect was much smaller in the case of trifluralin efficacy and only a slight increase in application rate was needed for weed control. Atrazine is more mobile than trifluralin and therefore was in greater contact with biochar particles, allowing a more efficient deactivation by biochar compared with the more stationary trifluralin (Martin et al., in press).

For both herbicides, the adjustment needed was higher in the calcarosol than in the ferrosol. These results suggest that the effect of biochar in reducing herbicide efficacy would vary with herbicide and soil type. However, these results from laboratory studies need to be verified under field conditions and over longer time frames as it is possible that the detrimental effect of biochar on herbicide efficacy may disappear rapidly as the biochar surface undergoes weathering.