A review of evidence for the potential role of biochar to reduce net GHG emissions from New Zealand agriculture

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SOIL SCIENCE

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A review of evidence for the potential role of biochar to reduce net GHG emissions from New Zealand agriculture

A report prepared for the New Zealand Ministry of Primary Industries and the New Zealand Agricultural Greenhouse Gas Research Centre
by

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Glossary of terms

AC ($ t^{-1} CO_2e)	Abatement Cost (value in $ of 1 tonne CO_2e reduction)
ACVM	Agricultural Compounds and Veterinary Medicines Act
ANZBIG	Australian and New Zealand Biochar Industry Group
BC	Biochar (a charcoal created from waste biomass for the purpose of a C-sink)
BC_{+100} %	The % of biochar carbon content that will last in the soil for over 100 years
BECCS	Bioenergy with C capture and storage
BQM	Biochar Quality Mandate
CCA	Copper-chrome-arsenate timber treatment
CDR	Carbon dioxide removal
Climate positive	or Carbon negative (actively decreasing atmospheric CO_2 concentration)
CO_2e	The GWP of GHG emissions expressed as carbon dioxide equivalent
Co-pyrolysis	Mixing two feedstocks for added benefit of heat or nutrient provision
C_{org}	The organic carbon content of a biochar
C-sink	A form of mineral or organic C storage that is unable to exchange with atmospheric CO_2
DACCS	Direct air C capture and storage
EBC	European Biochar Certificate
FC	Field capacity, the maximum water content of a soil after allowing for drainage of free water
Feedstock	The source of biomass for biochar or bioenergy production
GHG	Greenhouse gas
GWP	The global warming potential of greenhouse gases
HHT	Highest heating temperature
IBI	International Biochar Initiative
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardisation
Kiln	A large oven for burning biomass with limited ability to restrict oxidation of C to CO_2
LCA	Life Cycle Assessment
MS	Milk Solids
MSDS	Material Safety Data Sheet
NET	Negative emissions technologies
PAWC	Plant available water content
PWP	Permanent wilting point, the water content of a soil when plants are
QA/QC	Quality Assurance/Quality Control
Retort	An enclosed oven that allows thermal decomposition in the absence of air with ability to collect or burn volatile oil and gas distillates
RMA	Resource Management Act
SEPA	Scottish Environmental Protection Agency
t CO_2e t^{-1} BC	tonnes CO_2e per tonne biochar

1. Executive Summary

Biochar production to contribute to net GHG reductions

Biochar production is one of a limited number of negative emissions technologies (NET) considered by the Intergovernmental Panel on Climate Change (IPCC) meeting in Paris (4 November 2016) for carbon dioxide (CO_2) removal (CDR) from the atmosphere as a way of limiting global warming to 2 ^oC (Section 2).

Biochar is a carbon (C)-rich solid product generated by pyrolysis (thermal decomposition under limited supply of oxygen) of biomass (such as wood, prunings and crop, food or animal residues, and referred to as feedstock, Section 2.1). After manufacture, biochar is added to soils and can, depending on the biochar and soil properties, improve soil properties and growth of plants by acting as a fertiliser or liming material, or by improving soil drainage and aeration.

The climate-positive effect (CO_2 reduction) of biochar manufacture exploits the fact that plants and trees remove CO_2 from the atmosphere through photosynthesis (Section 2.1). Usually the resulting residual biomass degrades naturally, or through traditional management pathways (such as composting, burning and landfill, Figure 2.1) to carbon dioxide (CO_2), carbon monoxide (CO) or methane (CH_4). Many of the degradation pathways return these gases to the atmosphere in minutes (burning), days and months (crop residue decomposition in soils), or over years (compost and forest residue decomposition). However, if instead, a fraction of the carbon in this biomass is “locked up” in the form of long-lasting materials such as wood or biochar, it can contribute to a net reduction in greenhouse gases (GHG) in the atmosphere. In particular, biochar is rich in condensed, aromatic-C (Section 2.3.1), which because most microbes lack the required set of enzymes to degrade it, has degradation half-lives 100 to several 1000 times longer than the feedstock biomass. Thus, stockpiling biochar, or biochar application to soil, creates a long-lasting C sink, preventing the return of CO_2 or CH_4 to the atmosphere. Furthermore, as new plants and trees grow, they continue to fix CO_2 from the atmosphere via photosynthesis and, if the surplus biomass is converted to biochar on a regular basis, this will contribute to ongoing GHG mitigation.

Scientific evidence and protocols for including biochar production as a climate change mitigation

New Zealand’s primary industries are rich in biomass harvest residues, which are potential feedstocks for biochar manufacture. However, policy makers facing the challenge of reducing New Zealand’s GHG footprint need research evidence on the feasibility of a domestic biochar industry contributing to net GHG reductions.

This review searched over 18,000 research publications to synthesise research findings on: how feedstock type and pyrolysis conditions influence the C sink value of biochar, the energy value of pyrolysis co-products (heat, gas and oil), and the biochar properties that add value (e.g. fertiliser and liming value and reduction of non-CO_2 GHG emissions) when biochar is applied to soil (Section 2). Additionally, Life Cycle Assessment (LCA) studies of biochar production and use were analysed to understand the relative significance of different feedstocks, pyrolysis conditions, fate of co-products, and different uses of biochar in contributing to mitigating climate change (Section 3).

Published research into the concept of biochar production for climate change mitigation is less than 15 years old and currently there are few international examples of biochar production at scale. The main reason for the lack of adoption of biochar manufacture as a CDR technology is that the costs associated with biochar production are not currently covered by the C market, or by the biochar fertiliser and liming values. A lesser reason is that the bulk (70%) of the scientific research providing evidence for (i) the variation of degradability of biochar made from different feedstocks and its C-sink value, and (ii) the agronomic effectiveness of biochar as a fertiliser and liming value, has been published in only the last four years.

Confidence in the research evidence that biochar production can provide a cost effective GHG reduction strategy for low-income nations with biomass residues prompted the IPCC (in 2019) to propose a protocol for biochar to be included in national inventories as a soil amendment contributing to increases in soil C (Section 5). Accounting for biochar C in national GHG inventories and emissions trading schemes has been made practicable by voluntary associations such as the IBI (International Biochar Initiative), EBC (European Biochar Certificate) and ANZBIG (Australian and New Zealand Biochar Industry Group). They have been developing biochar certification schemes for biochar producers to demonstrate that the appropriate ethical, physical and chemical compliance standards of a C sink and environmental regulations are met (Section 2.2 and 2.3).

Biomass (feedstock) resources for a New Zealand biochar industry

New Zealand's biomass resources that could be diverted to biochar manufacture consist of forestry residues, wood processing residues, agricultural residues (e.g. orchard residues, stover, straws and animal manures), processing residues (juicing marc, packhouse rejects, peelings, paunch grass from meat processors) and municipal waste (e.g. yard waste, food waste, bio-solids or sludge). Research evidence indicates that woody residues (forestry residues, wood processing residue, orchard prunings) provide the greatest yields of biochar with properties most suited for use as a long-term C sink. The pyrolysis of food waste, bio-solids or sludge and animal manures yields less biochar C but greater value as fertiliser as they contain phosphorus, potassium, and magnesium (Section 2.1.2 and Section 4). However, determining the suitability of each of these feedstocks for biochar production should also take into account the total amounts available (Section 2.2), degree of aggregation at point of generation (i.e. concentrated versus distributed), and risks to be mitigated either in the current management regime, or associated with subsequent use of the biochar (e.g. non-CO_2 GHG emissions from wastes directed to landfill or composting, the risk of leachate from mismanaged composting of grape marc, abandoned logging residues being washed down rivers to populated areas, or poor selection of feedstocks containing heavy metals that then are applied to agricultural soils).

Concerns about biochar increasing soil and atmospheric contamination

Whilst concerns have been raised about the processing of municipal wastes into biochar and the risks of transferring unwanted heavy metal and organic contaminants to soil, this is not a concern with clean woody residues, crop wastes and aggregated packhouse or food processing residues that are pyrolysed under well-controlled conditions. In addition, the risks of soil contamination with biochars made from municipal wastes can be avoided by chemical analysis-audits, showing compliance with the IBI and EBC certification standards. If biochar meets these standards, it will also comply with the New Zealand regulations for avoiding soil contamination (Section 2.3).

Choosing the appropriate kiln, retort or converter for the type of feedstock and scale of operation influences: (i) the yield of biochar as a % of feedstock (10-30%), (ii) the quality of the biochar, (iii) the level of air pollution, and (iv) the GHG footprint of the pyrolysis operation (Section 2.2). For example, operation of simple kilns may result in significant emissions to soil in the form of tars, and to the atmosphere in the form of volatiles and non- CO_2 GHGs. When these emissions are not processed in a thermal oxidiser (high efficiency flare), they can negate any climate positive effect of the carbon in the charcoal. Therefore, it is not recommended that simple kilns be used at scale. Thermal oxidation of pyrolysis volatiles and gases requires a significant jump in technology complexity, which justifies the use of more complex reactors (retorts or converters) to minimise the emissions footprint of biochar production. These reactors can also utilise the exhaust CH_4, CO and other volatile organic compounds to generate heat to drive the pyrolysis reaction and/or recover gases and oils that can be further refined as fuels. Retorts and converters operating at scale have pollution control technology that enable atmospheric discharges to comply with clean air regulations.

Current carbon footprint and costs of biochar production

The C footprint of biochar production can be addressed by undertaking consequential Life Cycle Assessment (LCA). Such modelling compares the GHG emissions and other environmental impacts of the usual pathway for the biomass (e.g. landfilling, composting of municipal and horticultural wastes) with the proposed biochar pathway.

The vast majority of LCA studies of biochar production from forest residues, agricultural and horticultural residues, including subsequent biochar application to land, have calculated net climate-positive results (Section 3). The magnitude of the benefit depends upon the type of feedstock and whether the feedstock needs drying. Additional climate-positive benefits are gained from (i) using the heat and bio-gas/oil by-products to displace heat and electricity production using fossil fuels, and (ii) the reduction of non-CO_2 GHG emissions that may be associated with the usual fate of the biomass, or with the application of the biochar to soil.

A review of techno-economic assessments of biochar production indicates that the costs of biochar production currently range from 300 – 800 $NZ t^{-1} biochar (Section 4). In cases of co-production of heat, gas and oil, these costs can be defrayed to some extent by displacement of energy generation from existing fossil fuel use but only where a conversion from fossil fuel to biomass has occurred (e.g., converting a hospital boiler from coal to burning biomass): this defraying of energy costs is not possible on greenfield developments, which have no existing NZUs associated with the use of fossil fuels, for which offsets could be claimed and unused NZUs sold on the NZETS. The other major product of pyrolysis is condensable bio-oil, which is easily combusted to heat. Commercial technology for turning bio-oil into cost competitive transport fuels awaits large-scale development, and the most likely route is as a feedstock to oil refineries. Also, pyrolysis of waste streams (municipal wastes and horticultural processing wastes) where the traditional management pathway has landfill fees, may be able to completely offset the cost of biochar production. The existence of a large waste incineration industry in many European cities suggests that it is feasible to direct wastes to biochar production.

LCA combined with techno- economic assessment can be used to calculate the financial cost/benefit of mitigating one metric ton of CO_2e GHG emissions by producing biochar, relative to a traditional pathway. This is termed the GHG abatement cost and expressed as \$ t^{-1} CO_2e.

The GHG abatement cost and compliance need to be considered in the context of achieving a climate-positive result compared to the traditional pathway for the waste or residue. Examples range from simply being left on the ground to mulching *in situ*, or collection then land-spreading, composting, or burning (e.g., the fate of many orchard residues), or even landfilling. Making biochar is an alternate to these where the residue is redirected to approved commercially available kilns, which are retorts mounted inside a firebox, where the pyrolysis volatiles and gases are recycled to the firebox for oxidation. To reach the temperatures required to remove the products of incomplete combustion the kiln may need augmenting with a diesel or gas burner. Open flame curtain systems are not recommended as the curtain is unstable, especially in wind, and have high risk of fugitive GHG emissions, particularly in the early and late stages of operation. Also, these systems have a much lower ratio of biochar yield to biomass feedstock, usually less than 1:10. Together these factors make it unlikely that flame curtain systems will deliver a climate-positive impact. Nevertheless, they have gained popularity in United States for mitigation of forest floor fuel loads as part of forest fire prevention as they deliver benefit compared to open fires (Section 4.1). Typically adding technological complexity to mitigate pyrolysis emissions and their associated environmental impacts adds cost to biochar production and is more appropriate for compliance of large-scale operations.

Three case study examples illustrate the different scales of biochar production that could take place with on-farm wastes, horticultural processing wastes and forest residues. Firstly, the removal of prunings from kiwifruit orchards (Section 4.3). Based on the assumption that a kiwifruit orchard has 5.3 t ha^{-1} of air-dry pruning residues available per year (that are normally burnt), the yield of biochar (BC) from a flame curtain kiln would be 0.53 t BC ha^{-1}. It is estimated that the biochar would be 77% C but only 70% of this amount would last in the soil for over 100 years (BC_{+100}), i.e. 0.29 t C ha^{-1}(or 1.05 tCO_2e ha^{-1}). If the sum of emissions during feedstock collection and processing plus the non-CO_2 GHG emissions (CO, CH_4, particulates) during pyrolysis are equivalent to 34% of the biochar CO_2e, then biochar production is estimated to have a positive climate effect of -0.69 CO_2e t ha $^{-1}$. (The negative indicates a net sequestration of carbon, which is a positive climate effect.) This would offset 15% of the 4.6 t CO_2e ha $^{-1}$ of GHG emissions produced in normal operation of a kiwifruit orchard (see more detailed discussion in Section 4.3). The cost of producing the biochar was estimated to be about 1020 NZ\$ t^{-1} BC. After deducting the emissions due to production, then abatement cost was calculated as 777 NZ\$ $t^{-1}CO_2$ e, which is about ten times higher than the estimated, combined carbon and fertiliser and lime value of the biochar (53 NZ\$ t^{-1}BC). This illustrates the inefficiency of the flame curtain kiln in producing a small C-sink value relative to the equipment and labour costs. An additional issue is that with poor kiln management, the fugitive emissions of products of incomplete combustion (CO, CH_4, particulate) during pyrolysis are likely to be higher than 34% of the C sink (BC_{+100}) value.

Secondly, for grape marc (GM), of which Marlborough currently produces 46,000 tonnes per annum, the cost of biochar production using state-of-the-art kilns was calculated to be 495 NZ\$ t^{-1}BC. As comparison (Section 4), the net cost of composting is 18 NZ\$ t^{-1} GM and of biochar production is 56 NZ\$ t^{-1} GM; both are currently a net cost to winegrowers. When carbon footprints are compared, biochar is climate positive and significantly better at -225 kgCO_2e t^{-1} GM than the +20 kgCO_2e t^{-1} GM for composting. The economics become profitable when biochar is sold as charcoal heating fuel, returning a net revenue of +17.60 NZ\$ t^{-1}BC (+2.00 NZ\$ t^{-1} GM), but has a carbon footprint of +24 kgCO_2e t^{-1} GM, similar to composting. Where the economics and environmental benefits really come together is in replacing coal boilers with biomass boilers. New Zealand has many coal boilers in hospitals, schools and other public facilities, and the government has a policy to convert these to renewables. It is less costly to modify coal boilers to use biochar than building biomass burning boilers. Both economic and environmental benefits are enhanced when the offset value is monetised on the NZETS. For example, where the NZU price is 37.20 NZ\$ $t^{-1}$$CO_2$e (as on 8/12/20), the revenue rises to 120.00 NZ\$ t^{-1}BC (or 13.65 NZ\$ t^{-1} GM) and the carbon footprint rises to -256 kgCO_2e t^{-1} GM, which reflects the coal replaced. These are encouraging numbers, but it must be remembered that once all the coal boiler conversions have occurred in New Zealand, this business model no longer applies. Then, the only repurposing option that delivers net carbon footprint benefit is biochar applied back to the soil at -225 kgCO_2e t^{-1} GM.

Thirdly at a larger scale, currently 80% of the harvestable biomass residues in New Zealand are distributed forest harvest residues. Not all residues are easily harvested. Based on the assumption that 2.5 Mt of green harvestable forest residues are available per year (Section 4) and will yield 1.4 Mt dry wood which, when pyrolyzed by distributed medium-scale retort kilns fitted with best practice flare systems, could yield 0.4 Mt biochar and 0.17 Mt C as a long-lasting C sink (after process emissions have been deducted), then carbon dioxide removal is 0.62 Mt CO_2e y^{-1}. This is equivalent to reducing New Zealand's agricultural GHG emissions of 37.7 Mt CO_2e y^{-1} by 1.6% (Section 4.11). If kilns are centrally located near other processing facilities, the net benefits become even more climate-positive if the heat generated and biogas and oil are used to displace coal and/or gas energy generation. Even so, the benefits are reduced by the economic and environmental cost of transporting the biomass to the central location, and the onward transport of the biochar to the soil repository.

Costs of biochar production may be defrayed because biochar can confer other benefits to some soils, as a fertiliser, liming material and soil physical conditioner, which may increase crop yields and decrease non-CO_2 GHG emissions from soils (Section 2.1 and Section 4). The extent of these benefits are impossible to generalise and therefore difficult to monetise. Such benefits would need to be evaluated under New Zealand conditions on a case by case basis, because the value realised is not only dependent on the feedstock source and biochar properties, but also on the condition and management of the soil and crop grown. The fact that biochar can have added value as fertiliser or liming material provided that the soil nutrient or pH condition is either limiting root and/or crop, is consistent with the results of large meta-analyses of international research studies into the impact of biochar on crop yields, which show that positive yield increases occur predominantly on low fertility, low organic matter content, acidic soils.

Rather than direct application of biochar to the soil, the use of biochar as a conditioner in green-waste composting is consistently reported to reduce non-CO_2 GHG emissions during composting (Section 2.14). At a 10% (1:9) application rate of biochar to green waste, a reduction in GHG emissions equivalent to 1.94 t CO_2e $t^{-1}BC$ can be achieved during composting, adding a further 68 NZ$ $t^{-1}BC$, provided the emissions reduction could be traded on the NZETS at 35 NZ$ $t^{-1}CO_2e$.

Most promising pathways for a New Zealand biochar industry to contribute to net GHG reductions.

Retorts and converters operating with pollution control technology can produce biochar from waste biomass with climate positive benefits as a C sink (Sections 3 and 4). The high monetary cost of biochar production (Section 4) relative to the low C credit value of the biochar, however, currently limits the development of a biochar industry both internationally and in New Zealand. Development of improved technologies is being held up because investors (and others) are awaiting New Zealand-specific research evidence (proof) on whether there are consistent yield gains from biochar application to crops, pastures, orchards and forests that can offset production costs. To overcome this impasse, trials of promising pathways for biochar production from waste biomass should be encouraged, both to stimulate innovation in biochar production technologies and to address the research needs. These are:

(i) Portable, small to medium-scale pyrolysis operations producing biochar and useable heat from orchard and shelterbelt prunings that are normally incinerated, composted, or mulched. Use of this biochar should be targeted to application in orchard trials.
(ii) Medium-scale pyrolysis operations producing biochar and useable heat from horticultural and vegetable processing wastes that are normally composted. Use of this biochar should be targeted to replace fossil fuels and/or applied in crop trials.
(iii) Medium-scale pyrolysis operations producing biochar and useable heat from "clean" municipal wastes (untreated wood residue and food industry wastes) that normally incur both emissions and tipping fee costs when sent to landfill.
(iv) Small-scale pyrolysis operations producing biochar from clean crop and forest residues (cereal straws and landing site residues) that can be used in feeding trials to evaluate potential reduction in ruminant CH_4 emissions and increase animal growth rates.

Encouraging the implementation of small to medium-scale pyrolysis operations, which include contained thermal oxidation of pyrolysis volatiles and gases, will lead to innovation on how to reduce the costs of pyrolysis and allow wider experimental evaluation of how to derive added abatement and economic value from biochar.

Beyond the climate positive benefits of producing biochar, there is a need for more scientific evidence for the additional climate positive, or negative, attributes of biochar application to soils in temperate climates similar to that in New Zealand. In New Zealand, more information is needed with respect to the significance of effects such as positive or negative priming of soil organic C, soil N_2O emissions, and crop, pasture, orchard and forest yield responses.

Pathways to be avoided.

Essential work on measuring and monitoring undesirable emissions and unintended consequences of biochar production should be initiated. Encouraging the implementation of small- to medium-scale operations must not be done in the absence of pyrolysis certification standards that address unwanted emissions of gas, volatile organics and heavy metals to the soil and atmosphere. Kilns and feedstocks that have health and safety issues, unwanted environmental emissions and contamination risks should not be encouraged.

Policy implications for a pathway to reducing NZ emissions

Timely development of policy changes to allow biochar production and use to be included in forest, farm and waste industry GHG budgets will be needed to encourage the investment in, and construction of small- to medium-scale pyrolysers. At the national inventory scale, farm and waste industry GHG budget guidelines can be developed that reflect and complement the proposed IPCC protocols (Section 5) for accounting for biochar.

Recommendations for research

New Zealand needs research directed at lowering the abatement costs of biochar production by (i) providing evidence of the agronomic and environmental value of biochar, and (ii) capturing the value of co-products (heat, gas and transport fuels). In particular, New Zealand bioenergy researchers should evaluate the very recent developments of pyrolysers utilising auger reactors combined with catalytic conversion of bio oil to high quality biodiesel.

Research to calculate actual abatement costs will require biochar from monitored, small and/or medium pyrolysers to be used in long-term field trials, in which long-term soil and plant yield responses can be measured as well as the effects of positive or negative priming of soil organic C and soil N_2O emissions. Results from such trials will allow determination of accurate consequential LCA evaluation of the biochar C footprint and abatement costs. In addition to these trials, New Zealand-specific trials are required to test the methane emission reduction effect of (i) biochar addition to green-waste composting, and (ii) the inclusion of biochar in the diets of grass-fed cattle and sheep.

Carbon stock inventory research is needed to update the quantities of farm, forestry and horticultural residues available for biochar production (including their distribution both in time and space, current fate and risk management profiles).

Experts in biochar characterisation will be required to recommend which material aspects should be included in a New Zealand-based biochar certification scheme, balancing feasibility and accuracy for certification purposes and national GHG accounting.

2. Introduction

The Intergovernmental Panel on Climate Change (IPCC) meeting in Paris (4 November 2016) highlighted technologies for carbon dioxide removal (CDR) from the atmosphere as ways of limiting global warming to 2°C. A limited number of negative emissions technologies (NET) are being considered internationally as CDR pathways to sequester atmospheric carbon dioxide (CO_2) into stable carbon (C) sinks (Fuss et al. 2018). These NETs include direct air C capture and storage (DACCS), enhanced weathering, ocean fertilisation and the land management options afforestation, reforestation, soil C sequestration, bioenergy with C capture and storage (BECCS) and biochar technologies (Smith et al., 2019).

Biochar is a carbon-rich solid product generated by pyrolysis (thermal decomposition under limited supply of oxygen) of biomass (such as wood, prunings and crop, food or animal residues, and referred to as feedstock), which can improve soil properties while contributing to mitigate climate change (Woolf et al., 2010; 2018). The IPCC (2019a, b) has more specific definitions of biochar in their guide to the use of biochar to increase soil C stocks and as an energy offset to fossil fuels through syngas production (see Section 5).

This review focuses on the reduction of greenhouse gas (GHG) emissions that would otherwise occur if the feedstock degraded naturally, or through a traditional management pathway (Figure 2.1) to carbon dioxide (CO_2) or methane (CH_4). The negative GHG emissions balance created by biochar, primarily through CDR, relies on the fact that biochar is rich in condensed aromatic C, for which most microbes lack the required set of enzymes for degradation (Lehmann et al., 2015). As a consequence, biochar is resistant to decay (a long-lasting C sink) with a half-life 100 to several 1000 times longer than the feedstock biomass (Verheijen et al., 2009; Woolf et al., 2018). The amount of condensed aromatic C in biochar (which can be estimated by the $H:C_{org}$ ratio, Budai et al. (2013); Lehmann et al. (2015)) depends on the degree of carbonisation, which is strongly related to the highest heating temperature (HHT) of pyrolysis (Calvelo Pereira et al., 2011). To succeed with the primary purpose of providing a C sink for CDR, the biochar can be stockpiled or applied to soil. Application to soil can create additional benefits where it might improve the physical (structure, aeration, water holding capacity and drainage), chemical (soil acidity, cation exchange capacity, and major or trace element availability) or biological (microbial habitat, denitrification) properties of the soil. Improvement of the soil may lead to plant yield (food and fibre) increases that are able to offset the costs of biochar production. The ability of biochar to positively influence soil properties (Hardie et al. 2014) can be optimised through selection of the feedstock and pyrolysis conditions to deliver specific biochar characteristics, along with (i) determining the background properties of the soil receiving environment, and (ii) applying the right rate and right particle size, in combination with the required fertiliser input (Jeffery et al.,2011; Ye et al., 2020). The variation in the type and condition (e.g. moisture content) of feedstock and pyrolysis conditions (as well as the pre-treatments of the feedstocks and post-treatments of the biochars) can deliver a wide range of products (Camps-Arbestain et al., 2015). This has led the voluntary groups promoting biochar as a CDR technology to establish guidelines for selection and processing of feedstocks, type of pyrolyser and production standards for certification of the quality of biochar (e.g., International Biochar Initiative and European Biochar Certificate standards).

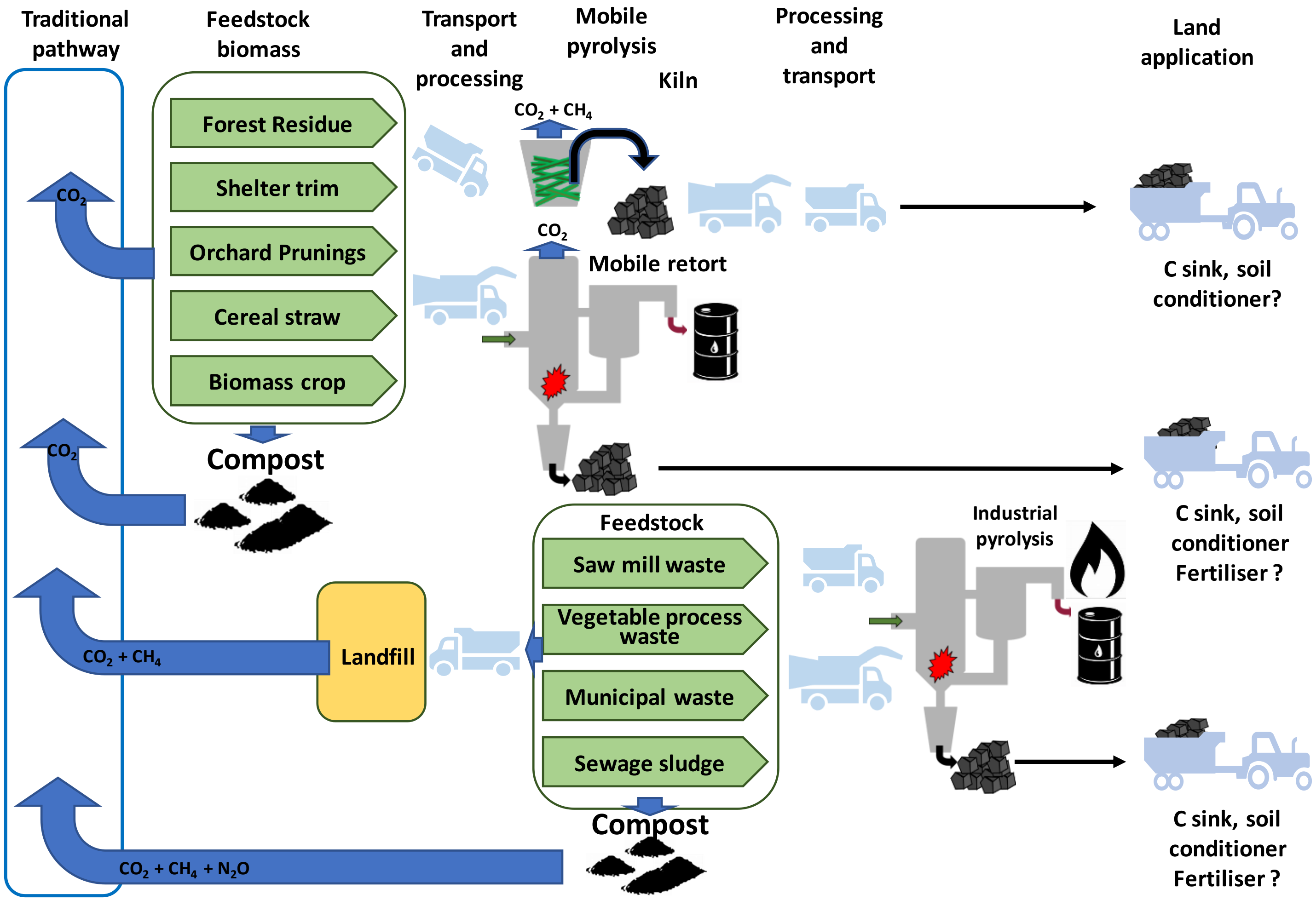


Figure 2.1 Examples of carbon pathways for natural or traditional processes in biomass degradation and biochar production and distribution. Illustrated are typical feedstocks, the nature of GHG emissions in the natural and traditional pathways (i.e. composting and landfill) of biomass decomposition and the operations required for the biomass to be utilised in biochar manufacture, local to or distant from the biomass source, the land application of, and the potential use of the biochar.

2.1 Guidelines for selection of feedstocks and potential biochar products

All voluntary biochar certification schemes (e.g. the Biochar Standards of the International Biochar Initiative (IBI,2015), the European Biochar Certificate (EBC, 2012), the Biochar Quality Mandate (2013) and the ANZ Biochar Industry Group (ANZBIG, 2020)) have codes of practice that make it necessary for the feedstock biomass to be harvested from a sustainable source.

For example, the EBC (2012) defines feedstock sustainability by assessing the C-neutrality of the biomass: “a feedstock material (biomass) for the generation of a C-sink is considered C-neutral if it is either the residue of a biomass-processing operation, or if the biomass removal did not, over the reference period, lead to the reduction of the total C stock of the system in which the biomass had been grown”. Residues from forestry (e.g., forest prunings and harvest residues), agriculture (e.g., cereal straws and stover), horticulture (e.g., tomato, potato and cabbage stalks, and leaves, or vine and tree prunings) and bioenergy crops (e.g., Miscanthus) can be considered C-neutral biomass (Table 2.1), provided the quantity of biomass removed does not decrease soil organic C stocks (Laird & Chang, 2013; Liska et al., 2014; O’Brien et al., 2020). Change in soil C stocks will be a balance between C input into the soil and soil C decomposition. Crop and farm management programmes can be selected to allow biomass harvesting yet maintain or increase soil C stocks. For example, the negative impact on soil C stocks of removal of dry summer crop residues (which are ideal feedstocks for biochar) can be remedied by introducing a strongly rooting cover crop (e.g. short rotation ryegrass or cool season green-manure such as oats) or re-application of the biochar to the site of feedstock harvesting (O’Brien et al., 2020).

Crop, orchard and vineyard residues (e.g., pomace, nutshells and fruit stones, grape marc, etc.) that are created away from the paddock during food processing are considered C-neutral input materials for bioenergy or biochar. This is on the basis that any GHG emissions associated with their generation during food processing are credited to the production of the main primary product or food (e.g. wine, olive or any other kind of oil, fruit juice) (EBC, 2013).

At the time of writing (October, 2020), a few countries in the European Union have narrower definitions of feedstocks that can be used to manufacture biochar that is to be applied to agricultural land. Switzerland, for example, allows only woody biomass as a feedstock for pyrolysis and Germany requires a minimum C content of 80% for biochar, essentially limiting feedstocks to woody residues and untreated wood processing wastes (EBC, 2012). Similarly, the Scottish Environmental Protection Agency (SEPA, 2012) has stated in a position paper that biochar can be manufactured only from untreated wood waste from agriculture, horticulture and forestry activities. In Scotland, the production of biochar from these feedstocks will not require a waste management licence and the biochar produced will be treated as a waste material in terms of the law and may be used only in accordance with the relevant waste management controls.

Table 2.1. The pH, yield relative to the dry feedstock mass (FS), carbon, nitrogen and ash content and molar ratios of biochar produced from a range of feedstocks by pyrolysis at different maximum temperatures and heating rates (O:C and H:C are molar ratios of oxygen, carbon and hydrogen which are used to predict the C sink value of biochar).

Feedstock	Highest Temp.	Heating rate	pH	Yield	C	N	Ash	O:C	H:C	Reference
	^{0}C	^{0}C min^{-1}		%FS	g kg^{-1}			Molar ratio		
Pine fragments	550	24	7.9	28.8	847.0	6.0	41.0	0.06	0.50	Calvelo-Pereira et al., 2011
Pine wood chips	550	100		25.4	859.0	9.0	7.0	0.08	0.43	Crombie et al., 2015
Pine wood chips	650	100		23.0	847.0	14.0	50.0	0.06	0.33	Crombie et al., 2015
Pine saw dust	680	76	9.7		909.0	11.0	10.1	0.12	0.01	Srinivasan et al., 2015
Poplar coppice	550	48	8.8	26.9	758.0	11.0	65.0	0.13	0.56	Calvelo-Pereira et al., 2011
Willow coppice	550	46	8.6	28.8	791.0	17.0	75.0	0.08	0.52	Calvelo-Pereira et al., 2011
Wheat straw	550	100		29.8	696.0	14.0	197.0	0.08	0.28	Crombie et al., 2015
Wheat straw	650	100		29.4	619.0	13.0	276.0	0.10	0.17	Crombie et al., 2015
Corn stover	550	40	9.9	28.0	743.0	7.8	108.0	0.12	0.44	Fuertes et al., 2010
Oil seed rape straw pellets	550	28	9.8		689.0	16.0	195.0	0.10	0.31	Cabeza et al., 2018
Miscanthus straw pellets	550	28	9.8		754.0	8.0	122.0	0.09	0.38	Cabeza et al., 2018
Grape marc	600	5	10.1	32.6	721.3	21.6	85.0	0.21	0.03	Ferjani et al., 2019
Glasshouse Tomato	550		10.4	38.0	549.0	16.0	22.0		0.05	Dunlop et al., 2015
Broiler litter (woodchip base)	680	76	10.1		867.9	13.0	111.6	0.12	0.01	Srinivasan et al., 2015
Paunch grass	680	76	8.8		647.9	18.0	287.3	0.19	0.04	Srinivasan et al., 2015
Sewage sludge	680	76	7.9		779.8	5.0	126.1	0.25	0.03	Srinivasan et al., 2015
Biosolids	550	16		56.0	261.0	26.6	64.8		0.70	Wisnubroto, 2015
Municipal greenwaste	550	16		41.0	452.0	19.6	46.4		0.70	Wisnubroto, 2015

The International Biochar Initiative (IBI, 2015) focusses on standards applied to the biochar product. It states that feedstocks may not contain more than 2% of contaminants (heavy metals, plastics and toxic hydrocarbons), or 10% by weight of diluents (e.g. stones and soil).

The ANZBIG (2020) requires that when processed timber or municipal waste is used as feedstock, all plastic, rubber, some metals, electronic scrap and treated timber (copper-chrome-arsenate, CCA) or timber with reactive coatings (such as laminates, glues, paints, varnishes, fungicide) must be removed from the feedstock for health and safety reasons. The ANZBIG guidelines for biochar standards are a blend of both IBI and EBC proposed biochar certification standards.

The potential presence of contaminants in feedstock triggers the need for biochar product analysis to ensure that final concentrations of any impurities are below the accepted thresholds of the local regulatory authority. The IBI, the EBC, and the ANZBIG publish threshold values (see Section 2.3) for heavy metals, plastics and toxic hydrocarbons (IBI, 2015; EBC 2020, ANZBIG;2020), which reflect, and may default to, contemporary soil toxicity and chemical content reporting requirements for soil amendments, composts and fertilisers of the local regulatory authority.

2.1.1 Carbon sink (application to forest, arable crop and horticultural wastes)

To create a C sink, the feedstock and the pyrolysis system should be geared to producing a high yield of strongly carbonised biochar (see Section 2.31), whilst minimising the C footprint of the collection of feedstock, processing, transportation, storage, pyrolysis and land application of the biochar (process emissions, Figure 2.2). Woody materials such as forest residues, saw dust and non-hazardous municipal greenwaste pyrolysed at high temperatures can create biochars more suited to a role as a C sink with a higher C content and less ash than those produced from human and animal waste (Table 2.1). However, it should be noted that the C yield (the ratio of C in the biochar and that in the original feedstock) also decreases as the temperature of pyrolysis increases.

To fulfil the CDR potential, each batch of biochar must be produced from a sustainable feedstock. Sustainability is most easily visualised for a short-rotation forest (e.g. *Pinus radiata* in New Zealand, 25-30 y rotation), where the C sequestered by prunings and harvest residues could be feedstocks for biochar and will be replaced by the next generation of replanted trees. Decisions on the selection of a feedstock and a biochar production and distribution system should be supported by, at least, a C footprint analysis (EBC, 2020) and, ideally, by a full LCA to ensure that GHG reduction is achieved with no adverse environmental impacts (e.g. Homagain et al.,2015; Matustík et al., 2020). A mandated rule of the voluntary biochar certification schemes is that feedstock biomass production, transport and processing, pyrolysis and biochar distribution creates a net reduction in GHG emissions (Meyer et al., 2017). The reduction in GHG emissions created by biochar production relies heavily on the C sink potential of the biochar (Section 2.31) and biochar's deployment as a soil or compost conditioner (Section 2.13

and 2.14). The C sink potential depends upon the degradation rate of the biochar C being several orders of magnitude slower than degradation rate of the biomass C, when left in its natural, or traditional, pathway of degradation of C. For example, Beyaert and Voroney (2011) found in field trials that corn stover and wheat straw have degradation rates of 0.6 y^{-1} under conservation tillage and 0.7 -0.75 y^{-1} under conventional tillage in the first year after surface application or incorporation into the soil, respectively. By year 15, 91-95% of the biomass C had been lost to atmospheric C under both forms of tillage. At a range of sites in New Zealand, Garrett et al. (2010) measured rates of decay of woody radiata pine residue ranging from 0.0481 to 0.1658 y^{-1} depending on regional climate. In the NZETS (exotic forest residues are assumed to have degradation rates of 0.1 y^{-1} for 10 y (MPI 2017). In contrast, the conservative degradation rate is 0.003 y^{-1} for certified biochar (EBC, 2020) pyrolyzed at high temperatures with H: C_{org} ratios below 0.4 (see section 2.31).

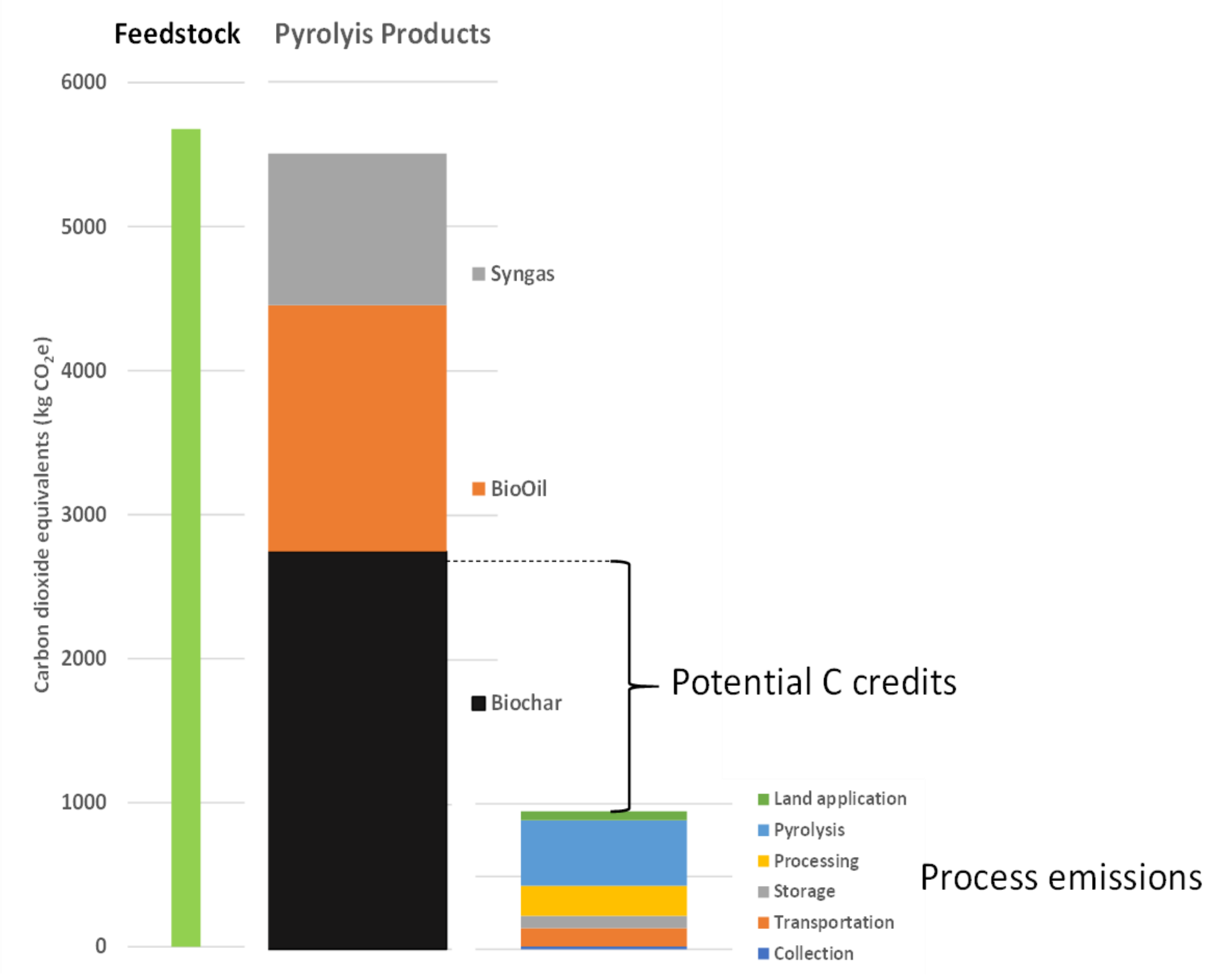


Figure 2.2 Carbon dioxide equivalent budget for producing 1 tonne Biochar (750 kg C) from Forest residue (3.1 t DM). (Data source Homagain et al., 2015).

2.1.2 Fertiliser and lime (co-pyrolysis, animal wastes, municipal green and biosolid wastes)

The essential element content of a feedstock is the single most important factor governing the potential fertiliser and liming value of a biochar. Due to the wide range of essential element concentrations in biochars, not all are suitable as fertilisers. For this reason, IBI has developed certification standards to classify biochars as eligible (or not) for fertiliser use based on criteria provided by Camps-Arbestain et al. (2015) (see Section 2.32).

Biochars manufactured from woody feedstocks are nutrient-poor sources, but those from nutrient-rich sources (Table 2.2) such as animal wastes, municipal greenwaste, biosolid wastes and sewage sludge (Racek et al., 2020; Paz-Ferreiro et al., 2018; Wang et al., 2012a), or co-pyrolysis with phosphate (P) sources (Tumbure et al., 2020), can be very effective P fertilisers. The P availability of high-ash biochars ranges from 26 to 93% of total P content extractable in 2% formic acid and depends on feedstock type and pyrolysis temperature, as shown by Wang (2012a) working with biochars made from dairy manure-wood mixtures and biosolid-wood mixtures produced at a range of temperatures up to 550 ^{0}C. Irrespective of feedstock type all biochars are less significant as available N sources because during pyrolysis at higher temperatures C and N form aromatic and heterocyclic N-ring structures, which are slow to degrade or are recalcitrant in soils (Wang et al., 2012b). For example, the percentage of hydrolysable total N in biochars that have $H:C_{org}$ ratios < 0.7 is low, ranging from 1 – 10% (Camps-Arbestain et al., 2015).

Table 2.2 Carbon, nutrient, ash and liming equivalent content of biochar produced from range of feedstocks by pyrolysis at different maximum temperature (HHT). (Data source adapted from Camps-Arbestain et al., 2015).

Feedstock	HHT (°C)	C%	N%	P%	K%	S%	Mg%	Ca%	Ash %	**% Ca CO_3-eq.**
Pine wood	450	82.0	0.35	0.04	0.28	<d.l.	0.11	0.53	1.1	3.9
Pine wood	550	87.2	0.48	0.05	0.29	<d.l.	0.11	0.61	1.6	5.0
Eucalyptus wood	350	83.6	0.4	0.22	0.51	<d.l.	0.11	0.42	4.4	7.2
Willow branch	350	73.9	1.36	0.35	0.73	0.41	0.27	2.87	5.6	6.5
Willow branch	550	77.4	1.78	0.42	1.07	0.35	0.43	4.29	10.8	18.2
BioSolid+Eucalypt.	450	36.8	1.85	4.78	0.71	0.31	0.32	2.11	47.0	16.6
BioSolid+Eucalypt	550	35.7	1.66	5.06	0.70	0.21	0.31	2.30	51.1	15.1
Cattle manure	550	16.5	1.08	1.46	3.25	0.4	1.20	2.24	73.0	4.2
Poultry litter	550	40.8	3.77	3.32	3.35	0.48	0.90	6.36	43.0	11.8

Other major plant nutrients in biochars include S, K, Ca, and Mg, which are mostly constituents of the ash content and thus their content and availability also depend on the type of feedstock and the pyrolysis temperature (Table 2.2). Whilst S (10-76%) has similar levels of extractability to that of P, the cations, K (52-100%), Ca (35 – 100%) and Mg (18-95%) can be more soluble with higher concentrations and solubility closely related to ash content.

Biochars can have a liming value depending on feedstock and the amount of ash generated during pyrolysis, reflecting the basic cation content of the feedstock and the degree of oxidation that has occurred. Typical biochars manufactured from woody material (Table 2.2) have low $CaCO_3$ equivalence (4-7%), whereas willow branch, biosolids and poultry manure can have higher values (12-18%). Whilst liming value can be a positive attribute for raising the pH of acidic soils (pH < 6), over-liming may occur when high biochar application rates (> 20 t ha^{-1}) to soils with pH values ≥ 6 are being considered (rates of $CaCO_3$ equivalence > 4 t ha^{-1}) and pH induced trace element deficiencies could occur (Mengel & Kirkby, 2001).

Biochar can have added value as a fertiliser or liming material (Ye et al., 2020, Jeffery et al., 2011) provided the soil nutrient or pH condition is either limiting root and/or crop growth. Hence most positive effects of biochar addition on crop yields are reported for experiments conducted in the tropics on highly weathered soils (Jeffery et al., 2017). Dokoohaki et al. (2019) used Bayesian networks for modelling yield response to biochar addition, based on the results generated from a meta-analysis of international experimental data. They concluded that crop responses to biochar addition are more likely to occur on soils low in clay, SOC, soil pH, and CEC. While these authors show an elegant pathway through which one can establish where (and what) biochar could be applied, the applicability of their model is limited by the database analysed. For example the meta-analysis included both pot and field experiments, there was no limit in the biochar application rate, and the comparison did not distinguish between fertilised or unfertilised controls, or consideration of the soils basal macronutrient availability (particularly N and P) in relation to crop nutrient demand, which are fundamental measurements made in examples of good soil fertility management and agronomic practice. In a recent meta-analysis, which did consider the level of N supply, Ye et al. (2020), found that the response of crop yield to biochar addition was less a result of climatic zones or soil types than the combined use of N fertiliser along with biochar. Generally, high-nutrient biochars increase yield substantially more than low-nutrient biochars, indicating that feedstocks producing biochars with high nutrient and ash contents (Table 2.2) can be selected if the added value from agronomic effectiveness is required to support biochar manufacture.

To increase the probability of obtaining a crop yield response to biochar application, good standard agronomic practice is required, in which the target soil properties for maximising the economic margins of the crop response are known, in addition to the nutrient and liming value of a specific biochar to be applied. Independent nutrient sources (e.g. nitrogen fertiliser, legume or previous crop residue) are recommended if the biochar application does not meet the target values. In soils with poor drainage or low in clay, SOC and CEC the biochar might have additional interactive benefits as a soil conditioner.

Currently there is a total absence of published field trial research from New Zealand that has analysed the fertiliser value and/or the interactive effect of biochar application to soil with respect to crop or pasture yield growth responses.

2.1.3 Soil conditioner (forest, arable crop and horticultural residues)

Influence of applications on soil physical properties

Biochar exiting a pyrolyser has a low density and is highly porous; this is particularly the case when produced from plant material. Feedstock type, pyrolysis conditions and/or post-pyrolysis processing influences particle size, density, porosity and the biochar hydrophobic or hydrophilic properties. As a low density, porous particulate material, when added to soils it can alter a range of soil physical properties including soil density, pore size distribution, total porosity, soil moisture content (e.g., water holding capacity (WHC) or plant available water content (PAWC)), infiltration, and hydraulic conductivity. Whether biochar influences these soil properties depends on the relative differences between the physical properties of the biochar (including its particle size) and the receiving soil, as well as its application rate and depth of application. An extensive review of laboratory and field experiments by Blanco-Canqui (2017) provides very useful information on the effect of biochar application on soil physical properties (Table 2.3). There are useful discussions of the theory and analysis of biochar's predicted effects on the important properties of soil water retention by Yi et al. (2020) and drainage and water-use efficiency by Fischer et al. (2019) and Hardie et al. (2014). Measuring the moisture release characteristics of biochar allows an estimate of its range of pore sizes and the likely effect on the soil receiving environment.

Biochar pore size is known to vary over several orders of magnitude depending on feedstock and pyrolysis temperature (Thies and Rillig, 2009). Careful description of the biochar particle size, density and pore size and that of the receiving soil can avoid time consuming "trial and error" experimental work, and allow models (Yi et al., 2020) to estimate the effect that a quantity of biochar may have on soil physical properties such as soil porosity and soil water retention (Thies and Rilling, 2009; Herath et al., 2013). For example, if a biochar has a higher percentage of pore sizes in the range between > 0.2 µm and < 30 µm in diameter (Appendix A, Figure A.1), when added to a sandy soil, it will make a significant contribution to PAWC. However, the same biochar added to a clay-dominant soil might cause little change in PAWC because the soil will already have a high proportion of pores in this range. Likewise adding a biochar to a clay-dominant soil that creates larger aggregates with dominantly large, continuous, macropores > 30 µm in diameter (Appendix A, Figure A.1) might speed its drainage rate but would have little effect on the drainage rate in a sand dominant soil. However, after addition of biochar to soil not all soil-water relationship changes have been found to be predictable. Hardie et al. (2014) added 47 t ha^{-1} of a biochar (feedstock acacia, whole-tree, green waste) to a sandy loam, which reduced bulk density and increased porosity but expected increases in soil plant-available water content, due to a biochar causing a direct increase in 0.2-30 µm pore space, were not significant. Instead, increased earthworm activity stimulated by the biochar application, increased macroporosity. Particle size is also important. In a study carried out with the Tokomaru soil (Manawatu, New Zealand), the burial of large particle size biochar (> 4 mm) at a rate of 20 t ha^{-1} through soil inversion impaired water drainage to a larger extent than the burial of small particle size biochar (< 2 mm). This was attributed to the need of a larger water potential at the contact zone between the two soil layers (in the presence of the large particle size biochar) in order to drain through the subsurface layer (Mahmud et al., 2018).

Table 2.3 The effect of biochar application on different soil physical properties (Adapted from Blanco-Canqui, 2017)

Soil Property	Biochar effect soil physical properties ?	% Change	No.Studies
Bulk density	Decreases by	3–31%	>22
Total porosity	Increases by	2–41%	>14
Percentage of water stable aggregates	Generally increases by	21–226%	13
Soil water repellency	Small and mixed effects		5
Water infiltration	Reduces in sandy soils and increases in clayey soils		6
Saturated hydraulic conductivity	Decreases in coarse-textured soils by	7–2270%	
	and increases in fine-textured soils by	25–328%.	28
Unsaturated hydraulic conductivity	Reduces in sandy soils and increases in clayey soils		3
Water retention	Increases in most soils (90%) but not clayey soils		19
Plant-available water	Increases in most soils (72%) from	4–130%	29

The majority of experimental evidence shows (Table 2.3) that on incorporation of biochar into soil rooting zones, bulk density will decrease, and total porosity and percentage of water stable aggregates will increase (Blanco-Canqui, 2017). However, as mentioned above, changes to the water relations of the soil will be dependent upon the textural and structural characteristics of the soil, the application rate, the depth of application, and the particle size of biochar. In most cases, biochar application has created increased water retention and plant-available water in sandy soils but not clayey soils. Eyles et al. (2015) found that the growth and water use of seedling apple trees was similar in soils with poor drainage, with and without the amendment of 47 t ha^{-1} of biochar. The authors indicated that measurable impacts of biochar-based soil amendments might require a longer timescale for study, which allowed biochar-soil interactions to occur.

As biochar application to soils changes a number of physical properties, there are longer-term interactions with plant establishment (e.g. porosity, moisture, aeration influence germination and root growth) that influence the water-use efficiency of crops. A review of crop experiments involving modelling of daily and longer-term water balances (Fischer et al., 2019), indicated that in 75% of cases, biochar applications increased long-term evapotranspiration rates which had led to increased crop yields and, in 35% of those cases, had also increased water-use efficiency.

Influence of applications on other soil physical properties and dustiness

Some concerns have been raised that many C-footprint assessments do not consider that biochar application to soils will increase the radiative forcing on bare soils, exposed during biochar application and/or seed bed preparation. This is because high application rates of biochar to soil can darken the soil surface. Dark coloured soils (soils with more organic matter content or more biochar) absorb more solar radiation than light soils, but dark surfaces also radiate less heat than light coloured soils, changing the albedo of the soils surface. Albedo is the ratio between the reflected energy (outgoing) and the incoming (incident energy). When albedo decreases, soils warm and there is a risk that the increased temperature will prime soil organic matter decomposition. In historic times, Māori modified soils with charcoal. One notable area was about 400 hectares of the Waimea West plains near Nelson with gravels and charcoal to darken the soils (Mitchell and Mitchell, 2004). Without these additives, the growing season was too short for kumara. The effect of the gravel was to warm the soil during the day and the charcoal to retain that warmth during the night (Furey, 2006). Woolf et al. (2018) have a thorough discussion of the effect if biochar was applied to the cropland of the world. They reviewed research and concluded that, after cultivation and weathering of the biochar for 1 year, the soil albedo returns to normal. To minimise this effect and, of course to increase soil organic matter, soils should be left without plant cover for the minimum time possible. These concerns have little application to New Zealand pasture and forest soils, where thermal properties are controlled by the plant canopy.

The dustiness of biochar is another concern. Airborne dust from biochar storage, transport, application, and wind erosion could contribute to tropospheric aerosols and black carbon (BC) deposition on snow and ice (Woolf et al., 2018). The health risks associated with the dust need to be mitigated by normal industrial health and safety protocols for dusty working environments. The global warming risks can be mitigated by ensuring that biochar handling is enclosed and biochar is incorporated into soils (i.e. ploughed under) after application to reduce wind erosion.

Influence of applications on soil N transformations

In New Zealand, approximately 19% of New Zealand's reported agricultural emissions are nitrous oxide (N_2O) emissions from agricultural soils (MFE, 2020). The peak of N_2O emissions from N-enriched soils (urine patches or around fertiliser granules) arises when soil moisture contents remain in the range 70-80% of water-filled pore space, or close to field capacity (Owens et al., 2017). In this range the microbial oxidation of NH_4^+ (nitrification) is limited by oxygen supply and the microbial decomposition of soil organic matter stimulates denitrification of nitrate (NO_3^-), which generates N_2O and dinitrogen (N_2) gas, the ratio of gasses depending on oxygen supply. As soils approach 100% water-filled pore space (> FC), more complete denitrification occurs and the $N_2O/(N_2O + N_2)$ ratio declines until N_2 is the major gas product. The presence of easily-biodegradable organic matter (the so-called labile C) in moist soils can stimulate microbial respiration and drive an oxygen limitation, which stimulates denitrification in the presence of NO_3^-. As biochar application influences soil pore space distribution, water retention and soil organic matter content, it commonly influences NO_3^- concentrations in soils and N_2O emissions.

Biochar properties such as pore-size distribution, organic C content, charcoal characteristics, nutrient and ash content, and lime equivalence all interact strongly with soil chemical properties and microbial activity. These interactions with the nutrient concentration and physical condition of the soil receiving environment create uncertainty in the ability to predict whether a biochar application will reduce N_2O emissions and NO_3^- leaching from arable and grazed pasture soils. This point was also raised in a review of biochar effect on GHG gas emissions from soils by Kammann et al. (2017). In a more recent meta-analysis of the effects of biochar application on NO_3^- leaching and cumulative N_2O emissions, Borchard et al. (2019) concluded that the use of biochar reduced both N_2O emissions and NO_3^- leaching in arable farming and horticulture, but did not affect these losses in grasslands and perennial crops (e.g. grapevine and fruit trees) and forest soils. The overall average N_2O emissions reduction (with all land uses studied considered) was 38%, but N_2O emission reductions tended to be negligible one year after biochar application – the latter was attributed to a decrease in the electron shuttle function of biochar as it weathers (Yuan et al., 2019). In terms of effect of feedstocks, biochars produced from woody and lignocellulosic material tended to reduce N_2O emissions consistently but biochars made from manure and biosolids did not. Biochars produced from lignocellulosic biomass at temperatures of >500 ^{0}C reduced NO_3^- leaching. Whereas biochar applications increased NO_3^- concentrations in experiments on grassland soils, N_2O emissions and NO_3^- leaching remained un-affected (Borchard et al., 2019). Clearly such experiments require additional intensive monitoring of soil property changes and climate to provide evidence of cause and effect, thus making their findings more portable and relevant.

In New Zealand, the effects of biochar amendment of soils on N_2O emissions from urine patches have been studied but so far with no consistent results showing clearly that type of biochar, rate of application and soil drainage conditions in the experiment have an effect both on the presence of the mineral N substrates for denitrification and the microbial activity association with N transformations in soils (Anderson et al. 2014). Taghizadeh-Toosi et al. (2011) found significant emissions reductions using pine chip biochar (30 t ha^{-1}), whereas lower rates of application of a pine chip biochar (5 t ha^{-1}) to urine patches in another experiment (Treweek et al., 2016) had no effect. Mahmud et al. (2018) working with Tokomaru soils, found that the influence of pine biochar (20 t ha^{-1}) on N_2O emissions was dependent on the method of placement and the biochar particle-size, with an increase in the emissions when burying biochar with a particle size > 4 cm, and was associated with the impaired drainage at the contact zone between the soil and biochar layers.

It should be noted that many experiments on N_2O emissions have been carried out in the laboratory or in a glasshouse, using small amounts of soil (e.g., in pot experiments). The experiment of Mahmud et al. (2018) using lysimeters showed that biochar may cause a discontinuity in the water column with the contiguous soil layer, which has an influence on N_2O emissions, and this would not be picked up using pot experiments. Scientists continue to evaluate the effectiveness of biochar on reducing GHG emissions from soils in incubations (e.g.

Brassard et al., 2018) but care needs to be taken to simulate field soil water relations if predictions are to be made of the effectiveness of biochar amendment of soil on GHG emissions (Brassard et al., 2016). Kammann et al. (2017) reported that the lack of significance effect of biochar on N_2O emissions observed in some field studies could partly be attributed to the greater soil and crop heterogeneity and the less homogeneous biochar particle distribution compared with pot experiments.

Mitigation of contaminants in biochars and soils

Under the Resource Management Act (RMA, 2011), New Zealand has soil standards (MFE, 2011a) to protect human health from ingestion of the following contaminants: (i) the heavy metals and metalloids, arsenic (As), boron (B), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), and (ii) the organics benzo(a)pyrene (BaP), DDT, dieldrin, dioxins and dioxin-like PCBs and pentachlorophenol. These soil standards apply to residential, lifestyle block and commercial land. The protection of agricultural land from contamination associated with application of animal feedstuffs, fertilisers and soil conditioners falls under the Agricultural Compounds and Veterinary Medicines Act (ACVM, 2011). The ACVM states that, when used as recommended, "the feedstuff, fertiliser or soil conditioner will not result in residues in primary products that exceed the limits prescribed in applicable food residue standards set in, or under any enactment, or be toxic to animals treated with, or exposed to, the compound to an extent that causes unnecessary or unreasonable pain or distress".

The application of biochar to soils will need to make sure that the soil receiving environment sustainably complies with the "Standards for Contaminants in Soil to Protect Human Health" and the ACVM 2011 regulations.

As discussed in Section 2.1, the voluntary codes of good practice (IBI, 2015; ANZBIG, 2020) aim to limit the concentration of contaminants in feedstocks. The ANZBIG (2020) requires that when processed timber or municipal waste is used as feedstock, all plastic, rubber, some metals, electronic scrap and treated timber (copper-chromium-arsenate, CCA) or timber with reactive coatings (such as laminates, glues, paints, varnishes, fungicide) should be excluded from the pyrolyser.

Organic contaminants can also be generated during pyrolysis. The benzo(a)pyrene group of organics represent (MFE, 2011a) the carcinogenic polycyclic aromatic hydrocarbons (PAHs), which are persistent organic contaminants with ≥ 2 aromatic rings. PAHs are incomplete combustion products formed on the surface of biochar when the organic feedstock is heated during pyrolysis (Wang et al., 2017). Two fundamental mechanisms lead to PAH formation during slow pyrolysis: (i) formation of PAHs occur at temperatures $< 600\ ^{o}C$, with a peak at 500 ^{o}C, through condensation, carbonisation, and aromatisation of the solid material (Ravindra et al., 2008; Bucheli et al., 2015), and (ii) pyrosynthesis, where gaseous hydrocarbon radicals generated at temperatures $> 500\ ^{o}C$ undergo reactions to form polyaromatic ring structures, with dominance of PAHs in tar and gas over the solid phase (Ravindra et al., 2008). Wood gasification has the highest PAH concentrations due to recondensation of PAHs pyrosynthesised at high temperature in solid residues and, therefore, PAHs in biochar can be minimised during production by diverting and collecting pyrolysis gases separately. After an extensive review of the factors (feedstock type and pyrolysis temperature and heating rate), Wang et al. (2017) conclude that a wide range of wastes biomass feedstocks can be pyrolyzed at temperatures > 500 ^{0}C to produce biochars that (i) have PAH concentrations that are under soil standard concentrations, and (ii) present low bioavailability of PAHs. Hale et al. (2012) demonstrated that the ability to control pyrolysis conditions with increasing pyrolysis time and temperature can produce biochars with reduced PAH concentrations. Mayer et al. (2016) indicated that although some biochars may have high PAHs values, even beyond the limits established in biochar standards, these are very poorly bioavailable and the surfaces of biochars tend to adsorb and act more as a sink than as a source of PAHs, hence some biochars can be used to remediate levels of PAHs in soils (next section).

Polychlorinated dibenzo-p-dioxins and -furans (PCDD/Fs) can form when biomass containing high concentrations of chlorine is pyrolysed (Hilber et al.,2017). For example, pyrolysis of food wastes containing salt (2.9 % Cl, Hale et al., 2012) produced a toxic dioxin concentration of 1.2 pg/g TEQ. It may follow that forage plant biomass growing in saline conditions or irrigated with saline grey water with elevated chlorine concentrations (e.g. > 1% Cl, Grieve et al., 2004) may produce biochar with elevated PCDD/Fs concentrations. Feedstocks without elevated chlorine concentrations can be selected and, in these cases, the formation of PCDD/Fs during pyrolysis can be managed to produce biochars that are generally lower than and/or comply with the organic contaminant standards (Godlewska et al., 2020; Hilber et al., 2017; Hale et al., 2012) published in the IBI and ANZBIG biochar certification standards. These standards report the maximum allowable contaminant concentration in biochar products (see Section 2.33) that meet international maximum threshold values based on standards for soil amendments or fertilisers from a number of jurisdictions (see Section 2.33).

Remediation of contaminants in soils

Adsorption sites for organic molecules on charcoal surfaces contributes to biochar being able to adsorb organic molecules like PAHs strongly and the liming value of ash may precipitate some cationic contaminants (Cu, Cd, Pb or Zn) from soil solution, thereby reducing the bioavailability and pollution risk of the contaminants (Hilber et al., 2017). Post pyrolysis, activation (oxidation) of the biochar surface to generate surface negative charge can chemically change biochar into strong cation exchangers that can be used to remove cation contaminants from water (Shaheen et al., 2019). Shaheen et al. (2019) also reviewed the prospects for producing “engineered-wood biochar” to improve removal efficiency of contaminants in contaminated water, and Qi et al. (2017) review the role of similar “engineered biochars” in contaminant immobilization in soils. Both reviews indicate that selection of high-quality woody feedstocks, with surface activation engineering of the biochar can produce biochars customised to remove both cationic and anionic contaminants during water treatment and remediate contaminated land at minimum cost. Yet it should be noted that the energy (CO_2e cost) invested in activating biochars may negate the value of biochar as a CDR C sink (Alhashimi et al., 2017; Oleszczuk et al., 2012).

In New Zealand, Gregory et al. (2015) conducted an experiment to demonstrate that application of a woody biochar (willow feedstock) was able to remediate organochlorine pesticide concentrations in old sheep dip sites co-contaminated with arsenic and a range of organochlorine pesticides. This is a niche role for biochars that may be able to be manufactured local to the estimated 50,000 historic sheep-dip sites throughout New Zealand.

The hydrophobic properties of freshly made biochar and biochar's large internal surface area are the reason why it has a strong affinity for organo-pesticides (Graber and Kookana, 2015). Similarly, soil organic matter adsorbs herbicides and pesticides and affects their efficacy. It is well known that application rates of herbicides and pesticides are adjusted to the soil organic matter content following the manufacturer's recommendations. Likewise, the herbicides and pesticides efficacy can be affected when high rates (> 20 t BC ha^{-1}) of biochar are present in soil (Graber and Kookana, 2015). However, the weathering of biochar in soil will lead to the attenuation of this effect over time as the biochar surfaces becomes oxidized and thus less hydrophobic. Recommendation rates for pesticide application in cropping systems will need to be developed.

Soil, material and environmental scientists need more experience with manufacturing and evaluating the longer-term effectiveness of "engineered biochars" targeted at reducing the bioavailability of selected contaminants in soils before biochars are deployed alone, or as part of an integrated solution with phytoremediation to clean up contaminated sites. Similarly, more experimental evidence is needed on optimal herbicide and pesticide rates for biochar-rich soils.

2.1.4 Inclusion of biochar in the composting of biological wastes

Biochar production, as an alternative to composting of biowaste, can reduce GHG emissions significantly. Composting of municipal green-waste, food waste, sewage sludges and animal manures results in emissions of CH_4 and N_2O. The extent of emissions vary with the decomposability of the waste biomass, nutrient content, particularly N, moisture content, and whether it can be maintained in an aerobic state (IPCC, 2006a). However, the high moisture content of many of these wastes make them unsuited to biochar production. Woody wastes, with a high C content and low moisture content suited to biochar production are often collected at the same waste transfer stations as the compostable wastes. This opportunity has stimulated research into the effect of pyrolyzing the woody wastes and including biochar in the composting process (see reviews by Akdeniz, 2019; Kammann et al., 2017; Sanchez-Monedero et al., 2018). The IPCC (IPCC, 2006a) default emissions for the composting of biological wastes are 10 g CH_4 and 0.6 g N_2O kg^{-1} waste (DM). The reviewed research indicates that both emissions of CH_4 and N_2O can be reduced by 0 to >80% depending on the biomass being composted, the rate and characteristics of the biochar added (e.g. 1- 20%), and the management of the process. Similar to the role of biochar discussed for soil conditioning (Section 2.13), the porous and reactive surfaces of the biochar provide aerobic surfaces and improve the structure of the compost, stimulating aerobic microorganisms, such as methanotrophs and nitrifiers, increasing oxidation of C and N to CO_2 and NO_3 (or N becomes immobilised by the biochar), respectively, and lowering CH_4 and N_2O emissions. With appropriate selection of biochar and compost management, a number of researchers have achieved at least a 50% reduction in CH_4 and N_2O emissions with a 10% addition of biochar mixed into the compost. The sum of default IPCC emission factors for CH_4 and N_2O are equivalent to 0.43 t CO_2e t^{-1} waste treated (DM). At a 10% (1:9) application rate of biochar then a reduction in GHG emissions equivalent to 1.94 t CO_2e t^{-1} biochar can be achieved. Provided the resulting biochar plus compost is applied to land then the C-sink value of the biochar can still be realised as well (see example Sections 3.4 and 4).

2.1.5 Animal feed supplements (high-quality biochar)

It is recommended (EBC, 2012) that the feedstock materials for biochar products used as feed additives should be made from bio-based materials such as natural and untreated trunk wood found in tree, vine and shrub prunings, energy crops (e.g. Miscanthus), and untreated wood processing wastes (saw dust and wood chips). The raw material composition should not vary by more than 15% (EBC, 2012). Additives such as lime, bentonite or clay should not make up more than 10% of the feedstock by weight.

In other regions where charcoal has been a long-term animal remedy, biochar is being produced from a wide range of feedstocks, such as wheat straw, crop residues, corn cobs, stover and wood waste. A recent review by Man et al. (2020) has considered research on the effects of incorporating biochar into animal feed for ruminants, pigs, poultry and fish. In general, biochar when used as dietary supplement (< 1 – 8% of diet) for livestock, had positive effects on animal growth (weight gain, feed conversion ratio, and nutrient intake) of cattle, goats, pigs, poultry and fish. New Zealand's interest in biochar feeding is to reduce rumen methane emissions of grass-fed ruminants (cows, cattle and sheep). It has been shown that pine and corn stover biochars had no negative effects on *in vitro* rumen activity (Calvelo-Pereira et al., 2014) and biochar is capable of reducing methane emissions from ruminants fed a range of rations (e.g. cassava, Leng et al., 2012). *In vivo* studies found that the emission of methane decreased by 11% to 13% when 1% w/w biochar was added into young cattle diets fed 2.5 kg DM $head^{-1}$ day^{-1} as cassava root chips and leaves (Leng et al., 2012). The biochar dietary supplement also increased the efficiency of live weight gain by 25%. There are no reports of biochar feeding effects on ruminants with fresh pasture-based diets. This is an obvious knowledge gap for New Zealand, which could be addressed by following up the concept of adding biochar to dairy cow diets via in shed concentrates or grass and maize silage supplements (Calvelo-Pereira et al., 2014).

2.2 The influence of manufacture and distribution on biochar GHG emissions budgets

Pyrolysis is the main thermochemical conversion technology for large-scale production of biochar. During the pyrolysis process, biomass is thermally decomposed at temperatures (often above 300°C) under oxygen-depleted conditions into non-condensable gas, condensable volatile vapour and solid product (biochar). Industrial pyrolysis reactors for biochar production are generally classified into three categories – slow, intermediate, and fast pyrolysis – according to the heating rate and residence time of volatiles inside the reactors, as shown in Table 2.4.

Slow pyrolysis reactors for biochar production include kilns and retorts. Wood logs or lump wood pieces are placed in these reactors to produce biochar. Long heating times are needed as the large dimensions of the feedstock limit the heat and mass transfer during pyrolysis, thus requiring days for each production cycle (including heating, decomposition, and cooling). The long residence time provides ample opportunity for secondary reactions which produce more high molecular weight tarry volatile matter and more biochar. Of these, the biochar is the only useful product (Sánchez Careaga, et al., 2020).

Commercially-available, intermediate sized pyrolysis reactors are rotary kilns (or drums) and auger reactors. Feedstocks include wood chips, hogged forestry residues and chopped agricultural crop residues. The moderate size of feedstock particles shortens the heating time to within seconds for ground particles (< 2mm), or minutes for wood chips (< 20mm), to reach the carbonisation temperature. These reactors are usually used to produce biochar as well as bio-oil condensates (see Sections 4.1 and 4.3 for case studies). Very recent developments of auger-reactors, combined with catalytic conversion of bio-oil to biodiesel, have created the opportunity for turning waste biomass into high quality fuel for automotive combustion engines (Schmitt et. al., 2019).

Industrial-scale fast pyrolysis reactors, which are mainly spouted or fluidised bed reactors, are generally used for bio-oil or pyrolytic liquid production and are not a viable method for producing biochar. Although a small amount of char residue is produced from the fast pyrolysis process, it is a by-product requiring additional operations and expense to remove it from the spouted or fluidised bed material. Neither of two of the largest biomass gasification systems currently in operation in the world, which are Finish plants processing sorted municipal waste and forest residues produce any biochar. Bed ash as the only solid by-product (Bolhar-Nordenkampf and Isaksson, 2016).

Table 2.4 Classification of pyrolysis reactors for biochar production (Sánchez Careaga, et al., 2020 ; García-Núñez, et al., 2017)

	Slow pyrolysis	Intermediate pyrolysis	Fast pyrolysis
Heating rate	<100°C/min	~100 °C/min	>1000°C/s
Vapour residence time	Hours to days	several seconds to minutes	~1 sec
Particle size	Logs or chips	Chips or small particles	<2 mm
Products	Biochar or charcoal	Biochar and/or bio-oil	Bio-oil
Typical Yield*	Biochar: 35% Bio-oil: 30% Gas: 35%	Biochar: 20% Bio-oil: 50% Gas: 30%	Biochar: 12% Bio-oil: 75% Gas: 13%
Reactors	Kilns, retorts	Rotary drums, auger reactors, moving beds	Spouted or fluidised beds

*Winsley, P. 2007. Biochar and bioenergy production for climate change mitigation. New Zealand Science Review, 64(1), 5-10

2.2.1 Pyrolysis kiln type and GHG emissions (simple flame-curtain, portable retort, industrial scale retorts with oil and gas recovery).

According to the modes of energy supply for biochar production and heating rates of the feedstock, the pyrolysis reactors for large scale biochar production are classified into kilns, retorts, and converters (García-Núñez, et al., 2017) in this survey.

Kilns are generally batch operated and simple in design without sophisticated treatment of gas and volatiles. Biochar kilns come in many forms, including earth kilns (pit and mound kilns), Casamance kilns, brick kilns and metal kilns, as shown in Table 2.5. Wood logs are piled in the kilns and are heated by energy released from the oxidation reactions of a fraction of the wood and limited air (see Sections 4.1 and 4.3 for case studies). Often a part of the feedstock is combusted to provide the initial heat, sufficient to reach pyrolysis conditions in a localised part of the kiln. More sophisticated kilns recycle the volatiles and combust these as well. A recent design innovation is the flame-curtain kiln (e.g. Kontiki - Schmidt & Taylor, 2014). Flame curtain kilns operate by having a surface combustion zone, hence a flame above the piled biomass. Their advantage over the basic brick kiln is that they purport to combust the volatiles in the flame (Cornelissen et al., 2016). They come in different forms such as, troughs, deep cone, metal kilns and metal-shield soil pit kilns. They can be either stationary or portable. Constant supervision is required to maintain a surface, burning-bed of flames that will combust the volatiles. This burning of feedstock reduces the biochar yield (Karananidi et al., 2020).

The char yield of kilns ranges from 10% to 30%. The quality of biochar heavily depends on the construction of kilns which affects the temperature profile and uniformity, the mass and heat transfer, and the residence time. Well-constructed brick kilns usually achieve high carbonisation temperatures, thus produce good quality charcoal with a high H/C ratio, high internal surface area and good C-sink characteristics. Due to their simple construction, the investment costs in the kiln are low, and their life span is long (Seboka, 2009; García-Núñez, et al. 2017).

However, in these simple kilns, the majority of gas and volatile products from wood carbonisation is directly released to the environment (Table 2.5). This results in significant emissions to soil from the tars and to the atmosphere from the volatiles and gases. All the kilns are operated in batch mode with production cycles lasting for hours, weeks or months, depending upon the size of the woody material and kiln type. Treating the emissions from the kilns during production, is possible, but requires a significant jump in technology complexity. Reduction in the emissions footprint of the biochar (Figure 2.2) can be achieved using a more complex reactor (Section 2.22) that can flame off, or utilise, the exhaust CH_4 and non-methane volatile organic compounds (NMVOCs, Table 2.5) to generate heat .

Table 2.5. Characteristics of biochar kilns(FAO Forestry Dept., 1987; García-Núñez, et al., 2017; Lohri, et al., 2016; Schure, et al., 2019; Woolf, et al., 2010; Emrich, 1985; Cornelissen, et al., 2016)

	Pit kilns	Mound kilns	Casamance kilns	Metal kilns	Brick kilns	Flame-curtain kiln
Construction	Wood logs in pit with grass and earth cover	Wood logs stacked on the ground surface with leaves and earth cover	Mound kilns with a chimney to improve gas circulation and carbonisation	Mobile or stationary, metal cylindrical kiln sections with chimney	The kiln is built completely with bricks, concrete, with or without iron or steel support	Troughs, deep cone metal kilns, etc, with a top combustion zone
Carbonisation temperature, °C	~280		500~700	600	900	680-750
Capacity, m^3	1~30	5~20	< 250	1~6	4~350	1~5
Production cycle	2~3 months	~ 1 month	20~30 days	2~7 days	9 ~ 35 days	1~3 days
Char yield, %	12~30	9~30	17~30	12~30	13~32	22±5
Charcoal quality	Varying degree in carbonisation	Varying degree in carbonisation	Higher fraction of fixed C and lower volatile matter than from pit or mound kilns	Good quality charcoal, relatively high fraction of volatile content in char from oil drums	High quality charcoal	Carbon content: 76±9%
Volatile and gas treatment	Volatiles condense into soil and gas is released to air	Volatiles condense into soil and gas is released to air	Collecting volatile condensate at the base of chimney, consisting of water and acids, or releasing through chimney as vapour to air	Volatiles condenses in the chimney and are baked to a hard pitch; gas is released to air	Untreated (for Brazilian and Argentine kilns) or treated by an after-burn system before release to air	In-situ burning pyrolysis gas in the flame curtain for heating
Emissions, g/kg charcoal		CO_2: 1058-3027; CO 143-333; CH_4: 32-62; TNMHC:60-124; TSP: 13-41		For a Magnien kiln, CO_2: 814-1741; CO: 241-475; PIC: 206-347 For a drum, CO_2: 1517; CO: 336; CH_4: 58; TNMHC/TNMOC: 72; TSP: 4.2; Condensable: 66	For Brazilian kiln, CO_2: 543-1533; CO: 152-373; CH_4: 37-57; TNMHC/TNMOC: 24-81 For Missouri, CO_2: 350-870; CO: 48-254; CH_4: 30-57; TNMHC/TNMOC: 51-229, TPM: 62-208	CO_2: 4.3±1.7 kg/kg biochar;CO: 54±35 g/kg biochar;CH_4: 30±60 g/kg biochar;NMVOC=6±3 g/kg biochar;PM_{10}: 11± g/kg biochar; PIC: 100±83 g/kg biochar; NO_x: 0.4±0.3 g/kg biochar
Costs	Low investment costs (approx. $27/ton of charcoal), low skill requirement for operation	Low investment costs (approx. $27/ton of charcoal), flexible in size, low skills needed to operate	Operation requires considerable skill and experience; kilns need to be rebuilt for each cycle	Life span 2~3 years, the manufacture cost of a transportable metal kiln is greater than a brick kiln of similar output (US$ 2000-5000 for a TPI kiln of 300-400 kg charcoal per cycle)	Long life span (6~10 years), low in capital cost ($500-1500 more than pit kilns for Brazilian kilns, US$15,000 for a 350 m^3Missouri kiln), moderate in labour requirements	Kiln cost: € 30-5000
Example			Swedish earth kilns	Tropical Products Institute (TPI) kilns, kilns made from oil drums	Missouri kilns, Half-orange Argentine kilns, Brazilian beehive kilns	Kontiki

Stationary retorts are more complex than kilns and allow a greater level of control over the process. As listed in Table 2.6, typical stationary retorts used for biochar production include the wagon retort, Lambiott retort, Lurgi process, multi-hearth furnace and Carbo twin retort. These biochar production systems are of medium to large scale (Stassen, 2015). Unlike kilns (Table 2.5), gaseous products from carbonisation in retorts (Table 2.6) are recovered or burned as a fuel to provide heat for biomass pyrolysis. As a result, the char yield for these pyrolysis reactors is higher, mostly around 30-35%. Most of retorts are continuous or semi-continuous with carbonisation temperature up to 600°C. Due to the high maintenance costs of steel wagons and the shell of the retort, only a limited number of batch wagon retorts were used in the latter period of the 20th century in Europe (FAO, 1985).

Emissions from biochar production in retorts (Table 2.6) are much lower than from kilns (Table 2.5) since the gaseous products from wood pyrolysis are burnt in-situ as a heat source. For example, emissions from a recently developed Carbo twin retort were compliant with the stringent standard of Netherlands Emission Guidelines for Air (NeR) (Reumerman and Frederiks, 2002). Due to this additional processing, the investment costs of retorts are generally higher than those of kilns.

Converters, which include rotary drums and auger reactors (Table 2.7), are intermediate pyrolysis reactors often used for large scale operations (see Section 4.1 and 4.2). In addition to biochar, (approximately 20-30% of yield), bio-oil is the main by-product (50% yield). The gaseous and part of the volatile products from pyrolysis are burned to produce heat for maintaining the process through direct heating or indirect heating. Electricity is also needed to rotate the moving parts of the reactors. These reactors are either stationary or portable, and suitable for continuous production such as sewage sludge/biosolids pyrolysis. Rotary drums or kilns can be used to carbonise woody biomass, forestry residues, and agricultural residues with a wide range of particle size, while auger reactors operate best with small to medium particle sizes. Other intermediate pyrolysis reactors, such as moving agitated reactors and paddle reactors, usually operate under vacuum conditions; more research is needed before these can be widely used for biochar production (García-Núñez, et al., 2017).

Table 2.6. Characteristics of biochar retorts (Stassen, 2015; Klavina, et al., 2015; Rautiainen, et al., 2012; Szymkowski and Bultitude-Paull, 1992; FAO Forestry Dept., 1985; Lohri, et al., 2016)

	Wagon retort	Lambiotte or French SIFIC	Lurgi process	Herreshoff (multiple-hearth) furnace	Carbo twin retort
Operating mode	Batch	Continuous	Continuous	Continuous	Batch or semi-continuous
Heating approach	External heating from under retort by burning wood gas	Heating by recirculating hot flue gas from burning wood gas or alternative fuels in an outside burner	Heating by recirculating hot flue gas from in-situ burning wood gas or volatiles in an incinerator	Heating by recirculating hot flue gas from in-situ burning wood gas or volatiles	External heating through retort wall by burning volatiles
Feedstock	Roundwood, split roundwood, slabs from sawmilling, average length 1-1.2m, moisture content <25% (ref 7)	Roundwood and slabs, length 250-350mm, diameter 80-150mm, moisture content <30%	Wood blocks, e.g. 150×150×250 mm, moisture content < 30%	Small-particle-size material, e.g., sawdust, bark, straw, and husk	Lump wood pieces, extruded briquettes from sawdust or agricultural residues, moisture content <25%
Dimensions	Diameter: 2.5m Length: 8-16m	Diameter: 3m Height: 18m	Diameter: 3m Height: 27m	Diameter: 1.4-7.5m, 6-17 hearth	
Volume:	35-60 m^3	Typically, 130 m^3	200 m^3		2×4.5m^3 per each module
Input capacity	9-18 tons of dry wood per charge or 270 tonnes per month	Typically, 7000 tons, up to 20,000 tons of dry wood per year	6.2 t/h	4-10 tons of sawdust/bark per hour	900 tons of air-dried wood per year per module
Char yield, %	30-33	30-35	30-37	25-30	30-33
Other product	Pyroligneous acid: 20-25%	~1 MW electricity			
Carbonisation temperature, °C		547-560	Heating gas temperature 600	500-650	500
Char quality			Good quality, Fixed C content: 93%, volatile: 4.5%	Powdered charcoal	Good quality, C content 92%,
Emissions		CO: 1940±600 mg/m^3; Total VOC 650±20 mg/m^3; NO$_x$: 82±14 mg/m^3			CO$_2$: <1000 g/kg charcoal;CO: <0.144 g/kg charcoal; CH$_4$+NMVOC: <0.03 g/kg charcoal;particles < 0.015 g/kg charcoal; NO$_x$ <0.2 g/kg charcoal
Costs	High maintenance and operating costs	High capital costs, e.g. US$0.5-2 million	$10 million		Investment costs: €480,000 per module

Table 2.7. Converters (intermediate pyrolysis reactors) for biochar production (Campuzano, et al., 2019; Severy, et al., 2018; García-Núñez, et al., 2017; Lohri, et al., 2016; Meier, et al., 2013; Malkow, 2004)

	Rotary drum	Auger reactor
Feedstock	Wood, forestry and agricultural residue, wastes; Size 10-300mm	Small or medium size wood chips, agricultural crop or forestry residues dried sewage solids
Reactor size	Diameter: 0.3-1m Length: 4-12m	
Input capacity	Up to 300 t/day	Up to 50 t/day
Carbonisation temperature, °C	500-600	400-700
Char yield, %	25-30	20-30
Other products	Bio-oil	Bio-oil
Heating approach	Direct or indirect heating by recirculated flue gas	External heating by burning volatiles
Portability	Stationary or portable	Stationary or portable
Emissions		CO_2: 9-11 kg/kg biochar; CO: <3g/kg biochar; Unburned hydrocarbon: <2g/kg biochar TPM: ~6 g/kg biochar
Examples	3R Agrocarbon; MTK, Japan	Pyro-7 by Pro-Natura

In New Zealand, large-scale biochar production will need to meet the national ambient air quality standards, which are listed in Table 2.8 (MFE, 2011b). These standards were developed from the 2002 Ambient Air Quality Guidelines (MFE, 2014) and define the minimum requirements for outdoor air quality unless a resource consent is exercised. Recommendations in the 2002 Guidelines, including concentrations and averaging periods, still apply for those not covered by the ambient standards. Retorts and converters with low pyrolysis emissions and high air quality may allow processing of horticultural crop wastes and prunings and municipal green wastes during periods of fire bans (e.g. May – Aug for Napier or Hastings Airsheds and Canterbury Clean Air Zones) in the fruit growing regions of New Zealand with air quality problems associated with open fires.

In the European Union Directive issued in 2010, biomass pyrolysis was categorised as a “waste incineration plant” (EU, 2010). The emission limit values for the gaseous emissions from such plants are listed in Table 2.9. As the heat for biochar production is mostly sourced from the combustion of biomass or gaseous products of the pyrolysis process, regulations on biomass combustion plants can be used as a reference. For instance, the European Union enforces directives on the emissions from industrial-scale combustion plants including the emission limit values of several pollutants, as listed in Table 2.10, would define the compliance operating conditions for a kiln, retort or converter.

Table 2.8. Ambient air quality standards for gaseous pollutants (MFE, 2011b & 2014)

Pollutant	Averaging time	Ambient Standard from the National Environmental Standards for Air Quality 2004 (amended in 2011)		Ambient Air Quality Guidelines (2002)
		Value	Permissible Exceedances in a 12-month period	Value
Carbon monoxide (CO)	1-hour 8-hour	- 10 mg.m^{-3}	- 1	**30 mg.m^{-3}** 10 mg.m^{-3}
PM$_{10}$	24-hour Annual	50 mg.m^{-3} -	1	50 mg.m^{-3} **20 mg.m^{-3}**
Nitrogen dioxide (NO$_2$)	1-hour 24-hour	200 mg.m^{-3} -	9 -	200 mg.m^{-3} **100 mg.m^{-3}**
Ozone (O$_3$)	1-hour 8-hour	150 mg.m^{-3} -	0 -	150 mg.m^{-3} **100 mg.m^{-3}**
Sulphur dioxide (SO$_2$)	1-hour 1-hour 24-hour	350 mg.m^{-3} 570 mg.m^{-3} -	9 0 -	350 mg.m^{-3} 570 mg.m^{-3} **120 mg.m^{-3}**

Table 2.9. Emission limit values (at 11% O$_2$) for waste incineration plants in EU (EU, 2010)

		Daily average	Half-hourly average, no breaches allowed	Half-hourly average, 3% breaches allowed	Averaged over 0.5–8 hours for heavy metals, 6–8 hours for dioxin and furans
Total dust	mg/Nm^3	10	30	10	–
Total organic C in gas or volatiles (TOC)	mg/Nm^3	10	20	10	–
Hydrogen chloride (HCl)	mg/Nm^3	10	60	10	–
Hydrogen fluoride (HF)	mg/Nm^3	1	4	2	–
Sulphur dioxide (SO$_2$)	mg/Nm^3	20	200	50	–
NO$_x$ (as NO$_2$), > 6t.h^{-1} new plants	mg/Nm^3	200	400	200	–
NO$_x$ (as NO$_2$), < 6t.h^{-1}	mg/Nm^3	400	–	–	–
Carbon monoxide (CO)	mg/Nm^3	50	100 (150 as 10-min average)		–
Cd+Tl	mg/Nm^3	–	–	–	0.05
Hg	mg/Nm^3	–	–	–	0.05
Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V	mg/Nm^3	–	–	–	0.5
Dioxin and furans	ng/Nm^3	–	–	–	0.1

Table 2.10. Emission limit values (mg/Nm^3 at 6% O_2 for biomass and 3% O_2 for gas) for medium and large biomass combustion plants in EU (EU, 2015; EU, 2010)

	Fuel	Rated thermal input, MW_{th}	SO_2	NO_x	Dust	CO
Existing plants	Biomass	1-5	200	650	50	
		5-50	200	650	30	
		50-100	200	300	30	
		100-300	200	250	20	
		> 300	200	200	20	
	Gas	> 50	35	100	5	100 (for natural gas); none for other gases
New plants	Biomass	1-50	200	300	20	
	Biomass	50-100	200	250	20	
	Biomass	100-300	200	200	20	
	Biomass	> 300	150	150	10	
	Gas	> 50	35	100	5	100

2.2.2 Scale of feedstock processing, pyrolysis and distribution

The biomass resources in New Zealand mainly consist of forestry residues, wood processing residue, agricultural residues (e.g., orchard residues, stover, and straws), and municipal waste (e.g. wood waste, food waste, bio-solids or sludge). Scion (Hall, 2017) estimated the annual gross volumes of key residual biomass resources potentially available in New Zealand every 5 years from 2017 to 2042. As shown in Figure 2.3, the estimated annual gross supply of these biomass resources varies between 4.6 and 7.1 Mt, which is equivalent to 30~47 PJ of energy per annum. However, not all these resources can be exploited or recovered due to the economic and/or physical limitations (Hall, 2017; Kumar et al., 2021). For high recoverability, the annual total amount of the key residual biomass resources ranges from 2.6 to 4 Mt, or 18 to 28 PJ of energy (Hall, 2017). At low recoverability, the volume of the biomass resources reduces to 2~3 Mt y^{-1} (Hall, 2017).

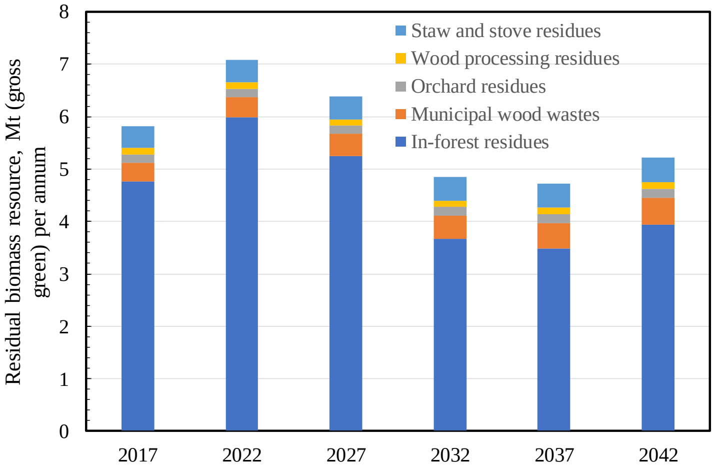


Figure 2.3. Prospective gross volumes of key residual biomass resources in New Zealand (Hall, 2017)

In addition to the above-mentioned lignocellulosic (woody) feedstock, food waste can also be used to make biochar. According to WasteMINZ (2018), household food waste that reaches landfill through domestic kerbside collections was about 229,000 t y^{-1}. The “avoidable” food waste is estimated to be about 49% of the average household’s food waste WasteMINZ (2018). Physical waste audits conducted by Otago University at NZ supermarkets in 2017 revealed that food waste from supermarkets in New Zealand was about 60,500 t y^{-1} (Goodman-Smith, 2018), of which 23% was sent to landfill. Another significant food waste stream is the biological by-products from the agricultural and horticultural sector. The Bioresource Processing Alliance (BPA) estimated that the waste from the agricultural and horticultural sector, such as skins, seeds, spent grains from breweries, reached 350,000 t y^{-1}, which is sent to landfill, spread on land or used as fertiliser or animal feed. Directing wastes to biochar production is feasible, based on the large waste incineration industry already in existence. In many large international cities incineration of municipal waste is a preferred option. About 8% of municipal waste in Canada is incinerated, one of the lowest proportions among developed countries. In comparison, percentages for other developed countries include: 59& in Switzerland, 54%in Belgium, 48% in Denmark, 47% in Sweden, 42% in France, and 16% in the United States. https://www.wasterecyclingmag.ca/feature/incineration-in-canada/.

Directing some of these food wastes and agricultural residues to biochar production could be a cost neutral alternative to existing waste processing pathways. A recent study in New Zealand compared the repurposing options for grape marc, comparing composting to thermal processing of which one option was making biochar (Jones et al., 2020). The cost of biochar production was similar to composting, with the added benefit of significant volume reduction of the residue and conversion of labile carbon to recalcitrant carbon, which makes it useful as a carbon storage mechanism. (See also Sections 3.4 and 4.3).

2.3 Biochar characteristics, certification and use.

To ensure that biochars are fit for purpose either for CDR as a C sink and can claim added value as fertilisers, liming material, a contaminant remediation agent or as an animal feedstuff, the large amount of research into these various roles has been summarised by the voluntary biochar promotion initiatives to create biochar certification standards. There are four separate voluntary associations with their own biochar standard guidelines: the International Biochar Initiative, Biochar Standards (IBI-BS, International Biochar Initiative, 2015), European Biochar Certificate (EBC) (European Biochar Foundation, 2012; updated 2020), Biochar Quality Mandate (BQM) (British Biochar Foundation, 2014), and the ANZ Biochar Industry Group (ANZBIG). In this review, we provide in Appendix B, Table B.1, the comparison between IBI-BS and EBC carried out by Camps-Arbestain et al. (2015). The ANZBIG is to a large extent based on these two documents.

For application to soil as a C sink, fertiliser, liming agent or soil conditioner both the EBC and IBI standards require the declaration of pH, bulk density, water content and electrical conductivity of the biochar as well as making sure that the biochar contaminant analyses falls within the maximum limits for the following contaminants (i) the heavy metals and metalloids, arsenic (As), boron (B), cadmium (Cd), chromium(Cr), copper (Cu), lead (Pb), mercury (Hg), and (ii) the organics benzo(a)pyrene (BaP), DDT, dieldrin, dioxins and dioxin-like PCBs and pentachlorophenol. (Appendix B, Table B.1).

2.3.1 Carbon sink (stockpile, application to forest, arable and pasture systems)

Biochar fit to be used as a CDR C sink when applied to soil must remain undegradable for as long as possible. Modelling the degradation of biochar in soils is the main method of stability assessment (Leng et al.,2019).

The IBI Stable C Protocol (Camps-Arbestain et al., 2015) was developed by Budai et al. (2013), and is based on the analysis of the relationship between H/C_{org} ratios of biochars and their measured degradation rates in medium-term (3 to 5 years) incubation experiments in soil carried out under controlled and optimal environmental conditions. A regression equation (Eq 1) was developed to estimate the percentage of organic C (C_{org}) in biochar that remains stable in soil for more than 100 years (BC_{+100}) (Figure 2.4).

$$(BC_{+100})\% = -74.3 \times H/C_{org} + 110.2 \qquad (r^2\ 0.50) \qquad \text{Eq 1}$$

The choice of a measuring unit of $(BC_{+100})\%$ of C_{org} in biochar which remains stable in soil for 100 years is based on the fact that global warming potentials of GHGs are assessed over a 100-year time horizon (IPCC, 2007). Therefore, if biochar manufacture and sequestration in soils was accepted as a method of CDR it would be compatible with national GHG inventory calculations. More recent relationships for estimating the BC have been provided by (Lehmann et al., 2015) but they appear to give no improved prediction of $(BC_{+100})\%$.

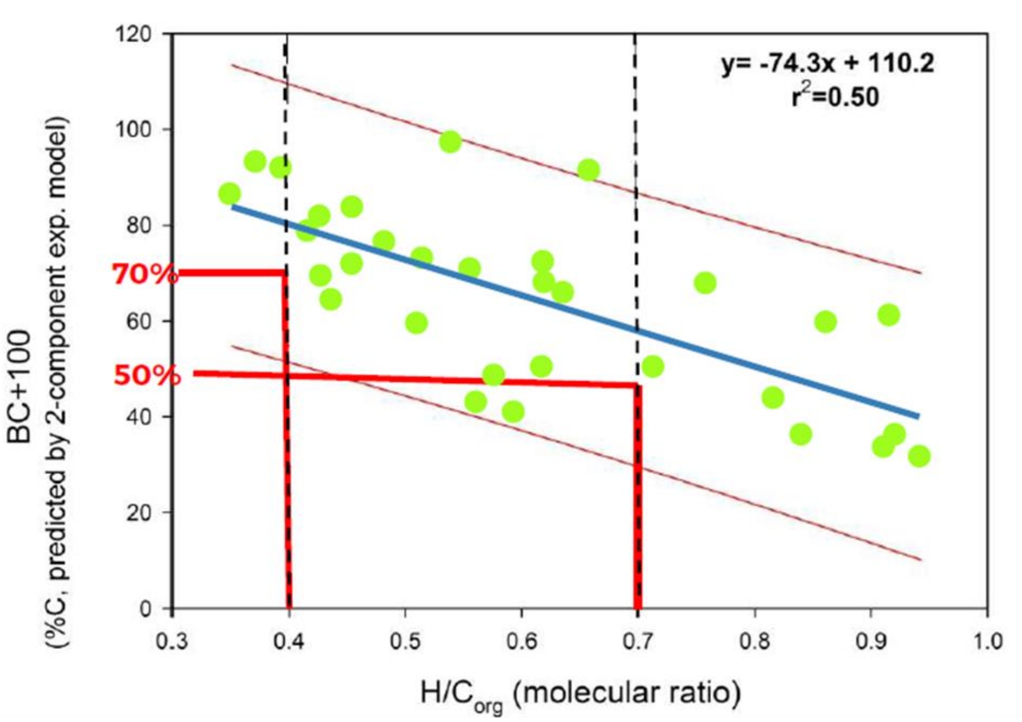


Figure 2.4 The correlation between H/ C_{org} and % biochar C predicted to remain after 100 years (redrawn from Budai et al., 2013)

Currently the IBI Stable C Protocol uses Eq 1 (Budai et al., 2013) Thus, a biochar sample with an H/C_{org} value of 0.7 would be predicted to have a BC_{+100} of 58.2 % the measured C_{org} in that biochar remaining after a century. Budai et al. (2013) calculated a range of BC_{+100} percentages from H/C_{org} values and developed conservative cut-off values (Table 2.11, adapted from Camps-Arbestain et al. (2015)). Budai et al. (2013) recommended that, for biochars with an atomic H/C_{org} ratio between 0.4 and 0.7, the BC_{+100} value to be used should be 50%, and that for biochars with an atomic H/C_{org} ≤ 0.4, this should be of 70%. The atomic H/C_{org} and therefore BC_{+100} is strongly associated with the aromaticity of the biochar.

Table 2. 11 Molar H/C_{org} ratios, percentage of aromaticity based on Wang et al (2013) and predicted BC_{+100} % values using Eq1 and recommended cut off values for BC_{+100} class based on 95% confidence limit (Camps-Arbestain et al., 2015)

H/C_{org} (mol mol^{-1})	Aromaticity (%)	BC_{+100} %			
		Mean	Lower Limit	Upper Limit	Cut-off values
0.4	92	80.5	72.6	88.2	70
0.5	87	73.1	67.1	78.9	50
0.6	81	65.6	60.5	70.6	50
0.7	74	58.2	52.5	63.8	50

The European Biochar Certificate (EBC,2012), requires biochar used for the purpose of CDR to meet both H/C_{org} and O/C_{org} standards. The upper O/C_{org} limit of 0.4 is required by EBC for a standard-compliant biochar in addition to the H/C_{org} ratio of <0.7. This is similar to the recommendation of Schimmelpfennig and Glaser (2012) that the upper O/C_{org} limit be 0.4 and H/C_{org} be 0.6.

To account for biochar C contribution to soil C stocks, the IPCC (2019a) accept predictions of BC_{+100} from the biochar property measurements, H/C_{org} and O/C_{org}, using the IBI and EBC protocols. The IPCC (2019b) also have developed a method (see Section 5) for predicting the fraction of biochar C remaining in soil (unmineralised) after 100 years ((BC_{+100}) %). The method is applied to biochar produced at temperatures > 350^0C and uses pyrolysis temperature (Table 2.12), which is more easily available than biochar property measurements, H/C_{org} and O/C_{org}, to predict (BC_{+100} %).

Table 2. 12 The prediction of the percentage of Biochar C remaining after 100 years BC_{+100}% (adapted from IPCC, 2019b)

Values for the % Biochar C remaining after 100 years	
Production (Pyrolysis temp.)	BC_{+100}%
High temperature pyrolysis and gasification (> 600 °C)	89 ± 13%
Medium temperature pyrolysis (450-600 °C)	80 ± 11%
Low (350-450 °C)	65 ± 15%

Whilst it is possible to utilise data from carefully conducted laboratory and field incubations of biochar in soil to develop the above relationships enabling the prediction of BC_{+100}, Camps-Arbestain et al. (2015) added the caveat that the effect (e.g., positive or negative priming) of the biochar on native soil C (SOC) stocks remains unpredictable for many soils and are not considered in these protocols. In the experiments reviewed, the direction and magnitude of SOC change in incubations and field trials varied with soil and biochar type. For example, Palviainen et al. (2018) reported that 2 years after a biochar application of 5 and 10 tBC ha^{-1} there was no reduction in native soil C stocks, and then 3 years after they reported increased pine tree growth had resulted from the 10 tBC ha^{-1} application (Palvianen et al., 2020). In contrast, Zhang et al. (2019), found that biochar amendment of 20 -40 t BC ha^{-1} to an orchard soil significantly increased soil respiration in the first and third years after biochar application. Clearly the detail of the biochar type, background soil properties and seasonal climate conditions need to be monitored and understood to develop an understanding of the potential for the biochar/production system interactions to produce both biochar and SOC gains rather than losses. With the availability of longer field trial evaluations and more scientific data, methodologies for predicting both BC_{+100} and soil C stock changes can be reviewed and updated to include the influence of specific biochar-soil type interactions, as well as the effects of biochar amendments on the decomposition or stabilization of native organic C (‘priming effects’) and plant productivity.

2.3.2 Classification of fertiliser and liming values of biochar

Producers and end users of biochar have been encouraged to use standard methods to measure the total N, P, K, S, Mg content of a biochar (Appendix B, Table B.1) and easily-extractable amounts of these elements using standard methods of fertiliser analysis (Camps-Arbestain et al., 2017). The results of these analyses can be used to place biochars in their fertiliser class (Camps-Arbestain et al. 2015) but, more importantly, can be used to calculate the fertiliser replacement value, costs and GHG offsets (see Section 4, Figure 4.1) when biochars are applied in sufficient quantities to substitute for other nutrient inputs. In addition, the nutrient analysis might indicate that biochar application rates should be limited because heavy biochar applications for CDR result in nutrient doses becoming excessive and environmentally damaging.

Similarly, using standard methods of measuring the $CaCO_3$-equivalence of biochars (Tables 2.2 Appendix B, Table B.1) has been recommended to establish the biochar liming value (Camps-Arbestain et al., 2015). This is both to determine the extent of liming value when biochar is applied to remedy acid soils and to avoid over-liming caused by heavy applications of biochar. Methodologies for these and other biochar properties are detailed in the book of Singh et al., (2017) which is a guide to analytical methods for the characterisation of biochars.

2.3.3 Classification of biochar for acting as a soil conditioner

There are currently no standards published for biochar to be used specifically as a soil conditioner to improve soil drainage conditions or to remediate the concentrations of contaminants in soils such as heavy metals and organic residues from pesticides. Instead the standards applied to land application of biochars are designed to ensure that the application of biochar does not add to the level of soil contamination. As mentioned earlier, the guiding legislation in NZ is the Resource Management Act (1991) and the Agricultural Compounds and Veterinary Medicines Act (ACVM, 2011). Codes of practice for land application of biochar can be developed from the New Zealand soil standards (MFE, 2011a) and the Guidelines for the Safe Application of Biosolids to Land in New Zealand NZWWA (2003) in a similar manner to the guidelines proposed by ANZBIG (2020) that draw on the Australian Compost Standard AS4454-2012 and both the EBC and IBI (2015) standards

For application as a soil conditioner, both the EBC and IBI Standards require the declaration of pH, bulk density, water content and electrical conductivity of the biochar as well as making sure that the biochar contaminant analyses falls within the maximum limits for recognised contaminants. Camps-Arbestain et al. (2015) further provides a classification system for biochars based on their particle sizes using ternary diagrams.

2.34 Animal feed supplement

The EBC has developed product criteria for feed-grade biochar, which are more restrictive than those for biochar to be applied to land (EBC, 2012, Chapter 9). For example, the C content of biochar for use as animal feed must contain at least 80% C (dry matter). Biochar with EBC-feed certification meets all requirements of the EU parliament feed regulations and may be used as feed or feed additive in animal husbandry authorised as vegetal carbon, a feed additive. In New Zealand Biochar used as a feed additive must comply with the Agricultural Compounds and Veterinary Medicines Act (ACVM, 2011).

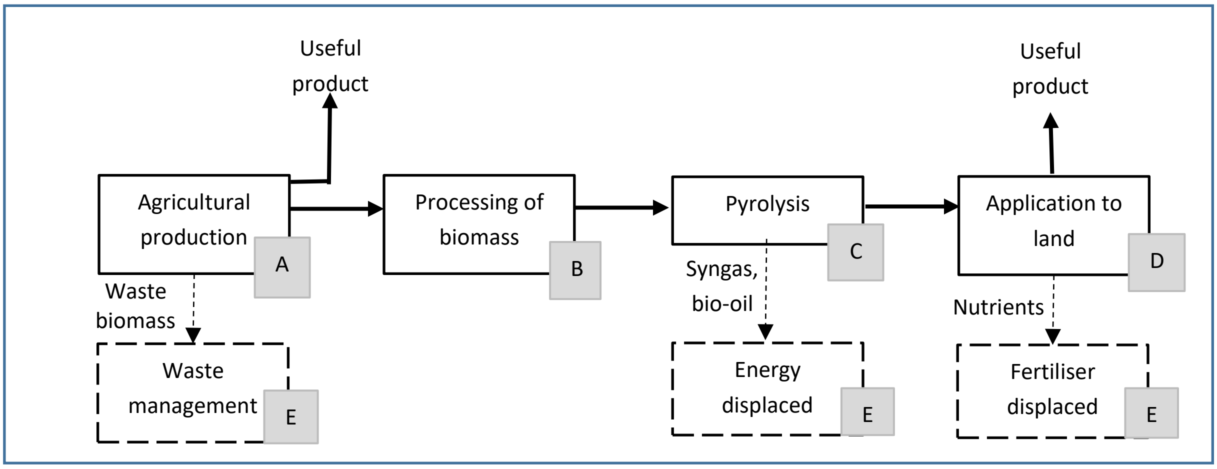
3. Life cycle assessments of biochar case studies appropriate to New Zealand.

3.1 Overview of Life Cycle Based Modelling for Biochar Systems

There have been a number of Life Cycle Assessment (LCA) studies of biochar. Matustik et al. (2020) identified 27 LCA studies on biochar produced from pyrolysis that is subsequently applied as a soil amendment. The studies included use of feedstocks such as human sewage sludge, municipal green waste, animal manure, cereal/oilseed straw, other agricultural residues (e.g. rice husk, cocoa shells), wood, as well as dedicated crops (e.g. miscanthus, switchgrass).

The generalised system that is studied in an LCA of biochar produced from waste agricultural biomass (as opposed to a dedicated crop or sewage sludge or municipal green waste) is shown in Figure 3.1. In addition to the choice of input biomass, other key modelling aspects that influence the LCA results are indicated on this diagram by the letters A to E in grey boxes.

Figure 3.1 Generalised system modelled in LCA of biochar systems using biomass from agricultural systems Note: dotted lines indicate avoided activities.



For ***by-products from agricultural production (A),*** some studies allocate the environmental impacts of the agricultural production between the main product and the co-products. For example, Robb and Dargusch (2018) divide the emissions from oil palm cultivation by mass across the different co-products). However, if the byproduct is regarded as a waste, then at its source it is usually it is treated as a “free” input into the biochar system.

The ***degree of processing of the agricultural biomass (B)*** is an important variable. For example, Jones et al. (2020) showed that the net climate change impact of pyrolysis of one tonne fresh grape marc varied from -12 to -213 kg CO_2e depending upon the initial moisture content of the grape marc (modelled at 75, 67 and 59%, Figure 4.13, Jones et al., 2020). Transportation of either the waste biomass and/or the biochar for subsequent utilisation is generally included and can make a significant contribution to the climate change result. For example, Llorach-Massana et al., (2017) found that the efficiency benefits of larger pyrolysis units could be cancelled out by the greater transportation distances compared with smaller decentralised units.

The pyrolysis conditions affect ***the amount of, and C-sink value of biochar produced (C).*** Matustik et al. (2020) noted that most studies assume that the operating temperature is somewhere between 450 to 600°C, and that the yield of biochar is between 20 and 35% (of the feedstock dry matter mass).

When it is applied to the land ***(D), biochar may be more or less persistent*** depending upon a number of attributes (discussed in Section 2.3). The percentage of stable C in the biochar (% biochar C remaining after 100 years, (BC_{+100})) is calculated using IBI protocol (Table 2.11), despite the fact that Wang et al. (2016) concluded in their meta-analysis of biochar stability in soil that, “Ninety-seven percent of the added biochar can persist in soils on a centennial scale.” In future it is expected that the IPCC (2019a) protocol (see Section 2.31) will be used for predicting BC_{+100} % from the biochar property measurements, H/C_{org} and O/C_{org} (IBI and EBC Protocols) or pyrolysis temperature (IPCC, 2019b; Table 2.7).

Other aspects that are sometimes modelled in LCAs of biochar application include: increased availability of nutrients in soil (see also E below), decreased toxicity due to adsorption of contaminants, reduced methane (particularly in paddy fields) and N_2O emissions from soil, reduced NO_3- leaching, increased soil biota, and increased crop yields due to improved soil quality. Other environmental impacts that have been identified by Woolf et al. (2010, 2018) but which do not appear to have been included in LCA studies to date include effects on mineralisation of the soil, change in the soil’s albedo effect, and indirect land use change (due to, for example, increased crop yields when biochar is applied). It should also be noted that the quantity of biochar applied to land varies widely across different studies, generally ranging from 1 to 30 t ha^{-1} (Matustik et al., 2020).

System expansion (E) is used in a number of studies to account for the activities that are avoided when biochar production takes place. This is termed a ***“Consequential”*** assessment. These are shown as dotted boxes and lines in Figure 3.1. The main avoided activities are:

- Displaced waste management of the waste stream from agricultural production when it is used in a biochar system.
- Displaced energy generation when the syngas and/or bio-oil are used to generate energy (usually heat and/or electricity)
- Displaced fertiliser production due to nutrients applied in biochar onto agricultural land, and due to increased availability of nutrients in the soil (see Sections 2.22 and 2.32).

When reviewing and comparing LCA studies of pyrolysis with biochar production, it is important to be aware of the modelling differences between the various studies as they can potentially affect the results.

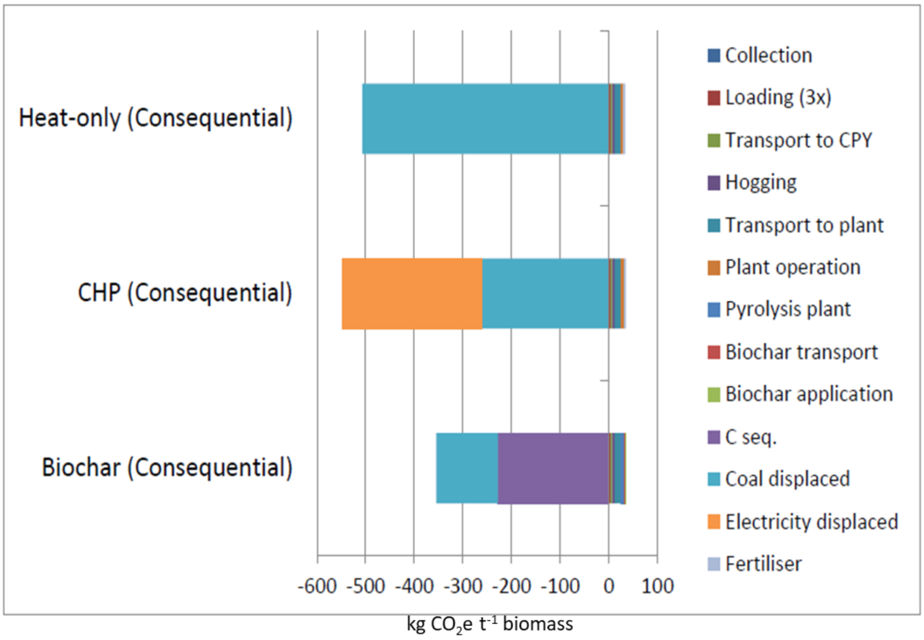
3.2 Review of studies on use of forest residue for biochar.

The use of logging residues in central North Island was investigated by Anaya de la Rosa (2013). For this study, the displaced baseline was leaving the residues on site where they decay and are recycled back into the soil. Climate change results were calculated for combustion, combined heat-and-power, and for pyrolysis. The modelling assumptions included:

- One tonne of logging residues results in biochar (comprising 74% stable C) with a C sequestration value of -228.3 kg CO_2e; 6% of the biochar is lost during transportation and application.
- Moisture content of logging residues is 53%
- Heat and electricity generated displace coal-generated energy production.

The results in Figure 3.2 show that all the scenarios are climate-positive (473, 511 and 321 kg CO_2e t^{-1} processed logging residues, collected at the landing site, respectively). The results are dominated by the displaced coal combustion and electricity generation, and the C sequestered in biochar (for the biochar scenario). In a sensitivity analysis where factors such as moisture content, stability of the C in the biochar, biochar migration, and bio-oil boiler efficiency, were varied to represent pessimistic and optimistic assumptions for the pyrolysis scenario, it was found that all results remained climate-positive (ranging from 26 to 938 kg CO_2e t^{-1} processed logging residues) (Table 31, Anaya de la Rosa, 2013).

Figure 3.2. Climate change results for combustion, combined-heat-and-power (CHP) and biochar production from 1t logging residues (Anaya de la Rosa, 2013)



Azzi et al. (2019) considered use of forestry residues in Sweden for biochar production, including changing from the current combined-heat-and-power (CHP) plants to pyrolysis plants producing biochar plus a reduced amount of heat and power relative to the CHP stations. They further modelled subsequent use of the biochar both in cattle feed and mixing it with manure and incorporating it into agricultural soil. In their analysis they accounted for displacement of coal-and/or gas-fired heat and power, decreased decay of soil organic matter due to addition of biochar to the soil, and reduced use of mineral N fertilisers and lime associated with application of biochar to the soil. The biochar yield from pyrolysis was either 21% or 36% (as % of total dry mass of forestry residues) and consisted of 80% stable C; it was applied to dairy farms at a rate of 0.8 t ha^{-1} y^{-1}. When assuming substitution of gas-fired heat and power, they found that the pyrolysis scenarios always outperformed the conventional power plant fuelled by wood chips but that the conventional CHP plant's climate change result could be better or worse depending upon the modelling assumptions for the pyrolysis scenarios (e.g. biochar yield). Similar to Anaya de la Rosa (2013), the biochar C sequestration and heat/power substitution were the biggest contributors to the climate change result; however, production and transport of wood chips, and agricultural soil effects did make more significant contributions to the pyrolysis scenario results (each contributing about 15% of the total GHG emissions) than were assessed in Anaya de la Rosa (2013). Furthermore, when the displaced sources of heat and/or electricity were varied (i.e. to represent the Stockholm-specific scenarios in Azzi et al., 2019), both the conventional power plant fuelled by wood chips and the conventional CHP plant had more climate-positive results than the pyrolysis scenarios (with one exception where the most optimistic assumptions were made about the pyrolysis scenario) in 2020, and less climate-positive results than the pyrolysis scenarios in 2040, when more renewable options for power generation were predicted.

Puettmann et al. (2020) analysed pyrolysis of wood chips in three types of portable pyrolysis units (BSI Biochar Machine, Air Curtain Burner, and Oregon Kiln), displacing burning of forestry residues *in situ* in forestry operations. They assumed a yield of 11 to 21% biochar from dried forestry residues, and 58 to 89% fixed C in the biochar (Table 4, Puettmann et al., 2020). Across the three pyrolysis units, and assuming various transportation distances to the pyrolysis plant and different operating conditions, the net climate change result varied from –1700 to -2800 kg CO_2e t^{-1}marketable biochar. In particular, they found that use of medium-chipped wood as opposed to ground-up residues increased the fixed C storage by approximately 25% in the BSI Biochar Machine, and that transporting the wood chips for four hours before pyrolyzing increased the GHG emissions by up to 195 kg CO_2e t^{-1}marketable biochar (Table 6, Puettmann et al., 2020).

3.2.1 Summary

The functional unit for the three studies discussed in this section was "1 t fresh forestry residues" (Anaya de la Rosa, 2013, assuming 53% moisture content), "1 t (dry weight) forestry residue/woodchips" (Azzi et al., 2019; Puetmann et al., 2020), "1 t marketable biochar" (Puetmman et al., 2020), and 1 tonne fixed C in biochar (Puettmann et al., 2020).

For Anaya de la Rosa (2013) and Azzi et al. (2019), the displaced activity, when the forestry residues were collected and pyrolyzed, was leaving the residues in the forested area. Anaya de la Rosa (2013, p.157) assumed this caused no difference in the C sequestered in the forest because decomposition of residues occurs as part of the natural C cycle. Azzi et al. (2019), on the other hand, assumed a loss of 89.1 kg CO_2e t^{-1} woodchips removed due to long-term change in forest C (and this contributed 4 to 7% of the total GHG emissions across all the pyrolysis scenarios). Puettmann et al. (2020) did not account for the alternative fate of the forestry residues in their baseline study but undertook a sensitivity analysis where the residues were burned in situ.

In summary, then, these studies indicate that:

1. Pyrolysis of forestry residues and subsequent application of biochar to agricultural soil is always climate-positive.
2. The magnitude of the climate change result is largely dependent upon the biochar yield and displacement of fossil-fuel sourced heat and power. Regarding the latter, these values vary widely depending upon assumptions about displacement of heat versus power, fuel source, and efficiency of the displaced CHP, heat and/or power plants.
3. Additional contributions to the climate change result for pyrolysis, i.e. production and transport of wood chips, and agricultural effects, have been modelled to contribute from 5% to 27% of the total GHG emissions depending upon the local conditions (e.g. distances between forests and pyrolysis plants), and other modelling assumptions. However, if the displaced heat and power is from renewable sources, then the contribution of production and transport of wood chips, and agricultural effects, increase to approximately 40% of the total GHG emissions for the pyrolysis scenarios (2040 Stockholm pyrolysis scenarios in Azzi et al., 2019).
4. Regarding the relative performance of CHP, combustion and pyrolysis, alternative CHP and power plants burning wood chips may be more or less climate-positive than pyrolysis and subsequent biochar use in agriculture, depending upon the displaced energy sources (i.e. fossil fuels versus renewables). If the displaced energy source is renewables, then it is likely that pyrolysis will provide a more climate-positive result than CHP or power plants burning wood chips.

3.3 Review of LCA studies on use of municipal green wastes for biochar.

Although no NZ studies of using domestic garden waste have been identified, there are a limited number of international studies on this feedstock:

- Roberts et al. (2010) found that pyrolysis of yard waste had a net climate-positive result of -885 kg CO_2e t^{-1} dry yard waste (with 63% of this result due to C sequestration in the biochar).
- Miller-Robbie et al. (2015) considered the pyrolysis of yard trimmings at a wastewater treatment facility (with heat use in the wastewater treatment facility), and then land application of the biochar after mixing with digested sewage sludge (bio-solids). The baseline scenario was composting and/or landfilling of the yard trimmings. In the two scenarios representing use of varying amounts of yard trimmings, and landfilling or incineration of excess biosolids, one scenario was climate-negative and one scenario was

climate-positive; the main reason for this difference was the amount of biochar used in each scenario (the functional unit being one tonne dry biosolids).

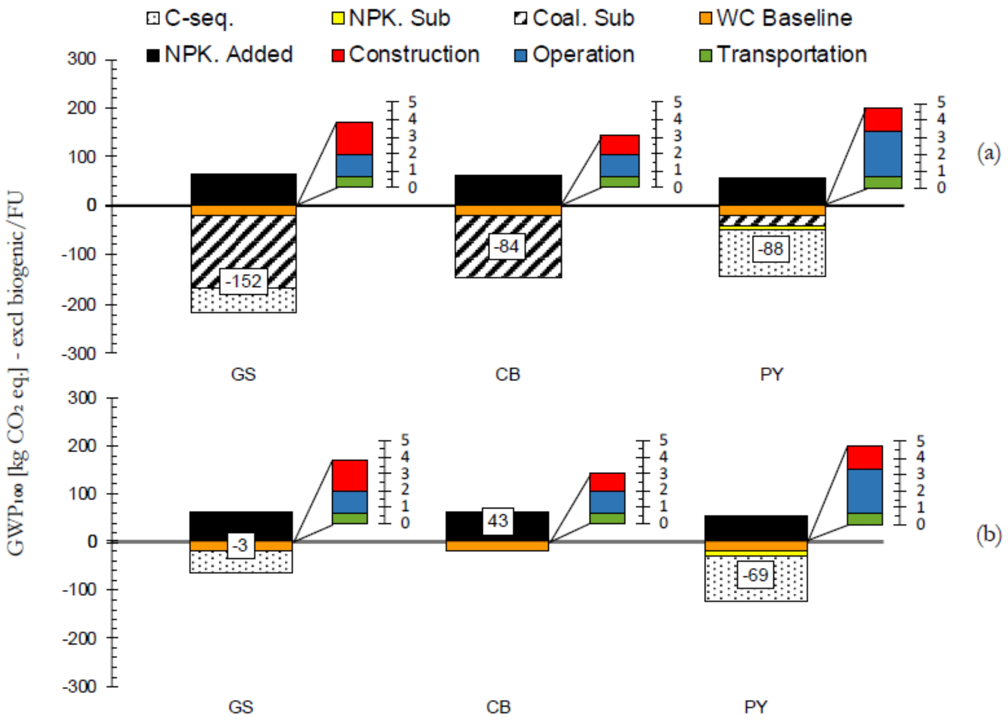
Use of domestic garden waste for use as a feedstock in pyrolysis merits further consideration given that collection systems already exist in many places.

3.4 Review of LCA studies on use of crop and horticultural residues for biochar.

LCA studies have investigated production of biochar from oilseed and rice straw, arable straw and seed hulls, grape marc and maize cobs (see review in Matustik et al., 2020). In New Zealand, studies have been undertaken of grape marc (Jones et al., 2020) and apple orchard prunings (Anaya de la Rosa, 2013).

Jones et al. (2020) undertook a C and economic analysis of pyrolysis, gasification, combustion and composting of grape marc produced in the Marlborough region of New Zealand. They found that the three thermal treatments were climate-positive when including the benefits due to displacement of heat from coal-based energy generation, and electricity from conventional (for combustion) or coal-fired (for gasification) electricity generation. When no displacement of coal-based energy generation took place, pyrolysis was climate-positive, gasification was marginally climate-positive, and combustion was climate-negative. Note that the baseline assumed the grape-marc would have otherwise been composted in a well-managed composting facility and reapplied to horticulturally productive land (e.g. vineyard). The climate change result for pyrolysis was dominated by C sequestration in the biochar, followed by addition of N fertiliser to provide equivalent N to that of the grape marc compost in the baseline, the displaced baseline (composting for this study), and displaced coal-based energy generation. Transportation of grape marc for processing, and construction of processing facilities, made a negligible contribution to the final results. In summary, the results showed that pyrolysis had the most climate-positive result out of the three thermal treatments when excluding displaced alternative energy generation, and gasification had the most climate-positive result when including displaced alternative energy generation; however, in both cases the pyrolysis result was at least 70% better than the displaced composting baseline (mainly due to C sequestration in the biochar). It should be noted that this result was highly dependent upon the initial moisture content of the grape marc; if it increased to 75% from the original assumed 67% moisture content, then the pyrolysis system was only marginally climate-positive (-12 kg CO_2e compared with -88 kg CO_2e t^{-1} incoming fresh grape marc at 67 %). If the moisture content decreased to 50%, which is achievable with mechanical dewatering, the climate impact result changed to -213 kg CO_2e t^{-1} incoming fresh grape marc.

Figure 3.3. Climate change results for gasification, combustion, and pyrolysis of 1t fresh grape marc (Jones et al., 2020) (a) With displaced heat and electricity generation; (b) excluding displaced heat and electricity generated.



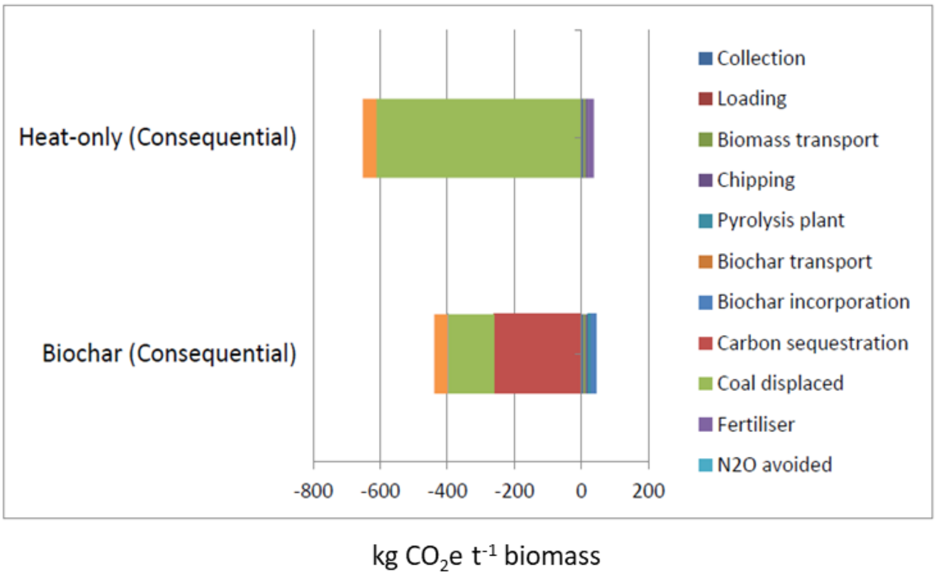
Note: GS=gasification; CB=combustion; PY=pyrolysis; NPK=synthetic fertiliser; WC=windrow composting

The use of apple orchard prunings (in Hawkes Bay) was investigated by Anaya de la Rosa (2013). For this study, the displaced baseline was chopping and reapplication of the orchard prunings back onto the orchard, and climate change results were calculated for combustion and for pyrolysis. The modelling assumptions included:

- one tonne fresh prunings results in 128 kg biochar (comprising 74% stable carbon); 1% of this is lost prior to application and 1% during tillage; it is spread at a rate of 444 kg biochar/ha,
- moisture content of the orchard prunings is 50%,
- application of biochar is associated with more efficient use of nutrients than in the baseline – hence there is a (small) credit for avoided fertiliser use,
- nitrous oxide emissions are suppressed by approximately 1 kg CO_2e t^{-1} processed prunings,
- heat generated displaces coal-generated heat.

The results in Figure 3.4 show that both the combustion ("heat-only) and pyrolysis ("biochar") scenarios are climate-positive (-614 and -393 kg CO_2e t^{-1} processed prunings, respectively). The results are dominated by the displaced coal combustion, and the C sequestered in biochar (for the biochar scenario). In a sensitivity analysis where factors such as moisture content of the prunings, stability of the C in the biochar, C content of the biochar, and boiler efficiency were varied to represent pessimistic and optimistic assumptions for the pyrolysis scenario, it was found that all results remained climate-positive (ranging from -36 to -973 kg CO_2e per one tonne processed prunings) (Table 26, Anaya de la Rosa, 2013).

Figure 3.4. Climate change results for combustion and pyrolysis of 1t apple prunings (Anaya de la Rosa, 2013).

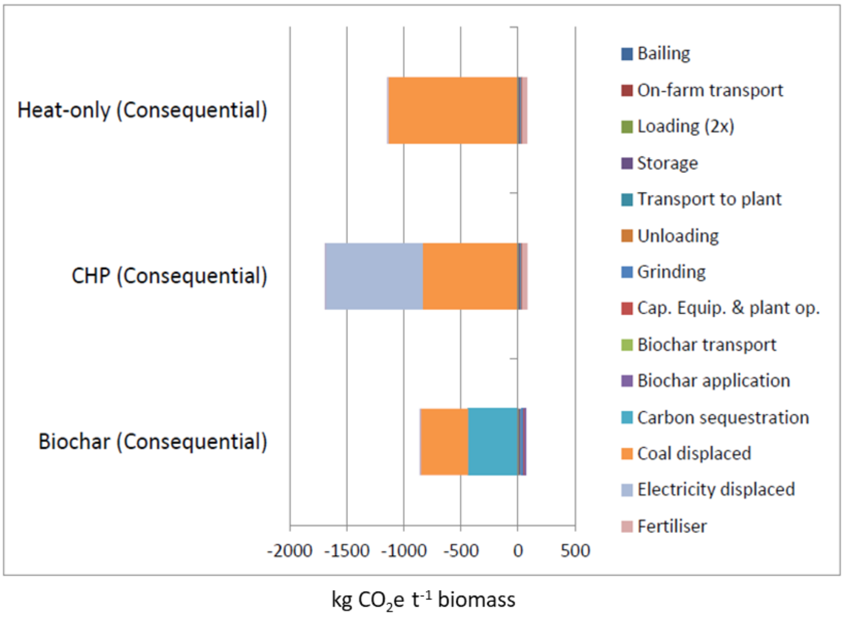


Anaya de la Rosa (2013) also assessed use of wheat straw from Canterbury to produce biochar. He assumed the displaced baseline was chopping and incorporation of the biochar into the soil, and climate change results were calculated for combustion, combined heat-and-power, and pyrolysis. The modelling assumptions included:

- one tonne fresh wheat straw results in 231 kg biochar (p.218, Anaya de la Rosa, 2013) (comprising 74% stable carbon); 1% of this is lost prior to application and a further 1% during tillage; it is spread at a rate of 973 kg biochar/ha,
- wheat straw has 13% moisture content
- application of biochar is associated with no change in use of nutrients compared with the baseline.

Heat and electricity generated displaces coal-generated heat and electricity. The results in Figure 3.5 show that all the scenarios are climate-positive (-1064, -1608 and -793 kg CO_2e t^{-1} processed wheat straw). As for the orchard prunings, the results are dominated by the displaced coal combustion and electricity generation, and the C sequestered in biochar (for the biochar scenario). In a sensitivity analysis where factors such as biochar migration, bio-oil yield, biochar yield from biomass feedstock, and stability of the C in the biochar, were varied to represent pessimistic and optimistic assumptions for the pyrolysis scenario, it was found that all results remained climate-positive (ranging from -205 to -1207 kg CO_2e t^{-1} processed wheat straw) (Table 38, Anaya de la Rosa, 2013).

Figure 3.5. Climate change results for combustion and pyrolysis of 1t wheat straw (Anaya de la Rosa, 2013)



Other studies that have considered use of agricultural by-products to produce biochar include:

- Muñoz et al. (2017) studied use of oat hulls to produce biochar, and subsequent application to barley fields. Based on field experiments, they calculated that the same barley yield could be maintained with a decreased application of nitrogen fertiliser (by 37.2 kg N/ha) when biochar was applied at a rate of 20 t ha^{-1}. However, they found that the climate change result was still dominated by the fixed C in biochar and avoided fossil fuels. The climate change result ranged from -2.59 to -2.70 t CO_2e t^{-1} biochar produced (depending upon the burning temperature). Net environmental benefits were also noted for the fossil depletion, human toxicity, and freshwater eutrophication impact categories – but these were related to displaced natural gas (due to syngas generated during pyrolysis) and reduced need for urea fertiliser (due to biochar application to the soil). However, use of forestry residues to produce the same quantity of biochar provided greater benefits than use of oat hulls across all the impact categories (Table 2, Muñoz et al., 2017).
- Thers et al. (2019) used an alternative functional unit of 1 tonne seed of winter oilseed rape, and found that producing biochar from the oilseed rape straw substantially mitigated the climate change impact of oilseed production, although this was not sufficient to offset the GHG emissions associated with oilseed cultivation.

In general, then, LCA studies of biochar production from agricultural and horticultural residues, including subsequent biochar application to land, have climate-positive. The magnitude of benefits depends upon the type of feedstock (and, in particular, its moisture content), and the avoided fossil fuels.

3.5 Review of LCA studies on use of sewage sludge for biochar.

There have been relatively few LCA or C footprint studies of producing biochar from human sewage sludge. In a recent review, Singh et al. (2020) refer to only two original studies: Hospido et al. (2005) and Ramachandran et al. (2017). Hospido et al. (2005) studied both anaerobic digestion and pyrolysis of sewage sludge. For all the scenarios, they calculated a net climate-negative impact ranging from about 250 to 630 kg CO_2e t^{-1} thickened mixed sludge (dry basis), and worse results for the pyrolysis scenarios. The main contributor to these results was thermal drying of the sludge prior to pyrolysis; however, it should be noted that C sequestration in the biochar was not included in the analysis as it was assumed that the biochar was burned for heat generation, displacing charcoal as a fuel source.

Ramachandran et al. (2017) examined the potential for co-gasification of sewage sludge with woody biomass; this was compared with the current system comprising anaerobic digestion followed by incineration of the dried sewage sludge in Singapore, and incineration of woody biomass. They found that the existing system had a small climate-positive result of -33 to -45 kg CO_2e t^{-1} feedstock (comprising 200 kg dried sewage sludge and 800 kg woody biomass). However, the co-gasification system had a climate-positive result of -438 kg CO_2e t^{-1} feedstock (Section 3.1, Ramchandran et al., 2017); displaced electricity made the biggest contribution to this result.

Two other LCA studies on pyrolysis of sewage sludge were identified for this review: Barry et al. (2019) and Cao and Pawlowski (2013). Barry et al. (2019) considered pyrolysis of sewage sludge in Canada using four scenarios: agricultural application of biochar, use of biochar as an energy source in a cement kiln, and incineration of biochar both with and without energy generation. They did not include a displaced baseline scenario. They found that only two of the scenarios had a net climate-positive result: agricultural application of biochar and use in the cement kiln (with the latter providing the best result). All the scenarios had freshwater toxicity impacts (determined mainly by the fate of heavy metals in the sewage sludge) – and the two non-incineration scenarios also had the lowest freshwater ecotoxicity results. Cao and Pawlowski (2013) analysed pyrolysis of sewage sludge with and without an additional anaerobic digestion step in the process. They found that pyrolysis combined with anaerobic digestion provided the most climate-positive result (34% more climate-positive than for just pyrolysis alone), and that the main contributors to this result were drying of the sludge, and the displaced energy sources. Carbon sequestration in biochar contributed only 7 or 13% of the total GHG emissions for pyrolysis with or without anaerobic digestion respectively (Table 3, Cao and Pawlowski, 2013, Table 3, total GHG emissions calculated by adding both avoided and emitted GHGs).

A further two LCA studies have included sewage sludge in their analysis of a range of feedstocks for pyrolysis: Ibarrola et al. (2012) and Cheng et al. (2020). Ibarrola et al. (2012) found that pyrolysis (and gasification) yielded climate-positive results for sewage sludge. The net climate-positive result for slow pyrolysis was -790 and -750 kg CO_2e t^{-1} feedstock, for scenarios where the biochar was either applied to the soil or burned for heat generation respectively. For the biochar applied to the soil, the main contributors to this result were C sequestration in biochar, soil effects, and electricity offset from pyrolysis (about 42, 27 and 21% of the result respectively). For the biochar burned for electricity generation, the largest contribution was from displaced electricity (about 65% of the result). The displaced baseline was landfilling, which contributed about 10% to the final result. In contrast, Cheng et al. (2020) found that pyrolysis of sewage sludge had a net climate-negative impact at all temperatures (varying from about 150 to 300 kg CO_2e using a functional unit of 1 tonne dry weight feedstock-see results in Section 3.6); however, their study did not account for soil effects or any displaced baseline scenarios.

It should be noted that the results of Hospido et al. (2005) are quite different from the other studies because that study did not account for C sequestration in biochar or for displaced electricity. However, the other studies vary widely from each other and it can be concluded that there is both a need for a more standardised approach to biochar consequential LCA and a need for more research on the climate change impact of using sewage sludge as a feedstock for biochar. It is also clear that the greatest hurdle to positive climate outcomes for sewerage biochar is the thermal drying of the sludge before pyrolysis.

3.6 Review of LCA studies on use of animal manures and litters for biochar.

Regarding livestock manures, again there are relatively few studies. Hamedani et al. (2019) studied pyrolysis of pig manure (but did not account for any displaced baseline scenario); pyrolysis yielded a net climate-positive result (-472 kg CO_2e t^{-1} biochar) but contributed to increased impacts for other environmental impact categories (except for aquatic eutrophication which was neutral). Struhs et al. (2020) evaluated pyrolysis of cattle manure versus other manure management practices, finding that pyrolysis had at least two-thirds less GHG emissions than the other practices (per one tonne dry manure). However, relatively few details are provided on the results of either of these studies.

3.7 Summary of insights relevant to New Zealand from existing LCA studies

3.7.1 Modelling of pyrolysis systems producing biochar

The LCA studies show that decisions about displaced activities (E in Figure 3.1) are an important part of the modelling process in analysis of pyrolysis. In particular, the choice of displaced energy source when producing heat or electricity during pyrolysis is critical in determining the magnitude of the final result. Also, specific characteristics of the displaced activities can make a big difference to the final results. For example, Jones et al. (2020, Figure 4.13) also showed that assumptions about methane production from the displaced composting system for grape marc had a big impact on the net climate change result: if the pyrolysis system displaced a badly managed composting system which produced high quantities of methane, its net climate-positive result could be 65% better than in the situation where it displaced a well-managed composting system.

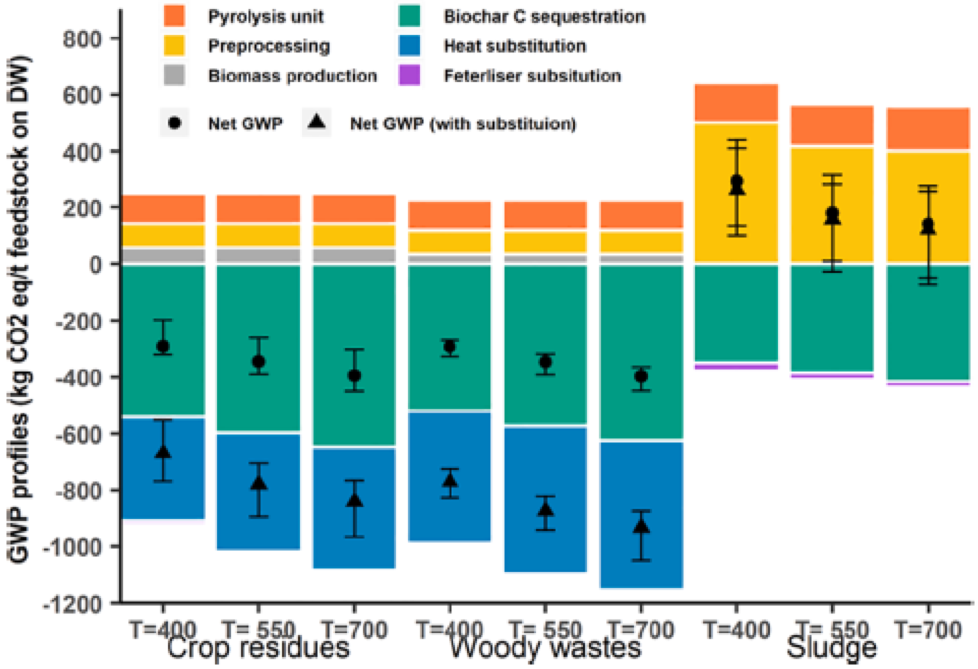
Modelling of the impacts on soil of applying biochar (e.g. increased availability of nutrients, reduced nitrous oxide emissions) make a smaller, and generally climate-positive, contribution to the net climate change impact. Assumptions about the quantity of biochar (C in Figure 3.1), and the percentage of stable C in biochar (D in Figure 3.1), also have a big influence on the climate change result. A smaller influence on the results is associated with varying parameters such as operating temperature of the pyrolysis unit, and degree of processing of the incoming biomass (B in Figure 3.1) with the exception of the biomass moisture content which has a big impact on the results (see, for example, Jones et al., 2020).

Generally, the biomass used as feedstock is regarded as a “free” input to the pyrolysis system (A in Figure 3.1). However, several studies note that removal of this biomass may have negative impacts on the agricultural soil; for example, there may be a long-term decline in soil organic matter content (Lee et al., 2020), as well as a higher fertiliser N requirement (Jones et al., 2020).

3.7.2 Comparison between feedstocks

Most LCA studies focus on use of one feedstock for pyrolysis. However, Anaya de la Rosa (2013) considered three different feedstocks: logging residues, apple prunings and wheat straw. He found that wheat straw provided the most climate-positive result followed by the apple prunings and then the logging residues (comparing the three feedstocks on a “one tonne fresh biomass” basis). This result is largely due to the low moisture content of the wheat straw (13% compared with 53% and 50% for the logging residues and apple prunings, respectively), and a higher biochar yield from the straw and logging residues than modelled for the apple prunings.

Cheng et al. (2020) used a machine learning method to model the energy and climate change impact of corn stover (representative of crop residues), forestry residues and sewage sludge as feedstocks for biochar production at different temperatures. The results are summarised in Figure 3.6. It can be seen that, at any given pyrolysis temperature, there was very little difference in the net climate change impact between crop residues and woody wastes but that sewage sludge had a net climate-negative impact at all temperatures (using a functional unit of 1 tonne dry weight feedstock).



Note: T=temperature in °C

Figure 3.6. Climate change results for pyrolysis of corn stover, forestry wastes and sewage sludge at different temperatures, using Monte Carlo simulations to calculate minimum and maximum values (Cheng et al., 2020, Figure 9)

The difference in relative results for the crop residues and forestry residues that is seen between Anaya de la Rosa (2013) and Chen et al. (2020) is at least partly due to the choice of functional unit. Anaya de la Rosa used a functional unit of one tonne fresh biomass, and Cheng et al. (2020) used a functional unit of one tonne dried biomass. Ibarrola et al. (2012) studied pyrolysis of wood waste, sewage sludge, green waste, food waste, anaerobic digestion digestate, dense refused derived fuel (DRDF), paper sludge, poultry litter, cardboard and whiskey draff. They found that pyrolysis of all biomass types was climate-positive, even for sewage sludge. In all cases, the single largest contributing activities was C sequestration in biochar, but soil effects and displaced electricity were also significant for all biomass types. The most climate-positive results was for wood waste, followed by green waste and food waste (where the functional unit was one tonne of biomass).

3.7.3 Activities making a material contribution to the climate change impact

Both Anaya de la Rosa (2013) and Jones et al. (2020) found that more than half of the climate change result for pyrolysis was due to C sequestration in the biochar. In Anaya de la Rosa (2013), the remaining climate change impact was largely due to displacement of fossil fuel-sourced electricity and/or heat. Other LCA studies have generally arrived at the same conclusion e.g. Matustik et al. (2020).

Given the contributions of different activities to the final result, Anaya de la Rosa (2013, Section 4.5.1) concluded that, “it might be pragmatic to disregard any potential soil-related GHG benefits offered by biochar application, such as fertiliser savings, suppression of N_2O emissions from soils, and higher crop productivity”. This is due to “the uncertainty attached to these benefits and the complexity of validation, monitoring, reporting and verification methods that would need to be in place in order to corroborate any claims made on these aspects.”

However, Jones et al. (2020, Figure 4.10) found that the second largest negative contribution to the climate change result for pyrolysis of grape marc was addition of N fertiliser to replace the N no longer available when applying biochar in the vineyard rather than composted grape marc. This was followed by the positive contribution made by displacing composting activities, which had almost the same climate change impact as the displaced coal-sourced heat. Along the same lines, Ibarrola et al. (2012) showed that the displaced baseline could make a big contribution to the final results when it involved incineration of sewage sludge, green waste or wood waste, or recycling of cardboard. Further, Lee et al. (2020) noted that there is an “outstanding argument” about the relative magnitude of the GHG emissions associated with loss of soil organic matter in the baseline scenario when biomass is removed for biochar production.

3.7.4 End use of biochar

When considering the relative benefits of producing biochar by pyrolysis of crop and forestry residues, compared with alternative fates of that biomass, the benefits mostly accrue from long-term C sequestration of fixed C via application of biochar to soil and through displacement of fossil fuel-sourced heat and/or power. All LCA studies indicate a climate-positive result from this type of system, regardless of the biomass and even if fossil fuels are not displaced. Based on their review, Matustik et al. (2020) conclude that, “biochar-soil amendment systems, as such, show a clear benefit, from the climate change perspective.”

If the biochar is instead burned as an energy source, the climate change benefits are largely lost. Lee et al. (2020), in their review of LCA studies of pyrolysis followed by burning the biochar to produce energy, showed that most studies (using feedstocks varying from rice husk to forestry residues) calculate a small climate-positive result for this type of system (less than 10% reduction in the climate change impact compared with the displaced baseline scenario) and noted that there may be negative impacts on soil quality when biomass is removed. Ibarrola et al. (2012), in their study of ten different feedstocks, found that there was relatively little difference in the net climate change impact between applying biochar to soil and burning it for electricity generation; however, in all cases but one, the scenarios where biochar was applied to soil were more climate-positive than those where it was burned to generate electricity.

3.7.5 Other environmental impacts associated with pyrolysis

Most LCA studies of biochar production focus on the climate change impact but few estimate the wider impacts of trade-offs on air, soil and water quality, added values or penalties that need to be considered when evaluating waste to energy or biochar operations.. However, Lee et al. (2020, Table 4), in their review of LCA studies of pyrolysis followed by burning the biochar to produce electricity, summarise the results of six studies that include other impact categories. These results suggest that, although all the studies found climate-positive results associated with pyrolysis, the results for other impact categories were mixed i.e. for impacts such as acidification, eutrophication and toxicity, some results were better and some were worse than the baseline scenario in different studies. For example, the existence of trade-offs in environmental impacts is particularly noted in studies assessing use of sewage sludge as feedstock, where the fate of heavy metals is a concern from a toxicity perspective.

3.7.6 Pyrolysis versus other thermal treatment of biomass

The studies by Anaya de la Rosa (2013) and Jones et al. (2020) found that two alternative thermal treatments (combustion with heat recovery, and gasification) had more climate-positive results for all three feedstocks than pyrolysis with subsequent soil application of biochar - except for combustion of grape marc where the results were similar. However, these results were dependent upon displacement of coal-sourced electricity and/or heat. In the absence of such displacement, pyrolysis provided the most climate-positive result. Azzi et al. (2020) obtained a similar result: alternative combined heat-and-power (CHP) and power plants burning wood chips are more or less climate-positive than pyrolysis with subsequent biochar use in agriculture, depending upon the displaced energy sources (i.e. fossil fuels versus renewables, respectively). If the displaced energy source is renewables, then it is likely that pyrolysis will provide a more climate-positive result than CHP or power plants burning wood chips.

3.7.7 Scale of analysis

Anaya de la Rosa (2013) noted that the potential for climate-positive results at the national scale depends upon the availability of different feedstocks. At the regional scale investigated in the study, the logging residues in the Central North Island (CNI) region provided the highest total climate-change mitigation potential out of the three studied options (i.e. compared with use of apple prunings in the Hawkes Bay and wheat straw in Canterbury), despite having the lowest climate-positive result per tonne biomass processed. This is due to the larger quantities of logging residues that are available in the central North Island region, which can potentially be converted into more energy and/or biochar than the orchard prunings and the wheat straw in their respective regions (Section 4.5.3, Anaya de la Rosa, 2013).

4. Economic feasibility of biochar case study examples

Establishing the economic feasibility of biochar manufacture (Meyer et al., 2011; Bach et al., 2016) and application requires customising the financial analysis to each scenario of feedstock type, collection (harvesting) and pre-processing, pyrolysis, pyrolysis products and biochar post-processing, transport and use. As with LCA analysis the financial feasibility will be the difference between the costs and revenue associated with a traditional or alternative pathway for the feedstock (e.g. as a fuel, compost, organic fertiliser or landfill) and the biochar pathway (Woolf et al., 2018).

In this review we are considering the potential role of biochar to reduce net GHG emissions from New Zealand agriculture and this mainly revolves around the C sink value of the biochar and any additional value as a fertiliser, liming material or a soil conditioner, when applied to the soil. Similar, to biochar production, some GHG reduction technologies for agriculture (e.g. use of denitrification inhibitors, methane inhibitors and vaccines) have a cost. Therefore, a useful financial feasibility indicator to compare GHG mitigation strategies is the GHG abatement cost (AC). AC ($ t^{-1} CO_2e) measures the average financial cost of reducing one metric ton of CO_2e of a GHG. With respect to biochar production from biomass residues, AC is a relativity tool, comparing one treatment process to reduce emissions with another treatment process for the same biomass residue. In a wider context, AC may be useful to policy makers making decisions on the relative cost of GHG emissions reduction strategies for New Zealand primary industries. For example, dairying and on-farm forestry can be compared this way. When New Zealand dairy farms (cost structure 2016-2019, 6.59- 7.40 NZ$ kg MS^{-1}) reduced N leaching by 22 to 30 per cent by reducing inputs of N fertiliser and purchased feed, with an associated reduction in stocking rate, the average reduction in farm GHG emissions was 9 to 24 per cent and farm profitability fell (3-5%) producing an average abatement cost of NZ$100 t^{-1} CO_2e (Beukes et al.,, 2019). On-farm forestry can achieve far larger emission reductions (3–96%), depending on the percentage of the farm planted. However, this also can be an expensive option for dairy farms with an implied C cost in excess of NZ$100–600 t^{-1} CO_2e, mainly because of the large lost opportunity cost incurred when taking land out of dairy grazing.

The disposal of grape marc in Marlborough is an example case study. Current practice in Marlborough is land-spreading of raw grape marc, not composting. Land-spreading occurs at 42 t ha^{-1} of raw GM. While this complies with the Marlborough Environment Plan with respect to nitrogen loading, it does not follow recommended practise (equivalent to 9 t ha^{-1} raw GM) to avoid possible biological oxygen demand overload in soils[1]. Jones et al (2020) examined the feasibility of bioenergy, biochar production and composting solutions for the grape marc.

When considering the AC of biochar, the Marlborough grape marc study by Jones et al (2020) provides enough process detail to compare a range of thermal options to best-practice composting for the repurposing of the grape marc. For biochar compared to composting, AC (Eq.2) is the difference between the cost of producing and applying biochar compared to

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[1] Laurenson S., Houlbrooke, D., Review of guidelines for the management of winery wastewater and grape marc. A Report (by AgResearch) prepared for Marlborough District Council, June 2012.].

compost, divided by the difference between the carbon footprints, adjusted to give the AC units of NZ$ t^{-1} CO_2e.

$$AC = -\left(\frac{\left(\frac{\$\,activity\ cost}{t\ raw\ GM}\right)_{biochar} - \left(\frac{\$\,activity\ cost}{t\ raw\ GM}\right)_{compost}}{\left(\frac{C\ footprint\ (kg)}{t\ raw\ GM}\right)_{biochar} - \left(\frac{C\ footprint\ (kg)}{t\ raw\ GM}\right)_{compost}}\right) \times \frac{kgC}{tC} \times \frac{tC}{tCO2e} \qquad \text{Eq 2}$$

For this comparison, abatement costs range from NZ$40 - $44 per tonne of CO_2e, which is not far above the current NZU price on the NZETS (37.20 NZ$ t^{-1} CO_2e). This demonstrates that if biochar were a permitted carbon sequestration vehicle within the NZETS, biochar becomes a viable commercial activity compared to composting for repurposing of grape marc.

When reviewing the financial feasibility of biochar's use in agricultural soils (33 relevant publications), Robb et al. (2020) found that the degree of mechanisation and scale of biochar production and the gross national income per capita (GNI) have a large influence on the value of the feedstock, the cost of biochar production and the net value of biochar as a soil amendment. When expressed as an average AC, biochar applied to land in 'lower-income countries' (GNI <US$ 995) was - 58 US$ t^{-1} CO_2e and financially feasible, compared with + 93 US$ t^{-1} CO_2e in 'higher-income countries' (GNI>US$ 12,055; note NZ's GNI is ~ US$42,670), and not financially feasible unless average national C prices exceeded this value.

It important that international case study examples used to review the economic feasibility of biochar manufacture in New Zealand are either New Zealand case studies or reflect the same level of GNI, soil fertility, C price and biochar market value as a soil amendment as in New Zealand. In the lower GNI countries much of the cost of biochar production was offset by increases in crop yields when the biochar was applied to the soil. In these scenarios the infertility or physical limitations of the soils and the high local cost of manufactured fertiliser were a major player in the degree of crop response and its value (Robb et al., 2020).

As discussed in Section 2.12 the influence of feedstock type and pyrolysis conditions has a major effect on the C, available nutrient content and liming value of the biochar (Table 2.2). These in turn strongly influence the potential fertiliser value and C credit value of the biochar when added to the soil. Using the biochar chemical characteristic information and nutrient availability data gathered by Camps-Arbestain et al. (2015) it is possible to calculate the BC_{+100} value of resistant biochar C in $ $t^{-1}CO_2e$, based on an NZU C value of 35 $NZ $t^{-1}CO_2e$, the nutrient and lime value based on current ex-works domestic fertiliser and lime prices in New Zealand (Figure 4.2). It should be noted that non-CO_2 emissions during pyrolysis were not measured in these biochar characterisation studies and their value in tCO_2e have not been subtracted from the BC_{+100} tCO_2e. Based on the results of Sparrevik et al. (2015) non-CO_2 emissions during pyrolysis (see example in Figure 2.1) could range from 18 to 35% of the biochar product CO_2e for mobile retort and open kiln pyrolysers, respectively, with no air pollution control. Similarly, Pires de Campos (2019) found that when no emissions controls were used when pyrolying *Pinus radiata* woodchips, CO levels peaked at 15,000 ppm and CH_4 at 4,000 ppm, also with significant visible particulate emissions. With the introduction of an effective flare system, peak values were reduced to 200 ppm for CO and 80 ppm for CH_4 and the flue gas was smokeless. The use of a flare lowered the emissions well below the US EPA limit for biomass suspension burners, which is 2400 ppm for CO, and also just below the limit of 230 ppm for the more efficient fluidised bed burners. When emissions were averaged over the production cycle and CO and CH_4 quantities were multiplied by the IPCC 20-year horizon global warming potentials (GWPs) of 8.8 (CO) and 84 (CH_4), respectively, the real cost of poor emissions control becomes apparent. In Figure 4.1, Pires de Campos (2019) showed that when the GWPs are included and summed, the net emissions from the study (Massey) pyrolyser without any abatement (i.e., no flare system) was 25.0 kilograms of carbon (as CO_2e) per 1.0 kg carbon in the biochar. When the flare was used, emissions dropped to 2.0 kg C (as CO_2e) per 1.0 kg carbon in the biochar. In comparison to other simple kilns, which have partial combustion, the Kenyan Earth Mound emits 8.9 kg C (as CO_2e) and the Adam retort 5.2 kg carbon (as CO_2e) per 1.0 carbon in the biochar.

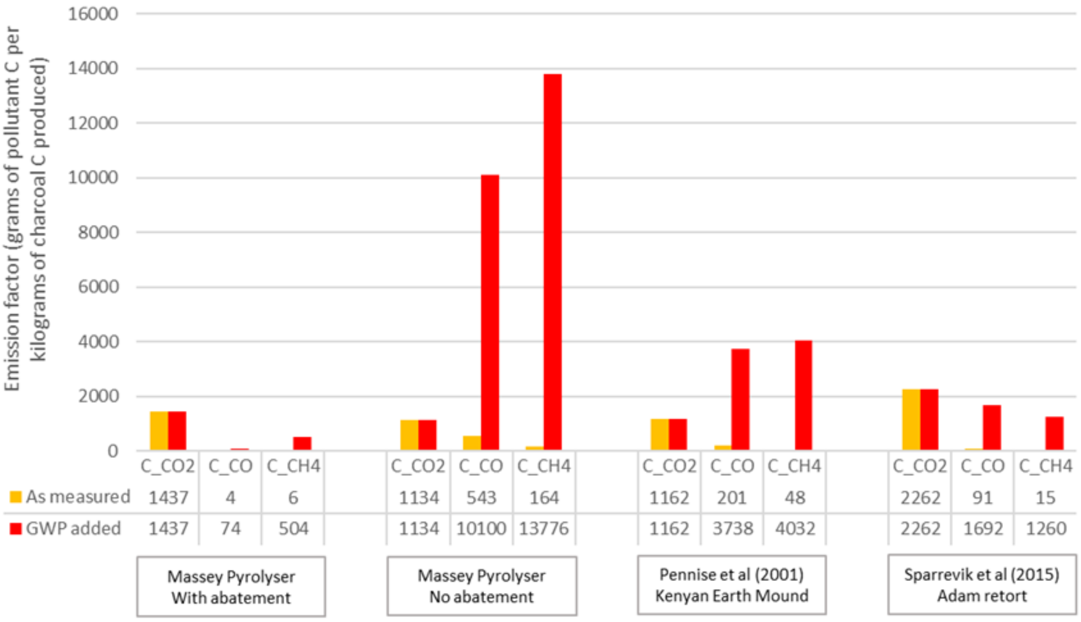


Figure 4.1. Comparison of the emission factor for CO_2, CO and CH_4 carbon emission from the Massey pyrolyser, Kenyan Earth Mound and Adam retort kilns, per kg of char carbon produced. Orange bars represent the rate of emission without adding the GWP values for the gases. The red bars represent the carbon equivalent production of each gas. The numbers below the bars are the production rate of each gas. The 20 year global warming potential horizon is used in these calculations (from Pires de Campos (2019)).

The emissions during biochar production and application to land need to be compared to biomass undergoing natural aerobic decay. For the case study used by Pires de Campos, 3.48 kg of dry biomass delivers 1 kg of carbon into the biochar. With *P. radiata* containing 51.49% carbon (Bridges, 2013) this means that the limiting emission by aerobic decay is 1.79 kg C (as CO_2). Only when using a flare does the net emission do better than aerobic decay, i.e., 2.0 kg C emitted (as CO_2e) - 1.0 kg C in the biochar = 1.0 kg net emission of carbon (as CO_2e), which is an abatement value of 1kg C (as CO_2e) kg^{-1} C as biochar. This highlights that closed pyrolysis systems with effective flaring of the pyrolysis gases are essential for GHG reduction, but also essential economically if biochar producers were required to purchase NZUs (currently 35 NZ\$ t^{-1} CO_2e) to account for the excess emissions during production. Provided the carbon sink value of the biochar in NZU's exceeds the non- CO_2 pyrolysis emissions, then the C credit can offset biochar production costs.

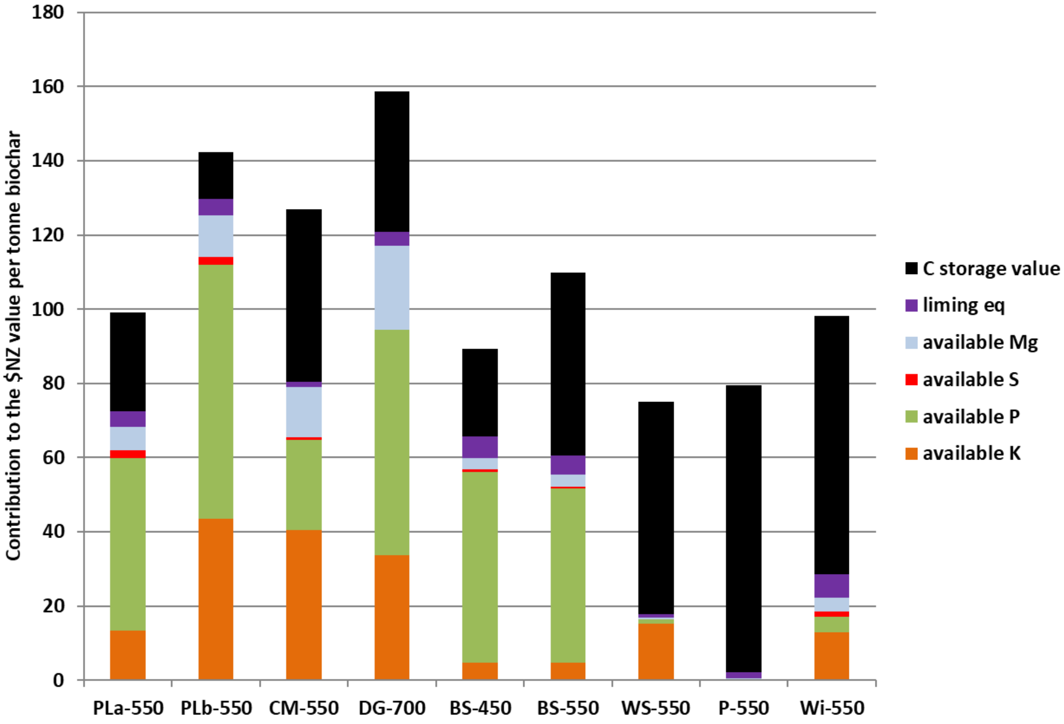


Figure 4.2 The C, available nutrient content and liming value of the biochars produces from Poultry litters (PLa-550 and PLb-550), Cattle manure (CM-550), Biosolids Digestate (DG-700), Biosolids + eucalyptus wood BS-450, BS-550, Wheat Straw (WS-550), Pine Chip (P-550) and Coppiced Willow (Wi-550). (The chemical data was sourced from Camps-Arbestain et al. (2015) and Medynska- Jurazek et al. (2020).The nutrient values (\$/kg element ex-works) were determined from the value of Superphosphate, Sulphur Super 30, KCl, MgO, and lime in the July 2020 Ravensdown fertiliser Price list. Values used were 2.55 NZ\$ kg^{-1}P, 1.30 NZ\$ kg^{-1} K, 0.55 NZ\$ kg^{-1}S, 1.30 NZ\$ kg^{-1}Mg , and Lime 0.03 NZ\$ kg^{-1}Ca CO_3. The Carbon storage value is based on a calculated BC_{+100} using IBI protocols (Camps -Arbestain et al., 2015) and a C price of 35 NZ\$ $t^{-1}$$CO_2$ e)

In addition, the fertiliser value of nutrients in biochar, and its liming value may be realised when added to acidic, low fertility soils. It is worth noting the large range of values and the contrasting contributions of nutrient value for biochars made from poultry litter, manures, biosolids, wheat straw and pine chips (Figure 4.2). The biochar classification system designed by Camps-Arbestain et al. (2015) would not class the biochars made from wheat straw (WS-550) and pine chips (P-550) as having fertiliser value because very large impractical quantities would need to be added per hectare to meet crop nutrient demands. This range of values in Figure 4.2, emphasises why biochars need to be characterised by a certifying authority before assumptions are made about their agricultural value, fertiliser GHG offsets and C sink values. In addition, the degree of non-CO_2 emissions (e.g CH_4, N_2O, NMVOC) during pyrolysis will need to be measured or estimated to adjust the CO_2e value of the BC_{+100} C content of the biochar.

When considering AC and renewable biomass fuels, similar or lower abatement values can be achieved when the bioenergy can replace fossil fuels. For example, Frank et al. (2020) show that fast pyrolysis of corn stover (>1000°C.s^{-1}) producing mostly fuels and small amounts of biochar had a zero NPV (Net present value, calculated over 20 years plant life) when the minimum C price (MCP) was 61 US\$ t^{-1} CO_2e, but for slow pyrolysis of pine wood (< 100°C min^{-1}) producing biochar only MCP was US\$ 123 t^{-1} CO_2e. The MCP is effectively the AC in year 1 of the 20-year NPV calculation.

In New Zealand, whereas 80% of electrical supply is from renewable sources, 60% of its domestic and commercial energy requirements are derived from fossil fuels. One area with opportunity is that New Zealand has many coal boilers in hospitals, schools and other public facilities; the government has a policy to convert these to renewables. Both economic and environmental benefits are enhanced when the coal offset value is monetised on the NZETS, where the NZU price is 35 NZ\$ t^{-1} CO_2e. Including the offset value for grape marc charcoal to generate heat for public buildings returns a profit of 7.10 NZ\$ t^{-1} GM (using the baseline cost of industrial heat generation of 10 NZ\$ GJ^{-1}). In contrast, the cost of producing biochar and returning it to soil is 56.70 NZ\$ t^{-1} GM when biochar is not a recognised C sink, or 48.80 NZ\$ t^{-1} GM if it is a recognised C sink and permitted into the NZETS as a C-sink mechanism. This converts to a break-even sale price for the biochar of 500 NZ\$ t^{-1} BC and 430 NZ\$ t^{-1} BC, respectively. Therefore, from a financial perspective, coal replacement is a superior option to making biochar and returning it to soil. From an emissions perspective, there is little difference between biochar added to soil or grape marc used as a coal offset; when the coal offset is included, the carbon footprint increases from -225 kgCO_2e t^{-1} GM (for biochar only) to -256 kgCO_2e t^{-1} GM. While coal replacement value is encouraging, it must be remembered that once all the coal boiler conversions have occurred in New Zealand, this business model is no longer possible, and heat generation from charcoal becomes a net cost of 5.00 NZ\$ t^{-1} GM and the carbon footprint reverts to +24 kgCO_2e t^{-1} GM, reflecting the fact that there is a net energy requirement to run the process plant. It must be noted that these figures calculated above, costs or profits, are the margin required to deliver a net present value of zero over twenty-five years with a cost of capital of 5%.

A more substantial opportunity in New Zealand is conversion of the dairy industry from its high reliance on coal to heat spray driers. Indeed in 2019, for the food industry, where most low-grade process heat is required, coal supplied 18.70 PJ of energy, representing *ca*. 620 kilo tonnes of coal (MBIE, 2019). Biomass (e.g. straw or wood chips) can easily supply the required low-grade heat, < 250°C (as compared to high-grade heat required for metals smelting, >1000°C). However, there are several difficulties; (i), coal boilers need to be replaced rather than retrofitted especially for 100% conversion to biomass as they operate at temperatures that will cause corrosion and slagging of the alkali metals in biomass; (ii), distributed collection of biomass is more costly than point-source mining of coal; (iii), biomass moisture content is both variable and expensive to remove prior to combustion; and (iv), the tonnages required are large and so significant infrastructure is required for materials handling and feedstock preparation. While not investigated specifically in Jones et al. (2020), because dairy heat replacement is at a far larger scale than the grape marc tonnages available, the profitability is likely to be similar to the figure calculated above at 7.10 NZ$ t^{-1} GM, or 2.27 NZ$ GJ^{-1} of heat produced. This indicates that biomass charcoal in retrofitted boilers is competitive at scale as long as the cost of coal heat generation does not drop lower than 7.73 NZ$ GJ^{-1}. Specific detailed studies are required.

Other opportunities for biomass replacement of fossil fuel derived energy are limited because of high demands for transport fuels in New Zealand. Currently emissions from fossil fuel in NZ are 30,167 kt CO_2e with 50% (15071 kt CO_2e) in road transport (MFE, 2020). Therefore, there are opportunities for fuels derived from pyrolysis of biomass to substitute for gas and liquid fuel use in NZ. Several researchers have shown that co-production of biochar for C sequestration and bio-oil for transportation fuels is cost-competitive for a pyrolysis facility (van Schalkwyk, et al., 2020; Frank, et al., 2020; Brown, et al., 2011). The limitation of these projections appears to be that technologies for producing transportation fuels from bio-oil through biomass fast pyrolysis are not proven at a commercial scale (Perego and Bosetti, 2011) and current methods of refining deliver refined fuel costs that are not competitive with current petrol and diesel costs. The main limitation for refining bio-oils derived from biomass pyrolysis is that they corrode equipment and rapidly coke and so deteriorate the expensive catalysts, which reduce refinery operating hours before catalyst renewal is required (Mostafazadeh et al., 2018). As mentioned in Section 2.2, however, very recent developments have created the opportunity for pyrolysing waste biomass into high quality fuel for automotive combustion engines (Schmitt et. al., 2019). The feasibility of implementing such technology on a regional basis in New Zealand now needs to be explored.

4.1 Forest residues to soil C sink – with and without oil and gas recovery.

4.1.1 Without oil recovery (medium to small scale)

Examples of biochar production systems using forest residues providing simulations of the cost productions have been published for differing scale operations. Each biochar case study budget includes equipment, fuel and labour costs associated with transportation, pre-processing of the raw feedstock to a particle size suitable for the pyrolyser (large-scale: using forwarders and excavators with jib saws; medium-scale: using loaders and tractors with grabs and chainsaws), pyrolyser capital (amortised) and operational costs.

For the flame-curtain kilns it is assumed that pyrolysis takes place in a season when the woody residues have dried to 20% moisture content (Wilson, 2017), suitable for direct pyrolysis without further drying. Flame-curtain kilns can have a larger labour component than retorts because constant supervision is required to maintain a surface, burning-bed of flames that will combust the volatiles. Because these open kilns have no insulation, heat is readily lost and so yields of biochar are in the order of 10% or less. They should not be operated in climatic conditions where fugitive sparks may be carried to nearby vegetation. For the portable retorts, where biochar yields of 30% are typical, small volumes of diesel fuel are used to raise the feedstock temperature to begin pyrolysis and higher moisture contents (up to 35%) can be tolerated (Wrobel-Tobiszewska et al., 2015). On orchards and farms, however, both kilns and retorts can share labour costs with intervals of other daily duties. At forest landing sites duties can be shared with activities such as firewood supply or hogging chip for transport to bioenergy plants.

Wrobel-Tobiszewska et al. (2015) and Keske et al. (2019) both considered the operation of a portable retort pyrolyser that could process 12 t d^{-1} of cut to length air dry log from cutover and landing residues from eucalyptus forest in Tasmania and Canadian black spruce forest, respectively and produced biochar for reapplication to the soil (Table 4.1).

Despite using a similar scale of pyrolyser the cost for producing 1 t of biochar in the Canadian case study was 1004 CAD$ t^{-1} BC, whereas the cost in the Tasmanian case study was 191 AU$ t^{-1}BC; notably the labour and machinery costs were estimated to be much higher in Canada, even considering the effect of time on monetary value..

At a similar production scale, the lower technology case study of Wilson (2017), used a flame curtain kiln to process trees from a Siberian Elm shelter belt . This operates with manual labour using chain saws and a tractor or skid loader to fill the kiln. The biochar production cost, of 133 US$ t^{-1} BC, was in a similar range to that estimated in the Tasmanian case study. The biochar value in nutrients and C credits were likely to be similar to P-550, 80 NZ$ t^{-1}BC (55 $US t^{-1}BC) in Figure 4.2. If the biochar was locally re-applied to the shelter belt area approximately 30% of the C credit value (77 NZ$ t^{-1}BC (53 $US t^{-1}BC) needs to be deducted to account for the non- CO_2 GHG emissions during pyrolysis (CO, CH4, other hydrocarbons and particulate) and emissions associated with running the plant (estimated from data provided by Homagain et al. (2015) and Sparrevik et al. (2015)). After this correction the biochar has a nutrient plus C credit value of values 57 NZ$ t^{-1} BC (39 $US t^{-1}BC), which covers approximately 29% of the biochar production costs. It must be noted that in this analysis, the biochar yield as a % of feedstock and the pyrolysis losses in emissions values seems to be closer to those of a medium scale re-tort and do not reflect the larger scale of losses expected from a flame curtain kiln. Prior to investment in this type of biochar production, the abatement value and operational costs need to be determined in country specific conditions.

Approximately 80% of the harvestable biomass residues in New Zealand are forest residues (Hall, 2017). Not all are easily recovered. Based on the assumption that 2.5 Mt of green harvestable forest residues (low residue recovery assumption, Hall, 2017) are available per year and will yield 1.4Mt dry wood, which when pyrolyzed could yield 0.4 Mt biochar and 0.17 Mt C as BC_{+100} (a long-lasting C sink after process emissions have been deducted), then carbon dioxide removal is 0.62 Mt CO_2e y^{-1}. This is equivalent to reducing New Zealand's agricultural GHG emissions of 37.7 Mt CO_2e y^{-1} by 1.6%.

Table 4.1 Examples of Biochar production at different scales that have forecast Biochar production costs

Feedstock	Pyrolyser type (Capital cost)	Feedstock Value	Pyrolysis Scale	Biochar Production (BC t d^{-1})	Cost ex-pyrolyser (\$. tBC^{-1})	Cost includes \$ benefits of BC or avoidance \$	Reference
Eucalypts Harvest residue	CharMaker MPP20 mobile pyrolysis plant from the Earth Systems® (VIC, Australia) ($AU 250,000)	0 (AU$)	12 (FS t d^{-1})	3 (BC t d^{-1})	191 (AU$)	No	Wrobel-Tobiszewska et. al. (2015)
Canadian black spruce logs residue	Mobile pyrolyis plant ($CAD 200,000)	0(CAD$)	12 (FS t d^{-1})	2 (BC t d^{-1})	1004 (CAD$)	No	Keske et al. (2019)
Siberian Elm shelter belt	Flame curtain kiln (22 m^3, $US 5000)	0 US$	25 (FS t d^{-1})	5 (BC t d^{-1})	133 US$	No	Wilson (2017)
Green waste (Yard waste)	Large scale commercial rotary kiln (8-9 $M US)	0 (US$)	10 (FS t h^{-1})	3 (BC t h^{-1})	161 (US$)	No	Roberts et al. (2010)
Green waste (Yard waste)	Large scale commercial rotary kiln (8-9 $M US)	0 (US$)	10 (FS t h^{-1})	3 (BC t h^{-1})	- 4.22 ($US)	Yes (Tipping fee 49 US$ FSt^{-1})	Roberts et al. (2010)
Corn stover	Large scale commercial rotary Kiln (8-9 $M US)	-43.5 ($US)	10 (FS t h^{-1})	3 (BC t h^{-1})	161 (US$)	No	Roberts et al. (2010)
Kiwi fruit Orchard (10ha) Prunings	Flame curtain kiln (6 m^3, NZ$ 2,500)	0 NZ$	1 (FS t d^{-1})	0.1 (BC t d^{-1})	712 NZ$	No	This review
Grape marc	Large scale commercial dryers and rotary kiln (55 -98 $M NZ)	0 NZ$	192 (FS t d^{-1})	21 (BC t d^{-1})	500 NZ$	No	Jones et al. (2020)

4.1.2 With oil and biochar recovery (large scale bio-oil refinery)

The potential for recovering forest thinning residues and converting into products such as bio-oil and biochar through pyrolysis was modelled by van Schalkwyk, et al. (2020). Reviewing other examples of bio-oil production, van Schalkwyk, et al. (2020) noted that few studies had considered biochar as revenue generating. In most studies, biochar was not a product of fast pyrolysis or was consumed to drive the heat and energy requirements of operations. The aim of van Schalkwyk, et al. (2020) was to assess whether C sequestration in biochar would drive a C negative fuel producing process from forest residues under intermediate pyrolysis conditions. Their projections, for the rate of collecting forest residues and wood processing wastes (dry basis t d^{-1}) from within 100 (338 t d^{-1}), 200 (1655 t d^{-1}) and 300 km (2549 t d^{-1}) distance of a combined pyrolysis and conventional oil refinery plant in South Africa, were that co-production of biochar for C sequestration and bio-oil as a feedstock to the refinery for production of transportation fuels is cost-competitive for the pyrolysis facility. The Minimum Selling Price (MSP) of bio-oil (22% IRR) for co‑processing was US$ 1.09 l^{-1} (crude) and US$ 1.46 L^{-1} (refined). Where biochar was a co-product (yielding 16.5% of the FS biomass, in the upgraded (refined) oil production) the biochar MSP was assumed to be 357.14 US$ t^{-1} biochar. With biochar as a co-product the projected global warming potential (GWP100) of bio–oil production corresponded to −9.0 (crude) and −5.8 (upgraded) kg CO_2e l^{-1}.

At the 338 t d^{-1} forest residue scenario of van Schalkwyk, et al. (2020),, the cost of collecting and transporting the residues is 27% of the operating and 17% of the total production cost, where the cost of collecting was 10.5 US$ t^{-1} and transporting the waste 100km was 5.90 US$ t^{-1}. According to Hall (2017) the similar costs in New Zealand for forest harvesting landing residues are 26.8 NZ$ t^{-1} for collection and 39.3 NZ$ t^{-1} (dry wood basis) for 100 km transport. In the case of pre-processing the wood residues in New Zealand for bioenergy, Hall (2017) includes hogging (chipping) the wood (@26.80 NZ$ t^{-1}) in preparation for loading furnaces. This plus loading costs yields a total cost of approximately 140 NZ$ t^{-1} to deliver hogged wood from 100km. Care needs to be taken in transferring the case study values of van Schalkwyk et al.(2020) to New Zealand situations because these latter costs for chipping appear to be missing from their scenario, although chipping as a process was mentioned as low cost.

4.2 Municipal green and biosolids wastes to fertiliser.

Using a large-scale commercial rotary kiln capable of processing 10 tonnes of green waster per hour Roberts et al. (2010) estimated that the production cost for biochar was $US161 t^{-1} BC. However, because the municipal green waste normally went to landfill, pyrolysis of the green waste avoided a landfill gate fee (tipping fee) of 49 $US t^{-1} FS. When this fee was deducted per tonne of green waste, the full cost of biochar production was immediately covered with a remaining surplus of 4.22 $US t^{-1} BC. When Roberts et al. (2010) considered the C value of the biochar produced at 20 $US t^{-1} CO2e and valued the P and K content, avoided fossil fuel energy, the tipping fee and avoided composting charges the surplus rose to $60 $US t^{-1} BC.

The New Zealand Biochar Research Centre at Massey University has considered a biochar production scenario as an alternative handling route for the biosolids produced in municipal wastewater treatment plant (WWTP). Biosolids typically go to landfill, especially from large cities. Preliminary research has been undertaken on a combined hydrothermal-pyrolysis process to convert 47% of primary sludge organic C into stable recalcitrant carbon. The main advantages are much smaller plant is required to make biochar (although more technically complex) and less susceptibility to biological shocks that can easily upset anaerobic digestors. The drying and pyrolysis of 12 t of thickened primary sludge (at 20% solids) produces 1t of biochar. One tonne of biochar avoids approximately 12 tonnes of tipping fees, 12 x 140 NZ$ = NZ$1680, deductible by 140 NZ$ if the biochar cannot be used for agricultural purposes because of contaminant loading and must go also to landfill as a repository for C. The economic cost of producing the tonne of biochar by this new process been determined to be similar to the existing sludge/biosolids processing costs of NZ WWTPs of between 400-600 NZ$ t^{-1} biosolids (at 18% dry solids content). In many NZ urban areas, however, metal contamination of sludge/biosolids is low and the biochar is likely to be able to be certified for land application, realising the fertiliser value of the biochar. If the avoided emissions from the landfill are equivalent to 0.105 t CO_2e t^{-1} feedstock dry weight (Ibarrola et al., 2012) then avoided emissions are 0.252 t CO_2e t^{-1} biochar and have a value of 9 NZ$ t^{-1} biochar. In this case the fertiliser and C value would be 90 -110 NZ$ t^{-1} biochar, plus the full avoided emissions and the avoided tipping and levy fees of 150 NZ$ t^{-1} thickened primary sludge are likely to cover between 27 – 40 % of the biochar production costs.

4.3 Crop and horticultural residues to soil conditioners.

The case study on repurposing grape marc (70,000 t /y @ 67% moisture content) conducted by Massey University for the Marlborough District Council (Jones et al., 2020) shows that, compared to a traditional pathway of composting and re-application to land, or the dominant current practice of direct land-spreading of grape marc, biochar (BC) manufacture in a centralised retort (producing up to 7900 BC t^{-1} y^{-1}) has the best C footprint outcome for the Marlborough wine industry among all the options. Options assessed, included composting, drying to produce animal feed or solid fuel pellets, drying followed by combustion for electricity, drying followed by gasification for combined heat and power, and drying followed by pyrolysis for biochar production (See Section 3.4; Figure 3.3 and Section 4) . The C footprint analyses considered all aspects subsequent to the pressing of the grapes for their juice: feedstock, transport to a centralised facility, drying pyrolysis, pyrolysis emissions, fertiliser offsetting and replacement, and land application. The comprehensive techno-economic analysis was carried out by determining the breakeven revenue required for each option to achieve zero net present value (NPV) over a period of 25 years at a discount rate of 5%. The analysis shows that shortfalls exist between the attainable revenues and those required to breakeven for most of the options, as shown in Table 4.2 and Section 4. Compared with the composting pathway the abatement achieved by pyrolysis for biochar was -245 kg CO_2e t^{-1} raw grape marc (or -2,220 kg CO_2e t^{-1} biochar). The commercial viability of biochar manufacture at this scale relies on balancing the cost, C footprint and liability to wine growers. This cost would either have to be balanced by a levy of 54 NZ\$ t^{-1} raw grape marc paid by the vineyards for biochar production or by sale of the biochar at 500 NZ\$ t^{-1}(which is within the range shown in Table 4.2). On this basis the abatement cost is 225 NZ\$ t^{-1} CO2e. The abatement cost, which is the comparative cost difference between making and using biochar, which sequesters carbon, and the alternative of composting, which does not, is 40 - 44 NZ\$ t^{-1} CO_2e (See Section 2.14). If the abatement comparison was for biochar going to landfill (a hypothetical example, because the Marlborough Environment Plan does not allow this) compared to composting, then based on the Marlborough landfill gate fee of 135 NZ\$ t^{-1} the abatement cost becomes 59-60 NZ\$ t^{-1} CO_2e. Neither case is far from the current NZU price (35NZ\$ t^{-1} CO2e).

Table 4.2 The breakeven price of each option of the case study on repurposing grape marc (values in parentheses are for options with minimum drying requirement)

Scenario	Breakeven price	Units
Best practice composting	91	NZ\$/tonne compost
Biomass combustion for power generation	0.225 (0.080)	NZ\$/MJ electricity
Biomass gasification for combined heat and power	0.286 (0.172)	NZ\$/MJ electricity with
	0.086 (0.053)	NZ\$/MJ heat
Biomass pyrolysis for biochar and heat	890 (350)	NZ\$/tonne biochar, with
	0.051 (0.020)	NZ\$/MJ heat
or	499~773 (435)	NZ\$/tonne biochar only
i.e.	(0.015) *	NZ\$/MJ heat equivalent
Coal price as a reference	0.01**	NZ\$/MJ
Electricity retail price***	0.072~0.119	NZ\$/MJ electricity
among which	0.024~0.068	NZ\$/MJ electricity for lines component

*IPCC assumption – an energy content of 30MJ/kg charcoal (Reumerman and Frederiks, 2002))
** Denne, 2014
*** MBIE, 2020

Each year a New Zealand kiwifruit orchard will generate summer and winter prunings that can make up 5.3 t ha^{-1} of air-dry residues (Ferguson and Eiseman , 1983). These are normally burnt or mulched back into the ground. If stockpiled and left to dry these prunings could be converted to biochar (Table 4.1) using a small flame curtain kiln (6 m^3 , processing 1 oven dry tonne of prunings per batch). This operates with manual labour using chain saws and a tractor with a front loader to fill the kiln. Open flame curtain systems are not recommended as the curtain is unstable, especially in wind, and have high risk of fugitive GHG emissions, which is pronounced in the early and late stages of operation. Also, these systems have a much higher ratio of biomass feed to charcoal yield, usually in excess of 10 (Section 2.2.1). These factors combined flame curtain systems deliver a low or nil climate-positive impact. However, such simple kilns may be able to operate safely, when supervised in an orchard environment. The biochar yield will be low perhaps 10% of feedstock (0.53 t BC ha^{-1}). Costs are mainly associated with labour (2h per Kiln batch) and fuel costs. Therefore, despite the low-cost kiln the biochar production cost remains high at 1020 NZ\$ t^{-1} BC. It is estimated that the biochar would be 77% C but only 70% of this would last in the soil for over 100 years (BC_{+100}), 0.29 tC ha^{-1}(or 1.05 tCO_2e ha^{-1}). The biochar value in nutrients and C credits are likely to be similar to biochar made from coppice willow Wi-550, 98 NZ$ t^{-1} BC in Figure 4.2 (Note these values do not have production emissions subtracted). The kiwifruit pruning biochar with a BC_{+100} value of 1.05 t CO_2e ha^{-1}, will have a credit value of 36.7 NZ$ ha^{-1} , if biochar could be traded on the NZETS (NZU = 35 NZ$ t^{-1} CO_2e). If the biochar was locally re-applied to the orchard area approximately 34 % of the C credit value of 36.7 NZ$ ha^{-1} is expected to be lost in pyrolysis emissions plus fuel and equipment emissions, which leaves a positive climate effect of -0.69 t CO_2e ha^{-1} (and ETS value of 24.2 NZ$ t^{-1}BC). The biochar would have a combined nutrient and C residual value of 53 NZ$ t^{-1}BC, which would cover approximately 5.2% of the biochar production costs. On this basis the abatement cost is 777 NZ$ t^{-1} CO_2e (an example calculation is presented in Appendix C). The yield of biochar at 0.53 tBC ha^{-1} with a positive climate effect of -0.69 CO_2e t ha^{-1}, would offset 15% of the 4.6 t CO_2e ha^{-1} of GHG emissions from a kiwifruit orchard (Mueller et al., 2015).

Based on the assumption that 5.3 t.ha^{-1} of air-dry pruning residues are available per year on 12,692 hectares of kiwifruit in production in NZ, which when pyrolyzed could yield 6.7 kt biochar and 2.4 kt C as a long-lasting C sink (after process emissions of 34% of the C sink value have been deducted), then total carbon dioxide removal is 8.8 kt CO_2e y^{-1} across New Zealand. However, kiln management is important. Poor kiln management results in fugitive emissions of products of incomplete combustion (CO, CH_4, particulate) which can easily rise above the 34% of the C sink (BC_{+100}) value used in the above calculations.

4.4 Animal manures and litters to fertiliser and lime.

Few studies have determined the economic cost of drying and pyrolysing nutrient-rich cattle manures to produce nutrient-rich biochars. Struhs et al. (2020) have calculated the processing of feedlot cattle manure in the US to biochar costs approximately 237 US$ t^{-1}BC. The total emissions from production are 951 kg CO_2e t^{-1}BC, which are 3.3 fold lower than for the traditional pathway of composting of the manure. The biochar has a very high ash content of 58.9% unlike dairy collecting yard manure in New Zealand of 13.4 % (Camps -Arbestain et al., 2015), therefore C_{org} content of the US biochar is low at 23.2 % and the BC_{+100} is calculated to be 124 C kg t^{-1}BC, which would provide a C sink value of -455 kg CO_2e t^{-1}BC. In comparison with the traditional composting of feedlot cattle manure the climate positive effect (avoidance plus sink value) is approximately -2670 kg CO_2e t^{-1}BC. On this basis the abatement cost is 91 US$ t^{-1} CO_2e and therefore unlikely to be financially supported by nutrient values plus a C credit of 24 US$ t^{-1} CO_2e.

5. Methodologies for accounting for biochar carbon in National GHG inventories

The development an IPCC protocol for including biochar in National GHG Inventories will stimulate the commercial production of biochar production as a carbon dioxide removal (CDR) technology. Section 5.1 addresses how the IPCC has proposed biochar production can be accommodated within inventory methodologies.

To include biochar in a national inventory will require assurance that the biochar production pathway is climate-positive and causes no adverse environmental effects. This can be demonstrated through biochar certification to meet the standards of either the IBI, EBC or ANZBIG (as discussed in Section 2.3). The EBC has also published a specific guide to certifying the carbon sink potential of biochar (EBC, 2020).

In Section 5.2 the additional data and C footprint calculations that may be required by an assurance scheme for biochar are discussed.

5.1 International guidance

The IPCC (IPCC, 2019a) recently developed a “basis for future methodological development” to allow biochar to be included in national inventories as a soil amendment.

It provides a methodology for calculating the change in mineral soil organic C stocks due to biochar application (where the biochar has been produced from either pyrolysis or gasification processes) (IPCC, 2019a). For these calculations, IPCC (2019b) has defined biochar “as a solid material generated by heating biomass to a temperature in excess of 350°C under conditions of controlled and limited oxidant concentrations to prevent combustion. These processes can be classified as either pyrolysis (in which oxidants are excluded), or gasification (in which oxidant concentrations are low enough to generate syngas)”.

5.1.1 Accounting for biochar carbon added to soils

The IPCC Tier 2 methodology (IPCC, 2019b) calculates the change in biochar C stocks (ΔBC, Eq.3) and adds this value to the soil C stocks.

$$\Delta BC_{mineral} = \sum_{p=1}^{n} BC_{TOT_p} \times F_{C_p} \times F_{perm_p} \qquad \text{Eq. 3}$$

Where:

$\Delta BC_{mineral}$ = the total change in C stocks of mineral soils associated with biochar amendment, tonnes sequestered C yr^{-1}

BC_{TOT_p} = the mass of biochar incorporated into mineral soil during the inventory year for each biochar production type, p in tonnes biochar dry matter yr^{-1}

F_{C_p} = the organic C content of biochar for each production type, p, tonnes C tonne^{-1} biochar dry matter

F_{perm_p} = fraction of biochar C for each production type p remaining (unmineralised) after 100 years, tonnes sequestered C tonne^{-1} biochar C

n = the number of different production types of biochar

In this case the IPCC (2019b) endorses the methods that use the H:C_{org} and O:C_{org} atomic ratios certified by the IBI and EBC respectively to calculate F_{perm_p}, which is a metric identical to the IBI's BC_{+100}%. (See Section 2.32).

Knowing that a certain percentage of feedstock C has been carbonised into a chemically resistant biochar is one important part of the information required to estimate the CDR value of the biochar. However, for national GHG accounting, it is also necessary to account for the change in other C stocks caused by biomass and wastes being used for biochar production.

5.1.2 Accounting for feedstock biomass diverted to biochar production

The IPCC (IPPC, 2006b) requires the National Inventory to record the source of the feedstock. (Note the EBC and ANZBIG also require this documentation to demonstrate that the batch of biochar is produced from a sustainable feedstock).

Therefore, the use of residues as feedstock for biochar production must be recorded as a loss of biomass. For example, if forest wood biomass (or agricultural land with woody crops (e.g. orchards and vineyards) is harvested and used for biochar production, it will be recorded as part of reported National C stock changes associated with fuel wood removals. This is calculated using Eq.4 (Equation 2.11 from IPPC, 2006b):

$$\Delta C_L = L_{wood-removal} + L_{fuelwood} + L_{disturbance} \qquad \text{Eq. 4}$$

Where:

ΔC_L = annual decrease in C stocks due to biomass loss in land remaining in the same land-use category, tonnes C yr^{-1}

$L_{wood-removal}$ = annual C loss due to wood removals, tonnes C yr^{-1}

$L_{fuelwood}$ = annual biomass C loss due to fuelwood removals, tonnes C yr^{-1}

$L_{disturbance}$ = annual biomass C losses due to disturbances, tonnes C yr^{-1}

Similarly if plant residues and manures are used as biochar feedstocks, rather than applying them directly to land, this reduces the overall input of organic amendments to soil and thereby affects soil C stocks (IPCC 2006b, Volume 4). The reduction in application of crop residues, such as wheat straw used as a biochar feedstock, would be accounted for in annual changes in organic C stocks in mineral soils (Equation 2.25, IPCC, 2006b; Volume 4, Chapter 2, p 2.30.). In Tier 1 calculations, where the least information is available about biomass growth and losses, the inventory values for the organic matter stock change factor ($0<F_I>1$) would be moderated to a lower value (IPCC 2006c Volume 4, Chapter 5, TABLE 5.5. p 5.18. For more accurate country specific (Tier 2) predictions of soil organic matter change in the 0-30 cm layer of mineral soils in cropland that remains as cropland, the IPCC (2019d) endorse the use of a steady-state computer simulation of the change in three sub-pools of soil organic matter sensitive to changes in climate and the amount and quality of crop residue input. Therefore, diverting residue to biochar manufacture leads to a short-term reduction in C input to the soil. However, if the biochar is reapplied to soil, over time, because biochar C is recalcitrant unlike the crop residue, it will persist in the soil, and raise the soil C stock.

5.1.3 Accounting for displaced non- CO_2 emissions

If biochar production replaces the traditional burning of crop residues, which would otherwise produce non-CO_2 greenhouse gas emissions, the displaced impact can be determined using a Tier 1 calculation (IPCC 2006b, Volume 4, Chapter 2, p 2.42, Equation 2.27) by selecting the appropriate emission factors (G_{ef}, CO_2, CO, CH_4, N_2O and NO_X) for burning agricultural crop residues and adjusting M_B, the mass of fuel available for combustion.

The production of biochar may also result in the reduction of wastes in other waste streams such as landfill (IPCC 2006d, Chapter 3: Solid Waste Disposal), or full incineration (IPCC, 2006e). The displaced emissions can be calculated using the IPCC methods for the calculation of emissions from waste management.

5.1.4 Accounting for emissions during processing, pyrolysis and transport

Biochar production by pyrolysis and gasification of waste materials, such as wood processing wastes and vegetable wastes with no energy recovery, will emit CH_4 and N_2O ,which can be calculated by adapting new methodology for municipal solid waste (Tables 5.3a and 5.4a IPCC 2019d, Volume 5).

For all feedstocks fossil fuels will be used in the harvesting, transport (IPCC, 2006f) and pyrolysis of the feedstock and a potential release of other non-CO_2 greenhouse gases during the heating process (IPCC, 2006f, 2006g) which are included in the energy sector.

5.2 Additional considerations for inclusion of biochar in National GHG inventories.

5.2.1 Consequential life cycle assessment of the value of the C-sink

The existing IBI, EBC and ANBIG codes of practice were discussed In Sections 2.31, 2.32 and 2.33. Examples of simple calculations for the sustainable production of biochar have been presented by the EBC (2012). However, these calculations do not include the comparative analysis of the GHG footprints of the traditional or business as usual pathway for the biomass and the biochar pathway.

To assure the national inventory that the production pathway for the biochar is more climate positive than the traditional pathway requires additional information, most usually in the form of carbon footprint and life cycle assessment (LCA) studies. We recommend that consequential LCA (see Section 3) comparisons are conducted to provide assurance that the biochar pathway is more climate positive than the traditional (or another proposed) pathway for the harvested biomass. Examples of such consequential LCA comparisons have been reported in detail by Anaya de la Rosa (2013).

For the pyrolysis of the biomass, as well as modelling the displaced traditional pathway, the assessments involve consideration of the following life cycle stages:

1. Growing and harvesting the feedstock (for agricultural biomass)

   If fertilisers, machinery (e.g. tractors, excavators and trucks) consuming fuel are used to grow, collect or harvest biomass (Figure 2.1 and 2.2) to be used as a biochar feedstock then the GHG emissions (CO_2e) associated with the embedded energy in the machinery, the fuel and fertiliser need to be calculated per unit of BC_{+100} CO_2e biochar applied to the soil.
2. Transport, storage and processing of feedstock

   If vehicles, chippers and dryers consuming fuel are used to transport, place in storage and prepare the biomass for pyrolysis then the GHG emissions (CO_2e) associated with the production and use of the machinery, storage facility and the fuel need to be calculated per unit of BC_{+100} CO_2e biochar applied to the soil.
3. Pyrolysis

   If electricity is used to operate the pyrolysis plant and fuel is used to preheat the pyrolysis reactors, then the GHG emissions (CO_2e) associated with the production and use of the pyrolysis plant plus the electricity, fuel and any GHG emissions during pyrolysis need to be calculated per unit of BC_{+100} CO_2e biochar applied to the soil.
4. Transport, storage and application to the soil

   If machinery and vehicles are used to transport, place in storage and prepare the biochar and apply it to soil, then GHG emissions (CO_2e) associated with the production and use of the machinery, storage facility the fuel need to be calculated per unit of BC_{+100} CO_2e biochar applied to the soil.

5.2.2 Future data needs

As further scientific evidence accumulates supporting either the additional climate positive or negative attributes of biochar application to soils, such as positive or negative priming of soil organic C, soil N_2O emissions, and crop yield responses, then project-level accounting will need to be codified to include this in consequential LCAs.
For project level GHG accounting Woolf et al. (2018) stress that when methods are developed to assure that abatement has occurred, they must be accurate at the appropriate scale and cost-effective. This will encourage a high level of participation, implementation and therefore maximum abatement. Methods based on measurable emission factors linked to auditable quantity factors are often the most simple and robust.

Currently, for New Zealand primary industries, soils and climate, there is insufficient supporting evidence for the additional climate positive or negative attributes of biochar application to soils for this more detailed accounting system to be constructed. This is a knowledge gap that needs to be addressed. Based on the high proportion of climate positive results appearing in international research, the New Zealand research is likely to identify larger GHG abatement value when biochar application is targeted to responsive soils, crops and trees.

6. References

ACVM. 2011. Agricultural Compounds and Veterinary Medicines (ACVM) Act 1997. ACVM Regulations 2011. https://www.mpi.govt.nz/processing/agricultural-compounds-and-vet-medicines/fertilisers/.

Akdeniz, N. 2019. A systematic review of biochar use in animal waste composting. Waste Management 88, 291–300.

ANZBIG. 2020. The Australian and New Zealand Biochar Industry Group: Code of Practice for the Sustainable Production and Use of Biochar. Revision Draft – 2 June, 2020. https://anzbig.org/resources/.

Anaya de la Rosa, R.K. 2013. Biochar systems for carbon finance – an evaluation based on Life Cycle Assessment studies in New Zealand. PhD thesis, Massey University, Wellington, New Zealand. https://mro.massey.ac.nz/bitstream/handle/10179/5973/02_whole.pdf

Anderson, C. R.; Hamonts, K.; Clough, T. J. Condron, L. M. 2014. Biochar does not affect soil N-transformations or microbial community structure under ruminant urine patches but does alter relative proportions of nitrogen cycling bacteria. Agriculture Ecosystems & Environment 191, 63-72.

Alhashimi, Hashim A., and Can B. Aktas. 2017. Life cycle environmental and economic performance of biochar compared with activated carbon: a meta-analysis. Resources, Conservation and Recycling 118 , 13-26.

Azzi, E.S.; Karltun, E.; Sundberg, C. 2019. Prospective Life Cycle Assessment of Large-Scale Biochar Production and Use for Negative Emissions in Stockholm. Environ. Sci. Technol., 53, 8466–8476.

Bach, M., Wilske, B., & Breuer, L. (2016). Current economic obstacles to biochar use in agriculture and climate change mitigation. Carbon Management, 7(3-4), 183–190. https://doi.org/10.1080/17583004.2016. 1213608

Barry, D., Barbiero, C., Briens, C., Berruti, F. 2019. Pyrolysis as an economical and ecological treatment option for municipal sewage sludge. Biomass and Bioenergy, 122, 472-480. DOI: 10.1016/j.biombioe.2019.01.041

Beyaert, R., Voroney R.P. 2011. Estimation of decay constants for crop residues measured over 15 years in conventional and reduced tillage systems in a coarse-textured soil in southern Ontario. Can. J. Soil Sci., 91 (2011), pp. 985-995, 10.4141/cjss2010-055.

Beukes, P., Romera, A., Hutchinson, K., Van Der Weerden, T., de Klein, C., Dalley, D., Chapman, D., Glassey, C., Dynes, R., 2019. Benefits and trade-offs of dairy system changes aimed at reducing nitrate leaching. Animals 9, 1–11. https://doi.org/10.3390/ani9121158.

Blanco-Canqui, H. Biochar and Soil Physical Properties. Soil Sci. Soc. Am. J. 2017, 81, 687.

Bolhar-Nordenkampf, M., Isaksson, J. 2016. Operating experiences of large scale CFB-gasification plants for the substitution of fossil fuels. In Faaij, APC; Baxter, D; Grassi, A; Helm, P (Eds.) Papers of the 24th European Biomass Conference: setting the course for a biobased economy Pages: 375-381.

Borchard, N., Schirrmann, M., Cayuela, M.L., Kammann, C., Wrage-Mönnig, N., Estavillo, J.M., Fuertes-Mendizábal, T., Sigua, G., Spokas, K., Ippolito J.A. 2019. Biochar, soil and land-use interactions that reduce nitrate leaching and N2O emissions: a meta-analysis. Sci. Total Environ., 651 (2019), pp. 2354-2364, 10.1016/j.scitotenv.2018.10.060.

Brassard, P., Godbout, S., Palacios, J.H., Jeanne, T., Hogue, R., Dubé, P., Limousy, L.,Raghavan, V., 2018. Effect of six engineered biochars on GHG emissions from two agricultural soils: a short-term incubation study. Geoderma 327, 73–84. https://doi.org/10.1016/j.geoderma.2018.04.022.

Brassard, P., Godbout, S., Raghavan, V., 2016. Soil biochar amendment as a climate change mitigation tool: key parameters and mechanisms involved. J. Environ. Manag. 181, 484–497. https://doi.org/10.1016/j.jenvman.2016.06.063.

Biochar Quality Mandate, 2013. https://biocharbraf.files.wordpress.com/2013/06/bqm-v-1-0-version-for-public-consultation1.pdf (accessed Dec. 2020).

Bridges, R. 2013. Design and Characterisation of an 'Open Source' Pyrolyser for Biochar Production, Masters thesis, Massey University, New Zealand.

Bucheli, T.D., Hilber, I., Schmidt, H.P., 2015. Polycyclic aromatic hydrocarbons and polychlorinated aromatic compounds in biochar. Biochar for Environmental Management: Science, Technology and Implementation, pp.595-624.

Budai A., Zimmerman A.R., Cowie A.L., Webber J.B.W., Singh B.P., Glaser B., Masiello C.A., Andersson D., Shields F., Lehmann J., Camps Arbestain M., Williams M., Sohi S., Joseph S. 2013. Biochar carbon stability test method: an assessment of methods to determine biochar carbon stability. https://www.biochar-international.org/wp-content/uploads/2018/04/IBI_Report_Biochar_Stability_Test_Method_Final.pdf.

Cabeza, I., Waterhouse, T., Sohi, S., Rooke, J. A. 2018. Effect of biochar produced from different biomass sources and at different process temperatures on methane production and ammonia concentrations in vitro. Animal Feed Science and Technology, 237, 1-7.

Calvelo-Pereira, R., Kaal, J., Camps Arbestain, M., Pardo Lorenzo, R., Aitkenhead, W., Hedley, M., Macías, F., Hindmarsh, J., Maciá-Agulló, J.A., 2011. Contribution to characterisation of biochar to estimate the labile fraction of carbon. Org. Geochem. 42, 1331–1342. https://doi.org/10.1016/j.orggeochem.2011.09.002.

Calvelo-Pereira, R., Muetzel, S., Arbestain, M. C., Bishop, P., Hina, K., Hedley, M. 2014. Assessment of the influence of biochar on rumen and silage fermentation: A laboratory-scale experiment. Animal Feed Science and Technology, 196, 22–31. doi:10.1016/j.anifeedsci.2014.06.019

Camps-Arbestain, M., Amonette, J. E., Singh, B., Wang, T., Schmidt, H. P. 2015. A biochar classification system and associated test methods. In Lehmann J. and Joseph S., eds. Biochar for Environmental Management: Science, Technology and Implementation, 165-194.

Camps-Arbestain, M., Lehmann, J., Singh, B. (Eds.), 2017. Biochar: A Guide to Analytical Methods, 1st ed. CSIRO PUBLISHING, Boca Raton, London and New York.

Camps-Arbestain, M., Shen, Q., Wang, T., Van Zwieten, L., Novak, J.M. 2017. Available nutrients in biochar. CSIRO Australia Griffith NSW. 109-125. In Biochar: A Guide to Analytical Methods, Camps-Arbestain, M., Lehmann, J., Singh, B. (Eds.), 1st ed. CSIRO PUBLISHING, Boca Raton, London and New York.

Campuzano, F., Brown, R. C., Martinez, J. D. 2019. Auger reactors for pyrolysis of biomass and waste. Renewable and Sustainable Energy Reviews, 102, 372-409.

Cao, Y., Pawłowski, A. 2013. Life cycle assessment of two emerging sewage sludge-to-energy systems: Evaluating energy and greenhouse gas emissions implications. Bioresource Technology, 127, 81-91. DOI: 10.1016/j.biortech.2012.09.135

Chao, L.; Zhang, W. D.; Wang, S. L. 2018. Understanding the dominant controls on biochar decomposition using boosted regression trees. European Journal of Soil Science, 69. 521-520.

Chen, T., Zhang, Y., Wang, H., Lu, W., Zhou, Z., Zhang, Y., Ren, L. 2014: Influence of pyrolysis temperature on characteristics and heavy metal adsorptive performance of biochar derived from municipal sewage sludge. Bioresour. Technol. 164, 47–54.

Cheng, F., Luo, H., Colosi, L.M. 2020. Slow pyrolysis as a platform for negative emissions technology: An integration of machine learning models, life cycle assessment, and economic analysis. Energy Conversion and Management, 223, art. no. 113258. DOI: 10.1016/j.enconman.2020.113258

Cornelissen G., Pandit N.R., Taylor P., Pandit B.H., Sparrevik M., Schmidt H.P. 2016. Emissions and Char Quality of Flame-Curtain "Kon Tiki" Kilns for Farmer-Scale Charcoal/Biochar Production. PLoS ONE 11(5): e0154617. doi:10.1371/journal.pone.0154617

Crombie, K., Masek, O., Cross, A., Sohi, S., 2015 Biochar - synergies and trade-offs between soil enhancing properties and C sequestration potential. Global Change Biology Bioenergy, 7, 1161-1175.

Denne, T. 2014. Coal prices in New Zealand Markets – 2013 update. Ministry of Business, Innovation & Employment, https://www.mbie.govt.nz/assets/9b10f9b98c/coal-prices-in-new-zealand-markets-2013.pdf, last accessed in November, 2020.

Dokoohaki, H., Miguez, F. E., Laird, D., & Dumortier, J. 2019. Where should we apply biochar? Environmental Research Letters, 14, 044005.

Dunlop S.J., Camps-Arbestain, M., Bishop, P.A., Wargent, J. Closing the Loop: Use of Biochar Produced from Tomato Crop Green waste as a Substrate for Soilless Hydroponic Tomato Production. Hortscience 50, 1572–1581

EBC (2012) 'European Biochar Certificate - Guidelines for a Sustainable Production of Biochar.' European Biochar Foundation (EBC), Arbaz, Switzerland. (http:European-biochar.org). Version 9.1E of 25th September 2020.

EBC (2013) 'European Biochar Certificate   EBC Guidelines & Documents for the Certification. Positive list of biomass feedstock approved for use in producing biochar. https://www.european-biochar.org/biochar/media/doc/feedstock.pdf.

EBC (2020) 'European Biochar Certificate   Certification of the carbon sink potential of biochar, Ithaka Institute, Arbaz, Switzerland. (http://European-biochar.org). Version 1.0E of 1st June 2020

Emrich, W. 1985. Handbook of Charcoal Making. Solar Energy R&D in the European Community. Series E, Energy from biomass, v. 7. Springer-Science.

Enders, A., Hanley, K., Whitman, T., Joseph, S., Lehmann, J. 2012 Characterization of biochars to evaluate recalcitrance and agronomic performance. Bioresource Technology 114, 644-653

Entec UK Ltd, 2005. Preparation of the Review Relating to the Large Combustion Plant Directive: A Report for European Commission, Environmental Directorate General.

EU, 2010. Directive 2010/75/EU of the European Parliament and of the Council on industrial emissions (integrated pollution prevention and control). Official Journal of the European Union, L334/17.

EU, 2015. Directive (EU) 2015/2193 of the European Parliament and of the Council on the Limitation of Emissions of Certain Pollutants into the Air from Medium Combustion Plants. Official Journal of the European Union, L313/1.

Eyles, A., Bound, S.A., Oliver, Corkrey, R., Hardie, M., Green, S., Close, D.C. 2015. Impact of biochar amendment on the growth, physiology and fruit of a young commercial apple orchard. Trees 29, 1817–1826. https://doi-org.ezproxy.massey.ac.nz/10.1007/s00468-015-1263-7.

FAO Forestry Department, 1985. Industrial charcoal making. FAO Forestry Paper 63, Rome, Italy. ISBN 92-5-102307-7

FAO Forestry Department, 1987. Simple Technologies for Charcoal Making. Food and Agriculture Organization of the United Nations, Rome. ISBN 92-5-101328-1.

Ferguson A.R., Eiseman J.A. 1983 Estimated annual removal of macronutrients in fruit and prunings from a kiwifruit orchard, New Zealand Journal of Agricultural Research, 26:1, 115-117, DOI: 10.1080/00288233.1983.10420960

Ferjani, A.I., Jeguirim, M., Jellali, S., Limousy, L., Courson, C., Akrout, H., Thevenin, N., Ruidavets, L, Muller, A., Bennici, S. 2019. The use of exhausted grape marc to produce biofuels and biofertilizers: Effect of pyrolysis temperatures on biochars properties. Renewable and Sustainable Energy Reviews, 107, 425-433.

Fischer, B.M.C.; Manzoni, S.; Morillas, L.; Garcia, M.; Johnson, M.S.; Lyon, S.W. Improving agricultural water use efficiency with biochar—A synthesis of biochar effects on water storage and fluxes across scales. Sci. Total Environ. 2019, 657, 853–862.

Frank, J.R., Brown, T.R., Malmsheimer, R.W., Volk, T.A., Ha, H. 2020. The financial trade-off between the production of biochar and biofuel via pyrolysis under uncertainty. Biofuels bioproducts & biorefining-biofpr 14, 594-604. DOI: 10.1002/bbb.2092

Fuertes, A. B., Arbestain, M. Camps., Sevilla, M., Macia-Agullo, J. A., Fiol, S., Lopez, R, Smernik, R. J., Aitkenhead, W. P., Arce, F. Macias, F.) 2010. Chemical and structural properties of carbonaceous products obtained by pyrolysis and hydrothermal carbonisation of corn stover. Australian Journal of Soil Research, 48, 618-626.

Furey, L., 2006. Maori Gardening – An Archaeological Perspective, www.doc.govt.nz/documents/science-and-technical/sap235.pdf, p49.

Fuss, S., Lamb, W.F., Callaghan, M.W., Hilaire, J., Creutzig, F., Amann, T., Beringer, T., Garcia, W.D., Hartmann, J., Khanna, T. Luderer, G., Nemet, G.F, Rogelj, J., Smith, P., Vicente, J.L.V., Wilcox, J., Dominguez, M.D.Z., Minx, J.C. 2018. Negative emissions-Part 2: Costs, potentials and side effects. Environmental Research Letters 13, Issue: 6 .Article Number: 063002. DOI: 10.1088/1748-9326/aabf9f.

García-Núñez, J.A., Pelaez-Samaniego, M.R., Garcia-Perez, M.E., Fonts, I., Abrego, J., Westerhof, R.J.M., Garcia-Perez, M. 2017. Historical developments of pyrolysis reactors: A review. Energy & Fuels, 31, 5751-5775.

Garrett, L. G., Kimberley, M. O., Oliver, G. R., Pearce, S. H., & Paul, T. S. H. 2010. Decomposition of woody debris in managed Pinus radiata plantations in New Zealand. Forest Ecology and Management, 260, 1389–1398.

Godlewska, P., Ok, Y.S., Oleszczuk, P. 2020. The dark side of black gold: Ecotoxicological aspects of biochar and biochar-amended soils, Journal of Hazardous Materials (2020), doi: https://doi.org/10.1016/j.jhazmat.2020.123833.

Goodman-Smith, F. 2018. A quantitative and qualitative study of retail food waste in New Zealand. Master thesis of the University of Otago, Dunedin.

Gregory, S. J., Anderson, C. W., Camps-Arbestain, M., Biggs, P. J., Ganley, A. R., O'Sullivan, J. M., and McManus, M. T. 2015. Biochar in co-contaminated soil manipulates arsenic solubility and microbiological community structure, and promotes organochlorine degradation. PloS One 10, e0125393.

Graber, E.R., Kookana. R. 2015. Biochar and retention/efficacy of pesticides. In: Biochar for Environmental Management - Science and Technology and Implementation, 2nd edition. Lehmann J.and Joseph S. (eds.). Routledge Publishers

Grieve, C.M., Poss, J.A., Grattan, S.R., Suarez, D.L., Benes, S.E., Robinson P.H. 2004. Evaluation of salt-tolerant forages for sequential water reuse systems: II. Plant–ion relations. Agric. water Manage., 70, 121-135.

Hale, S.E. Lehmann, J., Rutherford, D., Zimmerman, A.R., Bachmann, R.T., Shitumbanuma, V., O'Toole, A., Sundqvist, K.L., Arp, H.P.H., Cornelissen, G. 2012. Quantifying the Total and Bioavailable Polycyclic Aromatic Hydrocarbons and Dioxins in Biochars, Environ. Sci. Technol. 46 (2012) 2830–2838. https://doi.org/10.1021/es203984k.

Hall, P. 2017. Residual Biomass Fuel Projections for New Zealand – Indicative Availability by Region and Source. Scion, Rotorua.

Hamedani, S.R., Kuppens, T., Malina, R., Bocci, E., Colantoni, A., Villarini, M. 2019. Life cycle assessment and environmental valuation of biochar production: Two case studies in Belgium. Energies, 12 (11), art. no. 2166. DOI: 10.3390/en12112166

Hardie M, Clothier B, Bound S, Oliver G, Close D (2014) Does biochar influence soil physical properties and soil water availability? Plant Soil 376, 347–361.

Herath, H. M. S. K., Camps-Arbestain, M.; Hedley, M.J. 2013. Effect of biochar on soil physical properties in two contrasting soils: An Alfisol and an Andisol. GEODERMA, 209, 188-197 .

Hilber, I. Bastos, A.C. Loureiro, S. Soja, G. Marsz A., Cornelissen G., Bucheli, T.D. 2017. The different faces of Biochar: contamination risk versus remediation tool, J. Environ. Eng. Landsc. Manag. 25 (2017) 86–104. https://doi.org/10.3846/16486897.2016.1254089.

Hill, A. J., Di, H. J., Cameron, K. C., Podolyan, A. 2015. Comparison of dicyandiamide and biochar for reducing nitrate leaching under winter forage grazing in Canterbury, New Zealand. New Zealand Journal of Agricultural Research 58, 121-130. DOI: 10.1080/00288233.2014.983614.

Homagain, K. , Chander S., Luckai, N., Mahadev S. 2015. Life cycle environmental impact assessment of biochar-based bioenergy production and utilization in Northwestern Ontario,Canada. Journal of Forestry Research, 26, 799-809 .

Hospido, A., Moreira, Ma.T., Martín, M., Rigola, M., Feijoo, G. 2005. Environmental evaluation of different treatment processes for sludge from urban wastewater treatments: Anaerobic digestion versus thermal processes. International Journal of Life Cycle Assessment, 10 (5), 336-345. DOI: 10.1065/lca2005.05.210.

Ibarrola, R., Shackley, S., Hammond, J. 2012. Pyrolysis biochar systems for recovering biodegradable materials: A life cycle carbon assessment. Waste Management, 32 (5), 859-868. DOI: 10.1016/j.wasman.2011.10.005

IBI, 2015. International Biochar Initiative: Standardized Product Definition and Product Testing Guidelines for Biochar That Is Used in Soil https://www.biochar-international.org/wp-content/uploads/2018/04/IBI_Biochar_Standards_V2.1_Final.pdf.

IPCC 2006a. 2006 Guidelines for National Greenhouse Gas Inventories. Volume 5, Chapter 4: Biological treatment of solid waste. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_4_Ch4_Bio_Treat.pdf

IPCC 2006b. 2006 Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use; Chapter 2. Generic methodologies applicable to multiple landuse categories. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_02_Ch2_Generic.pdf

IPCC 2006b. 2006 Guidelines for National Greenhouse Gas Inventories. Chapter 5: Cropland https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_05_Ch5_Cropland.pdf.

IPCC 2006d. 2006 Guidelines for National Greenhouse Gas Inventories. Chapter 3: Solid Waste Disposal. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_3_Ch3_SWDS.pdf.

IPCC 2006e. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 5, Waste. Chapter 5: Incineration and Open Burning of Waste. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_5_Ch5_IOB.pdf

IPCC 2006f. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 2 Energy Chapter 3: Mobile Combustion. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_3_Ch3_Mobile_Combustion.pdf

IPCC 2006g. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 2 Energy. Chapter 2: Stationary Combustion. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_2_Ch2_Stationary_Combustion.pdf

IPCC, 2007. Climate Change: The Physical Science Basis. Working Group I Contribution to the Fourth Assessment Report of the IPCC. Cambridge Univ Press, Cambridge, UK.

IPCC 2019a. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Published: IPCC, Switzerland. Chapter 2 Generic methodologies applicable to multiple landuse categories. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch02_Generic%20Methods.pdf.

IPCC 2019b. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Published: IPCC, Switzerland. Chapter 2 Generic methodologies applicable to multiple landuse categories. Appendix4: Method for Estimating the Change in Mineral Soil Organic Carbon Stocks from Biochar Amendments: Basis for Future Methodological Development. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch02_Ap4_Biochar.pdf.

IPCC 2019c. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Chapter 5: Cropland. Published: IPCC, Switzerland. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch05_Cropland.pdf

IPCC 2019d. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 5: Incineration and Open Burning of Waste. Published: IPCC, Switzerland. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/5_Volume5/19R_V5_5_Ch05_IOB.pdf.

Jeffery, S., Abalos, D., Prodana, M., Bastos, A.C., van Groenigen, J.W., Hungate, B.A., Verheijen, F., 2017. Biochar boosts tropical but not temperate crop yields. Environmental Research Letters Volume: 5 Article Number: 053001.

Jeffery, S., F.G.A. Verheijen, M. van der Velde, A.C. Bastos. 2011. A quantitative review of the effects of biochar application to soils on crop productivity using met-analysis. Agriculture, Ecosystems and Environment 144 (2011): 175-187.

Jones, J., McLaren, S., Chen, Q., Seraj, M. 2020. Repurposing Grape Marc. Research Report to Marlborough District Council. Palmerston North: Massey University. https://www.marlborough.govt.nz/services/recycling-and-resource-recovery/rubbish-and-recycling-projects/grape-marc-grass-and-greenwaste-repurposing-project.

Kammann, C., Ippolito, J., Hagemann, N., Borchard, N., Cayuela, M.L., Estavillo, J.M., Fuertes-Mendizabal, T., Jeffery, S., Kern, J., Novak, J., Rasse, D., Saarnio, S., Schmidt, H.P., Spokas, K., Wrage-Mönnig, N. 2017. Biochar as a tool to reduce the agricultural greenhouse-gas burden–knowns, unknowns and future research needs. J. Environ. Eng. Landsc., 25,114-139.

Karananidi, P., Som, A. M., Loh, S. K., Bachmann, R. T. 2020. Flame curtain pyrolysis of oil palm fronds for potential acidic soil amelioration and climate change mitigation. Journal of Environmental Chemical Engineering, 8(4), 103982.

Keske C., Godfrey T., Hoag D.L.K., Abedin J. 2019. Economic feasibility of biochar and agriculture coproduction from Canadian black spruce forest. Food Energy Secur. https://doi-org.ezproxy.massey.ac.nz/10.1002/fes3.188

Klavina, K., Karklina, K., Blumberga, D. 2015. Charcoal production environmental performance. Agronomy Research, 13 (2), 511-519.

Kumar, A., Adamopoulos, S., Jones, D., Amiandamhen, S. O. 2021. Forest biomass availability and utilisation potential in Sweden: A review. Waste and Biomass Valorization, 12, 65-80.

Laird, D.A. & Chang, C.W. 2013. Long-term impacts of residue harvesting on soil quality. Soil & Tillage Research 134, 33-40.

Lee, M., Lin, Y.-L., Chiueh, P.-T., Den, W. 2020. Environmental and energy assessment of biomass residues to biochar as fuel: A brief review with recommendations for future bioenergy systems. Journal of Cleaner Production, 251, art. no. 119714. DOI: 10.1016/j.jclepro.2019.119714

Lehmann, J., Abiven, S., Kleber, M., Pan, G., Singh, B.P., Sohi S.P, Zimmerman, A.R, Joseph, S. 2015. Persistence of biochar in soil." Biochar for environmental management: Science, technology and implementation 2:233-80.

Leng, R. A., Preston, T. R., Inthapanya, S. 2012. Biochar reduces enteric methane and improves growth and feed conversion in local “Yellow” cattle fed cassava root chips and fresh cassava foliage. Livestock Research for Rural Development, 24: Article#199.

Leng, L.J., Xu, X.W. , Wei, L., Fan, L.L., Huang, H.J., Li, J.A., Lu, Q., Li, J., Zhou, W.G..2019. Biochar stability assessment by incubation and modelling: Methods, drawbacks and recommendations. Science of the Total Environment, 664, 11-2.

Liska A.J., Yang H.S.,Milner M., Goddard S., Blanco-Canqui H., Pelton, M.P., Fang, X.X., Zhu, H.T., Suyker, A.E. 2014. Biofuels from crop residue can reduce soil carbon and increase CO2 emissions. Nature Climate Change 4, 398–401.

Llorach-Massana, P., Lopez-Capel, E., Peña, J., Rieradevall, J., Montero, J.I., Puy, N. 2017. Technical feasibility and carbon footprint of biochar co-production with tomato plant residue. Waste Management, 67, 121-130. DOI: 10.1016/j.wasman.2017.05.021

Lohri, C. R., Rajabu, H. M., Sweeney, D. J., Zurbrugg, C. 2016. Char fuel production in developing countries – a review of urban biowaste carbonisation. Renewable and Sustainable Energy Reviews, 59, 1514-1530.

Mahmud, A.F., Camps-Arbestain, M., Hedley, M., 2018. Investigating the Influence of Biochar Particle Size and Depth of Placement on Nitrous Oxide (N_2O) Emissions from Simulated Urine Patches. Agriculture, 8:175.

Malkow, T. 2004. Novel and innovative pyrolysis and gasification technologies for energy efficient and environmentally sound MSW disposal. Waste Management, 24, 53-79.

Man, K.Y., Chow, K.L., Man, Y.B., Mo, W.Y., Wong, M.H. 2020. Use of biochar as feed supplements for animal farming. Critical Reviews in Environmental Science and Technology. DOI: 10.1080/10643389.2020.1721980.

Matustik, J., Hnatkova, T., Koci, V. 2020. Life cycle assessment of biochar-to-soil systems: A review. Journal of Cleaner Production, 259, Article Number: 120998.

Mayer, P., Hilber, I., Gouliarmou, V., Hale, S.E., Cornelissen, G., Bucheli, T.D., 2016. How to determine the environmental exposure of PAHs originating from biochar. Environmental Science & Technology, 50:1941-1948.

MBIE (Ministry of Business, Innovation & Employment), 2019.Report. Energy in New Zealand – Energy Balances. https://www.mbie.govt.nz/building-and-energy/energy-and-natural-resources/energy-statistics-and-modelling/energy-publications-and-technical-papers/energy-in-new-zealand/, last accessed in November 2020.

MBIE (Ministry of Business, Innovation & Employment), 2020. Quarterly survey of domestic electricity prices. https://www.mbie.govt.nz/building-and-energy/energy-and-natural-resources/energy-statistics-and-modelling/energy-statistics/energy-prices/electricity-cost-and-price-monitoring/, last accessed in November 2020.

Medynska-Juraszek, A., Cwielag-Piasecka, I., Jerzykiewicz, M., Trynda, J. 2020 Wheat Straw Biochar as a Specific Sorbent of Cobalt in Soil. Materials 13, Issue: 11.Article Number: 2462. DOI: 10.3390/ma13112462.

Meier, D., van de Beld, B., Bridgwater, A. V., Elliott, D. C., Oasmaa, A., Preto, F. 2013. State-of-the-art of fast pyrolysis in IEA bioenergy member countries. Renewable and Sustainable Energy Reviews, 20, 619-641.

Mengel K, Kirkby EA. 2001. Principles of plant nutrition. 5th edn. Dordrecht: Kluwer Academic Publishers. 849 pp.

Meyer, S., Genesio, L. , Vogel, I., Schmidt, H.P. , Soja, G., Someus, E , Shackley, S , Verheijen, FGA (Verheijen, F.G. A., Glaser, B. 2017. Biochar standardization and legislation harmonization . Journal of Environmental Engineering And Landscape Management, 25, 175-191.

Meyer S, Glaser B, Quicker P. 2011. Technical, economical, and climate-related aspects of biochar production technologies: A literature review. Environ. Sci. Technol. 45, 9473–9483. MFE 2011a. Ministry for the Environment: Methodology for Deriving Standards for Contaminants in Soil to Protect Human Health. Wellington. https://www.mfe.govt.nz/land/risks-contaminated-land/managing-contaminated-land/contaminated-land-management-guidelines

MFE 2011b. Ministry for the Environment: Resource Management (National Environmental Standards for Air Quality) Amendment Regulations 2011 (SR 2011/103).

MFE 2014. Ministry for the Environment. 2011 User's Guide to the Revised National Environmental Standards for Air Quality.

MFE 2019. Ministry for the Environment: New Zealand's Greenhouse Gas Inventory, 1990-2017. Volume 1, Chapters 1-15. ISSN: 1179-223X. Publication number 1411.

MFE 2020. Ministry for the Environment: Emissions tracker. https://www.mfe.govt.nz/climate-change/state-of-our-atmosphere-and-climate/emissions-tracker.

Miller-Robbie, L., Ulrich, B.A., Ramey, D.F., Spencer, K.S., Herzog, S.P., Cath, T.Y., Stokes, J.R., Higgins, C.P. 2015. Life cycle energy and greenhouse gas assessment of the co-production of biosolids and biochar for land application. Journal of Cleaner Production, 91, 118-127. DOI: 10.1016/j.jclepro.2014.12.050.

Mitchell, H., Mitchell J. 2004. Te Tau Ihu o Te Waka: A History of Maori of Nelson and Marlborough. Vol I. The people and the land. pp51. Huia Publishers with the Wakatu Incorporation.

MPI 2017. Ministry for Primary Industries: A guide to Carbon Look-up Tables for Forestry in the Emissions Trading Scheme https://www.teururakau.govt.nz/dmsdocument/4762-A-guide-to-Look-up-Tables-for-Forestry-in-the-Emissions-Trading-Scheme.

Mostafazadeh, A. K., Solomatnikova, O., Drogui, P., Tyagi, R, D. 2018. A review of recent research and developments in fast pyrolysis and bio-oil upgrading. Biomass Conversion and Biorefinery, 8: 739-773.

Mueller, K., Holmes, A., Deurer, M., Clothier, B.E. 2015. Eco-efficiency as a sustainability measure for kiwifruit production in New Zealand. Journal of cleaner production 106, 333-342.

Muñoz, E., Curaqueo, G., Cea, M., Vera, L., Navia, R. 2017. Environmental hotspots in the life cycle of a biochar-soil system. Journal of Cleaner Production, 158, 1-7. DOI: 10.1016/j.jclepro.2017.04.163

NZWWA (New Zealand Water and Wastes Association). 2003. Guidelines for the safe application of biosolids to land in New Zealand. Wellington, New Zealand Water and Wastes Association. 177 p.

O'Brien, P.L., Sauer, T.J., Archontoulis, S., Karlen, D.L., Laird, D. 2020. Corn stover harvest reduces soil CO_2 fluxes but increases overall C losses. Global Change Biology Bioenergy, DOI:10.1111/gcbb.12742.

Oleszczuk, P., Hale, S.E., Lehmann, J. and Cornelissen, G. 2012. Activated carbon and biochar amendments decrease pore-water concentrations of polycyclic aromatic hydrocarbons (PAHs) in sewage sludge. Bioresource technology, 111, pp.84-91.

Owens,J., Clough, T.J., Laubach,J., Hunt,J.E., Venterea,R.T. 2017.Nitrous Oxide Fluxes and Soil Oxygen Dynamics of Soil Treated with Cow Urine. Soil Sci. Soc. Am. J., 81, 289.

Palviainen, M., Aaltonen, H., Lauren, A., Koster, K., Berninger, F., Ojala, A., Pumpanen, J. 2020. Biochar amendment increases tree growth in nutrient-poor, young Scots pine stands in Finland. Forest Ecology and Management, 474. Article Number: 118362. DOI:10.1016/j.foreco.2020.118362.

Palviainen, M.,Berninger,F.,Bruckman, V.J.,Köster, K.,Moreira,Ribeiro, deAssumpção, C., Aaltonen, H., Makita, N., Mishra, A., Kulmala, L., Adamczyk, B., Zhou, X., Heinonsalo, J., Köster, E., Pumpanen, J., 2018. Effects of biochar on carbon and nitrogen fluxes in boreal forest soil. Plant Soil 425, 71–85.

Paz-Ferreiro, J., Nieto, A., Mendez, A., Askeland, M.P.J., Gasco, G. 2018. Biochar from Biosolids Pyrolysis: A Review. International Journal of Environmental Research and Public Health 15, Article Number: 956. DOI: 10.3390/ijerph15050956.

Perego, C., Bosetti, A. 2011. Biomass to fuels: the role of zeolite and mesoporous materials. Microporous Mesoporous Mater 144:28–39.

Pires de Campos, A. C. 2019. Guidelines for a small scale biochar production system to optimise carbon sequestration outcome. Masters Thesis. Massey University, New Zealand.

Puettmann, M., Sahoo, K., Wilson, K., Oneil, E., 2020. Life cycle assessment of biochar produced from forest residues using portable systems. Journal of Cleaner Production 250, Article Number: 119564, DOI: 10.1016/j.jclepro.2019.119564.

Qi, F.J., Kuppusamy, S., Naidu, R., Bolan, N.S., Ok, Y.S., Lamb, D., Li, Y.B. Yu, L.B. Semple, K.T. Wang, H.L. 2017 Pyrogenic carbon and its role in contaminant immobilization in soils. Critical Reviews in Environmental Science and Technology 47, 795-876. DOI: 10.1080/10643389.2017.1328918.

RMA 2011. Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011. http://www.legislation.govt.nz/regulation/public/2011/0361/latest/DLM4052231.html.

Racek, J., Sevcik, J., Chorazy, T., Kucerik, J., Hlavinek, P. Biochar - Recovery Material from Pyrolysis of Sewage Sludge: A Review. Waste and Biomass Valorization 11, 3677-3709. DOI: 10.1007/s12649-019-00679-w.

Rajkovich, S., Enders, A., Hanley, K., Hyland, C., Zimmerman, A.R., Lehmann, J. 2012. Corn growth and nitrogen nutrition after additions of biochars with varying properties to a temperate soil. Biology and Fertility of Soils, 48, 271– 284.

Ramachandran, S., Yao, Z., You, S., Massier, T., Stimming, U., Wang, C.-H. 2017. Life cycle assessment of a sewage sludge and woody biomass co-gasification system. Energy, 137, pp. 369-376. DOI: 10.1016/j.energy.2017.04.139

Rautiainen, M. Havimo, M., Gruduls, K. 2012. Biocoal production, properties and uses. The Development of the Bioenergy and Industrial Charcoal (Biocoal) Production (Report of BalBiC - Project cb46), 1–28.

Ravindra, K., Sokhi, R., Van Grieken, R., 2008. Atmospheric polycyclic aromatic hydrocarbons: source attribution, emission factors and regulation. Atmospheric Environment, 42(13), pp.2895-2921.

Rayment, G. E. and Higginson, F. R. 1992. Australian Laboratory Handbook of Soil and Water Chemical Methods. Inkata Press, Melbourne

Reumerman, P. J., Frederiks, B.2002. Charcoal production with reduced emissions. 12th European Conference on Biomass for Energy, Industry and Climate Protection, Amsterdam.

Robb, S., Dargusch, P. 2018. A financial analysis and life-cycle carbon emissions assessment of oil palm waste biochar exports from Indonesia for use in Australian broad-acre agriculture. Carbon Management, 9 (2), 105-114. DOI: 10.1080/17583004.2018.1435958

Robb, S., Joseph, S., Aziz, A.A., Dargusch, P., Tisdell, C. 2020. Biochar's cost constraints are overcome in small-scale farming on tropical soils in lower-income countries. Land degradation & development 31, 1713-1726. DOI: 10.1002/ldr.3541

Roberts,K.G.;Gloy,B.A.;Joseph,S.;Scott,N.R.;Lehmann,J. 2010. Life Cycle Assessment of Biochar Systems: Estimating the Energetic, Economic,andCl imate Change Potential.Environ.Sci.Technol. 44, 827–833.

Sanchez Careaga, F.J., Porat, A., Briens, L., Briens, C. 2020. Pyrolysis shaker reactor for the production of biochar. The Canadian Journal of Chemical Engineering 98, 2417-2424 .

Sanchez-Monedero, M.A., Cayuela, M.L., Roig, A., Jindo, K., Mondini, C., Bolan, N. 2018. Role of biochar as an additive in organic waste composting. Bioresource Technology 247, 1155–1164.

Schmidt, H.P., Taylor, P. 2014. Kon-Tiki - the democratization of biochar production. Biochar Journal, Arbaz, Switzerland. ISSN 2297-1114, www.biochar-journal.org/en/ct/39, accessed in November 2020.

Schmitt, N., Apfelbacher, A., Jager, N., Daschner, R., Stenzel, F., Hornung, A. 2019 Thermo-chemical conversion of biomass and upgrading to biofuel: The Thermo-Catalytic Reforming process - A review . Biofuels Bioproducts & Biorefining-Biofpr 13 822-837.

Schimmelpfennig, S., Glaser, B., 2012. One step forward toward characterization: some important material properties to distinguish biochars. J. Environ. Qual. 41, 1001. https://doi.org/10.2134/jeq2011.0146.

Schure, J., Pinta, F., Cerutti, P. O. Kasereka-Muvatsi, L., 2019. Efficiency of charcoal production in Sub-Saharan Africa: Solutions beyond the kiln. Bois et Forets des Tropiques, 340, 57-70.

Seboka, Y. 2009. Charcoal production: Opportunities and barriers for improving efficiency and sustainability. In Bio-carbon Opportunities in Eastern & Southern Africa: Harnessing Carbon Finance to Promote Sustainable Forestry, Agro-Forestry and Bio-Energy. United Nations Development Programme: New York. http://www.environmentportal.in/files/Bio-carbon%20in%20Africa.pdf#page=113, last accessed in February 2021.

SEPA. 2012. A position paper on the Manufacture and use of Biochar from Waste. https://www.sepa.org.uk/media/156613/wst-ps-031-manufacture-and-use-of-biochar-from-waste.pdf.

Severy, M. A., Carter, D. J., Palmer, K. D., Eggink, A. J., Chamberlin, C. E., Jacobson, A. E. 2018. Performance and emissions control of commercial-scale biochar production unit. Applied Engineering in Agriculture, 34(1), 73-84.

Shaheen, S.M., Niazi, N.K., Hassan, N.E.E., Bibi, I., Wang, H.L., Tsang, D.C.W. Ok, Y.S. Bolan, N., Rinklebe, J. 2019. Wood-based biochar for the removal of potentially toxic elements in water and wastewater: a critical review. International Materials Reviews 64, 216-247. DOI: 10.1080/09506608.2018.1473096.

Shen, Q., Hedley, M., Arbestain, M.C. Kirschbaum, M.U.F. 2016. Can biochar increase the bioavailability of phosphorus? Journal of Soil Science and Plant Nutrition 16, 268-286.

Singh, B., Camps Arbestain, M. Lehmann, J.2017. Biochar :A Guide to Analytical Methods. ISBN 9781498765534. Published by CRC Press, Boca Raton, FL, USA, 320 pp.

Singh, S., Kumar, V., Dhanjal, D.S., Datta, S., Bhatia, D., Dhiman, J., Samuel, J., Prasad, R., Singh, J. 2020. A sustainable paradigm of sewage sludge biochar: Valorization, opportunities, challenges and future prospects. Journal of Cleaner Production, 269, art. no. 122259. DOI: 10.1016/j.jclepro.2020.122259

Smith, P., Adams, J., Beerling, D.J., Beringer, T., Calvin, K.V., Fuss, S ., Griscom, B., Hagemann, N., Kammann, C., Kraxner, F., Minx, J.C., Popp, A., Renforth, P., Vicente, J.L.V., Keesstra, S. 2019.

Land-Management Options for Greenhouse Gas Removal and Their Impacts on Ecosystem Services and the Sustainable Development Goals. In Annual Review of Environment and Resources, (Eds.Gadgil, A., Tomich, T.P). Vol. 44 Pages: 255-286.

Sparrevik M, Adam C, Martinsen V, Cornelissen G. 2015. Emissions of gases and particles from charcoal/biochar production in rural areas using medium-sized traditional and improved “retort” kilns. Biomass Bioenergy 72, 65-73.

Srinivasan, P., Sarmah, A.K., Smernik, R., Das, O., Farid, M., Gao, W. 2015. A feasibility study of agricultural and sewage biomass as biochar, bioenergy and biocomposite feedstock: Production, characterization and potential applications. Science of The Total Environment, 512, 495-505. DOI: 10.1016/j.scitotenv.2015.01.068.

Stassen, H.E. 2015. Current issues in charcoal production and use. in van Swaaij, W., Kersten, S., Palz, W. (Eds.) Biomass Power for the World. Pan Stanford Publishing, Boca Raton.

Struhs, E., Mirkouei, A., You, Y., Mohajeri, A. 2020. Techno-economic and environmental assessments for nutrient-rich biochar production from cattle manure: A case study in Idaho, USA. Applied Energy, 279, art. no. 115782. DOI: 10.1016/j.apenergy.2020.115782

Szymkowski, C.J., Bultitude-Paull, J.M. The production of high-quality silicon metal at Simcoa. INFACON 6, Proceedings of the 6th International Ferroalloys Congress, Cape Town, Vol. 1, Johannesburg, SAIMM, 1992, 185-191.

Taghizadeh-Toosi, A., Clough, T.J., Condron, L.M., Sherlock, R.R., Anderson, C.R., Craigie, R.A. 2011 Biochar Incorporation into Pasture Soil Suppresses in situ Nitrous Oxide Emissions from Ruminant Urine Patches. Journal of Environmental Quality 40, 468-476. DOI: 10.2134/jeq2010.0419.

Thers, H., Djomo, S.N., Elsgaard, L., Knudsen, M.T. 2019. Biochar potentially mitigates greenhouse gas emissions from cultivation of oilseed rape for biodiesel. Science of the Total Environment, 671, 180-188. DOI: 10.1016/j.scitotenv.2019.03.257

Thies J., Rillig M.C. 2009. Characteristics of biochar: biological properties. In: Lehmann J, Joseph A (eds) Biochar for environmental management: science and technology. Earthscan, London.

Tisserant A. Cherubini F. 2019. Potentials, Limitations, Co-Benefits, and Trade-Offs of Biochar Applications to Soils for Climate Change Mitigation. Land 2019, 8, 179; doi:10.3390/land8120179.

Treweek, G., Di, H.J., Cameron K.C., Podolyan, A. 2016. Effectiveness of the nitrification inhibitor dicyandiamide and biochar to reduce nitrous oxide emissions, New Zealand Journal of Agricultural Research, 59:2, 165-173, DOI: 10.1080/00288233.2016.1161651.

Tumbure,A., Bishop, P. A., Bretherton, M.R., Hedley M.J. 2020 Co-Pyrolysis of Maize Stover and Igneous Phosphate Rock to Produce Potential Biochar-Based Phosphate Fertilizer with Improved Carbon Retention and Liming Value. ACS Sustainable Chem. Eng. 8, 4178–4184.

van Schalkwyk, D.L. , Mandegari, M., Farzad, S., Gorgens, J.F. 2020. Techno-economic and environmental analysis of bio-oil production from forest residues via non-catalytic and catalytic pyrolysis processes. Energy conversion and Management 213, Article Number: 112815.DOI: 10.1016/j.enconman.2020.112815

Verheijen, F.G.A., Jeffery, S.A., Bastos, C., van der Velde, M., Diafas I. 2009. Biochar Application to Soils – A Critical Scientific Review of Effects on Soil properties, Processes and Functions. EUR 24099 EN. Office for the Official Publications of the European Communities, Luxembourg (2009). 149pp. accessed via Google Scholar.

Wang, C. Wang, Y. Herath, H.M.S.K. 2017. Polycyclic aromatic hydrocarbons (PAHs) in biochar – Their formation, occurrence and analysis: A review, Org. Geochem. 114, 1–11. https://doi.org/10.1016/j.orggeochem.2017.09.001.

Wang, J., Xiong, Z., Kuzyakov, Y. 2016. Biochar stability in soil: Meta-analysis of decomposition and priming effects. GCB Bioenergy, 8 (3), 512-523. DOI: 10.1111/gcbb.12266

Wang, T., Camps-Arbestain, M., Hedley M.J., Bishop P.A. 2012a. Predicting phosphorus bioavailability from high-ash biochars. Plant & Soil 357, 173–187.

Wang, T., Camps-Arbestain, M., Hedley M.J., Bishop P.A. 2012b. Chemical and bioassay characterisation of nitrogen availability in biochar produced from dairy manure and biosolids. Organic Geochemistry 51, 45-54.

Wang, T., Camps-Arbestain, M. and Hedley, M.J. 2013. Predicting C aromaticity of biochars based on their elemental composition. Organic Geochemistry 62, 1–6.

WasteMINZ, 2018. New Zealand Food Waste Audits. https://lovefoodhatewaste.co.nz/wp-content/uploads/2019/02/Final-New-Zealand-Food-Waste-Audits-2018.pdf, last accessed in Oct 2020.

Wilson, K. 2017. Converting Shelterbelt Biomass to Biochar: A feasibility analysis by Wilson Biochar Associates for North Dakota Forest Service. North Dakota State University – North Dakota Forest Service, Bismarck, North Dakota, February 10, 2017. https://www.ag.ndsu.edu/ndfs/documents/wba-converting-shelterbelt-to-biochar.pdf.

Wisnubroto, E. I. 2015. Investigation on the effect of biochar addition and the use of pasture species with different rooting systems on soil fertility and carbon storage : a thesis presented in partial fulfilment of the requirements for the degree of Master of Philosophy (MPhil) in Soil Science at the Institute of Agriculture and Environment, Massey University, Palmerston North, New Zealand. http://hdl.handle.net/10179/7180.

Woolf, D., Amonette, J.E., Street-Perrott, F.A., Lehmann, J. and Joseph, S. 2010. Sustainable biochar to mitigate global climate change. Nature Communications 1, 56. https://doi.org/10.1038/ncomms1053.

Woolf, D., Lehmann,J., Cowie,A., Cayuela,M., Whitman,T., Sohi,S. 2018. Biochar for Climate Change Mitigation Navigating from Science to Evidence-Based Policy. Chapter 8, p218-249. In Lal,R and Stewart B. A. (eds.) Soil and Climate. Pp. 448. CRC Press. Accessed https://cpb-us-e1.wpmucdn.com/blogs.cornell.edu/dist/9/7784/files/2018/11/20181108-Biochar-for-Climate-Change-Mitigation-Woolf_et_al-2mqma9i.pdf

Wrobel-Tobiszewska, A., Boersma, M., Sargison, J., Adams, P., & Jarick, S. (2015). An economic analysis of biochar production using residues from Eucalypt plantations. Biomass and Bioenergy, 81, 177– 182. https://doi.org/10.1016/j.biombioe.2015.06.015

Ye, LL; Camps-Arbestain, M; Shen, QH; Lehmann, J; Singh, B; Sabir, M. 2020. Biochar effects on crop yields with and without fertilizer: A meta-analysis of field studies using separate controls. Soil Use and Management, 36, 2-18 .

Yi, S.S., Chang, N.Y. Imhoff, P.T. 2020. Predicting water retention of biochar-amended soil from independent measurements of biochar and soil properties. Advances In Water Resources 142, Article Number: 103638. DOI: 10.1016/j.advwatres.2020.103638.

Yuan, H., Zhang, Z., Li, M., Clough, T., Wrage-Mönnig, N., Qin, S., Ge, T., Liao, H. and Zhou, S. 2019. Biochar's role as an electron shuttle for mediating soil N2O emissions. Soil Biology and Biochemistry, 133, pp.94-96.

Zhang, J.B., Li, Q., Wu, J.S., Song, X.Z. 2019. Effects of nitrogen deposition and biochar amendment on soil respiration in a Torreya grandis orchard. Geoderma, 355 Article Number: 113918 DOI: 10.1016/j.geoderma.2019.113918.

Appendix A: Description of the role of macro- and micropores in soil water retention and drainage.

This discussion provides supporting material for Section 2.13 Soil conditioner (forest, arable crop and horticultural residues) in the main report.

Water retention, drainage and aeration of porous systems, such as soils and biochar, are governed by the diameter and number of pores (pore-size distribution). Large macropores >30 µm in diameter and contiguous throughout the soil profile are responsible for drainage and aeration of a soil. These large pores are the gaps between soil particles, sand size and larger, or large aggregates (aggregates are made up of smaller silt and clay sized soil particles bonded together by surface chemical reactions of metal oxides and soil organic matter). Even larger pores are also created biologically by burrowing soil organisms and plant roots. Large macropores, > 30 µm in diameter, allow water to freely drain (Figure A.1) by gravity at normal atmospheric pressure leaving the soil at field capacity (FC). Clay-dominant soils will have greater pore space in the range < 30 µm in diameter than a sand-dominant soil and therefore will have a higher water content at FC.

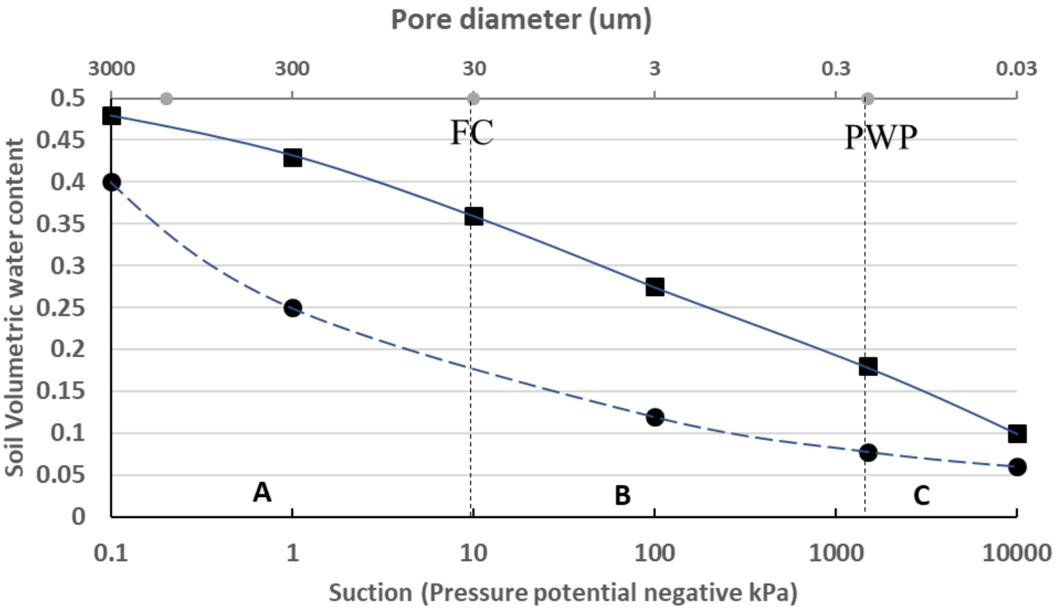


Figure A.1. The moisture release curves of a clay dominant soil (■) and a sand dominant soil (●) under different negative pressure potential (suction) and indicating the diameter of pores that will have drained.

Plant available water content (PAWC) is the difference in soil water content between a recently wetted soil at field capacity (FC, Figure A.1) that has just drained via pores >30 µm in diameter and a soil that is at permanent wilting point (PWP). Most agricultural plant roots can no longer extract water from soil at PWP. At FC, water is held by capillarity in medium to small pores < 30 µm in diameter (Figure A.1) and, at PWP, plants cannot extract water from that remaining in pores < 0.2 µm in diameter (Figure A.1). The ability of soils to form pores within this size range is a function of the soil’s total internal surface area and reactivity, which is mostly a function of soil texture, mineralogy and organic matter content. Soils dominated by sand-sized particles (sands and sandy loams) have relatively low surface area, and low PAWC, whereas silt loam and clay loams have higher more reactive surface areas and a higher PAWC.

Appendix B: Description of biochar properties and associated test methods used to classify and characterise biochar.

Table B.1 provides supporting material for the discussion in Section 2.3. Biochar characteristics, certification and use. in the main report.

Table B.1. List of properties and associated test methods that satisfy the criteria for biochar as defined by either the IBI Biochar Standards or the EBC Standards (Camps-Arbestain et al., 2015).

		European Biochar Certificate	EBC Test Method	IBI Guidelines	IBI Test Method
C content	**Requirement**	Total C	Total C , H, N analysis by dry combustion IR-detection (DIN 51732, ISO 29541). Inorganic C analysis by determination of carbonate-C content with HCl, as outlined inDIN 51726, ISO 925. Organic C calculated as total C minus carbonate C	Organic C (C_{org})	Total C and H analysis by dry combustion-IR detection. Inorganic C analysis by determination of carbonate-C content with HCl, as outlined in ASTM D4373-02. Organic C calculated as Total C – Inorganic C.
	Criteria	- ≥ 50% → **Biochar** - < 50% → **Biocarbon minerals (BCM)**		-≥ 60% → **Class 1** -≥ 30% - < 60% → **Class 2** -≥ 10% - < 30% → **Class 3** -< 10 % **NOT** classified as biochar	
Molar H/C ratio	**Requirement**	Molar H/C_{org} ratio	Molar ratio	Molar H/C_{org} ratio	Molar ratio
	Criteria	- < 0.7 → **Biochar** - ≥ 0.7 → **Not considered biochar**		- ≤ 0.7 → **Biochar** - >0.7 → **Not considered biochar**	
Total Ash	**Requirement**	Total ash	DIN 51719, 550 °C, ISO 1171 (or EN 14775)	Total ash	ASTM D1762-84
	Criteria	Declaration		Declaration	
Molar O/C ratio	**Requirement**	Molar O/C ratio	O calculated from ash content, C, H, N, S (DIN 51733, ISO 17247)	None	N/A
	Criteria	- < 0.4 → **Biochar**			

		European Biochar Certificate	EBC Test Method	IBI Guidelines	IBI Test Method
Main Elements / nutrients	**Requirement**	Total N, P, K, Mg, S, Na, Si, Fe, Mn, Ca content	**Total N**: Dry combustion-IR detection following the same procedure for total C and H (DIN 51732) **Other elements:** Digestion with Litium metaborate on ash 550 °C according DIN 51 729-11 and determination with ICP-OES according DIN EN ISO 11885 (DIN EN ISO 17294-2 is also possible – ICP-MS)	Total N required for **Test Category A** (compulsory) Total P and K required for **Test Category C** (optional) Mineral N (ammonium and nitrate) required for **Test Category C** (optional) Available P required for **Test Category C** (optional) Liming equivalence (if pH > 7) required for **Test Category A** (compulsory)	**Total N**: Dry combustion-IR detection following the same procedure for total C and H **Total P and K**: Modified dry ashing followed by ICP (Enders and Lehmann 2012) **Mineral N**: 2M KCl extraction, followed by spectrophotometry (Rayment and Higginson 1992) **Available P**: 2% formic acid followed by spectrophotometry as described by Wang et al (2012) after Rajan et al (1992) and AOAC (2005) **Liming equivalence**: Rayment & Higginson (1992)
	Criteria	Declaration		Declaration	
Heavy metals, metalloids and other elements	**Requirement**	Heavy metals: Pb, Cd, Cu, Ni, Hg, Zn, Cr	**All metals/metalloids**: microwave acid digestion with HF/HNO_3 and determination of the metals with ICP-MS (DIN EN ISO 17294-2) The determination of Hg is also possible with H-AAS according DIN EN 1483	Heavy metals: Pb, Cd, Cu, Ni, Hg, Zn, Cr, Co, Mo Metalloids: B, As, Se, Others: Cl, Na	**All metals/metalloids except Hg**: US Composting Council TMECC Sections 04.05 and 04.06 (USCC & USDA, 2001) **Hg**: US EPA 7471 (2007)
	Criteria	Two grades have been established: (i) **Basic Quality Grade** (following the Germany's Federal Soil Protection Act or BBodSchV)[1], and (ii) the **Premium Quality Grade** (Switzerland's Chemical		IBI Guidelines indicate in the disclaimer that it is the responsibility of the user of the Guidelines to determine the applicability of any national, state or provincial, and local regulatory limitations prior to use.	

Risk Reduction Act or ChemRRV[2]) on recycling fertilisers. Maximum threshold values are reported in table below.

	Basic	Premium
	(g t^{-1} = mg kg^{-1})	
Pb	150	120
Cd	1.5	1
Cu	100	100
Ni	50	30
Hg	1	1
Zn	400	400
Cr	90	80

Abrasion in connection with the use of Cr-Ni steels in the construction of pyrolysis reactors may lead, especially in the first weeks of production, to an increased Ni contamination of biochar. An exemption can be granted for biochars with a Ni contamination < 100g t^{-1}. Such biochars shall only to be used for composting purposes since the valid threshold are complied with in the finished compost.

[1] Deutsche Bundes-Bodenschutz – und Altlastenverordnung (BBodSchV) (1999). Latest amendment 31.7.2009.

[2] Schweizerische Chemikalien-Risikoreduktions-Verordnung (ChemRRV, SR 814.81) (2005)

Maximum threshold values are given as a range of values (in mg kg^{-1})based on standards for soil amendments or fertilisers from a number of jurisdictions: EU (A)[1], Australia (B)[2], Canada (C)[3], USA (D)[4] and Quebec (E)[5]. These entities were chosen as standards because they all have a long history of regulations addressing these toxicants in soils and other substrates.

Element	Symbol	A	B	C	D	E	range
Arsenic	As		100		41	13	13-100
Cadmium	Cd	1.4	20		39	3	1.4-39
Chromium	Cr	93	100		1200	210	93-1200
Cobalt	Co		100			34	34-100
Copper	Cu	143	1000		1500	400	143-1500
Lead	Pb	121	300		300	150	121-300
Mercury	Hg	1	10[1] 15[2]		17	0.8	0.8-17
Molybdenum	Mo			5	75	5	5-75
Nickel	Ni	47	600		420	62	47-600
Selenium	Se				36	2	2-36
Zinc	Zn	416	7000		2800	700	416-2800

[1] Methyl mercury
[2] Inorganic mercury

B, Cl and Na: Declaration

[1] http://www.bvsde.paho.org/bvsacd/cd43/used.pdf (accessed January 2012).

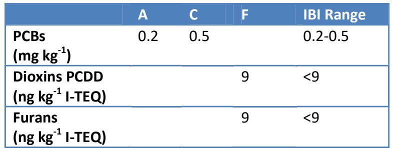
[2] www.ephc.gov.au/contam (accessed January 2012).

[3] http://st-ts.ccme.ca (accessed January 2012).

[4] http://water.epa.gov/scitech/wastetech/biosolids/503pe_index.cfm (accessed January 2012).

[5] http://www-es.criq.qc.ca/pls/owa_es/bnqw_norme.detail_norme?p_lang=en&p_id_norm=8184&p_code_menu=NORME (accessed January 2012).

		European Biochar Certificate	EBC Test Method	IBI Guidelines	IBI Test Method
pH	**Requirement**	pH	according DIN ISO 10390 with $CaCl_2$-solution	pH	US Composting Council TMECC Section 04.11, following dilution and sample equilibration methods from Rajkovich et al (2011)
	Criteria	Declaration If > 10, the delivery slip must feature appropriate handling information		Declaration	
Bulk density and water content	**Requirement**	Bulk density and water content	**Bulk density:** DIN 51705 **Water content**: DIN 51718 (106°C)	Water content (referred to as moisture in IBI Biochar Standards)	ASTM D1762-84
	Criteria	Declaration		Declaration	
Surface area	**Requirement**	Specific surface area (BET)	DIN 66132/ISO 9277	Total surface area and external surface area (BET)	ASTM D 6556-10
	Criteria	Declaration, but preferably higher than 150m^2 g^{-1}		Declaration (optional as Test Category C)	
Water holding capacity	**Requirement**	Water holding capacity	Water holding capacity determining by soaking and drying a sample (E DIN ISO 14238) WHC calculated as mass percentage of saturated and dry mass	None	N/A
	Criteria	Optional			
PAH content	**Requirement**	PAH content	Soxhlet-extraction with toluene and determination with GC-MS according DIN EN 15527 or Soxhlet-extraction with toluene and determination according DIN ISO 13877 with HPLC or Soxhlet-extraction with toluene und determination with GC-MS according DIN CEN/TS 16181	PAH content	US EPA 8270 (2007) or US EPA 8275 (1996)
	Criteria	Basic grade <12mg kg^{-1} Premium grade < 4mg kg^{-1}		6-20mg kg^{-1} 6 (A) and 20 (B)	

<table>
<tr><th></th><th></th><th>European Biochar Certificate</th><th>EBC Test Method</th><th>IBI Guidelines</th><th>IBI Test Method</th></tr>
<tr><td rowspan="3">PCBs, dioxins and furans</td><td>Requirement</td><td>PCBs, dioxins, furans</td><td rowspan="3">Soxhlet-extraction with toluene and determination with HRGC-HRMS based on US EPA 8290 (2007-02)</td><td>PCBs, dioxins, furans</td><td rowspan="3">PCBs: US 8082 (2007) or US EPA 8275 (1996)
PCDD/Fs: US EPA 8290 (2007)</td></tr>
<tr><td>Criteria</td><td>PCBs < 0.2mg kg^{-1}
Dioxins < 20ng kg^{-1} (I-TEQ OMS)
Furans < 20ng kg^{-1} (I-TEQ OMS)</td><td>
<table>
<tr><th></th><th>A</th><th>C</th><th>F</th><th>IBI Range</th></tr>
<tr><td>PCBs (mg kg^{-1})</td><td>0.2</td><td>0.5</td><td></td><td>0.2-0.5</td></tr>
<tr><td>Dioxins PCDD (ng kg^{-1} I-TEQ)</td><td></td><td></td><td>9</td><td><9</td></tr>
<tr><td>Furans (ng kg^{-1} I-TEQ)</td><td></td><td></td><td>9</td><td><9</td></tr>
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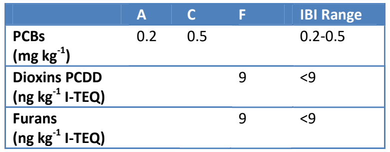
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</table></td></tr>
<tr><td>Criteria</td><td>Declaration</td><td>Declaration</td></tr>
<tr><td rowspan="2">Electrical conductivity</td><td>Requirement</td><td>Electrical conductivity</td><td rowspan="2">Method of the BGK (Federal quality community compost), volume 1, method III. C2 in analogy to DIN ISO 11265:</td><td>Electrical conductivity</td><td rowspan="2">US Composting Council TMECC Section 04.10, following dilution and sample equilibration methods from Rajkovich et al (2011)</td></tr>
<tr><td>Criteria</td><td>Declaration</td><td>Declaration</td></tr>
<tr><td rowspan="2">Particle size distribution</td><td>Requirement</td><td>None</td><td rowspan="2">N/A</td><td>Particle size distribution</td><td rowspan="2">Progressive dry sieving with 50mm, 25mm, 16mm, 8mm, 4mm, 2mm, 1mm, and 0.5mm sieves.</td></tr>
<tr><td>Criteria</td><td></td><td>Declaration</td></tr>
<tr><td rowspan="2">Germination inhibition</td><td>Requirement</td><td>None</td><td rowspan="2">N/A</td><td>Germination inhibition</td><td rowspan="2">OECD methodology (1984) using three test species, as described by Van Zwieten et al (2010).</td></tr>
<tr><td>Criteria</td><td></td><td>Pass/Fail</td></tr>
<tr><td>Volatile matter</td><td>Requirement</td><td>Thermogravimetry</td><td>LECO-method</td><td>Volatile matter</td><td>ASTM D1762-84</td></tr>
</table>

Appendix C. Example calculation of the cost of using a flame curtain kiln to produce biochar from kiwi fruit orchard prunings

Oregon Flame curtain Kiln			
Pyrolyser Capital cost (Oregon bin, 6 m3)	$2,500		
Useful life (y)	5		
Interest rate (%)	6		
Principle and Interest (Repay/y)	$593.49		
Working days per year	53		
Annual Share of capital cost (310 days in working year)	$101.47		
Labour operating cost (assumption time share with other orchard duties)			
Labour $/h	$25		
Labour to collect prunings and deliver to pyrolyis site (h/tFS)	1		
Total Hours to deliver prunings to pyrolyser (h)	53		
Total labour cost to deliver prunings to pyrolyser ($)	$1,325		
Pyrolyser operation Number of batches per Orchard	53.0		
Working hours to load feedstock in kiln/batch	0.5		
Working hours to supervise pyrolysis/batch	1		
Working hours to quench kiln/batch	0.5		
Working hours to empty kiln and store biochar/batch	0.5		
Working hours (per batch)	2.5		
Daily routine Kiln time (Empty 8 am)			
Batches per day	1		
Number of operating days /year	53.0		
Person Cost per Batch	$63		
Total labour cost for operating Kiln	$3,313		
Total Labour cost per Orchard	$4,638		
Pyrolyser Maintenance costs (assumes orchard tractor with loader and trailer used)			
Maintenance cost as % of Capex	4		
Maintenance cost per annum	$4		
Number of pyrolyser movements/y	1		
Cost of pyrolyser transport per move	$200		
Annual cost of moving pyrolyser	$200		
Machine hire to feed/empty pyrolyser per h	$0		
Hours of feeding /emptying pyrolyser	53.0		
Annual cost of machine cost to feed/empty pyrolyser	$0.0		
Total Maintenance costs/orchard	$204		
Fuel costs			
Cost diesel per litre	$0.95		
Diesel used to collect prunings (l/t)	0.53		
Cost diesel for collecting prunings	$26.69		
Diesel Used per batch of pyrolyser fill and empty (l) (Tractor with front loader)	0.53		
Cost diesel for filling and emptying pyrolyser per Orchard	$27		
Chain saw fuel per tonne feedstock (l/t FS)	1.9		
Petrol cost ($/liter)	$1.94		
Chain saw 2 stroke oil (1:40) (l/tFS)	0.0475		
Two stroke oil ($/Litre)	$11.40		
Chain bar oil (l/tFS)	0.396		
Chain bar oil ($/l)	$8.73		
Total fuel cost Orchard	$461		
Summary of Costs - biochar Capital and operating)			
Annual finance cost of pyrolyser	$101		
Annual Labour per Orchard	$4,638		
Annual Maintenance cost of pyrolyser	$204		
Annual Fuel costs	$461		
Total Annual cost per Orchard	**$5,404**	**Cost t biochar**	**$1,019.56**
Cost of CO_2e (corrected for BC +100 and emissions during pyrolyisis)		**Cost t CO_2 e**	**$777.61**

Kiwifruit orchard prunings and biochar quantities			
Number of ha in Orchard (ha)	10		
Tonnes of wood processed per ha	5.3		
Tonnes of wood processed per Orchard	53		
Average Batch size (t)	1		
Number of batches	53		
Number tonnes processed/day	1		
Number tonnes processed/y	53		
	per t FS	**per day**	**per t BC**
Biochar(t) produced per tonne of wood	0.1	0.1	1
Carbon fraction	0.774		0.774
Tonnes C	0.077	0.077	0.774
Tonnes BC+100 (70%)	0.054	0.054	0.542
Tonnes of CO_2 per tonne Biochar C (coor. BC+100)	0.20	0.199	1.99
	per orchard		per t BC
Tonnes of CO_2 Biochar per orchard (coor. BC+100)/53tFS	10.53		
Pyrolysis non CO_2 emissions as fraction of Biochar CO_2e	0.34		
Tonnes CO_2e credit per orchard (minus non-CO_2 emissions)	6.95		1.31
Tonnes of biochar in Orchard per ha/y	0.53		
Tonnes of biochar made in Orchard /y	5.30		