Biochar systems for carbon finance – an evaluation based on Life Cycle Assessment studies in New Zealand

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Abstract

Char produced from the pyrolysis of biomass and applied into soils (biochar) can, under some conditions, improve soil functions and sequester carbon (C) over millennia. In New Zealand, if 80% of the available biomass residues were converted into biochar, about 1.7 Mt CO_2 could be sequestered annually. This represents ~2.4% of NZ's total annual greenhouse gas (GHG) emissions. However, the trade-offs associated with alternative uses of biomass need to be assessed from a life cycle perspective, particularly when considering policymaking.

The biomass feedstocks evaluated using Life Cycle Assessment were orchard prunings, logging residues, and wheat straw. The goals were i) to compare alternative management scenarios and ii) to determine the use of biomass that can achieve the largest amount of carbon credits in order to support policymaking. The biomass for heat-only (HO) scenario could mitigate 276 – 1,064 kg CO_2-eq per t biomass; the combined heat and power (CHP) scenario could reduce 410 – 1,608 kg CO_2-eq per t biomass; and the biochar scenario could abate 271 – 792 kg CO_2-eq per t biomass. Ranges vary according to the type of feedstock assessed and the type of fossil fuel (coal or natural gas) displaced. The assessment of the HO and CHP systems giving greater GHG emission reductions than the biochar system can be misleading as these only involve fossil-fuel offsetting whereas the biochar system would sequester some carbon irrespective of the other activities assumed to be displaced. The biochar carbon stability factor is the key component that affects its capacity to mitigate climate change. A distinctive C accounting, reporting and crediting approach should be developed for biochar to have high economic potential in carbon-pricing mechanisms.

Several approaches for incentivising biochar carbon sequestration were explored. These include using conservative carbon-accounting estimates, issuing temporary credits, establishing buffer funds, creating carbon credit multipliers, and inventing a new unit such as ppm CO_2 reductions for recognising atmospheric CO_2 removals as opposed to avoiding GHG emissions. While biochar technology is currently facing numerous barriers for acceptance in carbon markets, its future is promising since biochar production also offers potential in the agriculture, energy and waste management sectors.

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Abbreviations and acronyms

AAU	assigned amount unit
ADE	Anthropogenic Dark Earths
AIJ	activities implemented jointly
ALCA	attributional Life Cycle Assessment
AOS	Alberta Offset System
A/R	afforestation/reforestation
BAU	business as usual
BCSF	biochar carbon stability factor
BECCS	bioenergy with carbon capture and storage
BMF	biochar migration factor
BP	before present
C	carbon
C_{org}	organic carbon
Ca	calcium
CAN	calcium ammonium nitrate
CCS	carbon capture and storage
CDM	clean development mechanism
CEC	cation exchange capacity
CER	certified emission reduction
CF	carbon footprint
CH_4	methane
CHP	combined heat and power
CLCA	consequential Life Cycle Assessment
CMF	carbon maintenance fee
CMS	Carbon Market Solutions
CNI	Central North Island
CO	carbon monoxide
CO_2	carbon dioxide
CO_2-eq	carbon dioxide equivalent

COP	conference of the parties
CPY	central processing yard
CROPS	crop residue oceanic permanent sequestration
EECA	Energy Efficiency and Conservation Authority
ELB	end-of-life biomass
ERU	emission reduction unit
ET	emissions trading
ETS	emissions trading scheme
EU	European Union
FAO	Food and Agriculture Organisation
FSC	Forest Stewardship Council
GHG	greenhouse gas
GJ	gigajoule
GWP	global warming potential
H_2	hydrogen
H_2O	water
ha	hectare
HB	Hawke’s Bay
HO	heat only
HTC	hydrothermal carbonisation
HWP	harvest wood products
IBI	International Biochar Initiative
IEA	International Energy Agency
ILCD	International Reference Life Cycle Data System
IMF	International Monetary Fund
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organisation for Standardisation
JI	joint implementation
K	potassium
KCl	muriate of potash
kWh	kilowatt hour

LCA	Life Cycle Assessment
LCI	life cycle inventory
LCIA	life cycle impact assessment
LUCAS	Land Use and Carbon Analysis System
LULUCF	land-use, land-use change and forestry
LPG	liquefied petroleum gas
MAF	Ministry of Agriculture and Forestry
MBIE	Ministry of Business, Innovation and Employment
MfE	Ministry for the Environment
Mg	magnesium
MJ	megajoule
MPI	Ministry for Primary Industries
MSC	Marine Stewardship Council
MSDS	material safety data sheet
MSW	municipal solid waste
Mt	megatonne
MW	megawatt
N	nitrogen
N_2O	nitrous oxide
NER300	New Entrants' Reserve
NETs	negative emission technologies
NZ	New Zealand
NZAGRC	New Zealand Agricultural Greenhouse Gas Research Centre
NZFOA	New Zealand Forest Owners Association
O-I	Owens-Illinois
P	potassium
PAHs	polyaromatic hydrocarbons
PCRs	Product Category Rules
PET	polyethylene terephthalate
PETRA	PET Resin Association
PJ	petajoule

ppm	parts per million
REDD	reduced emissions from deforestation and forest degradation
RSB	Roundtable on Sustainable Biomaterials
S	sulphur
SEM	scanning electron microscopy
SETAC	Society of Environmental Toxicology and Chemistry
SO_2	sulphur dioxide
SOC	soil organic carbon
SOM	soil organic matter
SP	superphosphate
t	tonne
TRV	total recoverable volume
TP	terra preta
UK	United Kingdom
UN	United Nations
UNEP	United Nations Environment Program
UNFCCC	United Nations Framework Convention on Climate Change
USA	United States of America
VCS	Verified Carbon Standard
VMRV	validation, monitoring, reporting and verification
WFPS	water filled pore space
WHS	wood harvest and storage
WWF	World Wildlife Fund
Zn	zinc

Table of contents

List of figures

List of tables

CHAPTER 1: INTRODUCTION

The extraction of fossil fuels (coal, oil, and gas) from the Earth's crust and their subsequent combustion has increased drastically the levels of carbon (C) in the atmosphere (IPCC, 2013). The global atmospheric concentration of carbon dioxide (CO_2) has increased from a pre-industrial level of approximately 280 parts per million (ppm) to about 400 ppm in May 2013 (Williamson, 2013a). This human interference with natural systems is *extremely likely* to be the main cause of climate change, whose impacts could be distressing for the environment and society in general (IPCC, 2013). In order to stabilise atmospheric CO_2 concentrations, emissions would need to peak and decline thereafter. A series of mitigation scenarios were developed for different long-term stabilisation targets in the Fourth Assessment Report (AR4) of the Intergovernmental Panel on Climate Change (IPCC, 2007c). Besides reducing emissions, the removal of a portion of atmospheric CO_2 (negative emissions) is required to happen before the end of this century if the lower (and safer) stabilisation levels of 350-450 ppm are to be achieved and temperature rise constrained below $2^{o}C$. The mitigation scenarios of the AR5 will be published in 2014.

A scientific consensus to set greenhouse gas (GHG) pollution limits on the economies of the most polluting countries and industries has been reached. The Kyoto Protocol, the first international treaty giving birth to the carbon markets, was signed in 1997 and entered into force in 2005. It was developed for industrialised countries (defined as Annex I countries in the protocol) to reduce their GHG emissions by agreed targets. The USA, which initially fostered the replacement of the proposed judicial system of pecuniary penalties with market-based mechanisms (Werksman, 1998) withdrew from negotiations and refused to ratify the treaty. The rest of the Annex I countries were legally bound to reduce their GHG emissions by an average of five per cent against 1990 levels between 2008 and 2012. Now that this 'first commitment period' has ended, the future replacement of the Kyoto treaty is uncertain but will probably continue out to 2020. However, several signatories including Japan, Canada, Russia and New Zealand have pulled out (Metz, 2013) and only around 11% of total global emissions are now covered by the treaty.

New Zealand (NZ) was a signatory to the Kyoto Protocol. Although GHG emissions of NZ represented only 0.14% of the global GHG footprint in 2010, it was ranked as the fifth highest polluter among Annex I countries based on emissions per capita (MfE, 2013). The NZ government developed an emissions trading scheme (ETS) to help the country meet its Kyoto commitments, which was to involve all sectors of the economy and all GHGs. Due to political and economic pressures, the dates at which sectors were/are included in the ETS has slowed and for "agriculture" has been suspended indefinitely.

Agriculture is the most significant sector to the NZ economy and, uniquely for a "developed" country, is responsible for almost half of NZ's total GHG emissions (MfE, 2013). For this reason, ways to reduce NZ agricultural emissions have become the focus of a lot of research. One of the mechanisms is to balance emissions with sequestration in indigenous forests and biomass plantations. However, such sequestration is temporary until the forests become mature after which the sequestration and decay rates reach equilibrium. In contrast, a technology called biochar offers a route to continuously store carbon for longer periods of time by carbonising a fraction of the harvested residue and incorporating it into the soil.

Biochar is carbonised biomass obtained from sustainable sources and sequestered in soils to sustainably enhance their agricultural and environmental value under present and future management. Due to its relatively high stability, biochar has been suggested as a carbon-negative strategy to mitigate climate change (Hansen *et al*., 2008). The full potential climate change effects of biochar, however, need further research before they are included in carbon markets and before there is large-scale deployment of biochar systems.

Because of its porous nature, biochar has the potential to retain water, nutrients, and pollutants commonly found in soils. Depending on the application, the effects of biochar could result in soil remediation, waterways protection, increased crop production, fertiliser savings, carbon sequestration in soils and renewable energy generation (Lehmann and Joseph, 2009). The production and use of biochar also offer several opportunities in the energy and waste management sectors.

Biochar is produced by heating biomass in the total or partial absence of oxygen. Pyrolysis is the technology most commonly employed for producing biochar. Bio-oil and gas can be co-produced in pyrolysis systems. These could be refined to a range of chemicals (Bridgwater, 2003) and/or used as sources of renewable energy if derived from sustainably produced biomass.

To date, short-term laboratory, greenhouse, and small field plot experiments on biochar have led to interesting but preliminary results. Many of the studies suggest that biochar has potential to reduce N_2O and CH_4 emissions from soils. However, it is recognised that the mechanisms to achieve such GHG emission reductions are not fully understood and need further research (Atkinson *et al*., 2010; Spokas 2013). Extrapolation of how short-term, laboratory-scale trials results compare with real life conditions should be done with care.

Soils vary widely and under different climates are complex systems. Different types of biomass feedstocks, each with different production, post-production, and application parameters, further complicate the biochar-soil dynamics. Biochar production methods from NZ's potential range of biomass feedstocks are in the process of characterisation to determine the needs specific of the numerous types of NZ soils.

Concerning C sequestration, the amount of C locked up in biochar over a certain timescale can vary depending on a matrix of factors. Evidence from '*Terra Preta'* (TP) soils in the Amazon Basin, where Pre-Columbian additions of charcoal into soils are still found, indicate that the stability of biochar ranges from hundreds to thousands of years (Lehmann, 2007). However, TP soils took centuries to form and build up under unknown conditions. This raises questions about the effectiveness of modern biochar application in NZ and whether the assumed carbon-negative status of biochar is realistic and can be captured.

Bruun and Luxhøi (2008) argued that comparable CO_2 emissions would effectively result if biomass, instead of being used exclusively for bioenergy that consequently displaced fossil fuels, was converted into biochar. Biomass resources are limited and evaluating the multiple environmental benefits offered by biochar systems requires the use of comparative analytical methods to support decisions about the best use of biomass.

The purpose of this study was to evaluate and compare, using Life Cycle Assessment (LCA) methodology:

- the climate-change impact potential of alternative uses of biomass for a range of different feedstocks in NZ;
- potential consequences of implementing alternative systems; and
- analyse the technical issues needed for biochar to gain acceptance in carbon markets.

The study aims at answering the following research questions:

- What is the use of biomass that can achieve the largest amount of carbon credits in order to support policymaking?;
- What are the key components of the biochar system that affect its capacity to mitigate climate change?; and
- What are the scientific and policy barriers to including biochar in carbon markets?

Despite criticisms of carbon trading and uncertainties in biochar knowledge, various researchers advocate for the inclusion of biochar in carbon markets. In contrast, a declaration to keep biochar and soils out of carbon trading has been signed by about 150 organisations concerned with land grabbing and carbon offsetting (Rainforest Rescue, 2009). Raising awareness about the criticisms of carbon markets is important.

With these aims in mind, the literature review is presented in chapter two. In chapter three, NZ's most prominent end-of-life biomass resources are reviewed and three feedstocks are selected for further analysis. In chapter four, alternative uses of these feedstocks are evaluated in case studies and compared with each other using LCA methodology. Chapter five describes the issues that affect the hypothetical inclusion of sustainable biochar systems in carbon markets. Conclusions and recommendations are elaborated in chapter six.

CHAPTER 2: LITERATURE REVIEW

The literature review is structured in five main sections. Section 2.1 provides the introduction to the issue of climate-change and current policies. In section 2.2, the biochar opportunity is explored. In section 2.3, Life Cycle Assessment (LCA) methodology is examined together with the methodological issues affecting the results of the LCA studies of biochar systems. Carbon markets are critically reviewed in section 2.4. Finally, in section 2.5, the rationale for the use of biochar in New Zealand is discussed.

2.1. Climate change and current policies

Planet Earth, our system boundary, is pushing its human dwellers to look for ways to live sustainably "within the ecological limits of a finite planet" (Jackson, 2009). Despite the large and still accumulating body of evidence indicating that anthropogenic global warming is occurring, the challenge remains in communicating clearly and transparently the science and imminent consequences of climate change to the global population (Hansen, 2009). Policies to reduce and sequester carbon emissions therefore need to be developed at a local and global level against which all countries can measure their performance in efforts to reduce the global warming effect.

The first United Nations summit on sustainable development took place in Rio de Janeiro in 1992 and gave birth to a legally binding convention aimed at preventing global warming: the United Nations Framework Convention on Climate Change (UNFCCC). The convention entered into force in 1994. "*Concerned"* that human activities have increased atmospheric CO_2-eq concentrations that result in global warming, the convention notes that developed countries are responsible for "the largest share of historical and current global emissions of greenhouse gases" (United Nations, 1992).

The ultimate objective of the Convention (Article 2) is to stabilise atmospheric CO_2-equivalent concentrations "at a level that would prevent *dangerous anthropogenic interference* with the climate system" (United Nations, 1992). Moreover, article 3.3

mentions that “the Parties should take precautionary measures to anticipate, prevent or minimize the causes of climate change and mitigate its adverse effects. Where there are *threats of serious or irreversible damage*, lack of full scientific certainty should not be used as a reason for postponing such measures…”.

The scientific evidence for climate change, with regard to climate science, adaptation and mitigation, is summarised in the publications of the Intergovernmental Panel on Climate Change (IPCC), which discuss uncertainties and provide the scientific consensus or continuing debate. The summaries for policymakers of the three different working groups (IPCC, 2007a; IPCC, 2007b; IPCC, 2007c) of the Fourth Assessment Report (AR4) were each approved sentence by sentence by negotiating officials, usually from around 110 to 150 governments, and with many observers representing civil society. In 2009, a government agreement to keep global average temperature rise below 2^oC was reached at Copenhagen (see section 2.1.1). The volume I providing the physical science basis of the Fifth Assessment Report (AR5) was published recently (IPCC, 2013). Volumes II and III of the AR5 will be published in 2014.

The cause of climate change is the rising atmospheric concentrations of a number of greenhouse gases (GHGs) that reduce the reflective ability of the Earth to incoming solar radiation. The global warming indicator can be defined in parts per million (ppm) by atmospheric concentration of GHGs in the atmosphere. It is also reported as carbon dioxide equivalent (CO_2-eq) by including the global warming potentials for each gas. The global CO_2 concentrations recently surpassed 400 ppm (Williamson, 2013a). The pre-industrial level of about 280 ppm atmospheric CO_2 is used as comparison (IPCC, 2007a). The CO_2-equivalent includes all GHGs and climate forcings such as solar irradiance but excludes slow feedbacks such as the heat being trapped in the ocean, which could thaw Arctic ice sheets and release methane (CH_4) from the permafrost. Moreover, the biggest uncertainty in climate models is the exact degree of the negative forcing (cooling effect) due to water vapour, cloud cover, and aerosols in the atmosphere (IPCC, 2007a).

According to the IPCC (2007a):

“Global atmospheric concentrations of carbon dioxide, methane and nitrous oxide have increased markedly as a result of human activities since 1750 and now far exceed pre-industrial values determined from ice cores spanning many thousands of years. The global increases in carbon dioxide concentration are due primarily to fossil fuel use and land use change, while those of methane and nitrous oxide are primarily due to agriculture.”

Based on observations and model predictions, climate change might provoke heavy storms, severe droughts, intense hurricanes and floods, strong heat waves, ocean acidification, habitat destruction and species extinction (IPCC, 2007b). Ice caps, mountain snow, icebergs, and glaciers are melting probably due to the higher levels of heat being trapped in the atmosphere by increased GHG concentrations. This also results in sea level rise. Climate change is also likely to lead to a reorganization of social economic systems due to possible reductions in food production and drinking water in some regions, health ailments, human deaths, unemployment, migration, material losses, and political unrest.

The Montreal Protocol is an international treaty developed to phase out the production of a number of substances believed to cause ozone depletion. It was signed in 1987, became effective in 1989 and is considered the most successful environmental agreement, resulting in the displacement of 97% of the ozone depleting substances mentioned in the treaty and allowing the ozone layer to recover by 2050 (Molina *et al.*, 2009). It has been suggested that experience from the Montreal Protocol could be used in climate change policy. Mascarelli (2010) argued that one common requirement in dealing with climate change and protecting the ozone layer is to understand the whole life cycle of a polluting activity.

Despite common characteristics, regulating climate change is economically, technically, politically, and socially more complicated than phasing out ozone-depleting gases. The GHGs responsible for climate change come from a wide variety of sources embedded directly or indirectly in all sectors of the global economy. Zhang (2009) argued that while the Montreal Protocol uses an approach based primarily on incentives and complemented with sanctions, certain climate change policies on the table give stronger emphasis to regulations (“sticks”) rather than incentives (“carrots”). Therefore, care must be taken when transposing lessons from the Montreal Protocol to the climate change context.

The Kyoto Protocol, linked to the UNFCCC, is an international treaty designed to combat climate change. It was signed in 1997 and entered into force in 2005. Excluding the USA, industrialised countries (Annex I countries) were required to reduce their GHG emissions by an average of five per cent against 1990 levels between 2008 and 2012, known as the ‘first commitment period’.

To help meet targets, Annex I countries may use the Kyoto’s three flexible market-based mechanisms: joint implementation (JI), clean development mechanism (CDM), and emissions trading (ET). The CDM, defined in Article 12 of the Kyoto Protocol, provides for Annex I countries to invest in projects that reduce emissions and promote sustainable development in non-Annex I countries, in return for offsets. Unlike the CDM, JI projects take place in other Annex I countries. ET allows Annex I countries that have emission permits to spare to sell this surplus to other countries exceeding their targets. Carbon, an intangible commodity, is traded in the form of emission reductions or removals, each equivalent to one tonne of CO_2 (see section 2.4).

2.1.1. Climate stabilisation targets

The targets to achieve climate stabilisation and avoid climate catastrophe appear in many forms and numbers in the literature. The selection of different stabilisation targets is important because it leads to different strategies. This makes climate change science very difficult to communicate to policymakers and global population.

The stabilisation targets appear in the literature in terms of global mean temperature change, CO_2 atmospheric concentrations, GHG emissions, radiative forcing, climate change impacts, and economic costs. All these factors are interdependent but it is uncertain to what extent they interact, mainly due to their relation with climate sensitivity, which is difficult to predict (IPCC, 2007a).

The target has been retained from the late 1980s when, based on the knowledge at that time, the global average increase of 2°C above pre-industrial levels was recommended as the limit for dangerous anthropogenic interference. The Copenhagen Accord, a 3-page non-binding agreement reached at the final day of the 15^{th} Conference of the Parties (COP), did not include a CO_2 atmospheric concentration target but seeks to limit global mean temperature rise to 2°C above pre-industrial times (Torney and Greup, 2010).

The Copenhagen Accord is often seen as a 'half-empty' or 'half-full' glass depending on the benchmark against which the outcome is measured. Torney and Greup (2010) discuss that based on "existing political realities", it is possible to view the Copenhagen Accord in positive terms but if climate science is the point of reference then there is "good reason to be deeply pessimistic".

The synthesis report of the IPCC's AR4 (IPCC, 2007d) revealed that a rise of about 1.5° - 2.5°C compared to pre-industrial levels "poses significant risks to many unique and threatened systems including many biodiversity hotspots". Ramanathan and Feng (2008) argued that the world has already "committed" to a global warming ranging from 1.4 to 4.3°C, where 2.4°C is most likely. Schellnhuber (2008) added that it is risky to aim at 2+*X*°C because tipping points are likely to occur when the temperature increase approaches 2^{o}C. "The greatest threats are tipping the Arctic-sea ice and the Greenland ice sheet" and probably other surprising "tipping elements" could be triggered within this century (Lenton *et al*., 2008). The probabilities of exceeding 2°C global warming have been calculated based on GHG emissions (Meinshausen *et al*., 2009) and the authors "recognize that 2°C cannot be regarded as a safe level".

The other most common target found in the literature is the atmospheric CO_2 concentrations. CO_2 concentrations and global mean temperature rise are directly but non-linearly linked. Hansen *et al*. (2008) and supported by Rockström *et al*. (2009), suggested aiming at an initial maximum of 350 ppm CO_2 target within decades, which represents a probability higher than 75% of staying below a 2°C rise (Molina *et al*., 2009).

The 450 ppm CO_2 concentrations often cited in the literature as a climate-stabilisation target means approximately a 50% probability of staying below 2°C (den Elzen *et al.*, 2010). The 450 ppm CO_2 target has been used to develop a series of robust energy and policy scenarios (IPCC, 2007c; IEA, 2009). For such a target, developed countries would have to reduce their emissions between 25 and 40% by 2020 and between 80 and 95% by 2050, while developing countries would need to substantially deviate from the baseline scenario (compared to 1990 levels).

Solomon *et al.* (2009) highlighted that if CO_2 concentrations are allowed to ratchet up to a peak of 450-600 ppm over the next 100 years, climate change impacts will be "irreversible". They explained that even if CO_2 emissions stop, temperature increases caused by atmospheric CO_2 concentrations are not expected to decline significantly. This is because CO_2 released from fossil fuel combustion will stay in the surface climate system for thousands of years (Hansen *et al.*, 2013). This leads to the observation that avoiding GHG emissions has to be complemented with removing CO_2 already resident in the atmosphere.

2.1.2. Measures

There is no "one-size-fits-all" solution to climate change. On the contrary, an extensive variety of approaches has been proposed. A number of GHG mitigation scenarios through a wide spectrum of measures in the energy, industry, agricultural, and forest sectors have been analysed (IPCC, 2007c). The scenarios assumed population and economic growth and "range from structural changes in the energy system and replacement of carbon-intensive fossil fuels by cleaner alternatives (such as a switch from coal to natural gas, or the enhanced use of nuclear and renewable energy) to demand-side measures geared towards energy conservation and efficiency improvements" (Fisher *et al.*, 2007).

Carbon sequestration options such as carbon capture and storage (CCS) during energy conversion processes and afforestation, reforestation, and conservation of forests were also taken into account in IPCC's AR4. Biomass plantations for energy are also mentioned but should be handled with care due to the possibility of increasing GHG emissions if net deforestation takes place.

Furthermore, IPCC's AR4 included lifestyles change in global population. However, they played a role in the baseline and not in the mitigation scenarios. For example, a 25% increase in global meat consumption was expected together with high-emitting agricultural intensification practices by 2030 (Fisher *et al*., 2007).

Global population lifestyles exacerbate or lessen the impacts of climate change. One of the major drivers of climate change is the human demand and use of products and services. GHG emissions occur from the extraction of resources and processing of raw materials to transport and disposal of products after being used. In this sense, Life Cycle Assessment (LCA) has been brought forward to determine the GHG emissions through the whole supply chain of a product or activity (Baumann and Tillman, 2004). If demand for products and services decrease, GHG emissions decrease. Hence, global population must be informed first in order to make the most sustainable decisions.

After IPCC's AR4 in 2007, climate scientists continue showing to policymakers and global population that action against climate change is urgent. Some of them have engaged in the complex field of climate policy (Hansen *et al*., 2008). They believe that current climate policies do not match completely the urgency of the problem. Their proposals, however, do not substitute IPCC's recommendations and should be seen as complementary.

Hansen *et al.*, (2008) proposed to phase out coal use that does not incorporate CCS over the next 20-25 years. This is, as they put it, a "herculean" task and unlikely to be implemented. Furthermore, they introduced carbon sequestration in soils in the form of charcoal as a potential measure. This form of charcoal sequestered in soils is called biochar.

Biochar (and inherent carbon) can last in soils for hundreds or thousands of years (Lehmann, 2007). Referred to as a carbon-negative strategy, biochar is seen as a low-cost measure to remove CO_2 from the atmosphere, and raise soil productivity (Read, 2009). Moreover, the production and use of biochar offer several opportunities in the energy, waste, and land management sectors. However, the full potential of biochar remains to be proven.

2.2. The biochar opportunity

Although the term biochar is relatively new, charcoal has been used in agriculture to improve soil conditions in the past. In modern times, the biochar technology is not limited to agriculture and has attracted international interest due to its multiple potential benefits. These are discussed below.

2.2.1. What is biochar?

The definition of biochar (or bio-char) is not clear in the literature. Ambiguously, different authors use different terms to describe biochar: char; charcoal; agrichar; biocarbon; and elemental, pyrogenic, active, activated or black carbon (Lehmann *et al*., 2003; Demirbas, 2006; Glaser, 2007; Renner, 2007; Harris and Hill, 2007). The disparities are very subtle since they are all produced from carbon-rich materials (Sohi *et al*., 2009).

In this study, biochar is defined as carbonised biomass obtained from sustainable sources and sequestered in soils to sustainably enhance their agricultural and environmental value under present and future management. This distinguishes it from charcoal that is used as fuel for heat, as a filter, as a reductant in iron-making or as a colouring agent in industry or art (Lehmann and Joseph, 2009). Since the sources, treatments, and uses of biochar can vary to a great extent, a more comprehensive definition would be helpful.

The International Biochar Initiative (IBI) recently released version 1.1 of the *"standardized product definition and product testing guidelines for biochar that is used in soil"* (IBI, 2013a). The "*IBI Biochar Standards*" provide recommendations to ensure that biochar is safe to produce and use but do not prescribe parameters for production and feedstock handling, nor do these provide thresholds or terms for defining the sustainability of the feedstocks or biochar products (IBI, 2013a). Cowie *et al*., (2012a) suggested elaborating on existing voluntary benchmarks such as the Forest Stewardship Council (FSC) and the Round Table on Sustainable Biomaterials (RSB), formerly known as the Roundtable on Sustainable Biofuels, to assess the life cycle sustainability of biochar systems.

Biochar, a porous material, helps retain water and nutrients in the soil for the plants to take up as they grow (Lehmann *et al.*, 2003). Due to its adsorption ability, some biochars have the potential to immobilize heavy metals, pesticides, herbicides, and hormones (Winsley, 2007; Uchimiya *et al.*, 2010; Wang *et al*., 2010; Sarmah *et al*., 2010); prevent nitrate leaching and faecal bacteria into waterways; and reduce N_2O and CH_4 emissions (Lehmann, 2007). Moreover, based on evidence from the Amazon region, Lehmann (2007) argued that the stability of biochar in soils ranges from centennial to millennial timescales, and therefore could be considered as a long-term C sink.

2.2.2. *Terra Preta de Indio*

The application of biochar into soils is not a new concept. Native pre-Columbian civilisations in the Amazon Basin added biochar to their soils and increased their fertility (Glaser *et al.*, 2001). Whether they did it on purpose or by accident is uncertain (Glaser, 2007). Because of their black colour and origins, these soils are known in Portuguese as *Terra Preta de Indio* (Indian black earth).

Typically, *Terra Preta* (TP) soils are found in patches of approximately 20 ha containing higher soil organic matter (SOM), greater concentrations of nutrients such as N, P, K, and Ca, and up to 70 times more biochar than surrounding infertile soils (Glaser *et al.*, 2001). Biochar alone is not responsible for the formation of these highly fertile soils. Research shows that nutrients in TP soils were incorporated in the form of human and animal excrements, aquatic and terrestrial biomass, and food residues such as mammal bones, fish bones, and turtle backs (Glaser, 2007). TP soils built up their fertility for centuries and are currently being removed and sold to farmers.

2.2.3. Biochar production and by-products

Biochar is produced by heating biomass in the total or partial absence of oxygen. Pyrolysis is the most common technology employed to produce biochar, and also occurs in the early stages of the combustion and gasification processes (Bridgwater, 2003). Besides biochar, bio-oil and gas can be collected from modern pyrolysers (Laird, 2008). These could be refined to a range of chemicals and/or used as sources of renewable energy if derived from sustainably produced biomass. There are three main thermal processes that convert biomass to biochar: hydrothermal carbonisation (HTC), gasification, and pyrolysis (Table 1). Each byproduct has a value, so process selected depends on end uses.

Table 1. Typical product yields obtained by different biochar-production methods (adapted from Bridgwater, 2006 with data from Funke and Ziegler, 2010, and Titirici et al., 2007)

Process	Conditions	Liquid (%)	Char (%)	Gas (%)
Hydrothermal carbonisation	Low temperature (180- 220^{o}C) Long residence time (4-24 hours)	-	~100	-
Gasification	High temperature (≥800 ^{o}C) Long residence time	5	10	85
Fast pyrolysis	Moderate temperature (~500 ^{o}C) Short hot vapour residence time (~1s)	75	12	13
Intermediate pyrolysis	Moderate temperature (~500 ^{o}C) Moderate hot vapour residence time (~10-20s)	50	20	30
Slow Pyrolysis	Low temperature (~400 ^{o}C) Long residence time	30	35	35

HTC is a dehydration and decarboxylation process involving relatively low temperatures (180- 220^{o}C) over an aqueous solution of biomass under saturated pressure and weakly acidic conditions for 4-24 hours (Titirici *et al.*, 2007; Funke and Ziegler, 2010). HTC, mostly under development in Germany, has been presented as a promising process to produce biochar from wet feedstocks such as algae, fruit peels, leaves, sugar-beet, manure, and sludge (Titirici *et al.*, 2007). Even though the HTC process has the highest char yield and does not require energy to dry feedstocks, the resulting biochar (hydrochar) has lower resistance to decomposition (Hu *et al*., 2010). Soil responses and the residence times of hydrochars need to be researched to determine the potential value for long term C sequestration (Fuertes *et al*., 2010).

Gasification systems produce very small amounts of biochar and higher quantities of gas (Table 1). This gas is mainly a mixture of carbon monoxide (CO) and hydrogen (H_2) with lower quantities of CO_2, CH_4, H_2O, and a range of volatile compounds (Bridgwater, 2006). Pyrolysis gas is costly to store and transport and therefore, it is usually combusted on-site to meet heat and electricity needs (Bridgwater, 2006; Laird *et al.*, 2009).

Current research on biochar production focusses on pyrolysis technologies. Pyrolysis is seen as a simple, flexible, and affordable technology which transforms biomass to biochar and renewable energy carriers: pyrolysis gas and bio-oil (Laird *et al*., 2009). Pyrolysis gas is generally used for internal processes. However, if the heat energy released during combustion of the pyrolysis gas exceeds the demand of a biochar system and opportunities exist nearby the plant, then the pyrolysis gas could be used to provide heat for external processes. If the energy provided by the pyrolysis gas is converted into electricity, then this can be fed into the grid.

Bio-oil consists of a complex mixture of oxygenated hydrocarbons with a considerable fraction of water (Bridgwater, 2006). Bio-oil is attractive for the energy industry due to the fact that it is transportable, storable, and a potential replacement of fuel oil or diesel in stationary applications including boilers, furnaces, engines, and turbines for electricity generation. However, it cannot be used directly as a transport fuel unless it is upgraded, which is technically feasible but expensive (Balat *et al.*, 2009). In addition, bio-oil can be a source of a number of valuable chemical products such as acetic acid, resins, sugars, food flavourings, slow release fertilisers, adhesives, and preservatives (Bridgwater, 2003).

Theoretically, pyrolysis takes place when biomass is heated above 300^{o}C without oxygen, producing a solid char, condensable hydrocarbons or tar and gases (Bridgwater, 2006). At lower temperatures, drying and roasting of biomass is called torrefaction. Torrefied biomass has favourable properties such as low moisture content, high energy density, and high ability to pulverise (Deng *et al*., 2009), which make it attractive in energy schemes rather than agricultural projects. In practice, however, biochars have been produced at temperatures ranging from about 200^{o}C (Demirbas, 2006) to 1000^{o}C (Kawamoto, 2005).

Slow pyrolysis reactors have been used to produce charcoal from woody biomass for thousands of years. In developing countries, traditional earth-mound, brick, and metal kilns are inefficient and usually do not include burning of the exhaust gases. Since they are regarded as an important source of deforestation and GHG emissions, alternative production scenarios have been modelled based on coppice management of native trees and improved kilns (Bailis, 2009). Moreover, small-scale pyrolysis stoves have been proposed to decrease fuel consumption and deforestation in developing countries, improve respiratory health, and increase soil fertility by incorporating the biochar into soils (Whitman and Lehmann, 2009).

Fast pyrolysis is of great interest to researchers in the energy field since it gives the highest yield of bio-oil. It is worth noting that in fast pyrolysis the biomass moisture needs to be brought down to around 10-15% and particle size should not exceed 2 mm (Bridgwater, 2003; Sohi *et al.*, 2009).

Different pyrolysis technologies are being assessed and tested in demonstration plants. These include bubbling fluid beds, entrained flow, rotating cone, ablative processes, vacuum pyrolysis, circulating fluid and transported beds. A detailed description of these technologies can be found elsewhere (Bridgwater, 2003; Bridgwater, 2006). According to Balat *et al.* (2009), commercial operation has only been achieved with the circulating fluid and transported beds and only for food and flavouring products.

From an exclusive energy point of view, the investment on a fast pyrolysis reactor varies from 10-15% of the total capital cost of an integrated system, where the rest consists of biomass reception, storage, handling, drying, grinding, bio-oil collection and storage and, when relevant, upgrading (Bridgwater, 2006). For biochar production, a comparative figure for a slow pyrolysis reactor is still unclear in the literature due to the fact that a biochar system is more dependent on the specific context than an energy scheme. Moreover, biochar research has not gone beyond laboratories, greenhouses, and small field trials.

Furthermore, microwave pyrolysis is recently being studied. The perceived advantages of microwave pyrolysis are energy efficiency, rapid and controlled heating, and the ability to operate from an electrical source (Robinson *et al*., 2010). Nonetheless, the latter could be argued to be a disadvantage based on availability, costs, and sources of electricity.

Two toxic compounds, polyaromatic hydrocarbons (PAHs) and dioxins can be also present in chars and bio-oils produced during pyrolysis (Garcia-Perez, 2008). Large quantities of PAHs are formed in chemical reactions at temperatures over 700^{o}C and evidence suggests that small amounts of PAHs can also be formed in pyrolysis reactors operating between 350 and 600^{o}C (Garcia-Perez, 2008). Dioxins predominantly form at temperatures above 1000^{o}C and are significantly reduced when chlorine and metals are not present. Biomass feedstocks such as switchgrass, miscanthus, and wheat straw could have a high content of chlorine (Samson *et al*., 2005) and therefore, dioxin levels in respective biochars should be carefully analysed.

Even though there is concern on the noxious direct impact and consequent leaching that these compounds may have in soils when biochar is applied, very little is reported in the literature (Sohi *et al*., 2009). It is important to note that in any future biochar manufacturing process these toxins must be avoided.

Biochar production technologies in New Zealand

Several organisations in NZ are developing biochar production technologies (Table 2).

Table 2. Biochar production technologies in New Zealand as of July 2013

Organisation	Location	Technology	Stage of development	Main interest
New Zealand Biochar Research Centre at Massey University	Palmerston North	continuous-based and batch-based slow pyrolysis	laboratory-scale and pilot-scale testing, respectively	biochar for various purposes
Carbonscape	Blenheim	microwave pyrolysis	prototype - seeking investment capital	biochar for various purposes, most recently as reductant for steel making
Waste Transformations Limited	Otaki	microwave pyrolysis and batch-based slow pyrolysis	demonstration plant	biochar for waste management of various streams
CQuest	Wellington	microwave pyrolysis	constructing a pilot plant	biochar for waste management
Lakeland Steel Limited	Rotorua	continuous-based slow pyrolysis	scaling up to commercial operation	byproducts for various purposes
The Wood Technology Research Centre at the University of Canterbury	Christchurch	fast fluidised bed pyrolysis	laboratory-scale testing	bio-oil and pyrolysis gas for input to Fischer-Tropsch for hydrocarbon fuels and hydrogen for fuel cells
Norske Skög	Kawerau	fast fluidised bed pyrolysis	funding approved	byproducts for various purposes
CRL Energy Ltd., OPUS International Consultants	Lower Hutt, Wellington	continuous-based slow pyrolysis and fast fluidised bed pyrolysis	pilot-scale and laboratory-scale testing, respectively	byproducts for various purposes

2.2.4. Sources of biochar

Lehmann *et al*. (2006) and Glaser (2007) have suggested using the charcoal residues from the already established charcoal production as biochar. They mentioned that almost all of the global charcoal-making takes place in developing countries (40 out of 41 Mt), and while charcoal residues are minimised in industrial production, 10-20% of the charcoal (4-8

Mt) is too small (<2 mm) to be sold and, therefore could be seen as a potential source of biochar. Traditional charcoal-making is highly inefficient and causes deforestation. Lehmann *et al*. (2006) and Glaser (2007) did not elaborate on methods of collection, storage, handling, transport, and distribution of this pulverised solid material. Moreover, they failed to acknowledge the fact that in many circumstances in developing countries charcoal production and trade is illegal (Post and Snel, 2003; Tabuti *et al*., 2003; Kituyi, 2004) and carried out by nomad workers making the potential of using charcoal residues as biochar impractical.

Instead, researchers have concentrated on making new biochar from diverse sources of biomass at different production parameters (temperature, heating rate, and residence time). Technically, any kind of biomass can be introduced in fast pyrolysis reactors and a lot of work has been done on wood because of its standard characteristics and high energy content (Bridgwater, 2006). However, choosing the type of feedstock for biochar is more important than for energy production due to complex soil dynamics.

Purpose-grown energy crops such as switchgrass, miscanthus, and corn have been studied in the USA (Roberts *et al*., 2010; Gaunt and Lehmann, 2008) but most research trials have used end-of-life biomass (ELB) such as chicken and cow manure, forestry and agricultural residues, and sludge (Ogawa, 2006; Camps Arbestain *et al*., 2009; McHenry, 2009; Sohi *et al.*, 2009; Lehmann, 2009; Tagoe, 2010; Cao and Harris 2010; Gaskin *et al*., 2010; Hossain *et al*., 2010; Van Zwieten *et al*., 2010).

Although the term ‘biomass waste’ as a renewable source is debatable because “nature does not produce waste, and a properly integrated society should not produce waste either” (Sims, 2002), ELB is generally regarded as sustainable. Yet, if the ELB source was controversial – Lehmann *et al*. (2006) included “tobacco waste” as a suitable feedstock – the debate on its use for biochar production would be intensified from a sustainability perspective.

Dedicated biomass plantations for the sole purpose of producing biochar are, under current prices, unlikely to be profitable (Lehmann, 2009). Biochar produced from indigenous forest clearing does not result in net emission reductions from a life cycle perspective and would also pose a risk to biodiversity conservation (Glaser, 2007). There is also competition between biomass resources (see section 2.2.8). The small amount of existing biochar plants are dedicated to specific ELB streams (Sohi *et al*., 2009) that are financially attractive.

Crop residues, when left in the field play an important role in carbon sequestration, conservation of soil and water, microbial activity, and agricultural productivity (Lal and Pimentel, 2009). Continuous removal of crop residues from the same land to produce biochar jeopardises these benefits, unless the biochar is returned to the same fields from where it was harvested (Laird, 2008; Sohi *et al*., 2009).

Biochar from sewage sludge produced in the treatment of household, municipal, and industrial sewage may contain heavy metals and/or organic pollutants that could contaminate the soil rather than ameliorate it. Because of this toxic possibility, this kind of source has been ruled out in certain studies (Lehmann *et al*. 2006) but is the focus of attention in others (Camps Arbestain *et al*., 2009; Hossain *et al*., 2010). Charcoal produced from sludge presented “some harmful substances including heavy metals” but did not exceed the acceptable level in Japanese standards (Shinogi *et al*., 2003). However, “the level of zinc (Zn) requires careful attention, as in this standard the zinc level is lax” (Shinogi *et al*., 2003).

In Australia, charcoal produced from sludge contained high content of heavy metals including arsenic, selenium, silver, cadmium, copper, lead, nickel, and zinc (Hossain *et al*., 2009). More recently, Hossain *et al*. (2010) reported that the application of biochar produced from sludge increased the yield of cherry tomatoes and “bioavailability of metals present in biochar was found to be below the Australian maximum permitted concentrations for food”. Notably, further research is needed to characterise different types of sludge under various conditions “as its level of contamination may be quite variable at different locations and different times” (Woolf, 2008). Moreover, wet feedstocks such as sludge raise the question of which kind of carbonisation process would be best to use since it would require significant energy to dry before going through pyrolysis.

2.2.5. Characterisation of biochar-soil dynamics

Leonardo da Vinci (1452-1519) once said: “We know more about the movement of celestial bodies than about the soil underfoot” (Montgomery, 2007). The challenge of understanding how soil, microbes, nutrients, and plant roots interact with each other is expanded when adding biochar into the equation. It will require many different tests under various conditions at a wide range of scales and probably many years (if not decades) to solve it.

The characterisation of biochars is a topic of relevant discussion in the literature. Seven physical properties have been identified to measure the quality of biochar: pH, volatile compound content, water holding capacity, ash content, bulk density, pore volume, and specific surface area (Sohi *et al*., 2009). The IBI (2013a) has formalised these and other properties into a proposed reporting standard.

The characteristics of feedstock and production parameters determine the physico-chemical properties and nutrient content of biochar (Demirbas, 2006; Lehmann, 2007; Tagoe, 2008; Sohi *et al*., 2009; Camps Arbestain *et al*., 2009; Van Zwieten *et al*., 2010). Based on surface area, pH, and cation exchange capacity (CEC), Lehmann (2007) proposed a temperature between 450-550^{o}C to optimise the characteristics of biochar (Fig. 1) and doubted that the use of biochars produced below 400^{o}C would improve soil fertility.

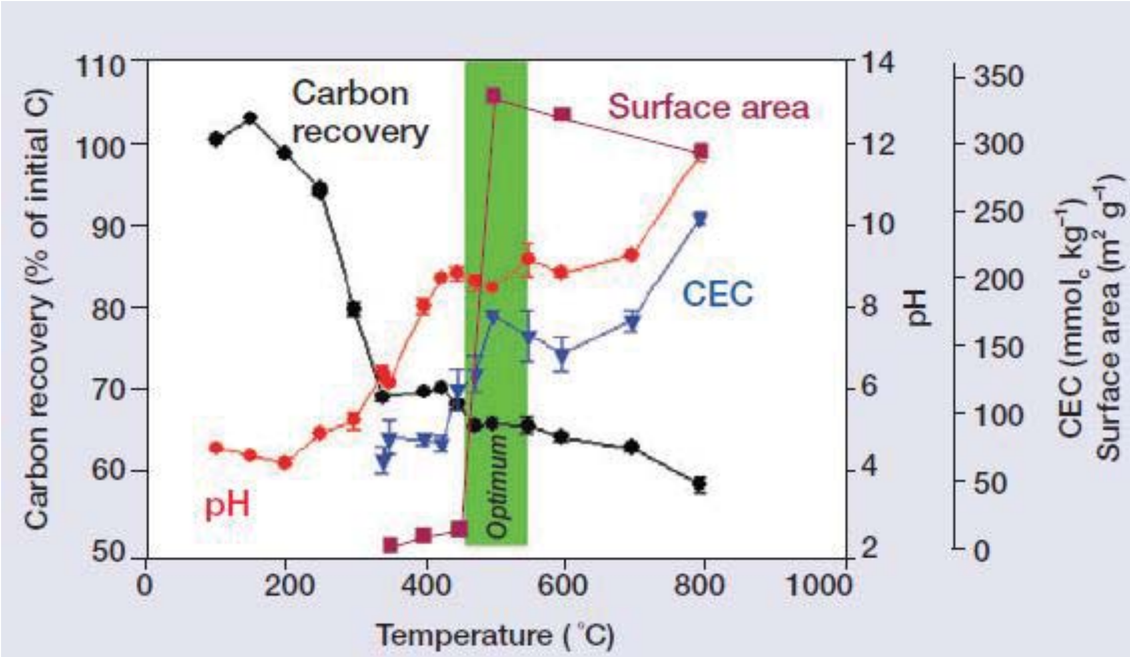

Fig. 1. Temperature effects on carbon recovery, cation exchange capacity (CEC; measured at pH 7), pH, and surface area for dried wood from Robinia pseudacacia L. (Lehmann, 2007).

Balancing parameters depend on what is desired. For example, the higher the process temperature the less biochar produced (less soil amendment and less C sequestered) but the higher its carbon stability (safer C sequestration) and co-products yield (more energy).

Since biochar science is relatively new, biochar publications correspond to short-term laboratory, greenhouse, and small plot experiments. Because of its origins, early studies focused on ancient TP soils and compared them to nearby soils in the Amazon Basin (Glaser *et al*., 2001; Lehmann *et al*., 2003). TP soils were found to have biochar that increased CEC (Liang *et al*., 2006), phosphorous nutrition and uptake, and decreased leaching of applied fertiliser N (Lehmann *et al*., 2003). Moreover, the latter showed that the amount of biochar in the soil is critical for the effects on plant growth and nutrition.

The porosity of biochar offers pore networks for water retention and microorganisms to thrive, but these can accelerate the decomposition of SOM and the biochar itself. Hamer *et al*. (2004) conducted a 60-day laboratory experiment, where they added glucose to different biochar-soil samples. After 60 days 0.78%, 0.72%, and 0.26% of carbonised maize, rye, and wood were respectively mineralised in the controls and glucose additions promoted the decomposition of black carbon by 58%, 72%, and 115% relative to the controls. The biochar made of wood at less than 200^{o}C proved to be more easily degradable because of its lower content of aromatic (chemically stable) carbon.

Liang *et al*. (2006) suggested that oxidation of biochars may not only mineralize organic C in soil but may also create negatively-charged surfaces causing higher CEC and nutrient retention in soil. A 120-day laboratory experiment (Cheng *et al*., 2006) compared uncrushed particles with finely-ground biochar, with and without inoculation with microbes or manure incubated at 30^{o} and 70^{o}C. The results showed that aliphatic (labile) C compounds were abiotically oxidized to CO_2 and while CEC increased, the pH decreased and aluminium saturation increased. This result contradicts the general assumption that CEC increases with higher pH (Lehmann, 2007).

According to Cheng *et al*. (2006), biochar contains a small fraction of labile compounds that degrade over time. Cheng *et al*. (2006) went on to explain that the abiotic processes were more relevant than biotic oxidation for fresh biochar. The rate of oxidation of biochar is surface dependent and finely ground biochar may oxidise faster. Long-term field studies are needed to elucidate this.

More recently, an aerobic incubation experiment showed that ageing (changes in biochar properties) of biochar can occur in any soil climate (-22^{o} to 70^{o}C) within a short period of 12 months (Cheng and Lehmann, 2009). Whether such changes will become more important over longer periods of time is not certain.

One of the longest field experiments in the literature illustrated that biochar in soils could decrease organic C in the form of humus in boreal forests in Northern Sweden (Wardle *et al.*, 2008a), and therefore increase CO_2 emissions. The experiment compared mesh bags filled with (i) humus from the forest, (ii) biochar, and (iii) a 50:50 mixture of humus and biochar. The bags were buried in three contrasting boreal forest sites and monitored over a 10 year period. The results indicated a considerable loss (of over 20%) of mass and humic

C in the mixture bags. The humus decomposition was attributed to the higher amount of microbial activity caused by adding biochar to the soil. Interactions of this type are typically described as priming, with increased or decreased turnover rates of native soil organic carbon referred as positive or negative priming, respectively. Both positive and negative priming effects due to biochar application have been reported in the literature (Woolf and Lehmann, 2012).

In response to Wardle *et al*. (2008a), Lehmann and Sohi (2008) commented that the decrease of mass and C in the mixture bags could have come also from the labile fraction of biochar and not from the humus alone. However, Wardle *et al*. (2008b) pointed out that the bags buried with only biochar were in close contact with humus from the forest for 10 years but presented negligible mass loss. And the high amount of mass lost in the mixture bags was incomparable to the ones observed elsewhere (Cheng *et al*., 2006). Wardle *et al.* (2008b) concluded that the strong advocacy to include biochar in carbon markets remains premature.

Based on the literature review, further characterisation of biochar-soil dynamics is required and could include analysis of the following factors:

- scale, baseline and test conditions, duration, and purpose of the study (eg. soil remediation, crop production, carbon sequestration);
- type of climate (eg. temperate, tropical, rainfall);
- type of soil (eg. sandy, clay, ferrosol, anthrosol);
- type of crop (eg. pine, corn, wheat, kiwifruit, apples, grapes);
- feedstocks for producing biochars (eg. orchard and vineyard prunings, logging residues, cereal straw, chicken and/or cow manure, sewage sludge);
- type of production process (eg. HTC, slow or fast pyrolysis) and respective parameters (eg. temperature, heating rate, residence time, pressure);
- post-production treatment (eg. inoculation with water, urine, manure, compost, synthetic fertilisers, microbes, lime, minerals);
- application into soils (eg. application rate, depth, particle size, tillage);

- follow-up and management (eg. watering, more fertilisers, extra biochar); and
- effects (eg. water holding capacity, pH, CEC, turnover rate, soil temperature, microbial activity, plant growth, nutrient leaching, toxins immobilization, GHG emissions, albedo).

2.2.6. Climate-change mitigation potential

Biochar is a multi-faceted strategy to mitigate climate change. The bio-oil produced during pyrolysis could replace fossil fuels and therefore avoid supplementary GHG emissions. The pyrolysis gas would represent additional GHG emission reduction potential if fossil fuels were displaced to provide heat, electricity or transport fuels in processes occurring outside the biochar system. The C sequestration potential of biochar is important and the application of biochar into soils could diminish CH_4 and N_2O emissions in agriculture. This has attracted the attention of policymakers in agriculture-based economies such as New Zealand (Winsley, 2007).

However, there is very limited research on the impact of biochar on GHG emissions from soils. On the one hand, one of the most cited studies on methane (Rondon *et al*., 2005) concerns a 50-day glasshouse pot experiment with soybeans and a tropical grass in very acid and low fertile soils from Colombian savannas. At a biochar application rate of 20 g kg^{-1} soil, the authors reported an almost complete suppression of CH_4 emissions and a 50 and 80% reduction of N_2O emissions on soybean and grass pots respectively. Lehmann *et al*. (2006) hypothesised that “these low emissions may be explained by better aeration (less frequent occurrence of anaerobic conditions) and possibly by greater stabilization of C. The lower nitrous oxide evolution may also be an effect of slower N cycling (possibly due to a higher C/N ratio).”

On the other hand, one of the most cited studies on nitrous oxide (Yanai *et al*., 2007) describes a 5-day Petri dish experiment, in which 30 g of air-dried soil samples were rewetted with distilled water at 64%, 73%, 78%, and 83% of the water filled pore space (WFPS) and then biochar from sewage sludge was added. Rewetting the untreated soil at

64% WFPS, N_2O emissions were not detected but at 73% and 78% WFPS they were, suggesting a high sensitivity to soil moisture. At 73% and 78% WFPS, the addition of 3 grams of biochar (10% weight of the soil sample) decreased N_2O emissions by 89%, while at 83% WFPS the addition of biochar increased N_2O emissions. Sohi *et al*. (2009) pointed out that the application rate of this study was relatively high (180 t ha^{-1} in topsoil) and wrote that not only this has an impact on N_2O emissions but also on the physical properties of the soil.

Biochar has decreased leaching of N from mineral fertilisers (Steiner *et al*., 2008; Lehmann *et al*., 2003); consequently less fertiliser would be needed and additional N_2O emissions could be avoided. Besides these direct N_2O emissions, indirect N_2O emissions resulting from "leached N leaving agricultural fields and entering water systems, and from volatilized N deposited onto natural ecosystems" (Crutzen *et al*., 2008) could also be mitigated.

Because of its high recalcitrance, Seifritz (1993) investigated how to store the carbon in charcoal but did not explicitly mention using it as a soil amendment. With the discovery of TP soils, a place to keep C from the atmosphere has been found. Radiocarbon dating suggests that TP soils were created between 7000 and 500 BP (Glaser, 2007). Hence, biochar is seen as a long-term and carbon-negative strategy (Lehmann, 2007).

Different biochars present different proportions of aliphatic and aromatic compounds, and this complicates the biochar-soil equation. As Lehmann (2007) expressed, "some biochars may decompose relatively rapidly in soils, while others persist for millennia" and "quantification of long-term stability requires long-term observations, exceeding the periods feasible in traditional experiments". Moreover, the introduction of biochar into soil ecosystems may increase their microbial activity and respective CO_2 emissions (Hamer *et al*., 2004; Wardle *et al*., 2008a).

Other than GHG emissions, another climate forcing that biochar may have an impact on is Earth's albedo, which is the ratio of sunlight that the Earth's surface reflects back into space. The application of biochar can significantly darken the colour of the soil (as happens in TP soils), and therefore decrease Earth's albedo and aggravate climate change directly and indirectly through secondary processes. Oguntunde et *al*. (2008) observed a 37% reduction in surface albedo in some of Ghana's haplic acrisols (sandy soils) where charcoal from inefficient kilns is left on the ground. Moreover, since dark soils absorb more solar energy, an average increase of 4^{o}C in soil surface was reported. Charcoal found in Māori soils in Nelson reputedly raised the soil temperature enough for kumara to grow (Rigg and Bruce, 1923). An increase in soil temperature would also likely speed the ageing of biochar (Cheng and Lehmann, 2009), extend the growing season (Sohi *et al*., 2009), increase microbial activity and produce GHG emissions.

The extent of how the Earth's albedo could be altered by biochar application depends on the scale of production and land application. Biochar may not have a significant impact on Earth's albedo at small scale. In contrast, large scale application requires more examination. Even if vegetation cover may offset the impact of biochar on Earth's albedo in the long run, measures should be put in place to minimise the amounts of pulverised char material that may be blown as dust during production, storage, transport and application.

2.2.7. Land use, scale and geo-engineering

If the impact of biochar on the Earth's albedo is addressed, there are still contradictions between the current focus of research and what would be needed to cool down the planet and feed its future population. If a biochar strategy was to make any difference in mitigating climate change and increase food production, then large amounts of biomass (not only ELB products) would need to be carbonised and sequestered across large areas of land. This involves trade-offs in land use for food, feed, fibre, timber, soil organic matter, biodiversity, recreation, and energy.

Read and Parshotam (2007) argued that an "urgency driven by an imminent abrupt climate change" exists. They assumed 1 Gha of non-arable land, mostly available in world's tropical regions, and recommended to establish, over 25 years starting in 2010, new biomass plantations and use half of the product as lumber and, of the residue, half as biochar. They added 0.72 Gha of temperate switchgrass and 0.43 Gha of sugar cane for energy use. Including caveats on land use improvement, sustainable development, and large-scale capacity building, they called these 2.15 billion ha of new biomass plantations the “holistic greenhouse gas management”.

Based on Read and Parshotam (2007), the Royal Society (2009) categorised biochar systems as geo-engineering options and recommended to not formally accept them as a method “for addressing climate change under the UNFCCC flexible mechanisms until their effectiveness, carbon residence time and impacts have been determined and found to be acceptable”. Biochar “may make a useful contribution at a small scale but require further assessment of the life cycle effectiveness, economic viability, and social and ecological sustainability” (Royal Society, 2009). This leads to the conclusion that at a small scale, biochar is not regarded as geo-engineering and a Life Cycle Assessment (LCA) approach to study biochar systems would be a useful approach.

Whether biochar implies geo-engineering or not is under debate. By sticking to ELB feedstocks, it seems that most biochar scientists currently prefer to remain small scale and stay away from the geo-engineering category and the respective criticisms concerning land use. One exception is Biochar Europe that advocates for the inclusion of biochar as a geo-engineering scheme (Glaser *et al*., 2009). Furthermore, instead of geo-engineering, biochar application in soils could be conceived as “agricultural development that aims to reverse the harm done by a geo-engineering project” (Bruges, 2009).

2.2.8. Competing processes

Land use is not the only contestant in the competition of biochar for natural resources. The use of biomass is of paramount importance. Factors such as climate change, peak oil, soil erosion, lack of uncontaminated water and sanitation, increasing prices of food and energy, species extinction, and overpopulation push humanity to make sustainable decisions on the best use of biomass. It must be noted that biomass is a local resource and as such, the best use of biomass depends on the specific local conditions of the baseline and project scenarios.

Crop residues and animal manure incorporated into soils combat erosion and boost soil organic matter and water holding capacity. They avoid the extra use of synthetic fertilisers and currently are commercialised as compost and soil improvers. Moreover, some crop residues are used to feed livestock. Therefore, these ELB represent no waste in agriculture and extraction of these ELB to produce biochar should be done carefully, i.e. the optimal amount of feedstock removal should be analysed. Biochar may be reincorporated to the same land where the feedstock came from, in a so-called closed system, to address environmental risks arising from biomass removal.

The most significant use of biomass is energy. According to Kaygusuz (2010), about 11% of the world's total primary energy supply is met by traditional biomass, which is also the largest energy source in rural areas of the developing world. Biomass is used for heating, electricity generation, and vehicle transport. Nowadays, the most important objective to use biomass for energy is arguably to avoid the use of fossil fuels and respective GHG emissions.

By maximising biochar production in slow pyrolysis technologies, biochar may represent an alternative to biomass for energy. Fowles (2007) developed a simple numerical model and compared these two options based on three variables: the biochar yield efficiency of pyrolysis systems, the energy conversion efficiency, and the carbon emission factor per unit of energy of different fossil fuels. He failed to include a labile fraction of biochar and estimated that only if biomass displaced electricity from coal at above-average conversion efficiencies (>33%) the GHG emission reductions would be greater in renewable energy generation. In other cases (natural gas, oil, LPG), biochar sequestration in soils could result in higher GHG emission reductions. Woolf (2008) claimed that if bioenergy with carbon capture and storage (BECCS) became practicable, then bioenergy would have even a greater potential in climate change mitigation than biochar production.

Contrasting the fossil GHG emissions that could be avoided if the biomass was combusted, Fowles (2007) argued that C sequestration from biochar may be permanent and not prone to rebound effects (additional demand of energy if saved). However, Bruun and Luxhøi (2008) questioned the carbon-negative status of biochar, that is, the capacity of biochar to remove CO_2 from the atmosphere. They specified that the energy that would be displaced with biomass would have to be supplied by burning fossil fuels “effectively resulting in comparable carbon emissions” if biochar was put in soils. Moreover, they pointed out that over relatively long timescales the C sequestered in biochar will be released back to the atmosphere as biochar decomposes.

Biochar has been introduced as a technique to sequester C and improve soil fertility while also giving a portion of energy in slow pyrolysis systems (Lehmann, 2007; Gaunt and Lehmann, 2008; Gaunt and Cowie, 2009; Roberts *et al*., 2010). Due to the fact that energy arrangements and financial support mechanisms are already in place, biochar has even been proposed as a co-benefit in fast pyrolysis systems in lieu of a primary innovation in soil management (Laird *et al*., 2009).

The use of biomass is currently encouraged to substitute fossil fuels and fight climate change. This interest in bioenergy is based on the argument that the CO_2 released during biomass combustion will be absorbed by plants in the future rotation resulting in a carbon neutral cycle. However, this argument has been recently contested. For example, based on the leakage resulting from land use change (Searchinger *et al*., 2008) and leaching of additional N fertiliser (Crutzen *et al*., 2008), fuels from biomass may potentially increase rather than decrease GHG emissions.

In summary, the mitigation potential for biochar systems is very project specific depending on a large number of factors. Their evaluation needs to include a comparison of alternative scenarios on a life cycle basis in order to minimise negative impacts. Life Cycle Assessment (LCA) methodology is appropriate for this purpose.

2.3. Life Cycle Assessment (LCA) methodology

Assessing the benefits and environmental impacts of biochar requires the use of comparative analytical methods and tools to inform decisions about the best use of biomass. In this context, Life Cycle Assessment (LCA) is a suitable methodology. In the following sections, the development and intended applications of LCA are briefly discussed. The structure of LCA methodology and the most common criticisms of LCA studies are explained. In addition, the LCA studies of systems that have included biochar production were identified from the literature and the methodological issues affecting the results are discussed.

2.3.1. Introduction to LCA

The Brundtland Commission, formerly the World Commission on Environment and Development (1987), first claimed that sustainable development results from the balance of economic, societal, and environmental issues (Martins *et al*., 2007). As far as the environment is concerned, Life Cycle Assessment (LCA) has emerged as a methodological tool used to evaluate the environmental impacts of products. Services are included in the term ‘products’ (ISO, 2006a). The full life cycle of a product extend from extraction of resources through to production, use, recycling, and/or ultimate disposal. Each of these life cycle stages can contribute to a broad range of impacts such as climate change, stratospheric ozone depletion, smog creation, eutrophication, acidification, toxicological stress on human health and ecosystems, depletion of resources, water use, land use, and noise (Rebitzer *et al*., 2004).

An unpublished study on Coca-Cola packaging conducted in 1969-1970 by the Midwest Research Institute in the USA is considered to be the first LCA study (Baumann and Tillman, 2004). However, at that time, the environmental analysis of products was not known as LCA, and there were no guidelines or standards for conducting such studies. Assessments based on the life cycle environmental concept had different names such as ecobalances, resource and environmental profile analysis, integral environmental analysis, and environmental profiles (Baumann and Tillman, 2004).

In August 1990, the Society of Environmental Toxicology and Chemistry (SETAC) organised the first workshop on ‘Life Cycle Assessment’ in Vermont, USA (Gabathuler, 1997). During the early 1990s, the term “Life Cycle Assessment” was widely adopted (Baumann and Tillman, 2004). Dutch researchers working for the Centre of Environmental Science, Leiden University (Centrum voor Milieukunde Leiden: CML) played a major role in the development of LCA methodology in these early years (Gabathuler, 1997).

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An important achievement during the 1990s was the publication of LCA standards in the ISO 14040 series: ISO 14040, 1997 (LCA – principles and framework); ISO 14041, 1998 (LCA – goal and scope definition, and inventory analysis); ISO 14042, 2000 (LCA – life cycle impact assessment); and ISO 14043, 2000 (LCA – life cycle interpretation). The updated ISO 14040 (2006a) and 14044 (2006b) replaced the previous standards and are regarded as the indispensable framework for LCA (European Commission, 2010a).

However, LCA is still under development (Finnvenden *et al*., 2009). A number of ongoing international initiatives, which aim at building consensus and giving recommendations on LCA, include the Life Cycle Initiative of the United Nations Environment Program (UNEP) and SETAC, the European Platform for LCA of the European Commission and the International Reference Life Cycle Data System (ILCD).

The International Journal of Life Cycle Assessment is the only journal devoted completely to LCA. *The Journal of Industrial Ecology* and the *Journal of Cleaner Production* also regularly publish LCA research, and for controversial topics such as the LCA of liquid biofuels there are hundreds of papers in numerous journals.

2.3.2. Use of LCA

The international standard ISO 14040 (ISO, 2006a) lists the following applications for LCA:

- identification of opportunities to improve the environmental performance of products at various points in their life cycle;
- information to decision-makers in industry, government or non-government organisations (e.g. for the purpose of strategic planning, priority setting, product or process design or redesign);
- selection of relevant indicators of environmental performance, including measurement techniques; and
- marketing (e.g. implementing an eco-labelling scheme, making an environmental claim, or producing an environmental declaration).

While LCA in public procurement policies is important, research to date has concentrated on the industrial sectors (Rebitzer *et al*., 2004).

2.3.3. Structure of LCA methodology

LCA methodology is divided into four phases: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and interpretation (Fig. 2).

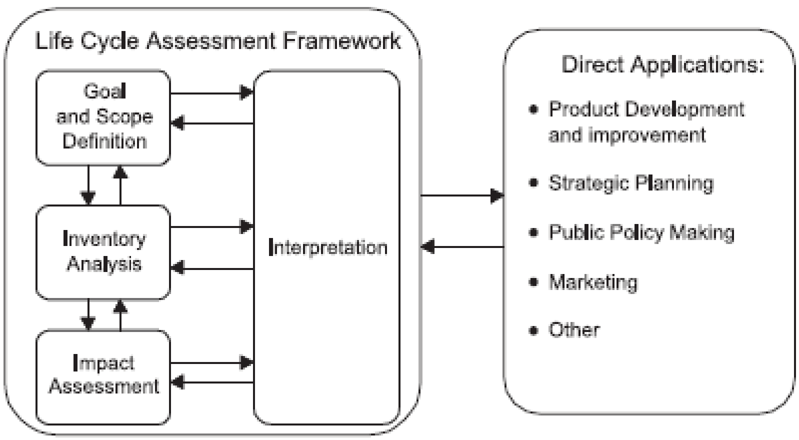

Fig. 2. Phases of LCA (ISO, 2006a)

Goal and scope definition

In the goal and scope definition phase, the product to be assessed and the objectives of the LCA are chosen. The goal of an LCA includes stating the intended application, the reasons for carrying out the study, and the intended audience (ISO, 2006a). According to Baumann and Tillman (2004), definition of the goal and scope is a collaborative work between the LCA commissioner and practitioner. Once the purpose of the study has been defined, scoping involves defining boundaries around the product system under analysis (Fig. 3), which is normally differentiated into the processes occurring in the foreground system and those impacting the background system (European Commission, 2010a). The foreground system may be defined as the system that is under control of the producer or user of the goods, or operator of the service. Contrastingly, the background system encompasses the processes that are affected by the system under analysis but are not under direct control of the person/s influencing the processes in the foreground system.

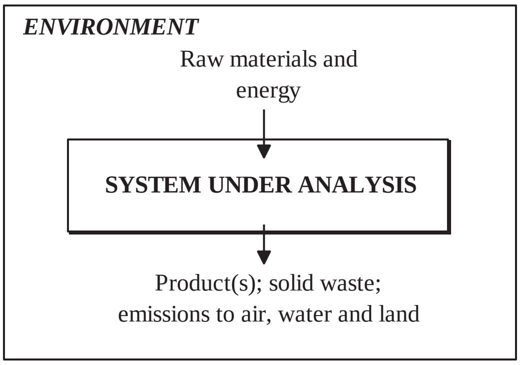

Fig. 3. Inputs and outputs across the system boundary in LCA (Cowell, 1998)

In the goal and scope definition phase, the environmental impacts to be considered, limitations, data quality requirements, and the functional unit are clearly defined (Baumann and Tillman, 2004; Finnveden *et al*., 2009). ISO (2006b) specifies a minimum of data quality requirements, which should address the following: time-related coverage, geographical coverage, technology coverage, precision, completeness, representativeness, consistency, reproducibility, sources of data, and uncertainty of the information.

The functional unit is the parameter that allows the comparison of the functions of alternative products in a quantifiable manner. In addition, the decision-context should be identified in the goal and scope phase. For example, if a decision is to be supported, then the study should "as good as possible" reflect the potential consequences of this decision (European Commission, 2010a).

Different decision situations and application categories for LCA were developed and categorised in the 1990s. Weidema (1998) found that these are related to time, space, and products, processes or interest groups affected. Wenzel (1998) further argued that the decision maker may also influence the results since it is not interesting to do an LCA if this is not used to support a decision and promote change. When a decision is taken based on the information from an LCA, a change will be induced somewhere in the society compared with the situation in which that decision was not made. This change (or consequence of the decision) should, therefore, be well foreseen and modelled in the LCA study. It should be noted, however, that there are cases in which LCA practitioners can claim that the study will not be used to support a decision (European Commission, 2010a).

The relatively recent International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010a) listed three types of decision-context: i) situation A: micro-level decision support; ii) situation B: meso/macro-level decision support; and iii) situation C: accounting. These mainly differ according to the intention of the study to support a decision and to the extent of the consequences arising in the background system (Table 3).

Table 3. General classification of the decision-context of LCA studies according to the European Commission (2010a)

<table>
<tr><td colspan="3">Are the results of the LCA study used to support a decision?</td></tr>
<tr><td>Yes</td><td>Yes</td><td>No</td></tr>
<tr><td colspan="2">How 'big' are the effects or changes in the background system and/or on other systems of the economy?</td><td rowspan="3">Situation C: accounting (C1 includes interactions with other systems and C2 excludes them)</td></tr>
<tr><td>None or small-scale (non-structural)</td><td>Large-scale (structural)</td></tr>
<tr><td>Situation A: micro-level decision support</td><td>Situation B: meso/macro-level decision support</td></tr>
</table>

If a decision is to be supported, the LCA study should reflect the potential consequences of this decision (situation A or situation B). If such consequences are expected to be large enough to cause structural changes outside the foreground system – e.g. production facilities would need to be constructed or dismantled – then the LCA practitioner shall follow the modelling guidance of situation B. Otherwise, if small-scale or no consequences are expected, situation A applies. The European Commission (2010a) acknowledges that there can be circumstances where it is not clear how to differentiate between situations A and B and provides further advice on how to handle such situations. If no decision support is pursued, then the system falls under situation C and should be analysed as it is. Furthermore, situation C is divided into C1, where the interactions with other systems are included through market effects, and C2 where the system is analysed in isolation without accounting for the interactions with other systems (European Commission, 2010a). The decision-context for the study is important as it can determine which modelling approach (attributional or consequential) is appropriate, according to the European Commission.

Attributional or consequential LCA

Depending on the goal and scope definition, two types of LCA studies can be defined: attributional (accounting) or consequential (change-oriented). The differences between the two approaches are quite pronounced (Table 4) and therefore the results of different approaches for assessing the same product can vary considerably (Thomassen *et al*., 2008).

Table 4. Differences between attributional and consequential LCA (adapted from Brander et al., 2008)

	Attributional LCA (ALCA)	Consequential LCA (CLCA)
The question to be answered	What are the total emissions from the processes and material flows directly used in the life cycle of a product?	What is the change in total emissions as a result of a marginal change in the production (and consumption and disposal) of a product?
Application	For consumption-based emissions accounting.	For informing consumers and policy makers on the change in total emissions from a

		purchasing or policy decision.
System boundary	The processes and material flows directly used in the production, consumption, and disposal of a product.	All processes and flows which are directly or indirectly affected by a marginal change in the output of a product.
Double-counting and accounting for absolute emissions	In theory, there would be no double-counting of emissions if ALCAs were conducted for all products using the same guidelines.	If CLCAs were conducted for all products, the sum of the results would be multiple times higher (or lower) than total emissions from consumption.
Marginal or average data	Uses average data.	Generally uses marginal data.
Market effects	Does not consider market effects.	Considers the market effects of the production and consumption of the product.
Allocation methods	Involves allocation.	Involves system expansion.
Non-market indirect effects	Does not consider other indirect effects.	Should include all other indirect effects, such as the interactions with existing policies.
Time-scales, means by which change is promoted, and magnitude of the change	Aims to quantify the emissions attributable to a product at a given level of production at a given time.	Aims to quantify the change in emissions which result from a change in production. It is necessary to specify the time-scale of the change, the means by which the change is promoted, and the magnitude of the change.

The most commonly-cited purpose for doing an ALCA is to simply understand or learn about the impacts of a product without considering indirect effects arising from changes in the output of a product (Tillman, 2000; Brander *et al*., 2008; Earles and Halog, 2011). This modelling usually involves use of more restricted system boundaries and less data collection.

In contrast, CLCA aims to provide information on changes arising as a consequence of an increase or a decrease in demand for the product in question – both inside and outside the foreground system boundaries. For example, if orchard prunings (which are generally mulched) are removed from orchard soils to be combusted for energy generation, the consequences of applying an extra amount of fertilisers to compensate for the loss of nutrients previously provided by mulch would have to be modelled in CLCA in order to demonstrate the change compared with the reference system. Further work is needed to reach consensus on when to use CLCA and how to standardise the procedure (Earles and Halog, 2011).

The identification of affected technologies, collection of marginal data (i.e., which technologies and to what extent they will be affected), and associated uncertainties are of particular concern in CLCA. Furthermore, the reference system against which the system under analysis is compared should be objectively defined in order to avoid biased results. For comparative LCA studies intended to guide future research, environmental policies or business strategy, the study may analyse a decision rather than a single system or process (European Commission, 2010a).

The choice between attributional and consequential LCA can affect the system boundaries defined in the goal and scope phase. In both consequential and attributional LCAs that analyse future systems or technologies, it is pertinent to model "scenarios" defined as "a description of a possible future situation relevant for specific LCA applications, based on specific assumptions about the future, and (when relevant) also including the presentation of the development from the present to the future" (Pesonen *et al*., 2000). The data collected during the inventory analysis must be consistent with the scenarios framed for the study.

Life cycle inventory analysis (LCI)

In the LCI, the inputs (resources) and the outputs (emissions) for each unit process are collected and attributed to the product over its life cycle in relation to a functional unit. Baumann and Tillman (2004) described the following actions as part of an LCI:

1. construction of the flow chart according to the system boundaries chosen in the goal and scope definition;
2. data collection for all the activities in the product system followed by documentation of collected data; and
3. calculation of the environmental loads (resource use and pollutant emissions) of the system in relation to the functional unit.

Construction of a flow chart in LCI

The input and output flows of the system are modelled and represented as a flow chart (Fig. 4). Transport of products and materials may occur between and during each life cycle stage. Each life cycle stage may consist of several unit processes, i.e. the smallest elements in LCI. For example, in an LCA study of biochar, the unit processes present in the manufacture life cycle stage may be: drying and shredding of biomass feedstock, pyrolysis process, and post-production treatment of biochar.

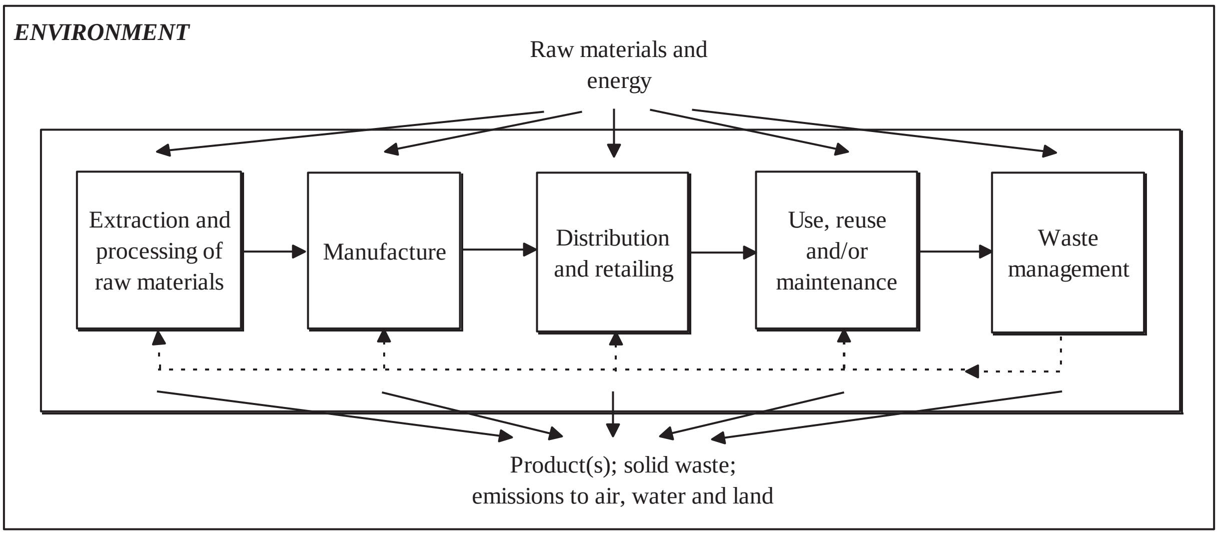

Fig. 4. Generic flow chart for LCA inventory analysis (Cowell, 1998)

In the real world, exploitation of resources and generation of emissions are likely to occur at multiple sites and regions of the world, at different times, and over different time periods, such as from a landfill. In LCA, these environmental exchanges are considered relevant irrespective of their geographical locations and the time period.

Data collection

Data collection demands the most work and time in LCA (Baumann and Tillman, 2004; Rebitzer *et al*., 2004; Finnveden *et al*., 2009) due to the fact that it may be difficult to access relevant data and/or some data may be unavailable. Because of the great efforts that data collection requires, Rebitzer *et al*. (2004) explained that the LCI is usually the focus of attention in streamlined LCA, and that it is simplified by deliberately excluding minor processes from the system (“cut-offs”). ISO (2006b) provides guidance on the use of several cut-off criteria based on mass, energy, and environmental significance. The inputs and outputs contributing less than a specified threshold proportion of the estimated total impact may be excluded. Moreover, the effect of the selection of the cut-off criteria shall be assessed through sensitivity analysis for comparative studies that will be disclosed to the public.

To help facilitate data collection a number of databases have been developed including public national or regional databases, industry databases, and consultants’ databases offered as a package with LCA software. Finnveden *et al*. (2009) listed a number of databases already available, mainly in Europe, USA, Japan, and Australia; others are under development in Brazil, Canada, China, Malaysia, and Thailand. Noticeably, an LCA database in New Zealand was absent from the list.

Data calculation

According to ISO (2006b), calculation procedures include:

- validation of data collected to confirm and provide evidence that the data quality requirements for the intended application of the study have been fulfilled;
- the relating of data to unit processes where an appropriate flow shall be determined for each unit process;
- the relating of data to the reference flow of the functional unit; and
- the refining of the system boundary based on a sensitivity analysis, which determines the significance of data to be included.

Allocation of flows and releases

Industrial processes usually give more than one single output and recycle intermediate or end-of-life products as raw materials. When a unit process has more than one functional flow, i.e., an input or output related to the function provided by the unit process, allocation refers to the issue of partitioning the inputs and outputs for that unit process between the functional flows. Allocation in LCI is one of the most debated methodological issues in LCA (Finnveden *et al*., 2009; Weidema and Schmidt, 2010; Suh *et al*., 2010).

ISO (2006b) suggests a stepwise allocation procedure:

1. if possible, allocation should be avoided by dividing the unit process in question into two or more sub-processes and collecting the input and output data related to these sub-processes, or by expanding the product system to include the additional functions related to the co-products and taking into account ISO requirements;
2. if allocation cannot be avoided, partitioning should reflect physical relationship, i.e., the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system; and
3. if physical relationship cannot be used as the basis for allocation, other relationships, such as economic value can be used instead.

Allocation is accepted in certain applications such as consumption-based emissions accounting (Brander *et al*., 2008), which is merely descriptive and does not imply any change in the quantity of the product produced. However, system expansion is recommended instead to deal with allocation problems (Weidema and Schmidt, 2010). In system expansion, the emissions of the products that could be replaced with the by-products of the product under analysis are accounted for, whereas in allocation these emissions, which are likely to happen in reality, are neglected.

Life cycle impact assessment (LCIA)

This phase connects data from the LCI to specific environmental impact categories and category indicators through the act of classification and characterisation. LCIA consists of mandatory and optional elements (ISO, 2006a). The mandatory elements include:

- selection of impact categories, category indicators and characterisation models;
- assignment of LCI results to the selected impact categories (classification); and
- calculation of category indicator results (characterisation).

LCIA uses technical terminology (Table 5).

Table 5. Terms typically used in LCIA with examples for assessment of climate change (adapted from ISO, 2006b)

Term	Definition and examples for the climate change impact category
Impact category	Climate change, stratospheric ozone depletion, smog formation, eutrophication, acidification, land use, water use, photochemical oxidant creation, human toxicity, ecotoxicity, noise, depletion of biotic and abiotic resources
Functional unit	Quantified parameter of a system for use as a reference unit
LCI results	Amount of a GHG emitted per functional unit
Characterisation model	Model of 100 years of the IPCC
Category indicator	Infrared cumulative radiative forcing (W/m^2)
Characterisation factor	GWP for each GHG (kg CO_2-eq. per kg gas)
Category indicator result	kg of CO_2-eq. per functional unit
Category endpoints	Coral reefs, forests, crops
Environmental relevance	Infrared radiative forcing is a proxy for potential effects on the climate

Impact categories have been organised and divided into three broad categories called "areas of protection": natural environment, man-made environment, and human health (Pennington *et al.*, 2004). LCIA is based on the concept of the cause-effect chain (or environmental mechanism), i.e., the primary effect of a pollutant may be the cause of several secondary effects. For each impact category, the cause-effect chain delineates the pathways between the emissions/resources to the area of protection (Fig. 5). LCIA may focus on the areas of protection (using endpoint modelling and indicators) or earlier in the cause-effect chain (using midpoint modelling and indicators).

Once the relevant impact categories are identified in the study and LCI results are assigned to these through classification, characterisation means that the magnitude of environmental impacts are calculated with the help of equivalency factors defined while modelling the cause-effect chains for each impact category (Baumann and Tillman, 2004). Midpoint characterisation modelling, i.e., choosing an indicator somewhere between emission and endpoint in the environmental mechanism, has been traditionally used (Finnveden *et al*., 2009).

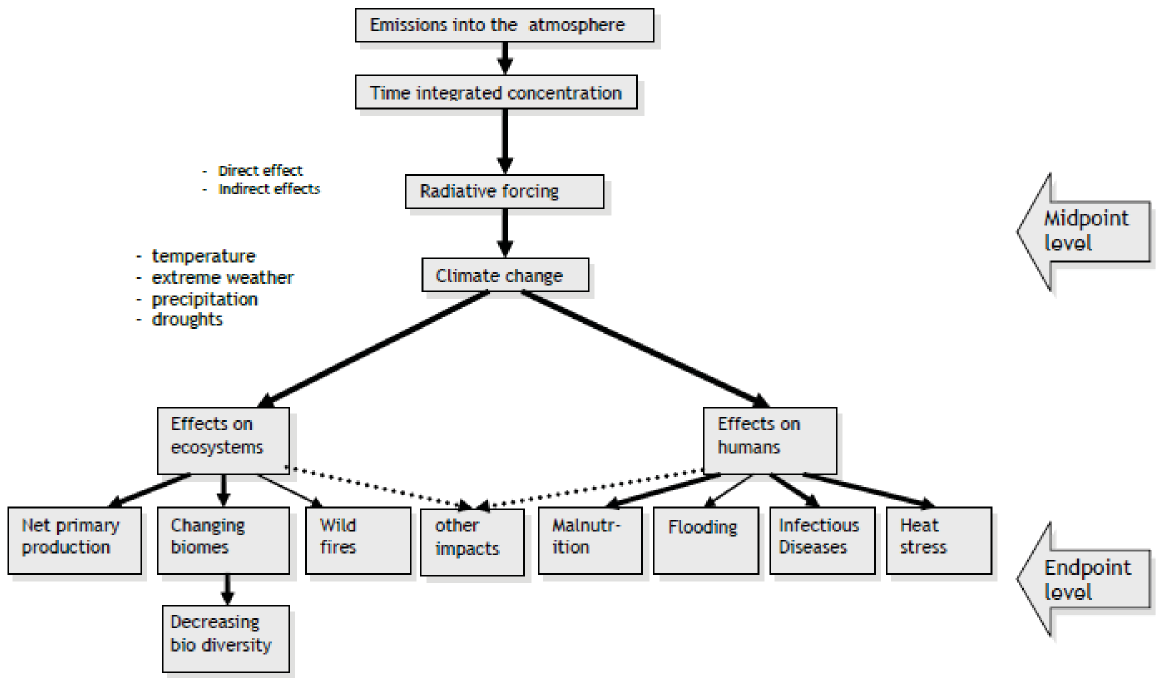

Fig. 5. Cause-effect chain for climate change (European Commission, 2010b)

Depending on the goal and scope definition, optional elements can be added as follows:

- calculation of category indicator results relative to reference information (normalisation);
- arrangement and possible ranking of the impact categories (grouping);
- conversion and possible aggregation of indicator results across impact categories using numerical factors based on value-choices (weighting); and
- better understanding of the reliability of the collection of indicator results (detailed data quality analysis).

Several LCIA methods have been developed and, in spite of similarities between them, the differences in their results can be important (Finnveden *et al*., 2009). Harmonisation of

LCIA methods is underway. For example, the Life Cycle Initiative, a partnership between the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) aims at standardising the impact categories. LCIA is also an iterative process; initial results may lead to review the system boundaries stated in the goal and scope phase.

Life cycle interpretation

Interpretation takes place at every phase in the iterative LCA methodology (Fig. 2). A more specific definition for the fourth phase in an LCA study is given by the ISO standard. "Interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are considered together or, in the case of LCI studies, the findings of the inventory analysis only. The interpretation phase should deliver results that are consistent with the defined goal and scope and which reach conclusions, explain limitations and provide recommendations" (ISO, 2006a).

Baumann and Tillman (2004) defined interpretation in LCA as "the act of analysing and presenting the results". They explained that interpretation should identify and analyse significant issues and test the robustness of the results. For this purpose, a range of analytic strategies exist, such as uncertainty analysis, sensitivity analysis, and variation analysis.

2.3.4. Criticisms of LCA studies

Four common criticisms of LCA studies are bias, time required, variability in methods, and limited inclusion of social and economic impacts. These are briefly discussed below.

Bias

In the past, LCA has been used to mislead consumers about the environmental characteristics of a company and/or product. Baumann and Tillman (2004) mentioned that the term "hired gun" has been used for biased studies that favoured the product assessed by those who sponsored the LCA study. They depicted an example from the late 1980s when an LCA study of a Danish company producing meat packaging trays made of foamed polystyrene showed that their product was environmentally better than other products, whereas a German study claimed that cellulose pulp trays were environmentally superior. Serious concerns about the inappropriate use of LCA in marketing claims led to the standardisation of LCA methodology in the mid 1990s, and publication of the ISO LCA standards.

However, even with LCA standards in place for several years, some people still argue that LCAs can be biased. Recent advertisements from the beverage packaging industry illustrate this point. Bio Intelligence Service (2007), an environmental research and consultancy firm, conducted a "cradle-to-grave" LCA study for Tetra Pak showing that carton had a lower global warming potential (GWP) than polyethylene terephthalate (PET) and glass containers in France. A couple of years later, Franklin Associates (2009) carried out a "cradle-to-grave" life cycle inventory (LCI) for the PET Resin Association (PETRA) in the USA. They compared PET containers with aluminium cans and glass bottles. In 2010, PETRA communicated to the public that a life cycle study demonstrated that PET bottles outperform aluminium and glass containers as far as the environmental footprint is concerned (Byrne, 2010). The Aluminum Association (2010) questioned the transparency and reliability of this LCA study.

More recently, Owens-Illinois (O-I), the world's largest producer of glass packaging, conducted an LCA study where they compared glass with aluminium and PET containers in the USA. They claimed that glass has the most favourable carbon footprint (O-I, 2010). The "cradle to cradle" LCA study of O-I was tested and validated by AMR Research, a supply chain research firm. All the companies argued that they strictly followed LCA methodology. The contrasting results of these LCAs are confusing to the public and put the credibility of LCA methodology in jeopardy.

Time required

In some applications, the time, costs, and human efforts spent on a detailed LCA study may not correlate with the usefulness of the results (Rebitzer *et al*., 2004). "Some have questioned whether the LCA community has established a methodology that is, in fact, beyond the reach of most potential users" (Todd and Curran, 1999). When decisions are to be taken fast, the LCA methodology is sometimes simplified into what is termed "streamlined LCA" (Rebitzer *et al*., 2004); the results of such studies, however, might be prone to criticisms such as masking the disadvantages of a product by omitting consideration of certain processes.

Variability in methods

In LCA, the results depend on the questions the LCA practitioner asks and what assumptions and methods are chosen throughout the whole LCA study. According to Reap *et al*. (2008), multiple issues arise at each of the four phases of LCA (Table 6), which can improve or reduce the accuracy of this methodological tool.

Table 6. Issues arising at each of the four phases of LCA methodology (Reap et al., 2008)

Phase	Issue
Goal and scope definition	Functional unit definition Boundary selection Social and economic impacts Alternative scenario considerations
Life cycle inventory analysis	Allocation Negligible contribution ('cut-off') criteria Local technical uniqueness
Life cycle impact assessment	Impact category and methodology selection Spatial variation Local environmental uniqueness Dynamics of the environment Time horizons
Life cycle interpretation	Weighting and valuation Uncertainty in the decision process
All	Data availability and quality

Limited inclusion of social and economic impacts

The focus of LCA has traditionally been restricted to the environmental impacts of products (ISO, 2006a). From a sustainable development perspective, the exclusion of the social and economic impacts of products "can limit the capability of LCA to support decisions" (Reap *et al*., 2008). Social LCA is in an early stage of development. The Guidelines for Social Life Cycle Assessment of Products from the task force under the UNEP-SETAC Life Cycle Initiative (Benoît and Mazijn, 2009) "provides a map, a skeleton and a flash light for stakeholders engaging in the assessment of social and socio-economic impacts of products life cycle”. Integrating these aspects into LCA is difficult though (Jorgensen *et al*., 2010).

2.3.5. Life Cycle Assessment (LCA) of biochar systems

Biochar systems have been evaluated from a life cycle perspective (Gaunt and Lehmann, 2008; McCarl *et al*., 2009; Gaunt and Cowie, 2009; Woolf *et al*., 2010; Shackley *et al*., 2012). However, evaluation of biochar systems with LCA methodology is recent and heterogeneous due to the highly-contextual characteristics of biochar production and application. With the exception of two LCA studies that included more impact categories than climate change (Sparrevik *et al*., 2012; McDevitt *et al*., 2013), the rest of the studies that encompassed biochar production focused mainly on the carbon footprint (CF) of the food, agro-fuel, or end-of-life biomass management systems under analysis (Roberts *et al*., 2010; Yu and Wu, 2010; Kameyama *et al*., 2010; Hammond *et al*., 2011; Cooper *et al*., 2011; Kauffman *et al*., 2011; Ibarrola *et al*., 2012; Ahmed *et al*., 2012; Mattila *et al*., 2012; Cao and Pawłowski, 2013; Field *et al*., 2013; El Hanandeh, 2013; Han *et al*., 2013; Huang *et al*., 2013; Lugato *et al*., 2013).

Different configurations of biochar systems can be assessed depending on scale (small, medium or large); location (fixed or mobile); type of production process (slow/fast pyrolysis or gasification); end-use of biomass (fuel or soil amendment); main system under analysis (food, fuel or waste management); and alternative systems for comparison. Using an LCA approach, the methodological issues associated with modelling of biochar systems concern the goal, scope and decision-context of the study; definition of the functional unit; recognition of multiple functions; selection of system boundaries and allocation approach; choice of impact categories; indirect consequences; and reference scenario with which the biochar system is compared. Each of these variables is described below.

Goal, scope and decision-context

The goal, scope and decision-context of an LCA study has been explained in section 2.3.3. Briefly, the goal of an LCA study includes stating the intended application, the reasons for carrying out the study, and the targeted audience (ISO, 2006a). Moreover, the decision-context shall be identified. If the LCA study is intented to support a decision, then the potential consequences of this decision should be identified. The decision-context of the study may lead to undertaking attributional or consequential LCA modelling, which can affect the results.

Definition of the functional unit

The definition of the functional unit is concerned with the service provided by the system and may vary according to the decision situation. In LCA studies of slow pyrolysis biochar systems, the functional unit may refer to the management of an area of land (where biomass feedstock is harvested or where biochar is applied); a quantity of biomass feedstock processed; or a quantity of biochar produced. For example, if biochar application into soils were to be compared with otherwise using the char to replace coal in steel making then, in this case, the most pragmatic functional unit would be 'the management of one tonne of char produced'. For slow pyrolysis biochar systems, Hammond *et al*. (2011) found that the climate-change mitigation potential per unit of energy delivered (t CO_2-eq/MWh) is not an appropriate unit since biochar (and not energy production) is the main product of the system. The functional unit referring to the energy delivered by the system, however, can apply to fast pyrolysis systems since energy (and not biochar) is the main product. Therefore, the multiple services provided by biochar systems must be recognised and the functional unit should be consistent and comparable among alternatives including the reference scenario.

Recognition of multiple functions

Biochar systems can deliver multiple functions, such as soil improvement; end-of-life biomass (ELB) management; energy production; waterways protection; and climate-change mitigation. For carbon accounting methodologies, the focus is on climate-change mitigation. However, concomitant services (e.g. energy generation) and the magnitude of the services (e.g. MJ) provided by biochar systems are very likely to be different across the different systems under analysis. Therefore, in order to make these functions comparable, LCA of biochar systems should consider, on a case-by-case basis, the functions supplied by the alternative systems. This can be done through product substitution and/or system expansion (see below).

Selection of system boundaries and allocation approach

The life cycle of a biochar system starts with the sourcing of the biomass feedstock and finishes with the effects of biochar application into soils. So, this pathway is generally what is included in the foreground system. GHGs emitted due to project implementation within the foreground system need to be calculated. If the feedstock is ELB, activities needed to produce it are usually not taken into account because ELB tends to be considered as a minor co-product relative to the main product or even as a waste stream. However, biochar produced from purpose-grown crops must include processes required for the production of the feedstock.

When comparing a biochar scenario with an alternative use of biomass feedstock (or char), the functions provided by both scenarios need to be equivalent. For this task, product substitution or system expansion can be conducted. In the case of product substitution, the magnitude of the service provided (e.g. MJ of energy delivered) can be included in the background system of the biochar scenario and the impacts arising from this process can then be subtracted from the foreground system. This form of product substitution would result in a C balance for the biochar scenario relative to the reference scenario. This approach, however, can mask the C sequestration potential of biochar systems if only the net result is presented and especially if long-term CO_2 removals are considered equivalent to avoided CO_2 emissions from fossil fuel combustion.

Slightly different but notably important for LCA of biochar systems, the reference scenario can be expanded and the equivalent services provided by the biochar scenario can be added to the background system of the reference scenario. Performing system expansion on the reference scenario would then allow the C balance of the biochar scenario to appear in absolute terms. System expansion of the reference scenario will then show that C sequestration in the form of biochar occurs regardless of the activities assumed to be displaced and will therefore confirm that biochar systems are carbon negative.

Furthermore, to account for GHG emission reductions achieved through factors such as avoiding fossil fuel combustion, reducing fertiliser use and avoiding methane release from biomass decomposition, the biochar scenario must be compared with a reference scenario, which assumes that GHGs would be avoided as a result of the implementation of the biochar scenario. The reporting of this CF will now be in relative terms, i.e. in relation to a reference.

The time horizon considered in LCA studies of biochar is also important for the definition of the system boundaries. In existing LCA studies of large-scale slow pyrolysis biochar systems (Roberts *et al*., 2010; Hammond *et al*. 2011; Ibarrola *et al*., 2012), the time scale of the decision to use a certain type of biomass for biochar production has been 20 years, whereas the time scale for biochar-C sequestration has been 100 years. In order to account for soil-related impacts other than C sequestration, it has been recently proposed to use the timeframe of 100 years (Verheijen *et al*., 2012). This proposition implies that the LCA practitioner should deal with large uncertainties and speculate about possible life cycle impacts because empirical evidence of the effects of biochar into soils cannot still be obtained for such a timeframe. Sensitivity analyses should then include conservative assumptions. Since the selection of the system boundaries depend on how the functional unit is defined and on the compared alternatives, which may not extend for 100 years, a time-dependent analysis may be required in the future. However, that is beyond the scope of this dissertation.

Choice of impact categories

The most important environmental impact categories that can be evaluated in LCA studies of biochar systems may include climate change; acidification; eutrophication; toxicological stress on human health and ecosystems; depletion of resources; water use; and land use. The choice of including certain impact categories will depend on the goal and scope definition and the type of system modelled. Since pyrolysis biochar systems are not widespread and a number of uncertainties about the impact of biochar application into soils exist, data are currently lacking for all impact categories.

Inclusion of indirect consequences

Indirect consequences associated with the implementation of biochar systems may be positive or negative with respect to climate change. These can be termed carbon leakage. In terms of GHG accounting, positive carbon leakage may include fertiliser and lime savings; avoidance of methane emissions due to the former anaerobic decomposition of biomass if this was not used for biochar production; displacement of fossil fuel and/or electricity formerly used for energy generation; and electricity saved due to less irrigation of crops. Conversely, negative carbon leakage may include displacement of biomass used as fertiliser, feed or fuel; land-use change; supplementary irrigation following biochar additions; and additional agricultural inputs required such as fertilisers and herbicides. Changes in the C balance of the system due to these indirect consequences can be represented through system expansion and/or product substitution.

Definition of the reference scenario

The definition of the reference scenario is usually done by assuming that current management practices, i.e. the business-as-usual scenario, will continue to the same point in the future as the proposed alternatives including the biochar scenario. However, biochar systems can deliver multiple functions such as energy production and soil improvement, among others. Therefore, if these multiple services are assumed to displace activities that formerly generated environmental impacts then the reference scenario would have to include these activities. This can be done by expanding the business-as-usual scenario in order to add background processes that would be assumed to be displaced as a consequence of implementing the biochar scenario. However, when system expansion/substitution is performed on the biochar scenario, the definition of the reference scenario may not be explicit since displaced and/or additional processes required to supply equivalent functions will be shown in the background system of the biochar scenario and not in the reference scenario. If consistency is maintained and long-term CO_2 removals continue to be considered equivalent to avoided fossil fuel GHG emissions, both ways of defining the reference scenario (system expansion in the reference scenario or system expansion/substitution in the biochar scenario) will lead to the same overall results.

2.4. Carbon markets

For biochar to be widely produced and applied into soils it must be economically feasible. The economic attractiveness of biochar could arise from a combination of factors such as the price of by-products in the energy and chemical industries, the value of biochar as a soil amendment, the cost of managing certain end-of-life biomass streams, and the value of carbon mitigated. This study focuses on the latter.

Greenhouse gas (GHG) mitigation policies include emission taxes, subsidies, technology and performance standards, voluntary measures (e.g. carbon labelling), and emissions trading (Gerber *et al*., 2010). Despite the number of uncertainties in biochar research (such as stability; interaction with different soil ecosystems; potential negative consequences; and lack of carbon accounting, validation, monitoring, reporting and verification methods), a number of researchers advocate for the inclusion of biochar in carbon markets (Ogawa *et al*., 2006; Lehmann *et al*., 2006; Gaunt and Lehmann 2008; Gaunt and Cowie, 2009; Laird *et al*., 2009; Roberts *et al*., 2010).

Carbon markets have been the object of serious criticisms (Lohmann, 2001; Lohmann, 2006; Pearson, 2007; Driesen, 2008; Wittman and Caron, 2009; Bullock *et al*., 2009; Lohmann 2009a; Gilbertston and Reyes, 2009; Böhm and Dabhi, 2009; Hansen, 2009; Docena *et al*., 2010; Spash, 2010; Bertram and Terry, 2010; Randalls, 2011; Paterson and Stripple, 2012; Descheneau, 2012; Bond, 2012; Childs, 2012; Böhm *et al*., 2012). The proposal to make biochar eligible in carbon markets has resulted in opposition to biochar by certain groups of civil and non-profit organisations with 'Biofuelwatch' being the most forceful. Others have argued that application of biochar in soils could work but not if linked to carbon markets (Bruges, 2009). There is an obvious need to analyse carbon markets before assessing the feasibility of biochar to become eligible for carbon finance.

2.4.1. Introduction to carbon markets

To minimise the impacts of climate change, humanity needs to stop its dependence on fossil fuels and move to more sustainable forms of energy as soon as possible. Arguably, it is believed that carbon markets could help in achieving this transition (World Bank, 2010). As an alternative to carbon markets, the carbon tax seems more difficult to implement based on the unpopularity of taxes (Schimmoller, 2007). However, it has also been claimed that a carbon tax could lower GHG emissions faster and at lower costs to the public than carbon markets (Wittneben, 2009). While the debate on a carbon tax is open, this study focuses on carbon markets since they are currently the preferred method of governments to address climate change.

Carbon markets are very complex systems, which, in essence, allow polluters to continue polluting by paying someone else to reduce or sequester carbon emissions at different spatial and temporal conditions in compensation for their emissions. Based on the so-called atmospheric carbon equivalence, social and environmental issues (e.g. water, biodiversity) at the local level are neglected for the purpose of trading an intangible commodity.

2.4.2. Brief history of carbon markets

The most powerful barrier to set comprehensive climate change policies is the close relationship between governments and multinational corporations:

- large polluters spend extraordinary amounts of money in lobbying against regulations;
- they lobby hard for choosing carbon trading over a carbon tax; and
- they weaken the already-flexible rules in carbon trading (Lohmann, 2006; Hansen, 2009; Bertram and Terry, 2010; Spash, 2010).

The Kyoto Protocol, the first international climate change treaty signed in 1997, was flexibly skewed in the interest of corporations and is regarded as the cornerstone of carbon trading. Fifteen years later, in 2012, the first commitment period of the Kyoto Protocol came to an end and a new legally binding treaty is planned for all countries by 2015, to become effective after 2020 (Metz, 2013).

The development of the Kyoto Protocol was reported by Michael Zammit Cutajar, former Executive Secretary of the UNFCCC:

“The sensitivity of the (Kyoto) Protocol to the market was largely instigated by the negotiating positions of the USA ... For example, the European Union – now fully committed to emission trading – was insistent (at first) that trading should be supplementary to domestic action to limit emissions, the latter seen as essential to the development of technologies that would open the way to a low-carbon future. The EU also frowned upon recourse to “sinks” for the same reason and because of the uncertainties surrounding that option. Yet these were among the final make-or-break issues for the US negotiators and it is not an exaggeration to brand the mechanisms of the Kyoto Protocol as ‘Made in the USA’” (Ainsworth, 2005).

Based on neoliberal economics, pollution trading was introduced into the Clean Air Act in the USA during the 1970s and became attractive until the mid-1980s when the lead-in-gasoline phase-down programme was considered successful (Ellerman, 2007). The sulphur dioxide (SO_2) trading, also known as the Acid Rain Programme, built on this experience and has been claimed the “poster child” of GHG emissions trading (Ellerman, 2007). Gilberston and Reyes (2009) contested this claim by saying that the North American context of controlling SO_2, a single substance emitted from a relatively small number of large fixed facilities already regulated in one developed country, cannot be compared with tackling climate change. Moreover, SO_2 trading did not include offsets (i.e. reductions outside the capped industry that are allowed to be traded for compensating for emissions released within the capped industry).

The notion of privatising the atmosphere by giving property rights (permits) to polluters who then can make business out of them in the market is argued to be a product of neoliberalism (Lohmann, 2006). In general, neoliberalism has a negative connotation in the literature of “creative destruction” (Harvey, 2006), leading to one of the criticisms to carbon markets. According to Harvey (2006), neoliberalism refers to “a theory of political economic practices which proposes that human well-being can best be advanced by the maximization of entrepreneurial freedoms within an institutional framework characterized by private property rights, individual liberty, free markets and free trade”.

Free markets can stimulate creation and innovation. They are believed to promote efficiency by letting consumers and producers agree on the price of a commodity (McFadden, 2006). Moreover, “they are a constant source of negative (sometimes irreversible) externalities which affect the existence of groups whose interests are not taken into consideration; they can do nothing or next to nothing about income inequalities; and they are not the best solution to guarantee everyone’s access to certain goods such as healthcare” (Callon, 2009).

Ochoa (2001) argued that the effects of neoliberalism have been well documented and include “severe cuts in social spending, the gutting of protective legislation, heightened foreign investment and privatization, growing income inequality, and the expansion of crime and violence”. He claimed that “neoliberalism is an attempt to realign and strengthen capitalist imperialism using the tools of the World Bank and the International Monetary Fund (IMF)”.

A memorandum that former World Bank chief economist Lawrence Summers wrote to his colleagues in 1991 is commonly cited to illustrate the economic reasoning behind carbon markets (Lohmann, 2006; Gilbertson and Reyes, 2009). The memorandum leaked. “I think the economic logic behind dumping a load of toxic waste in the lowest-wage country is impeccable and we should face up to that” and “I’ve always thought that under-populated countries in Africa are vastly underpolluted”, Summers wrote in the memorandum (Foster, 1993).

In 1992, the World Bank and the Norwegian government started financing a number of activities involving “carbon offset generation” (Gilbertson and Reyes, 2009). In 1995, this pilot work led to the establishment of the voluntary programme Activities Implemented Jointly (AIJ), which later became part of the Kyoto Protocol (Bumpus and Liverman, 2008; Larson and Breustedt, 2009). It is important to highlight that during the first year of the

Kyoto Protocol’s commitment period, the World Bank increased its loans for fossil fuel development by 94% –coal lending alone increased 256% – reaching over US$3 billion, whereas only US$476 million went to support “new” renewable energy technology (Redman, 2008). The World Bank, one of the major supporters of carbon trading, has been criticised for being inconsistent.

In 1997 during the early negotiations of the Kyoto Protocol, the majority of developing countries were not keen on this offsetting approach (Gupta, 2001). In particular, the Brazilian government took an official stand against AIJ projects based on the following concerns summarised by La Rovere (1998):

- the threat of continued increase in global GHG emissions as developed countries receive emission permits from investing in abatement projects in less developed countries;
- the problem of investor countries taking advantage of least expensive abatement options in recipient countries, leaving the recipient countries with only costly options to fund on their own in the future;
- the uncertainties associated with defining baselines;
- the difficulty of monitoring whether “avoided” GHG emissions have simply been shifted to other locations (positive carbon leakage);
- the difficulty of calculating the cost effectiveness of an abatement project;
- the difficulty of quantifying secondary social, environmental, and economic impacts of a project;
- the difficulty of verifying that abatement projects were not business as usual (additionality);
- the problem of recipient countries becoming dependent on investor countries for technology;
- the need to develop a management structure for abatement projects in recipient countries; and
- the risk of a country’s loss of sovereignty over territory that is set aside as a preserve in order to reduce GHG emissions.

As a parallel proposal to AIJ, the Brazilian government promoted a "Clean Development Fund" (Gupta, 2001), whereby money would be collected through fines imposed on industrialised countries that failed to comply with their GHG targets and then used to finance clean technologies in the global South. The Brazilian proposal was endorsed by the G-77 nations plus China; however, at the initiative of the USA and amid international disagreements during the last days of the intense Kyoto negotiations, the fund was transformed into the clean development mechanism (CDM) (Lohmann, 2006; Gilbertson and Reyes, 2009). Given the way it was introduced in the negotiations, this political twist is known as the "Kyoto surprise" (Werksman, 1998). "Fines were transformed into prices; a judicial system was transformed into a market" (Lohmann, 2006). Carbon markets were born. They are usually divided into two categories: 'cap and trade' and 'offsetting'.

2.4.3. Cap and trade

National governments or intergovernmental bodies (e.g. UNFCCC, European Commission) set a limit and hand out pollution permits to major industries. One polluter can, instead of tightening the belt, trade these permits with another who might make 'equivalent' changes more cheaply. The Kyoto Protocol includes emissions trading (ET) as a flexible mechanism and allows countries that have spare assigned amount units (AAUs) to sell this surplus to countries that are over their caps during the first commitment period (2008-2012). AAU trading has received little attention, whereas the European Union's emission trading scheme (EU ETS) is the world's largest carbon market and serves as a model for similar cap and trade schemes around the world.

The EU ETS was launched in 2005 to assist Member States in meeting the EU target defined in the Kyoto Protocol. The 15 States who were EU members in 1990 have different individual targets, but which combined make an overall target of reducing GHG emissions by 8% during 2008-2012 compared to 1990 levels. New Member States have individual targets except Cyprus and Malta which, as non-Annex I countries in the Kyoto Protocol, have no legal commitments.

There are about 11,500 power stations, factories and refineries covered by the EU ETS in the EU 27 member countries, plus Norway, Lichtenstein, and Iceland. These represent close to half of the EU's CO_2 emissions, including stationary emission sources. Direct emissions from road transport, aviation, shipping, forestry, and agriculture are not included (Gilbertson and Reyes, 2009).

The EU ETS has been criticised for handing out free permits to the most polluting companies based on historical GHG emissions in an approach known as 'grandfathering' (Martinez and Neuhoff, 2005). For the electricity sector, this was translated into major profits since they passed on the costs to consumers by raising electricity bills (Sijm *et al*., 2006) but these costs did not reflect the true cost of permits, but rather what electricity companies speculated the permits would cost (Gilbertson and Reyes, 2009).

In the first phase of the EU ETS (2005-2008), power companies in the UK were estimated to make windfall profits from the EU ETS amounting to £500 million a year or more, while in Germany, Eon, RWE, Vatenfall and EnBW would make profits of between €6 billion and €8 billion (Gilbertson and Reyes, 2009). In the second phase of the EU ETS (2008-2012), this problem was not corrected and windfall profits in this phase could reach €23 to 63 billion (Coelho, 2009).

Besides the power industry, polluters in other sectors (e.g. heat generation, pulp and paper, steel, cement and lime, oil refineries and metals) were granted an excess of permits. In total, over-allocation in the first phase of the EU ETS made the carbon market collapse with the permit price dropping to zero (Clò, 2010). The caps were lowered in the second phase but the price collapsed again during the most recent financial crisis. The EU ETS has been argued to fail to mitigate GHG emissions (Coelho, 2009; Gilbertson and Reyes, 2009). More recently, the EU committee has supported a "trading fix" to keep afloat the EU ETS (Williamson, 2013b).

For the third phase (2013-2020), it is planned that the cap will be tightened but even before its beginning, it was already criticised because surplus permits from phase two will still be tradable (Gilbertson and Reyes, 2009). It has been claimed that approximately 40% of what would be required to meet the cap of phase three could be provided by the banking of permits from phase two (Gilbertson and Reyes, 2009).

Furthermore, carbon leakage is a relevant topic. In the EU ETS, leakage from the European Union to countries, such as China, where pollution standards are less strict is a potential problem if the price of carbon becomes very high (Helm, 2009). Clò (2010) explained that ETS regulates the emissions linked to the production (rather than consumption) of goods within the EU. This means that the production of goods is the baseline scenario and, as such, polluters can meet their caps just by switching production activities to outside of Europe. Clò (2010) argued that leakage has already happened to some extent as imports from non-EU countries have increasingly replaced European production during the last decades.

In addition, the use of emission reductions issued outside the EU to comply with European GHG targets is permitted, and has been a controversial debate since the beginning of the EU ETS process (Flåm, 2009). These external carbon credits tend to be cheaper than European reductions and can prevent improvements in the domestic sectors. Since the EU expects that there will be an abundant supply of cheap credits in 2020, it has now set limits on the amount of credits that can be imported (Tol, 2009). Moreover, offseting is far from being free of criticisms.

2.4.4. Offsetting

Offsetting can take place through compliance and voluntary carbon markets.

Compliance offsetting

The two offsetting mechanisms of the Kyoto Protocol are: the clean development mechanism (CDM) and joint implementation (JI). Under the Kyoto Protocol, industrialised (Annex I) countries can meet their GHG commitments by purchasing Certified Emission

Reductions (CERs) from CDM projects based in developing (non-Annex I) countries and/or Emission Reduction Units (ERUs) from JI projects implemented in other Annex I countries.

Despite the term ‘reduction(s)’ embedded in the acronyms, an offset is not a reduction. An offset allows a party that is required to reduce its emissions to instead continue emitting while paying another party to undertake emission reductions elsewhere. This has been proven to be unsuccessful in numerous situations. Spash (2010) argued that carbon offset credits “could just as sensibly be called certified ‘emissions increase’ units”.

Most of JI projects take place in countries catalogued ‘economies in transition’, such as Russia, Ukraine, and Central and Eastern Europe, which tend to be the cheapest Annex I nations to host them, although JI projects have also seen activity in Germany, France, Sweden, and New Zealand. The main concerns about JI revolve around the risk of double counting (as a domestic reduction and as a reduction exported to offset emissions), especially within the EU ETS.

The CDM was built upon various factors that require careful assessment prior and during the implementation of the project such as additionality, sustainable development, local capacity building, equity issues, and public perception (Anaya de la Rosa, 2006). Furthermore, leakage (GHG emissions arising outside the project boundary as a consequence of project implementation) has proven to be significant (Schneider *et al*., 2010). In a considerable number of CDM projects and case studies, these factors have been overlooked (Lohmann, 2006; Pearson, 2007; Wittman and Caron, 2009; Bullock *et al*., 2009; Gilbertston and Reyes, 2009; Böhm and Dabhi, 2009; Lejano *et al*., 2010; Docena *et al*., 2010). This has “provoked strong, if diverse and confused, movements of societal self-defence” (Lohmann, 2010). Although active demonstrations are not uncommon in the development of CDM projects, “no CDM Project Design Document ever mentions community protests” (Lejano *et al*., 2010).

According to the World Bank (2010), CDM/JI projects can be successful if the following features exist: (i) a "committed champion" within the local company or government; (ii) a strong project design and planning from the beginning; (iii) solid project financing; and (iv) clear potential to reduce GHG emissions.

Voluntary offsetting

Similar to CDM but mostly with no governmental or intergovernmental control, voluntary carbon markets have been labelled the "wild west" of carbon trading (Nerlich and Koteyko, 2010; Böhm and Dabhi, 2009). Due to false claims about emission reductions and contributions to sustainable development, numerous standards have been developed in order to provide more confidence to the consumer. Recently, about 28 different voluntary standards were identified (Peters-Stanley and Yin, 2013). In 2012, the voluntary carbon markets transacted approximately 101 million tonnes of carbon credits at a total market value of US$523 million.

Just as for the CDM, the voluntary markets raise the same concerns of additionality, sustainable development, leakage, local capacity building, equity issues and public opinion, but also raises issues related to "motivation, ethical behaviour and social psychology" (Spash, 2010) in the minds of individuals who may have the good intentions to voluntarily do something about their emissions. Crowding out environmental motivations is possible since the efforts of people deciding not to pollute in the first place may be 'offset' by polluters who pay someone else to compensate for their actions.

Voluntary offsets are often compared with the indulgences sold by the Catholic Church in the medieval times (Spash, 2010). These moral sin offsets insinuated a kind of relief in the rich person's conscience and supposedly less time in purgatory without the requirement of questioning oneself, dealing with the problem or actually refraining from sinning. 'Cheatneutral.com', a British website, mocks carbon offsetting and invites cheaters to offset their infidelity by paying a fee that would be directed to loyal couples in return for psychological absolution of guilt. This would not only encourage cheating but would distract couples from taking appropriate measures to address the causes of cheating. Carbon offsetting is argued to be a distraction from advancing to a more sustainable society (Bachram, 2004; Spash, 2010).

2.4.5. Carbon markets and biochar

The production and application of biochar into soils could contribute to climate-change mitigation in different ways and therefore carbon markets are one option to channel funding to biochar projects. To become accepted, biochar advocates will have to address the following aspects:

- sustainability issues;
- public perception issues;
- additionality;
- proof of permanence;
- leakage;
- recognised carbon accounting, validation; monitoring, reporting, and verification methodologies; and
- compatibility with other type of credits.

Sustainability issues

All projects in carbon markets should promote sustainable development. If negative consequences are likely to arise, then project developers must take them into account and avoid them. Depending on specific local conditions, biochar projects may have detrimental effects on biodiversity and local populations due to the change of land use and crop production choices, which could result in social conflict (Ernsting and Smolker, 2009).

Public perception issues

Public perception is an essential factor to consider when developing a carbon project. Stakeholder meetings should take place prior to the commencement of any carbon project in order to collect and consider feedback from the community involved. Attitudes toward biochar are divided and can play a major role in the acceptance of biochar. Biochar advocates and pyrolysis companies push for the inclusion of biochar in carbon markets, whereas civil and non-governmental groups have signed a declaration to "keep biochar and soils out of carbon trading" (Rainforest Rescue, 2009). This declaration was signed by approximately 150 organisations. A Biofuelwatch briefing paper states (Ernsting and Smolker, 2009):

"Lobbying is underway for a massive scaling up of biochar production, and yet there is little to substantiate the many proclaimed benefits. It is critical that we address this issue with caution, especially given the many dire consequences associated with any technology that involves large biomass demand and manipulation of poorly understood soil ecosystems!"

Additionality

A project in carbon markets is additional if GHG emissions are reduced below those that would have occurred in a baseline scenario. On the one hand, studies have claimed that the economic viability of biochar projects highly depends on the revenue from carbon finance (Gaunt and Lehmann, 2008; McCarl *et al*., 2009; Roberts *et al*., 2010) suggesting their additionality. On the other hand, others claim that a biochar scheme "would need no subsidy: the farmer would make a profit" (Bruges, 2009, quoting James Lovelock, the author of the Gaia Theory). Baum and Weitner (2006) declared that "production and application costs of biochar may be fully recovered, even in the absence of a carbon market, based solely on crop production benefits and fertilizer cost savings". Sohi *et al*., (2009) stated that "for any biochar scenario it is possible that the agronomic value for biochar is sufficient to render the economic evaluation positive, without resorting to carbon markets or Government incentives." If these statements become reality and biochar turns out to be the powerful ingredient in soil management, then it should not be used as an additional permit to continue polluting through offsetting mechanisms.

Proof of permanence

Demonstrating permanence is a significant challenge for biochar projects. Concerning C sequestration in soils, the variation of labile fractions of different biochars under different soils and climates make it difficult to estimate an accurate stability of biochar. By comparison, temporary credits from forestry projects have primarily seen action in the voluntary carbon markets and have been traded at a negligible value because of the lack of compatibility reduces their attractiveness (Ristea and Maness, 2009).

Leakage

In carbon markets, leakage is defined as the net change of anthropogenic emissions by sources of GHG which occurs outside the project boundary, and which is measurable and attributable to the project activity (UNFCCC, 2012a). This definition is open to interpretation. It begs the question: ‘Who does the measuring and attribution, under whose rules and why?’ Leakage, just as baseline and project scenarios, is a hypothetical concept. In practice, project developers appear to generally dismiss leakage on the basis that it is difficult to calculate (Boyd *et al*., 2007; Millard-Ball and Ortolano, 2010).

The displacement of biomass used for energy to produce biochar could be seen as leakage (Bruun and Luxhøi, 2008). Gaunt and Cowie (2009) suggested another type of leakage concerning the production of biochar from the removal of crop residues left on the ground, which deprives soil of C build up. Furthermore, GHG emissions related to land use change is another kind of leakage that a biochar project could cause. For example, Roberts *et al*. (2010) hypothesised that the diversion of annual crops to perennial grass energy crops for biochar production would cause land conversion to cropland to replace the crops lost to biochar feedstock, and this would reduce the climate change benefit of the biochar system.

Recognised carbon accounting, validation, monitoring, reporting and verification methodologies

Currently there are no approved carbon methodologies for biochar projects. Carbon Gold (2009), a British company, presented a general biochar methodology to the Verified Carbon

Standard (VCS), formerly the Voluntary Carbon Standard Association, but it did not pass its double approval process. During the call for public inputs, there were nine submissions from stakeholders including the New Zealand Biochar Research Centre.

GHG emissions that could be avoided in a baseline scenario due to the combustion of bio-oil and pyrolysis gas for energy production are eligible and therefore, do not face major methodological barriers in carbon markets. Emission reductions attributed to the displacement of fossil fuels with liquid bio-oil produced from pyrolysis are accepted. If the pyrolysis gas and resulting heat are used to meet the energy needs of the pyrolysis plant, then the net GHG emissions associated to the combustion of the pyrolysis gas would be part of the project and would not represent any additional emission reductions. However, if the pyrolysis gas (or heat) is used to displace fossil fuels outside the project's boundary, then these reductions can also be recognised. If the feedstock is hypothesised to decompose anaerobically in the baseline scenario (Gaunt and Cowie, 2009), then the amount of CH_4 emissions avoided due to the production of biochar could also be claimed. Carbon accounting and monitoring methodologies for most of these activities are available at the CDM website.

Calculation of the emission savings that could be associated with the application of biochar into soils remain one of the greatest challenge that biochar faces in carbon markets. This is due to the large uncertainties concerning biochar-soil dynamics. Ogawa *et al*. (2006) and Gaunt and Cowie (2009) have highlighted the lack of methods for monitoring and verifying biochar projects.

Compatibility with other type of credits

The potential of biochar to sequester C in soils on a long-term basis has been explained (Lehmann, 2007). This form of C sequestration is rather unique and cannot be compared to other types of credits currently available in the market since it removes CO_2 from the atmosphere on a long-term basis. Some carbon credits refer to the avoidance of GHG emissions by displacing fossil fuels, whereas others stand for temporary C sequestration (e.g. forestry projects). The incompatibility between credits makes the price of temporary credits diminish (Ristea and Maness, 2009). Moreover, carbon credits issued to projects concerning avoided deforestation are also different and compatibility is a major concern (Neeff and Ascui, 2009). Further consensus on the need to distinguish biochar from other types of carbon abatement technologies is required to address the issue of compatibility for biochar credits.

2.4.6. An alternative approach

It seems that it would take many years to research and assess the considerable amount of uncertainty preventing biochar from being included in carbon markets. In this sense, an alternative way called "Carbon Maintenance Fee" (CMF), originally proposed by the Irish government, has been anticipated to finance biochar projects (Bruges, 2009). Under the CMF proposal, countries would be paid an annual fee based on the amount of C pools in plants, soils, and roots within their borders. The measurement and monitoring of these carbon pools would be done through remote sensing.

Bruges (2009) mentioned that a global soil database has been developed by the UN Food and Agriculture Organisation (FAO) and that experience from the Land Use and Carbon Analysis System (LUCAS) developed in NZ could be used as a model to other countries. The LUCAS programme was launched in NZ to account, monitor, and report land-use change and forestry activities in order to meet its Kyoto obligations. It involves mapping land use at 1990 and land-use changes for 1990-2007 and 2008-2012. The mapping has included extensive use of satellite imagery, some aerial photography and other spatial data.

Biochar benefits, if reflected on satellite imagery, could be measured on a yearly basis and this could help leapfrog some of the requirements in carbon markets. However, Bruges (2009) did not elaborate on potential problems that the CMF could represent. The CMF proposal to promote biochar projects needs further detailed analysis.

2.5. Rationale for biochar in New Zealand

The effects of biochar depend on soil and climate properties – among other factors (see section 2.2.5). New Zealand's history of land-use change has shaped the potential environments where biochar could be introduced. The NZ context needs to be understood prior to adapting the application of biochar into NZ's soils.

The economy and GHG emissions of NZ are closely tied to the performance of its agriculture and forestry sectors. Reducing GHG emissions and minimising the effects on its economy pose a great challenge for NZ. Moreover, NZ producers face the challenges of gaining access to water resources; improving water quality; managing the use of fertilisers, feed and agricultural chemicals; combating soil erosion; coping with the effects of climate-change impacts such as droughts; dealing with the disruption of harmful microbe or virus populations; and meeting the increasing demand of products with proven environmental performance credentials. The production and application of biochar into NZ's soils could help address these issues.

2.5.1. Background

NZ's geographic isolation and long history without human inhabitants has resulted in a unique environmental profile. NZ's land area of about 27 million ha (over two-thirds of land correspond to hilly and mountainous terrain), temperate climate with moderately high rainfall, rock type, and vegetation have interacted to create more than a hundred different soil types (MfE, 2007a). The soils in NZ traditionally have a high level of minerals and potential to release nutrients due to the fact that the rocks that formed them are geologically young.

The relatively recent arrival of the Polynesian ancestors of Māori (the indigenous people) in NZ transformed the pristine island ecosystems drastically. The causes for such alteration are attributed to the Pacific rat introduction, widespread faunal extinctions, soil erosion, and deforestation (McGlone, 1989). There is no consensus on the exact date of Māori arrival in

NZ. However, radiocarbon dating on rat-gnawed seeds and rat bones suggests that it was most likely to have taken place around 1280 A.D. (Wilmshurst *et al*., 2008).

Prior to human settlement most of the land in both islands of NZ was forested (Percival *et al*., 2000). McWethy *et al*. (2009) claimed that Māori burnt large areas of native forests deliberately and systematically, and these forests, with no previous history of fire, showed little resilience to this new disturbance. Māori started managing root crops on deforested land, which led to the beginning of soil classification in NZ (Hewitt, 1992). Māori introduced five food plants into NZ: the kumara, the yam, the taro, the gourd, and the coconut although the latter was not successful (Best, 1930). The use of human and animal manure as a fertiliser was considered disgusting by Māori (Cameron, 1964; Schaniel, 2001). The most common soil amendments used by Māori were gravel, sand, fish residues, seaweed, and wood ash. However, in some regions, such as the west Waimea plains near Nelson, Rigg and Bruce (1923) found charcoal in Māori gravel soils. The increase in soil temperature conferred by biochar is the principle reason given for the application of charcoal by Māori in this region as this was the most southern extent of kumara cropping.

By the time of widespread European colonisation in the 1840s and 1850s, close to half of the forest cover had been removed (McGlone, 1989). The European introduction of the potato into NZ caused major changes in Māori agriculture (Cameron, 1964). Potato growing displaced kumara cultivation because it was easier to grow and store than kumara. The great expansion in shifting cultivation over forest land to grow potatoes caused further deforestation. By 1920, most of the current agricultural land had been cleared (MAF, 2009a).

Grasses were introduced for grazing and superphosphate fertiliser and lime were applied to most of the agricultural land (Percival *et al*., 2000). A high percentage (>50%) of NZ soils feature an acidic (pH< 5.0) subsoil horizon and lime is commonly used in NZ to ameliorate soil acidity (Wheeler, 1997). In addition, it is estimated that over 50,000 sites, mostly sheep dips, have been contaminated with various chemicals and pollutants in NZ (MfE, 2007a) due to former, but now discontinued, activities involving pesticides, wood preservatives, and other chemical agents used in the rural economy.

2.5.2. Land and economic outlook

During a relatively short period of human settlement, land-use change has had a considerable impact on the economy of New Zealand. Tourism and primary production are NZ's top two export earners closely linked to land use. For the year ended March 2012, international tourism expenditure reached NZ$9.6 billion (MBIE, 2012). For the year ended June 2013, major primary sector commodities excluding seafood generated about NZ$22 billion in export earnings (MPI, 2013). Internationally, 66% of the world's lamb trade and 40% of traded milk products are supplied by NZ farmers and farmer-owned companies (Leslie *et al*., 2008). According to Wheeler *et al*. (2008), over 90% of dairy farmers in the country supply their milk to Fonterra, NZ's largest milk company.

The primary sector is expected to grow in the following years mainly due to the increased global demand of dairy and forestry products, especially from China. The magnitude of production required to meet future demand inevitably puts even more pressure on global and local environments.

2.5.3. New Zealand's greenhouse gas (GHG) emissions profile

Although NZ's total GHG emissions represented only 0.14% of the global GHG footprint in 2010, it was ranked fifth highest polluter per capita among 40 Annex I countries, at 16.4 t CO_2-eq per person (MfE, 2013). In 2011, NZ's total GHG emissions were 72.8 Mt CO_2-eq, 22.1% higher than the 1990 level of 59.6 Mt CO_2-eq. Despite this growth, the NZ government is on track to meet its Kyoto obligations of reducing GHG emissions between 2008 and 2012 to 1990 levels because of the 0.6 million ha of trees planted in the 1990s (MfE, 2009; MFE, 2013). However, these plantations are planned to be harvested in the 2020s, which will make it more challenging for NZ to meet future targets.

In August 2009, the NZ government announced the modest target of reducing GHG emissions in the range of 10-20% by 2020 compared to 1990 levels if the following conditions were met:

- the global agreement sets the world on a pathway to limit global temperature rise to not more than $2^{o}C$;
- developed countries make comparable efforts to those of NZ;
- advanced and major emitting developing countries take action fully commensurate with their respective capabilities;
- there is an effective set of rules for land use, land-use change, and forestry; and
- there is full recourse to a broad and efficient international carbon market.

According to the Ministry for the Environment (2009), some of the reasons for setting this relatively modest GHG target were:

- NZ has the third lowest GDP per capita among Annex I countries;
- NZ has the second highest population growth since 1990 among Annex I countries; and
- NZ faces high costs of reducing emissions due to its unique emissions profile.

More recently, in Doha, in December 2012, NZ announced that the country will not sign up to a legally binding second commitment period under the Kyoto Protocol (Metz, 2013). In August 2013, a target of 5% reduction of GHG emissions by 2020 compared to 1990 levels was announced by the NZ government. Since the successor of the Kyoto treaty is to be negotiated before it comes into force in 2020, NZ's position may revert back and forth in the coming years.

NZ's GHG emissions are dictated by the energy and the agriculture sectors, which in combination contributed to almost 90% of total emissions in 2011 (MfE, 2013). Industrial processes, waste, and solvent and other produce account for the rest. GHG emissions in the agriculture sector accounted for 47.2% of NZ's total emissions in 2011, whereas in other developed countries such emissions are on average 12% of the total (MfE, 2013). Between 1990 and 2011, GHG emissions in the energy sector grew over two times higher than those in the agriculture sector. This was largely due to the growth in emissions from increased use of motor vehicles.

The GHG emissions in the agriculture sector come primarily from CH_4 emitted by ruminant animals (sheep, cows, deer) and N_2O emitted during nitrification of animal excreted faecal and urinary nitrogen (N) and chemical fertiliser N applied on agricultural land (Leslie *et al*., 2008). Major research programmes are underway to reduce these emissions, or in the case of methane, to convert it to CO_2. The role of the New Zealand Agricultural Greenhouse Gas Research Centre (NZAGRC) is to find ways to meet the country's international GHG commitments without reducing agricultural output. The NZAGRC is a partnership between NZ's leading research providers working in the agricultural GHG arena and the Pastoral Greenhouse Gas Research Consortium.

2.5.4. Forestry and biochar in New Zealand

According to MAF (2009a), approximately 30% of NZ is covered by trees (1.8 million ha or 7% plantations and 6.3 million ha or 23% indigenous forests). Tree plantations (93% are privately owned) are dominated by radiata pine (89% by area). In the year to March 2008, 20.6 million cubic meters of roundwood were harvested and about 70% of this was exported.

Forestry products consist mainly of logs, chips, sawn timber, panels, pulp, paper and paperboard. For the year ended March 2013, about 65% of the forestry export revenue came from three countries: China, Australia and Japan (MPI, 2013). Despite the increasing global interest in wood pellets, they are still a relatively new product in NZ. Wood residues (bark, sawdust, shavings, and off-cuts) are already used extensively in sawmills and processing plants to generate heat and electricity, and harvest and thinning residues are of great interest as renewable sources of energy and biochar (MAF, 2009a). It seems that the widespread use of harvest residues will take several years to develop due to current transport and storage costs.

Since demand for forestry products and biomass may probably increase in the coming years, the sustainable production of trees is a major concern for global and local communities. The key soil indicators influencing tree plantation productivity and sustainability in NZ have been identified to be the C:N ratio and total P (Watt *et al*., 2008). Nutrients are added in the form of fertilisers. Moreover, there is pressure on water yields leading to limitations in the forestry sector (MAF, 2009a). It is considered that biochar in forest soils can assist forest managers by holding nutrients, improving soil conditions and increasing water holding capacity. Furthermore, biochar may enhance soil sorption and prevent leaching of the herbicide terbuthylazine, commonly applied in tree plantations and found in groundwater in some regions in NZ (Wang *et al*., 2010).

2.5.5. Agriculture and biochar in New Zealand

The agriculture sector is generally divided into edible crops and products derived from livestock management. The NZ government concentrates on certain categories based on economic, environmental, and export factors. Edible crops include horticultural (kiwifruit and pipfruit leading the export interest), viticultural, fresh and processed vegetables, and cereal grains (barley followed by wheat and maize).

Intensively cropped soils in NZ have low organic matter content and poor soil fertility (MfE, 2007a), which results in high demand of fertilisers and consequential nitrate leaching. Moreover, reducing the water and carbon footprints of edible crops produced in NZ is anticipated to be critical to meet local and global consumer demand (Deurer *et al*., 2010). Depending on soil type, the application of biochar in orchards, vineyards, and arable land in NZ could decrease fertiliser and irrigation requirements and prevent nutrients from contaminating waterways.

Pastoral livestock management includes dairy, meat (lamb, beef, and venison), and wool. The more intensive pork and poultry industries are generally considered separately because they are relatively small and focus almost entirely on the domestic market. The international competitiveness of the NZ livestock industry relies on low-cost pastures and a favourable temperate climate that allows animals to graze almost all year round. While this system is effective at producing livestock products, it is a major source of pollution which has contributed to water quality degradation in some rivers and lakes in NZ (Monaghan *et al*., 2008).

Nitrate can contribute to water eutrophication and is considered dangerous to human health if present at high concentrations in drinking water (Di *et al*., 2009). “About 30 percent of the country’s lakes are considered to have poor water quality due to excessive nutrient levels” (MAF, 2009a). The application of biochar in pastoral land could decrease fertiliser requirements and leaching of nutrients and faecal bacteria, especially to “sensitive catchments such as Lake Taupo” (Winsley, 2007). Sarmah *et al*. (2010) showed that biochar in soils can adsorb certain hormones used in beef production and impede their journey to waterways in NZ.

Beukes *et al*. (2010) proposed some strategies to mitigate GHG emissions on pastoral dairy farms in NZ. These include replacing non-productive animals, using animals with higher genetic merit, introducing maize silage as feed, increasing pasture quality, applying nitrification inhibitors, and putting cows on loafing pads to capture excreta. All these strategies aim at reducing N deposition on soils, nitrification and therefore N_2O generation. It is believed that biochar could help address these issues by adsorbing N and delaying nitrification. However, further studies are required to elucidate N dynamics when biochar is applied to pasture soils (Clough *et al*., 2010; Taghizadeh-Toosi *et al*., 2011; Taghizadeh-Toosi *et al*., 2012b).

In summary, using biochar as an underground storage mechanism for nitrate, heavy metals, faecal bacteria, herbicides, pesticides, and hormones and as a reservoir for water and nutrients is feasible in theory. But its capacity is not known, nor is it clear how much of these contaminants biochar can adsorb (or absorb) until it becomes saturated, nor how bio-available nutrients are once adsorbed.

In theory, biochar can increase crop yields by a variety of mechanisms including boosting the low pH often observed in NZ soils and improving their soil structure. Biochar also offers great potential in sequestering C and reducing CH_4 and N_2O emissions from agricultural land.

Because of all these benefits, biochar could be used to improve the environmental image of food products in NZ and be used as a convenient tool for NZ producers to continue expanding production. Nevertheless, if a picture of infinite growth through biochar was suggested, soils could eventually become packed with biochar and biochar could become saturated with toxic materials relatively fast. However, the timescale of reaching such 'point of no return' is very uncertain. Winsley (2007) implied that since many soil profiles in NZ are shallow, an "upper ceiling" of biochar addition might be reached much sooner than in the case of terra preta soils.

2.5.6. Biochar in New Zealand's carbon markets

NZ's activities in the clean development mechanism (CDM) and joint implementation (JI) during the first commitment period of the Kyoto Protocol are limited (Table 7). The application of biochar into soils is not recognised as a means to mitigate GHG emissions in these compliance markets.

Table 7. New Zealand's CDM and JI projects in the first commitment period (UNEP Risø Centre, 2013)

Title of the project	Type	Buyer/s	Status	$ktCO_2$-eq/year
Cattle waste management, Landhi cattle colony, Karachi, Pakistan	CDM	New Zealand (Empower)	Validation terminated	1,458
Methane capture and combustion from swine manure treatment for Corneche and Los Guindos, Chile	CDM	Japan, Canada, UK, and New Zealand	Registered	84

Methane capture and combustion from swine manure treatment for Peralillo, Chile	CDM	Japan, Canada, UK, and New Zealand	Registered	79
Methane capture and combustion from swine manure treatment for Pocillas and La Estrella, Chile	CDM	Japan, Canada, UK, and New Zealand	Registered	247
Te Apiti wind farm, Manawatu,	JI	Netherlands	Registered	106
Project White Hill, Southland,	JI	Switzerland	Registered	128
Tararua wind farm stage II, Manawatu	JI	France	Registered	46
Tararua wind farm stage III, Manawatu	JI	Japan	Registered	231
Awapuni landfill gas to energy project, Palmerston North, Manawatu	JI	Austria	Registered	30
Burwood landfill gas utilisation project, Canterbury	JI	United Kingdom	Registered	40

In NZ, the voluntary carbon market is mostly administered by three organisations. Permanent Forests International Limited sells only forestry credits (plantations and conservation of forests) from national and overseas projects. Landcare Research, through its ‘carbonzero’ online GHG calculator and certification programme, offers companies and individuals the option to measure, reduce, and offset their GHG emissions on a voluntary basis. The ‘carbonzero’ programme includes credits from wind farms, landfill gas, and native forest regeneration in NZ. Carbon Market Solutions (CMS) is a NZ based advisory, brokerage, and trading company which focuses on reaping the profits from compliance and voluntary markets. According to their website, in 2007, CMS became the first privately owned company to sell carbon credits on ‘Trade me’ (NZ’s largest online shopping website). Since voluntary carbon markets are more flexible than compliance schemes, a biochar project and respective methodology could be presented first to one of these organisations to seek approval in carbon markets in NZ.

While NZ’s offsetting mechanisms continue with little activity, the NZ Emissions Trading Scheme (ETS) has attracted international attention since it aims at covering all greenhouse gases mentioned in the Kyoto Protocol and all sectors of its economy including agriculture and forestry. Sectors are being introduced gradually (Table 8).

Table 8. Timeframe for sectors to enter the NZ's ETS

Sector	Voluntary reporting	Mandatory reporting	Full obligations
Forestry	-	-	1 January 2008
Transport fuels	-	1 January 2010	1 July 2010
Electricity production	-	1 January 2010	1 July 2010
Industrial processes	-	1 January 2010	1 July 2010
Synthetic gas	1 January 2011	1 January 2012	1 January 2013
Waste	1 January 2011	1 January 2012	1 January 2013
Agriculture	1 January 2011	1 January 2012	To be confirmed

Concerns with NZ's ETS are about the liquidity in, and access to, international carbon markets, leakage of production and emissions from trade-exposed sectors, and the overall impact on NZ's economy (Kerr and Sweet, 2008). Given that NZ is geographically and economically close to Australia, and that both countries intend to incorporate agriculture and forestry in carbon trading, there has been interest in linking their carbon schemes (Jotzo and Betz, 2009).

In NZ, the forestry sector has already started activities under the ETS. Participants have to monitor their GHG emissions and/or removals, report them periodically to the government, can choose to claim credit units for their removals but then surrender units to cover their reported emissions at harvest (MAF, 2010a). To calculate the amount of carbon sequestered, participants refer to carbon stocks already estimated and tabulated based on type of tree species, region, and age of the plantation (MAF, 2009b). Biochar and soil carbon in forest land are currently excluded from the forestry sector (MAF, 2010a). Moreover, the forestry sector has welcomed the inclusion of agriculture in the ETS (Kerr and Sweet, 2008).

The agriculture sector was to enter the ETS in 2015 but now there is no legal date for this to happen. When it does, the allocation of permits to the agriculture sector may be uncapped, i.e. there may be no limit on the amount of NZ units (NZUs) that may be allocated. Allocation is planned to be on an output intensity basis (i.e. emissions per unit of product). The plan had been for assistance level to start at 90% of the sector's allocation baseline and then phase out at 1.3% per year from 2016 (MAF, 2010b) but, without a foreseeable start date, these plans are currently suspended.

Rather than on individual farmers, liability is planned to fall on the shoulders of the agricultural processors such as meat, dairy and egg processors, live animal exporters, and fertiliser companies (MAF, 2010b). A default emission factor will be set for each agricultural product (Table 9).

Table 9. Unit or product used to calculate sector GHG emissions (adapted from MAF, 2010b)

Activity	Units (tonnes of CO2-eq per unit)
Dairy (processing)	kg milksolids
Beef, sheep, and deer (processing)	kg carcass weight
Cattle, sheep and pigs (live exports)	animal
Synthetic N fertiliser	tonne nitrogen

Numerous regulations still need to be developed over the period leading up to 2015 (MAF, 2010b):

- the default emission factors and methods of calculating emissions;
- exemptions and thresholds for participation;
- the allocation baselines; and
- unique emission factors.

It is not fully clear how the application of biochar into soils could be incorporated into the NZ's ETS. One of the first steps could be to account for the potential GHG emission reductions arising from the introduction of biochar in certain categories in the agriculture sector, and eventually integrate them in the emission factors to be developed. In order to do this, the Life Cycle Assessment (LCA) of agricultural and biochar systems is pertinent.

CHAPTER 3: SELECTION OF FEEDSTOCKS FOR BIOCHAR PRODUCTION

Theoretically, a broad range of biomass feedstocks are suitable for biochar production. However, the type of feedstock is only one factor that determines the characteristics of biochar. Technology parameters, post-production treatments and intended applications of resulting biochars for amendment of certain soils should be all considered alongside costs and local conditions.

It should be noted that, currently, some biomass resources may be potentially more appropriate for energy and other applications than for biochar production due to uncertainties in biochar research (see section 2.2.5). Moreover, the energy sector already has the access, infrastructure, networks and experience required for utilising biomass for energy purposes. Furthermore, a number of financial rewards exist for bioenergy, whereas biochar research is fairly new and lacks substantial economic incentives. In order to explore the opportunity that biochar represents in economic and sustainability terms, a series of case studies were undertaken. For this purpose, NZ's end-of-life biomass (ELB) resources were investigated and appropriate feedstocks that can be used as case studies for the production and use of biochar were identified.

3.1. Scope

Selection of feedstocks relies on a well-defined scope for biochar projects that includes sources of biomass, drivers for use of biochar and type of production process.

Sources of biomass must be sustainable and so, in order to avoid serious risks related to land use change that could be caused by purpose grown crops, only ELB streams were considered. In NZ, these include logging residues, wood processing residues, municipal wood residues, orchard and vineyard wood residues, corn stover, cereal (wheat and barley) straw, farm manures, and sewage sludge. Regardless of source, the greater work contained in this thesis was cognisant of the impact that monetary value for biochar may have, whether for soil amendment or for climate-change mitigation.

Possible drivers for use of biochar include increased crop productivity, improvement of degraded lands, waste management, renewable energy production, waterways protection, and climate-change mitigation. A combination of several drivers should be possible in selected applications although this study focuses on the ‘climate-change mitigation’ driver with the main objective to analyse the implications of biochar in carbon markets.

The type of biochar production process evaluated was limited to slow pyrolysis because it is the system that produces the highest amount of relatively stable biochar (see section 2.2.3). Slow pyrolysis is currently believed to be the most robust biomass processing technology when seeking long-term carbon (C) sequestration through biochar additions into soils (Thomsen *et al*., 2011).

3.2. Criteria for selection of feedstocks

The restricted scope outlined above led to three constraining requirements for analysis in this chapter: (i), that biomass is end-of-life; (ii), that production is by slow pyrolysis; and (iii), that the main driver for making biochar is climate-change mitigation.

- With respect to the end-of-life biomass, criteria relate to: total quantity of biomass available; distribution in terms of location and ownership, which affects the facilitation of feedstock supply; costs of feedstock delivered; and competing uses for the biomass.
- With respect to the production by slow pyrolysis, criteria relate to: efficiency of conversion into char; producing biochar that is safe to handle; and the economics of supply, production and delivery.

- With respect to climate-change mitigation, criteria relate to: the C-sequestration potential; the effect of monetary value for C-sequestration on the economics; and the potential for other environmental consequences of a biochar industry.

In addition, the study was constrained by data availability.

Criteria were ultimately defined by specific values, e.g., the biomass resources must be larger than X tonnes within a Y km radius. However, these differed for each application and arose out of detailed analysis. At the beginning, when likely case studies were selected, the ranges of the above criteria were explored for the most promising end-of-life biomass sources in NZ.

3.3. End-of-life biomass (ELB) feedstocks

Information on the most important ELB feedstocks considered for possible selection for biochar case studies in NZ was gathered from a variety of sources but data were mostly drawn from the series of the Bioenergy Options project led by the government organisation SCION (Hall and Gifford, 2008; Hall and Jack, 2008).

3.3.1. Logging residues

Tree plantations, mostly of radiata pine, are distributed throughout the country and logging residues are left on the ground after harvest in many regions.

Total quantity of biomass

Approximately 125,000 dry tonnes of logging residues – about 27% of the existing landing residues, or 7% of the total harvest residues – are already exploited each year as a source of energy in wood processing facilities located mostly in Central North Island and some in Nelson and Hawke’s Bay (MAF, 2009a). Volumes of the three types of logging residues will increase over time to 2030 (Table 10).

Table 10. Historic and projected availability of logging residues in New Zealand (adapted from Hall and Gifford, 2008)

Type of logging residues	2005-2010 (dry tonnes/year)	2016-2020 (dry tonnes/year)	2026-2030 (dry tonnes/year)
Cutover residues on steep terrain	725,000	828,000	1,690,000
Cutover residues on flat to rolling terrain	613,000	715,000	1,324,000
Landing residues	462,000	612,000	1,205,000
Total	1,800,000	2,155,000	4,219,000
Landing + flat to rolling terrain residues (considered easily accessible)	1,075,000	1,327,000	2,529,000

In 2016-2020 – bearing in mind current recovery of residual biomass for energy use – about 1,200,000 dry tonnes of logging residues were estimated to be available and easily accessible. In 2026-2030, the maximum quantity of potentially accessible residues is expected to be over 2.5 million dry tonnes a year, which equates to 200,000 truck loads, or enough feedstock for two large pulp mills.

Distribution and facilitation of feedstock supply

Over 30% of the national harvest comes from the Central North Island region and it is predicted to peak in 2026 to 2030 when more than 70% of logging volumes may possibly be concentrated in this area alone (Hall and Gifford, 2008). Northland and East Coast regions will see an important increase in harvest as well in the next 5 to 10 years. Affordable mechanisms to collect logging residues already exist. However, cutover residues left on steep terrain are not considered readily recoverable (MAF, 2009a).

Costs of feedstock delivered

The costs of delivering logging residues vary considerably from location to location and by region. In 2007, this ranged between \$24 and \$91/m^3 for solid wood delivered from tree plantations to points of use at a distance of between 25 and 100 km (Table 11).

Table 11. Delivered costs of tree plantation residues to a user in 2007 (Hall and Gifford, 2008)

Transport distance (one way)	Landing residues (NZ$/$m^3$ of solid wood)		Rolling cutover ground based harvest (NZ$/$m^3$ of solid wood)		Steep terrain hauler harvest (NZ$/$m^3$ of solid wood)	
	Low	High	Low	High	Low	High
25 km	$24	$34	$36	$50	$63	$78
50 km	$27	$39	$39	$55	$67	$83
75 km	$30	$43	$42	$59	$70	$87
100 km	$33	$47	$45	$63	$72	$91

Despite the possibility of increased interest in collecting all logging residues due to future prices of carbon and fossil fuels, the 'low-hanging fruit' is presently landing residues, followed by cutover residues left on flat to rolling terrain. The technical potential of integrated harvesting of logs and residues could be realised if the value of ELB compensated for any possible losses in log production (Baker *et al.*, 2010).

Ease of conversion into safe biochar

Not only is the quantity of accessible biomass relevant for biochar production, but also its quality. Traditionally, wood has been the preferred feedstock for pyrolysis conversion due to its consistency and comparability between tests (Bridgwater, 2006). Compared with other feedstocks, such as sewage sludge; food residues; municipal wood residues; and farm manures, logging residues may not require excessive drying prior to pyrolysis. Moreover, woody feedstocks are more energy dense compared to other feedstocks, and therefore can produce more energy during pyrolysis.

Generally, logging residues do not contain high amounts of heavy metals or any other pollutants that could compromise safety requirements of resulting biochars. Although logging residues could be mixed with significant amounts of soil and rocks that could decrease production efficiency or damage machines (Lehmann and Joseph, 2009), biochar production may withstand some dirt attached to the feedstock.

Carbon sequestration potential

The production of biochar from logging residues has been classified as a “challenging opportunity for additional carbon sequestration” in plantations in NZ (Wang, 2010), despite the high C content (50-90%) generally observed in wood biochars. This is due to the fact that biochar research in the forestry sector is limited (Wang *et al*., 2010) and the incorporation of biochar into plantation soils may not be easy due to the stumps and slash left on the soil area. However, biochar produced from logging residues could be applied elsewhere, if respective projects proved to be viable.

Competing uses of biomass

Biochar produced from logging residues competes directly with the use of biomass for energy generation. The second Bioenergy Options report (Hall and Jack, 2008) shows six different LCAs of possible pathways to energy-related products from logging residues (Table 12).

Table 12. Pathways to energy product conversion from logging residues (Hall and Jack, 2008)

Pathway name	Conversion technology	End product(s)
Combustion	Combustion	Heat
Cogeneration	Combustion	Heat and electricity
Ethanol	Enzymatic hydrolysis	Ethanol
Gasification – combustion	Gasification	Heat
Gasification – cogeneration	Gasification	Heat and electricity
Gasification – Fischer Tropsch	Gasification + Fischer Tropsch	Biodiesel

Economic returns as hog fuel and pellet fuel are already possible for the more easily accessible residues left on landing sites and on flat to rolling terrain. The feasibility of using logging residues for biochar production is likely to depend on the willingness of the energy sector to cooperate in a biochar initiative. Further work is needed to support possible partnerships at different scales of production. A small scale pilot project would be a good starting point.

3.3.2. Wood processing residues

The wood processing industry is one of the largest producers and users of biomass for energy (45 PJ per year) in NZ (Hall and Gifford, 2008). This is because wood processors have a high demand for heat and electricity, and residues, if not combusted, would have to be disposed of or used for other purposes.

Total quantity of biomass

Despite the fact that it is difficult to quantify the resource, it has been estimated that, every year, less than 400,000 dry tonnes of mixed processing residues are left unused (Hall and Gifford, 2008).

Distribution and facilitation of feedstock supply

Unexploited wood processing material comes mostly from small, scattered and sometimes remote sawmills in Southern North Island and Central North Island where supply of residues exceeds demand. The quantities of unutilised wood processing residues in isolated locations may be too small and dispersed to make the facilitation of supply for a large-scale plant feasible. Further research is needed to confirm the possibility of using small-scale reactors for this source.

Costs of feedstock delivered

Some residues currently without markets are sent to landfill. This makes them financially appealing since their use could avoid associated costs. However, at this moment, there is no market for biochar that could shift such expectations. Transport costs vary according to the type of residue (Table 13).

Table 13. Transport costs for wood processing residues (adapted from Hall and Gifford, 2008)

Type of wood processing residues	Transport costs from site to site
Sawdust	\$0.18 to \$0.25 per tonne km
Offcuts	\$0.18 to \$0.27 per tonne km
Dry shavings (very low density)	\$0.54 to \$0.81 per tonne km

Ease of conversion into safe biochar

Since large quantities of these woody residues are already being used for energy purposes, it can be assumed that they can be easily carbonised in pyrolysers. Yet, Hall and Gifford (2008) explained that the high variability of nature and quality of the material limit the understanding of how wood processing residues can be used in NZ. Therefore, it is uncertain to what extent the characteristics of these residues could affect the quality of biochar.

Carbon sequestration potential

The carbon contained in woody feedstocks is high and therefore the carbon sequestration potential of respective biochars is significant since their C content varies between 50% and 90%.

Competing uses of biomass

Wood processing residues (bark, sawdust, shavings, wood chips and off-cuts) from sawmills, timber-processing facilities and pulp and paper activities are widely used for animal bedding, landscaping, panels making, and energy production. Moreover, the demand for residues suitable for wood pellet manufacturing, such as sawdust and shavings, is growing. According to Hall and Gifford (2008), “there is limited opportunity to use this material outside the wood processing sector without affecting the level of energy self sufficiency within the sector, or the sector’s greenhouse gas emissions”. However, pyrolysis technologies can also be adjusted to supply significant amounts of renewable energy to the wood processing industry. Biochar, then, would become a by-product.

3.3.3. Cereal straw

Wheat and barley are the two cereal crops with highest production volumes in NZ.

Total quantity of biomass

During the year ended 30 June 2011, total harvested area of wheat and barley accounted for 52,600 ha and 64,900 ha, respectively (Statistics New Zealand, 2012). Wheat and barley straws are typically generated in the country at a rate of 7.4 and 6.5 dry tonnes ha^{-1} yr^{-1} respectively; but it was assumed that 50% of the straw would remain in the field to avoid compromising soil functions (Hall and Gifford, 2008). In total, close to 406,000 dry tonnes of cereal straw were available in 2011 (Table 14), or about 4 times the amount of corn stover (see section 3.3.7).

Table 14. Available dry tonnes of cereal straw for removal in 2010

	Wheat	Barley	Total
Available straw for removal (dry tonnes)	195,000	211,000	406,000

Distribution and facilitation of feedstock supply

Canterbury is the region where most wheat and barley crops are grown by many farmers. Otago and Southland are the next largest wheat producing regions (Statistics New Zealand, 2012). Convincing a sufficient number of farmers to participate in an energy scheme could be challenging not only due to logistics but also because of the perception that withdrawing residues is detrimental to soils (Hall and Gifford, 2008). If the straw-derived biochar was returned to the same soil, it may be easier to convince farmers to participate. Gathering half of the residues could be conducted by collecting all the residues every second year but needs further investigation.

Costs of feedstock delivered

The costs of recovering wheat and barley straw have been calculated at $22/tonne (Hall and Gifford, 2008). This figure does not include transport costs, which could be minimised if biochar was produced and used on farm.

Ease of conversion into safe biochar

The chlorine content of cereal straw is one reason to be cautious when producing and applying respective biochars into soils. During conversion, any chlorine may corrode the pyrolyser (Demirbas, 2006). Moreover, chlorine is a precursor of dioxins, toxic compounds, which could be found in straw-derived biochars if production conditions were not addressed properly. In general, these are minor issues that can be easily addressed.

Carbon sequestration potential

Hammond *et al.* (2011) assumed that all biochars made from different biomass sources, including wheat and barley straw, contained 75% carbon. More recently, the C content of biochars produced at 500^{o}C from wheat and barley straws was reported to be approximately 71% (Mani *et al*., 2011) and 66% (Mullen *et al*., 2010), respectively. On average, biochar made from straw was considered in this study to have 68% C content, though it varies with technology parameters.

Competing uses of biomass

Undesired straws are usually combusted or cut and incorporated into the soil – among other uses. Local governments, however, are implementing tougher measures to restrict burning of these materials. Therefore efforts concentrate on evaluating residues for energy production in this region. An LCA study of a combined heat and power (CHP) plant using straw as fuel considered that about 210,000 dry tonnes of surplus residue can be obtained from Canterbury every year (Forgie and Andrew, 2008). More than 90% of GHG emissions could be mitigated through straw to CHP compared to electricity supplied by the grid and heat produced by burning coal.

3.3.4. Municipal solid waste (putrescible)

Municipal solid waste sent to landfill comprises a mixture of various materials such as paper and organic residues, plastics, metal, glass, timber, and potentially hazardous substances. Sorting of municipal solid waste for biochar production would exclude plastics and focus on ELB streams. Due to the different carbon content, these were divided into putrescible and municipal wood residues (see section 3.3.5).

Total quantity of biomass

Putrescible or digestible residues consist mainly of domestic household refuse or kitchen waste. About 408,000 wet tonnes of putrescible solid waste were estimated for diversion from landfills in 2005 (Hall and Gifford, 2008). Considering that on average, food residues contain 70% moisture content (Zhang *et al*., 2007; Singleton, 2012; Powell, 2013) approximately 122,400 dry tonnes of putrescible waste were available in 2005.

Distribution and facilitation of feedstock supply

Municipal solid waste is concentrated in major population centres. Putrescible residues would need to be separated from the rest of the materials sent to landfills. This represents a challenge. Therefore, initiatives to minimise dumping of food residues into landfills include prevention and management of food scraps at the source of production, such as households; companies; farmers markets; supermarkets; restaurants; cafes; hotels; and universities. Due to its high moisture content, the biomass resource, which is already collected and sent to landfills, has been analysed for biogas production (Hall and Gifford, 2008).

Costs of feedstock delivered

The NZ Waste Strategy, through a number of tactics – increasing landfill levies is considered most effective – provides incentives to divert waste from landfills. The levy is currently set at \$10/tonne and in some European countries it is as high as ~\$100/tonne. This provokes excitement among biochar entrepreneurs who are keen to obtain feedstocks at no or even negative costs for taking care of the waste.

Ease of conversion into safe biochar

Food residues can contain pathogens; generate bad odour and putrid juices; easily attract pests such as rats, flies and cockroaches; and provide a breeding platform for maggots. Furthermore, directing putrescible residues to landfills would avert the nutrients from being recycled in agriculture, and could result in contamination of waterways with leachate and production of methane. The pyrolysis of this material could reduce these negative characteristics observed in the management of food residues. However, due to their high moisture content and sorting required for putrescible residues to be used as biomass feedstocks, it is not easy to convert this ELB stream into biochar.

Carbon sequestration potential

Biochars produced from the pyrolysis of kitchen waste were reported to have a carbon content of about 23% (Luo *et al*., 2010). The pyrolysis of kitchen waste, sewage sludge and farm manures produce the biochars with the lowest carbon content analysed in this study.

Competing uses of biomass

Composting and anaerobic digestion for production of biogas are the most common alternatives to directing putrescible residues to landfills. In addition, the use of food residues as animal feed is argued to have minimal environmental issues but it is not always viable (Singleton, 2012).

3.3.5. Municipal wood residues

Municipal wood residues in NZ are usually classified as green waste from gardens and timber waste resulting from demolition and construction practices.

Total quantity of biomass

More than 500,000 tonnes of municipal wood residues are dumped annually into landfills in NZ, making it an attractive feedstock for the energy and waste management sectors (Hall and Gifford, 2008).

Distribution and facilitation of feedstock supply

Although difficult to estimate an accurate figure, large amounts of these residues are concentrated around highly populated areas, notably the Auckland region. Municipal wood residues are already collected. This feedstock could be delivered to a pyrolysis plant as easily as to a landfill. Biochar enthusiasts speculate that a pyrolysis step to include biochar in composted material can lower the volume of material in the market or, if no market exists, reduce the volume transported to the landfill. Demand for municipal wood residues has been increasing but there is an ongoing challenge to provide a high level of quality product.

Costs of feedstock delivered

Good progress has been achieved on minimising landfilling of green waste, especially through composting or mulching at transfer stations and some landfills (MfE, 2007b). Overall, the number of operational waste disposal facilities has dropped from 327 in 1995 to 54 in 2010, according to the Online Waste Levy System (MfE, 2010), which increases the average transport distance and therefore cost per tonne of landfilled waste. The total cost of delivering municipal wood residues to the end user is estimated at $50/green tonne over a distance of 80 km (Hall and Gifford, 2008). This includes screening and blending costs, and assumes a final product having 50% moisture content.

Ease of conversion into safe biochar

Green waste can have high moisture content and is often mixed with soil and other unwanted materials. Timber waste consists of a mixture of treated and untreated wood. Treated timber contains chemicals that require careful disposal and is not suitable for making biochar. Materials segregation prior to or during landfill operations will be needed to overcome the quality barrier. Furthermore, blending of highly wet residues with drier biomass could improve moisture content enough to yield a favourable energy balance for drying and pyrolysis.

Carbon sequestration potential

The carbon sequestration potential of biochars produced from municipal wood residues can be considered important (C content ranging from 50-90%) due to the high carbon content of wood.

Competing uses of biomass

Currently, some municipal wood residues are converted into mulch or compost, and some companies are considering or trialling the option of using them as an energy source (Hall and Gifford, 2008).

3.3.6. Vineyard prunings

Wine making is a major industry in New Zealand worth about $1.2 billion in exports (New Zealand Winegrowers, 2012). Vineyards are mainly concentrated in Marlborough (59%) and, to a minor extent, in Hawke’s Bay (15%) and Gisborne (6%).

Total quantity of biomass

The country's area covered by vineyards was estimated at 33,400 ha in 2012 (New Zealand Winegrowers, 2012). Depending on numerous factors, including vineyard floor management, pruning weights can range from 1.2 to 2.8 kg per vine on a wet basis (Wheeler *et al*., 2005). Assuming 3,300 vines per ha and 50% moisture content, on average, about 110,000 dry tonnes of vineyard prunings can be assumed to be produced every year. Furthermore, trimming of shelterbelts and over-mature trees could provide additional feedstock for biochar production but data on these sources were not found.

Distribution and facilitation of feedstock supply

By and large, vineyard prunings are currently processed by a mulcher sitting on the back of a tractor. A baler, designed to fit between vine rows, could take the place of the mulcher making it relatively easy to collect the prunings as most of the equipment would have been used anyway. If prunings from a given vineyard are not abundant, aggregation of the material from different vineyards would have to be promoted in order to secure enough feedstock for a biochar project.

Costs of feedstock delivered

The costs of delivering prunings from vineyards would be analogous to that of ELB from orchards, which were estimated at $25/tonne or $3/GJ over a distance of 30 km (see below). If biochars were produced on site and incorporated into soils, transport costs would be low or negligible.

Ease of conversion into safe biochar

Vineyard prunings from managed land are promising for biochar production due to their low (or zero) content of pollutants and medium moisture content. Moreover, end of season prunings may be relatively dry and therefore a good feedstock. Little is known, however, about respective biochars and their application into vineyard soils.

Carbon sequestration potential

The C content of biochar produced from vineyard prunings would vary between 50% and 90% making it an attractive feedstock for climate-change mitigation.

Competing uses of biomass

Hall and Gifford (2008) did not include vineyard prunings in their assessment of biomass resources but it is known that most of this material is mulched and left within the vineyard. Alternatively, energy extraction from wood is of increasing interest. For example, a tractor has been successfully adapted to run through gasification of vine prunings instead of burning diesel (EECA, 2010) and an LPG-fired boiler has been replaced with wood-burning boilers, which use vineyard prunings as fuel during the winemaking process (EECA, 2011).

3.3.7. Corn stover

Corn stover consists of the stalks and leaf residues of the maize after harvest of the cobs. Due to significant volumes of production, corn stover has been acclaimed an important source of energy and biochar in the USA (Gaunt and Lehmann, 2008; Roberts *et al*., 2010).

Total quantity of biomass

In NZ, maize harvesting quantities vary from year to year. In 2011, the total area of maize grain harvested was 18,500 ha (Statistics New Zealand, 2012). On average, corn stover is produced at a rate of 10.8 dry tonnes ha^{-1} yr^{-1} in NZ, but again, only removal of 50% of the residues is suggested to comply with sustainable soil management requirements (Hall and

Gifford, 2008). Thus, a maximum potential of approximately 100,000 dry tonnes of corn stover could have been extracted from fields in 2011.

Distribution and facilitation of feedstock supply

Maize grown is a highly distributed crop – in terms of location and ownership – in the North Island; principally in Waikato, Bay of Plenty, Gisborne, Manawatu-Wanganui and Hawke's Bay regions. The processing plants are in Waikato and Bay of Plenty. Corn stover is easily collected through baling. However, this feedstock is of dispersed nature and owned by different individuals in NZ. Small scale projects involving corn stover may be feasible if co-operatives are established among cereal producers to raise awareness about the potential benefits of biochar and to coordinate field tests.

Costs of feedstock delivered

The cost of recovering corn stover has been estimated at $20/tonne based on past production of baling and handling straw (Hall and Gifford, 2008). Maize biochar closed-loop systems could keep transport costs to a minimum.

Ease of conversion into safe biochar

Corn stover is less energy dense than woody feedstocks and therefore produces less energy when pyrolysed. Moreover, harvest seasons play an important role in the moisture content of feedstocks. For example, the moisture content of corn stover obtained during the early harvest season was the reason for consuming a considerable amount of fossil fuels in a study of biochar systems (Roberts *et al*. 2010). Corn stover would generally be collected on a relatively dry state (moisture content of ~13%).

Regarding its quality, biochar produced from corn stover can retain a considerable amount of nutrients compared to its original feedstock, and therefore is considered a promising soil amendment (Fuertes *et al*., 2010).

Carbon sequestration potential

The C content of biochars produced from corn stover ranged from 64% to 74% and was reported to be relatively stable at higher temperatures (Fuertes *et al*., 2010). Although conclusive results on the effects of different biochars on maize germination have not been found in NZ, future work will examine the effects of biochars on maize establishment under field conditions (Free *et al*., 2010).

Competing uses of biomass

Some corn stover, after baling or grazing, is used for animal feed (roughage) and a little is used for heat generation. This, however, involves the removal of corn stover from the land, which is of concern as farmers perceive that withdrawing residues is detrimental to soils (Hall and Gifford, 2008). Some corn stover is returned to the soil by incorporation after grain harvesting. Maize biochar closed loop systems may look more promising.

3.3.8. Orchard wood residues

Orchards generate woody biomass through pruning, removal of over-mature trees, and trimming of shelterbelts. Furthermore, after the orchard has passed its production lifetime, it goes through a re-development stage when its trees are usually burnt in open fields.

Total quantity of biomass

More than 95,000 dry tonnes of orchard wood residues are available every year in NZ (Table 15). Note that only the estimates for wood residues from the two most important horticultural crops and respective regions (apple orchards in Hawke’s Bay and kiwifruit orchards in Bay of Plenty) include estimates for the prunings.

Table 15. Estimates of orchard wood residues in dried tonnes/year (adapted from Hall and Gifford, 2008)

region	apple / pear	peach / nectarine	cherry	avocado	citrus	kiwifruit	regional total (includes shelterbelt)
Northland	--	--	--	1,200	578	--	2,594
Bay of Plenty	--	--	--	2,280	--	20,000	43,420
Gisborne	614		--	--	1,216	--	2,440
Hawke's Bay	23,561	3,830	--	--	80	--	31,620
Tasman	6,965	--	--	--	--	--	8,706
Otago	1,493	2,784	1,350	--	--	--	6,618
Total	32,633	6,614	1,350	3,480	1,874	20,000	95,398

Pipfruit (apple and pear) is a major crop, whereas kiwifruit is the country's largest horticultural export. Management of kiwifruit orchards in Bay of Plenty results in about 40,000 dry tonnes of woody residues per year (New Zealand Clean Energy Centre, 2010). Shelterbelt trimmings account for approximately half of this.

Distribution and facilitation of feedstock supply

About one half of this feedstock resource is concentrated in Bay of Plenty. This is because this region hosts between 75 and 80% of NZ's kiwifruit orchards. Hawke's Bay is the other main region where approximately one quarter of residues, mostly from apple orchards, is concentrated.

Technically, orchard residues can be collected easily. Nonetheless, some orchardists tend to view the resource as a valuable soil conditioner that should not be removed from the property. However, if the wood is contaminated (e.g. as in the case of the recent outbreak of bacterial canker in kiwifruit orchards), removal from the land is advised. Some owners may also allow biomass removal for energy generation since perceptions of agricultural land and respective management practices vary widely from orchardist to orchardist (Hunt, 2010).

Costs of feedstock delivered

The costs of collecting orchard wood residues are estimated to be comparable to that of logging residues at landing sites ($3.40 to $4 per GJ of primary energy) since similar equipment would be used (Hall and Gifford, 2008). The New Zealand Clean Energy Centre (2010) estimated the costs of collecting and delivering unprocessed wood residues at $25/tonne or $3/GJ over a distance of 30 km. Transport costs could be minimised if biochar production and application into soils were done in situ.

Ease of conversion into safe biochar

Pollutants and moisture content of orchard wood residues can be assumed to be low enough to guarantee an easy conversion of biomass into biochar. Although biochar research related to kiwifruit (Holmes *et al*., 2010) and apple orchards (Sivakumaran *et al*., 2010a) has started, results are as yet unclear. Since it has been suggested that most kiwifruit soils are already fertile and may not need biochar additions, respective biochars could be applied elsewhere. However, the recent outbreak of canker bacteria in kiwifruit orchards in NZ prompted ideas about turning the diseased wood into biochar for hygiene and safety.

Carbon sequestration potential

Biochars produced from woody feedstocks have relatively high C content of around 50-90%.

Competing uses of biomass

Generally, residues are kept within the orchard; these are mulched, composted, and to a minor extent combusted (Milà i Canals *et al*., 2006). Furthermore, the removal of wood residues from orchards for external energy production is an expanding area of work. For example, about 10% of the available biomass residues from kiwifruit orchards have been proposed to be extracted to supply fuel to a 2 MW heat plant in Te Puke, Bay of Plenty (New Zealand Clean Energy Centre, 2010).

3.3.9. Sewage sludge

In NZ, several sewage treatment plants produce bio-solids (60% primary sludge and 40% secondary sludge), which are used for biogas production at different scales (Hall and Gifford, 2008).

Total quantity of biomass

About 73,000 dry tonnes of sewage sludge were available in 2005, and the total quantity of biomass available is expected to increase up to ~82,000 dry tonnes in 2020 (Table 16).

Table 16. Estimated available amount of municipal bio-solids per year (adapted from Hall and Gifford, 2008)

Region	Dry tonnes of municipal bio-solids in 2005	Dry tonnes of municipal bio-solids in 2020
Northland	1,095	1,204
Auckland	29,565	40,150
Central North Island	9,125	5,402
Gisborne	730	730
Hawke's Bay	2,920	2,774
Southern North Island	14,235	15,074
Nelson/Marlborough	2,190	2,445
West Coast	365	182
Canterbury	9,490	10,730
Otago/South Island	3,650	3,758
Total	73,365	82,449

Distribution and facilitation of feedstock supply

The amount of bio-solids generated within a city is proportional to its population density with the largest concentrations therefore being found in Auckland, Wellington, and

Christchurch. Sewage sludge for biochar production can be easily supplied at treatment plants. Odours from facilities could be unpleasant but this should be no more preoccupying than the existing system.

Costs of feedstock delivered

Sewage treatment establishments produce sludge whether a biochar industry exists or not. Collection of this resource for biochar production could potentially be conducted at zero or even negative cost since it would save on sludge disposal costs.

Ease of conversion into safe biochar

Management of sewage sludge is a ‘dirty’ business that some biochar researchers prefer not to address (Lehmann *et al*., 2006). The reasoning behind this exclusion is that respective biochars can contain heavy metals and/or other pollutants that could pose health risks to living beings if biochars from these materials were introduced into soils and food production chains. In NZ, for example, a recent survey showed that 55% of the respondents got rid of unused liquid medications via the toilet or sink, and conventional sewage treatment plants were not designed to deal with pharmaceutical compounds that can lead to their accumulation in treated sewage sludge (Braund *et al.*, 2009). Without comprehensive testing, it is possible that biochar produced from sewage sludge in NZ may have traces of these drugs.

Nevertheless, research on biochars produced from sewage sludge continues as the amount of harmful substances, including heavy metals, typically found in sludge biochars is argued to be acceptable under pollution standards in some countries (Shinogi *et al*., 2003; Hossain *et al*., 2010). Moreover, efforts to characterise a varied number of sludge-derived biochars produced under different conditions are needed since the level of pollution varies depending on location and time.

In addition, sewage sludge is generally too wet to feed pyrolysis reactors and would require extensive drying prior to its thermo-chemical degradation. For this reason, sewage sludge was omitted in one of the LCA studies of biochar (Hammond *et al*., 2011). However, in circumstances where the only disposal route for sewage sludge is drying followed by landfill, the collection and drying costs are already taken into account. The comparison then becomes pyrolysis for energy generation plus biochar versus straight landfill. The char, depending on its pollutant concentrations, can either be landfilled or used as biochar in soils. If landfilled, it is not unreasonable to assess potential complementary benefits, such as its adsorptive capacity to reduce leaching from the landfill.

Carbon sequestration potential

Depending on pyrolysis parameters, the amount of carbon inherent in biochar produced from Australian sludge varied between 20% and 25% (Hossain *et al*., 2011), which is significantly lower than plant-derived biochars, noticeably wood-derived biochar. Therefore, the carbon sequestration potential of sludge-derived biochar is relatively low.

Competing uses of biomass

Some installations already have bio-solids-to-biogas-to-electricity conversion systems and there is still potential to establish more biodigesters (Hall and Gifford, 2008). Biochar production could complement biogas projects if produced from spent sludge, i.e. the sludge remaining after biogas has been made. Producing biochar from spent sewage sludge offers great potential as an alternative and complementary waste management strategy.

3.3.10. Farm manures

In NZ, contrary to excrement from grazing animals, a large proportion of pig and poultry faecal matter is collected in manure management systems. In addition, a significant amount of dairy manure is collected from milking sheds.

Total quantity of biomass

Over 15 million dry tonnes of faecal matter are produced every year in NZ (Hall and Gifford, 2008). However, most of this material is found on grazed pastures and is difficult to collect. Over 4 million dry tonnes of farm manures were estimated to be available in 2005 with almost 98% of this being sourced from dairy cattle while at the milking shed (Table 17).

Table 17. Estimated available amount of farm manures per year (adapted from Hall and Gifford, 2008)

Region	Dairy faecal matter in 2005 (dry tonnes)	Piggery faecal matter in 2005 (dry tonnes)	Poultry faecal matter in 2005 (dry tonnes)
Northland	270,000	1,000	1,000
Auckland	96,000	--	18,800
Central North Island	1,608,000	5,000	15,000
Gisborne	--	--	200
Hawke’s Bay	65,000	1,000	700
Southern North Island	885,000	4,000	16,500
Nelson/ Marlborough	78,000	--	--
West Coast	112,000	--	--
Canterbury	477,000	19,000	1,400
Otago/Southland	400,000	2,000	2,000
Total	3,991,000	32,000	55,600

Distribution and facilitation of feedstock supply

Manures are distributed according to types of farm. Dairy manures are concentrated in the Central and Southern North Island regions; Canterbury; and Otago/Southland. Approximately 60% of NZ piggeries are concentrated in Canterbury. About 90% of poultry manure is concentrated in Auckland, Southern North Island and Central North Island. Note that different farms produce different amounts of residues. Provided that moisture content of farm manures is reduced to an acceptable level and farms (or small co-operatives) are large enough to supply biomass to a large pyrolysiser, then farm manures could be supplied to a centralised biochar plant.

Costs of feedstock delivered

Manures are a by-product of animal farming. However, collection, storage and handling costs could be important, especially if farm manures are excessively wet (Hall and Gifford, 2008). No specific cost estimates were found in the literature though.

Ease of conversion into safe biochar

Biochars produced from poultry litter and chicken manure have been evaluated under different conditions in Australia (Chan *et al*., 2008; Singh *et al*., 2010; Joseph *et al*., 2010), Japan (Tagoe *et al*., 2008; Tagoe *et al*., 2010), and the USA (Gaskin *et al*., 2008; Ro *et al*., 2010; Uchimiya *et al*., 2010). The agronomic value of these biochars can be higher than plant-derived biochars mainly due to an increased availability of nitrogen and phosphorus in soils. Moreover, due to possible pathogen contamination resulting from spreading poultry litter on agricultural land, carbonisation of the material before its application into food-producing soils is seen as a safer alternative (Chan *et al*., 2008).

Biochars produced from dairy manure offer potential for soil remediation through immobilisation of heavy metals and organic contaminants (Cao and Harris, 2010; Cao *et al*., 2011). Dairy manures have been also mixed with wood residues in NZ and the resulting biochar had a high P bioavailability (Wang *et al*., 2012a). Under similar production conditions, Cantrell *et al*. (2012) compared five different manure-derived biochars. It was found that biochar produced from pig manure had the greatest P, N and sulphur (S) content and the lowest values for pH and electrical conductivity.

Farm manures need to be dried before being fed into the pyrolysis reactor and this could be a barrier when developing a biochar scheme. Due to its high moisture content, the resource has been studied to produce biogas for local use (Hall and Gifford, 2008). In order to facilitate biochar production from manures, these could be mixed with agricultural and woody residues.

Carbon sequestration potential

Due to the broad production conditions found in the literature, ranges of carbon contents for poultry (27% - 35.5%), pig (17.7% - 33.8%) and dairy (23.2% - 34.7%) manure-derived biochars were taken from Cantrell *et al*. (2012), who used similar controlled pyrolysis parameters. The C sequestration potential of these biochars is lower than from plant-derived biochars.

Competing uses of biomass

Because of its high nutrient content, farm manures are used directly or mixed with compost as a fertiliser. Pathways to convert farm manures to energy are also studied (Hall and Gifford, 2008). Because of their high moisture content, the most promising energy conversion pathway is anaerobic digestion.

3.4. Selection of feedstocks for further analysis

Considering the above feedstocks in a highly optimistic scenario in which 80% of the available end-of-life biomass is sourced to produce biochar, about 1.7 Mt CO_2 could potentially be sequestered every year in NZ (Table 18). This equates to approximately 2.4% of the 72.8 Mt CO_2-eq reported as NZ's total (gross) greenhouse gas (GHG) emissions for 2011 (MfE, 2013). Note that these figures are rough estimates based on a number of assumptions (Table 19) and that biochar applications should not be justified based only on their C sequestration potential without understanding possible short and long-term interactions between biochars, soils, microbes and plant roots under different pedoclimatic conditions (Joseph *et al*., 2010). Furthermore, LCAs are needed to consider GHG emissions (e.g. arising from transport and/or displaced biomass used for energy production) and reductions (e.g. due to fossil fuel substitution, fertiliser savings and/or avoidance of soil-related GHG emissions) occurring along the whole supply chain. A full LCA would also consider environmental impacts beyond climate change.

While 2.4% is not particularly significant to NZ, it may be to any one sector of the economy, e.g., in determining the carbon footprint of agricultural products, or potential climate-change mitigation rewards to the waste management sector. It is important to reiterate that this study is restricted to residues in order to avoid competition with food and fibre production. Purpose grown crops or tree plantations focused on ‘carbon farming’, while raising ethical issues about land use and land-use change may substantially alter the sequestration impact to NZ.

Table 18. Annual carbon sequestration potential of biochar produced from the most prominent end-of-life biomass streams in NZ

Feedstock	Annual biomass available (dry t/yr)	Biochar produced (t)	Biochar applied (t)	Total C in biochar (t)	Stable C in biochar for ≥100 years (t)	CO_2 removed for ≥100 years (t)	Percentage of NZ’s total GHG emissions for 2011
Logging residues (easy to access)	1,200,000	240,000	230,400	172,800	138,240	506,880	0.70
Wood processing residues (without markets)	400,000	80,000	76,800	57,600	46,080	168,960	0.23
Cereal straw	373,000	74,600	71,616	48,699	38,959	142,850	0.20
Municipal wood residues	250,000	50,000	48,000	36,000	28,800	105,600	0.15
Municipal solid waste (putrescible)	122,400	24,480	23,500	5,405	4,324	15,855	0.02
Vineyard wood residues	110,000	22,000	21,120	15,840	12,672	46,464	0.06
Corn stover	95,000	19,000	18,240	12,403	9,923	36,383	0.05
Orchard wood residues	95,000	19,000	18,240	13,680	10,944	40,128	0.06
Sewage sludge	73,000	14,600	14,016	3,504	2,803	10,278	0.01
Farm manures	4,100,000	820,000	787,200	228,317	182,654	669,731	0.92
TOTAL						1,743,129	2.40

Table 19. Assumptions made to calculate the carbon sequestration potential of biochar produced from the most prominent end-of-life biomass streams in NZ

Assumptions		Percentage	References
Share of biomass used to make biochar		80	---------
Char losses in transport and application		4	Hammond, 2009
Char yield (weight %)		25	Hammond *et al*., 2011
Carbon content (weight %) of biochars produced from different feedstocks	Wood	75	Hammond, 2009
	Cereal straw (average between barley and wheat)	68	Mullen *et al*., 2010; Mani *et al*., 2011
	MSW (putrescible)	23	Luo *et al*., 2010
	Corn stover	68	Roberts *et al*., 2010
	Poultry litter (average)	31	Cantrell *et al*., 2012
	Dairy manure (average)	29	Cantrell *et al*., 2012
	Pig manure (average)	26	Cantrell *et al*., 2012
	Sewage sludge	25	Hossain *et al*., 2011
Stable fraction of carbon for ≥100 years		80	Roberts *et al*., 2010

A series of options were evaluated to determine the simplest pathways to produce biochar. The most promising feedstocks were the most likely to be used for biochar production and carbon sequestration in the near future. According to the relevance of their characteristics, feedstocks were arranged, for each criterion, into three classes in increasing order of importance meaning the most promising (Table 20). For example, annual volumes of biomass below 100,000 tonnes belong to class 1 (♦); between 100,000 and 1,000,000 tonnes belong to class 2 (♦♦); and above 1,000,000 tonnes belong to class 3 (♦♦♦). The same reasoning applies to other criteria. When assessing the feedstock qualitatively, the more pluses (+), the more promising the feedstock. Most promising feedstocks were then determined by weighting factors, in which classes were multiplied by ranking and added up into what was called 'total weighting' (right-hand column, Table 21).

The aim was to identify three promising feedstocks for further analysis using LCA. This was done using a weighting method for five screening criteria, which were ranked in decreasing order of importance:

5) carbon-sequestration potential;

4) total quantity of biomass available per year;

3) costs of feedstock delivered;
2) distribution and facilitation of feedstock supply; and
1) ease of conversion into safe biochar.

The C-sequestration potential was considered the main objective for producing and applying biochar into soils, followed by ‘total quantity of biomass available per year’ since significant volumes of biomass mean higher potential. The economics of acquiring feedstocks was ranked third. The criterion ‘ease of collection’, ranked fourth, is directly related to the ‘costs of feedstock delivered’ and to the ‘distribution of the resource’ criteria. Finally, the ease of transforming biomass into safe biochar was ranked fifth by default. The criterion ‘competing uses of biomass’ did not play a significant role in the selection of feedstocks due to the fact that these scenarios will be modelled in comparative carbon footprint studies of biomass management systems (see sections 4.2 - 4.4).

Table 20. Ranking of end-of-life biomass feedstocks based on criteria for selection of bio

	C-seq. (t CO_2/ dry t biomass)	Biomass available (dry t / year)	Costs of feedstock delivered ($/ fresh tonne)	Distribution & facilitation of feedstock supply
Logging residues (easy to access)	~0.42	~1,200,000	$24 -$63 (for solid wood; 25 - 100 km)	Concentrated
Class	♦♦♦	♦♦♦	♦♦	♦♦♦
Municipal wood residues	~0.42	~250,000	$50/green tonne over 80 km	Concentrated
Class	♦♦♦	♦♦	♦♦	♦♦♦
Vineyard prunings	~0.42	~110,000	$25 over 30 km	Dispersed
Class	♦♦♦	♦♦	♦♦	♦
Wood processing residues (without markets)	~0.42	~400,000	$0.18 - $0.81 / km	Dispersed
Class	♦♦♦	♦♦	♦♦	♦
Orchard wood residues	~0.42	~95,000	$25 over 30 km	Dispersed
Class	♦♦♦	♦	♦♦	♦
Cereal straw (wheat & barley)	~0.38	~373,000	$22 + transport costs	Dispersed
Class	♦♦	♦♦	♦♦	♦
Municipal solid waste (putrescible)	~0.13	~122,400	$0 or negative cost	Concentrated
Class	♦	♦♦	♦♦♦	♦♦♦
Farm manures	~0.16	~4,100,000	$0 but handling may be expensive	Dispersed
Class	♦	♦♦♦	♦♦	♦
Corn stover	~0.38	~95,000	$20 + transport costs	Dispersed
Class	♦♦	♦	♦♦	♦
Sewage sludge	~0.14	~73,000	$0 or negative cost	Concentrated
Class	♦	♦	♦♦♦	♦♦♦

[a] dispersed (♦) and concentrated feedstocks (♦♦♦); [b] feedstocks with high (♦) and low moisture con

Table 21. Weighting factors (ranking columns multiplied by classes (rows) from Table 20)

Feedstock	C-seq. potential	Total quantity of biomass available/year	Costs of feedstock delivered	Distribution & facilitation of feedstock supply	Ease of conversion into safe biochar	Total weighting
Ranking	5	4	3	2	1	
Logging residues (easy to access)	15	12	6	6	3	42
Municipal wood residues	15	8	6	6	1	36
Vineyard prunings	15	8	6	2	3	34
Wood processing residues (without markets)	15	8	6	2	2	33
Orchard wood residues	15	4	6	2	3	30
Cereal straw (wheat and barley)	10	8	6	2	3	29
Municipal solid waste (putrescible)	5	8	9	6	1	29
Farm manures	5	12	6	2	1	26
Corn stover	10	4	6	2	3	25
Sewage sludge	5	4	9	6	1	25

The objective of this process was to select three different feedstocks for detailed LCA analysis. Feedstocks with high moisture or contaminant contents could hinder the production or safe introduction of biochar into food supply chains, so municipal solid waste (putrescible), municipal wood residues, farm manures, and sewage sludge were discarded from this preliminary screening. Wood processing residues were not considered for further analysis because their use outside of the wood processing sector may affect the sector's energy self sufficiency or its GHG emissions. The use of orchard and vineyard wood residues for biochar production and application into soils should follow a similar pathway. Wood residues from apple orchards were selected for detailed evaluation due to better availability of data. More specifically, prunings from apple orchards were initially considered for biochar production and apple trees that are periodically removed from orchards were considered in the sensitivity analysis. Logging residues were chosen for further analysis because this is the most promising feedstock. Corn stover and cereal straw would also follow a similar life cycle pathway. Since data were initially collected for wheat straw, this was the third feedstock selected. Finally, alternative biomass management scenarios included the use of feedstocks as a source of energy and business as usual (e.g. orchard prunings are used as mulch, logging residues are left on soils, and wheat straw is cut and incorporated into soils). These scenarios are compared with biochar systems in the following chapter.

CHAPTER 4: CARBON FOOTPRINT OF THREE SELECTED BIOCHAR CASE STUDIES

The factors involved in the modelling of biochar systems are quite variable and context specific. There are currently no product category rules (PCRs), i.e., guidelines on how the environmental claims of a product are made, when undertaking Life Cycle Assessments (LCAs) of biochar systems and results from different studies may not be easy to compare. In the following section, the framing of the three carbon footprint (CF) studies that were selected in Chapter 3 is explained. Then, the three selected biochar case studies (prunings from apple orchards, logging residues, and wheat straw) are evaluated. Finally, an overview and further interpretation of the results is discussed.

4.1. Framing of the carbon footprint studies

This study follows Life Cycle Assessment (LCA) methodology as defined in the ISO LCA standards (ISO, 2006a; ISO, 2006b). This framework, however, leaves the LCA practitioner with a series of choices that may influence the results of the study. Therefore, further guidance was adopted from the International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010a) particularly for the identification of the decision-context.

Classifying the decision-context is relevant for this CF study of biochar systems in order to inform the modelling approach. The goals of this study are:

1) to compare future alternative management scenarios for the end-of-life biomass (ELB) feedstocks selected previously as case studies for production and use of biochar; and
2) to provide information to policy makers and stakeholders on the use of biomass that can achieve the largest amount of carbon credits.

The future comparative scenarios are: i) the business-as-usual (BAU) scenario, in which the ELB stream will continue to be managed as current practice; ii) the heat-only (HO) scenario, in which the ELB is used to generate heat-only; iii) the combined-heat-and-power (CHP) scenario, in which the ELB is used to generate combined heat and power; and iv) the biochar scenario, in which the ELB is subjected to slow pyrolysis at a centralised pyrolysis facility for biochar production and application back into the land where the biomass originated. The HO and CHP scenarios were selected as there is interest in using these types of biomass for energy generation in NZ (Hall and Gifford, 2008).

Since the goals are concerned with informing policymaking, the study belongs to the Situation A or Situation B ILCD categories (see section 2.3.3). As the consequences of the evaluated decision alone are expected to be too small to overcome thresholds and trigger structural changes of installed capacity elsewhere via market mechanisms, this study belongs to the "micro-level decision support" category (situation A). For cases of multifunctionality, which generally apply to biochar systems, the ILCD Handbook states that the system expansion approach shall be adopted, which could then lead to using a consequential modelling approach.

In this study, the results will be presented following both attributional and consequential modelling. Since there is continued debate about whether it is more appropriate to use marginal or average data in consequential LCA, coal was initially modelled as the displaced fuel in the case of the displacement of process heat. The displacement of natural gas was considered in the sensitivity analysis. For the consequential CHP scenarios, it was initially assumed that the coal-based electricity delivered from the grid would be displaced, whereas the average electricity grid mix was considered in the sensitivity analysis.

The functional unit used in this study was 'the management of one tonne of fresh biomass'. Activities needed to produce one tonne of biomass were not included as processes are identical in the compared alternatives, and the biomass feedstocks can be regarded as a minor co-product relative to the main products of the systems producing the biomass. The three alternative scenarios considered in each case study, however, deliver additional functions. In order to compare the three scenarios, then, the same functions need to be provided by the alternative scenarios for each case study. Therefore system boundaries were expanded for each scenario alternative to BAU to include background processes required and/or displaced to achieve the same functions. In doing so the alternatives become equivalent.

The consequential approach taken here is different from the attributional modelling with selective system expansion considered in some LCA studies of slow pyrolysis biochar systems (Roberts *et al*., 2010; Hammond *et al*., 2011). In those studies, the decision to use biomass to produce biochar and apply it into soils was assumed to be independent from options for using biomass for other purposes and alternative scenarios delivering different functions were not adjusted for equivalent comparison. In this study, however, the carbon abatement potential of biochar is recognised as one alternative use amongst others, and with consequences for other activities in the economy. Due to the infancy of biochar research as well as the complexity of biochar-soil dynamics, there is a lack of reliable data sets. Therefore, a conservative C accounting approach was followed and uncertainties were analysed in sensitivity analyses. For further description of the framing of the studies refer to each of the case studies presented in the following sections of this chapter.

4.2. Carbon footprint study of using orchard prunings to produce biochar in a closed-loop system

Wood grown in apple and kiwifruit orchards is the most promising feedstock for biochar production within NZ’s horticultural sector (see section 3.3.8). Since there is interest in withdrawing biomass from orchards to use it as feedstock for energy production (Hall and Gifford, 2008), orchardists, that usually mulch tree prunings on soils, may perceive that the removal of this wood from the land could jeopardise soil fertility due to nutrient depletion and higher risk of soil erosion. Biochar application, however, differs from biomass extraction for only energy purposes and could be better placed if the concept of biochar was well understood by orchard owners, sustainability guidelines were created and followed, and biomass was retained within the orchard in the form of biochar. The following assessment is intended to raise awareness about the climate-change mitigation potential of a biochar closed-loop system compared with alternative management practices for orchard prunings.

4.2.1. Goal and scope definition

Goal and objectives

The goal of this carbon footprint (CF) study is to compare three different future management options for the tree prunings from apple orchards in the Hawke's Bay (HB) region in NZ. The study intends to answer the question: what is the use of biomass that can achieve the largest amount of carbon credits? Therefore, the study is restricted to the climate-change impact category of Life Cycle Assessment (LCA) methodology. The results of this case study will provide useful information for orchardists, biomass stakeholders and policy makers interested in biochar deployment and climate-change mitigation in NZ.

Functional unit

According to LCA standards (ISO, 2006a), a functional unit is defined to set a reference against which the input and output data of the system are normalized. This allows the comparison of alternative products or systems. For this comparative study, the functional unit is 'the management of one tonne of fresh biomass'. Prunings from apple orchards in the HB region have been considered to have 50% moisture content (wet basis) at the time of cutting, and this level of moisture has been assumed to be retained throughout the supply chain.

System boundaries

The system boundaries extend from the instant when pruned biomass is dropped onto the orchard soils to when it is processed into alternative products. These are described in three different scenarios:

i) business as usual (BAU), in which prunings are mulched and left on orchard soils to recycle nutrients (Fig. 6);

ii) heat-only (HO), in which prunings are removed from orchards and used for heat generation elsewhere (Fig. 7); and

iii) biochar, in which prunings are removed from orchards and converted by slow pyrolysis into gas, bio-oil and biochar. The pyrolysis gas and bio-oil are burned to produce heat, whereas biochar is applied back into the same soil area of the orchard (Fig. 8).

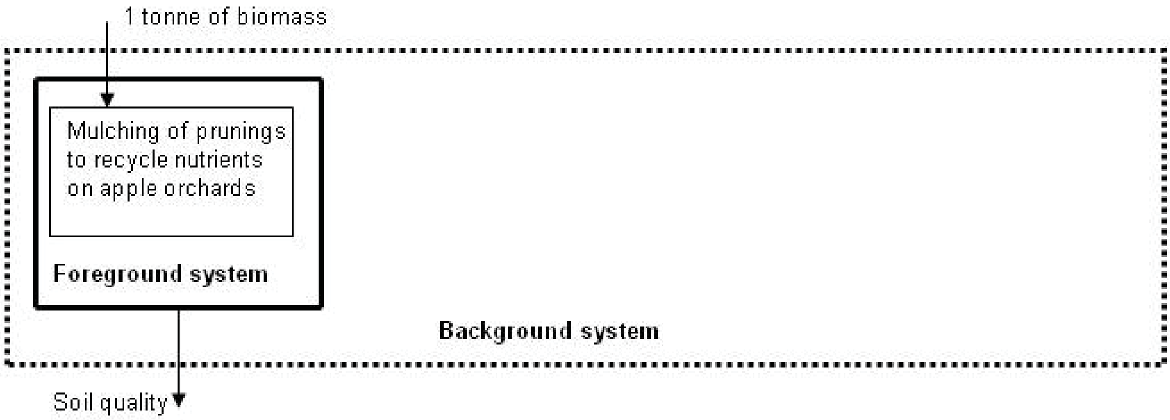

Fig. 6. Business-as-usual scenario for orchard prunings: prunings are mulched on orchards to recycle nutrients and maintain soil quality

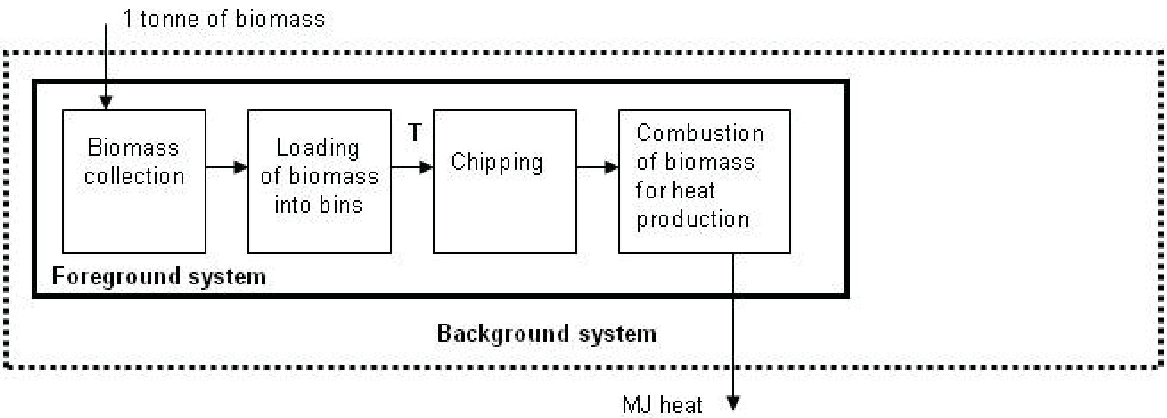

Fig. 7. Heat-only scenario for orchard prunings (attributional): prunings are removed from orchards and combusted elsewhere to deliver heat to a processing plant (T indicates transport)

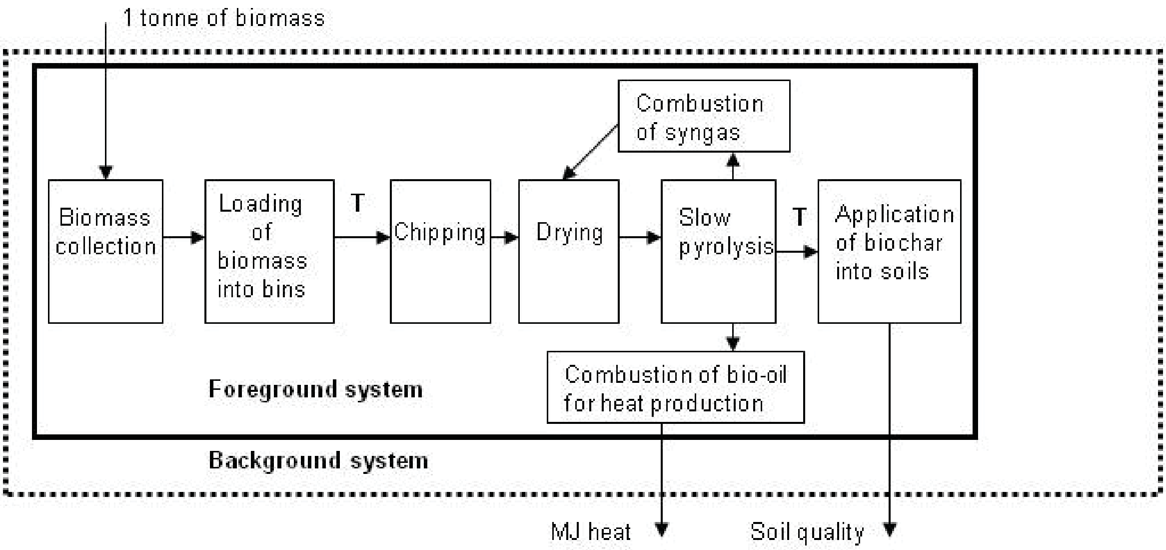

Fig. 8. Biochar scenario for orchard prunings (attributional): prunings are removed from orchards and converted by slow pyrolysis into gas, bio-oil, and biochar which is returned to the same soil area. The pyrolysis gas is combusted to dry the feedstock; bio-oil is burned to deliver heat to a processing plant; and biochar is assumed to improve soil quality (T indicates transport)

The life cycle study begins with the acquisition of one tonne of fresh biomass. Activities needed to produce one tonne of biomass were not included as prunings can be regarded to be a minor co-product relative to apples. Furthermore, these processes are identical in all of the analysed alternatives. In order to evaluate and report the CF of each of the alternative pathways, two different LCA approaches were considered: attributional and consequential.

The attributional analysis only accounts for the direct climate-change impacts produced by the activities enclosed in the foreground system (Fig. 6, Fig. 7, and Fig. 8). This attributional approach describes the absolute C balance of implementing the supply chain under analysis without comparing it with the alternative options. In contrast, the consequential approach was used to compare the alternative scenarios, which deliver different functions (see below). Therefore, system boundaries were expanded for the consequential HO scenario (Fig. 9) and the consequential biochar scenario (Fig. 10) in order to add and/or subtract background processes that would be additionally required or displaced as a result of project implementation. In doing so the alternatives become equivalent and can be compared.

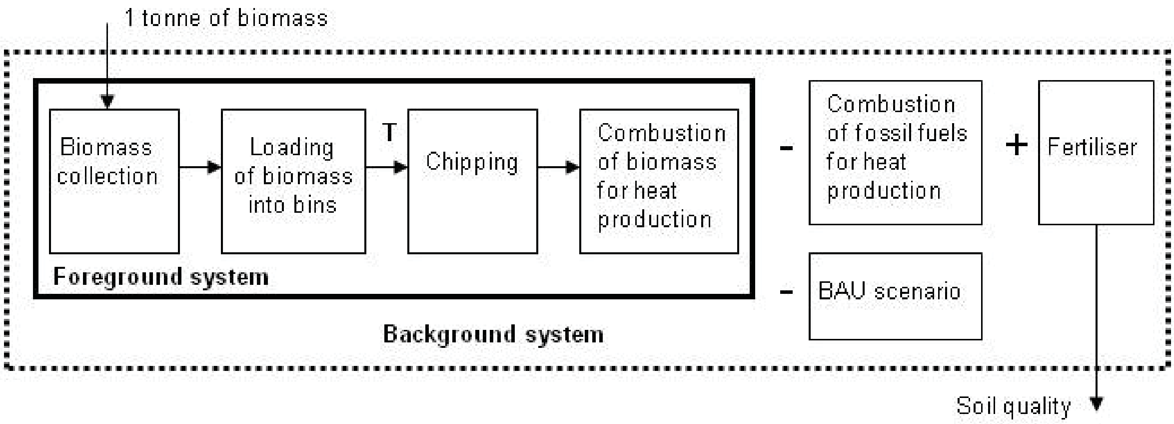

Fig. 9. Heat-only scenario for orchard prunings (consequential): prunings are removed from orchards and combusted elsewhere to deliver heat to a processing plant. As a result, the business-as-usual scenario and the use of fossil fuels for heat production are displaced whereas some fertiliser is added to replace nutrients (T indicates transport)

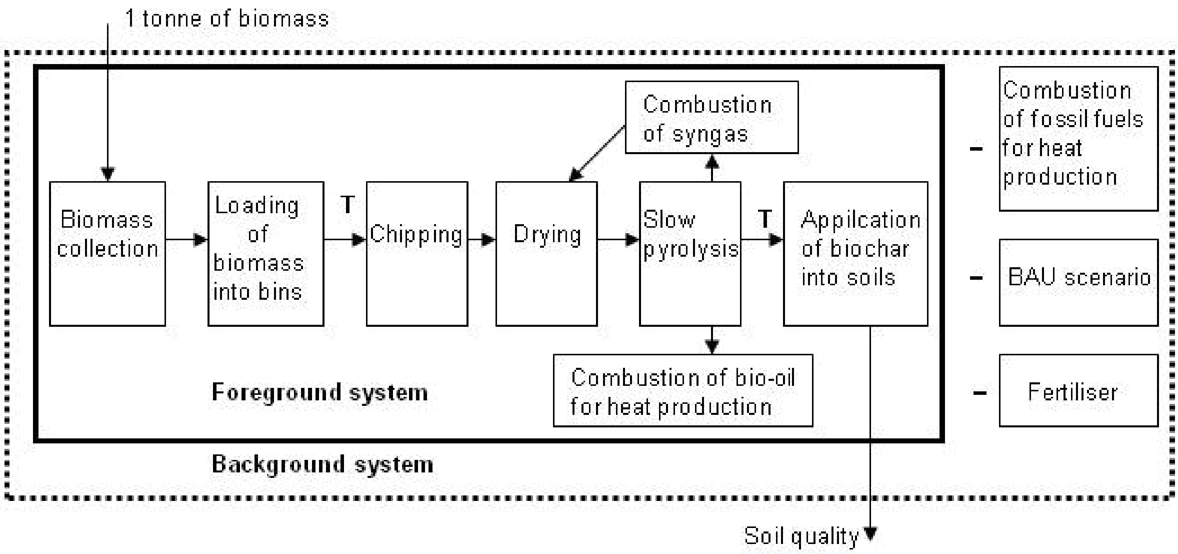

Fig. 10. Biochar scenario for orchard prunings (consequential): prunings are removed from orchards and converted by slow pyrolysis into gas, bio-oil, and biochar which is returned to the same soil area. The pyrolysis gas is combusted to dry the feedstock; bio-oil is burned to deliver heat to a processing plant; and biochar is assumed to improve soil quality. As a result, the business-as-usual scenario and the combustion of fossil fuels for heat production are displaced whereas fertiliser use is reduced (T indicates transport)

Multiple functions

The BAU scenario maintains soil quality; the HO scenario delivers process heat; and the biochar scenario improves soil quality and supplies process heat. The procedures for modelling these additional functions are described below.

- Soil quality

In the BAU scenario, prunings are mulched and placed under the tree rows to prevent weed growth and to recycle nutrients and organic matter which, in turn, can help to maintain sustainable levels of soil fertility and to protect soils from eroding (Milà i Canals, 2003).

In the HO scenario, prunings are removed from orchards to be combusted for energy purposes. Following a consequential approach, mulching of prunings would be displaced. Also, it was assumed that an additional amount of fertiliser would have to be applied in order to supply the nutrients previously supplied by the mulch. Therefore, the consequential HO scenario (Fig. 9) included the subtraction of fertiliser in the background system to make it comparable with the alternative scenarios. Furthermore, the combustion of fossil fuels for heat production was subtracted in the background system to account for potential GHG emission reductions resulting from the delivery of heat through biomass combustion.

In the biochar scenario, prunings are converted by slow pyrolysis into biochar, which is then introduced back into orchard soils. Mulching of prunings would be displaced. Note that it was assumed that biochar would be mixed and charged with some kind of nutrients such as chemical fertilisers, compost, urine and/or manure. Although the agronomic value of biochar tends to be more important for feedstocks with high nutrient content (e.g. biosolids and manures) than for wood (Chan and Xu, 2009; Singh *et al*., 2010; Wang *et al*., 2012a), fertiliser-use efficiency could also be improved due to the ability of biochar to retain nutrients in soils. Therefore, in the consequential biochar scenario (Fig. 10), it was initially assumed that orchardists applying wood-derived biochar into soils would consequently apply less fertiliser. Although soil tests of apple orchards under real conditions in the HB region are needed to analyse fertiliser savings due to the application of biochar produced from apple tree prunings, data evaluated in three CF studies of slow pyrolysis biochar systems were utilised. Note that the possible need for additional application of artificial fertilisers was also considered in the sensitivity analysis. Furthermore, the combustion of fossil fuels for heat production was subtracted in the background system to include the potential displacement of fossil fuel-based heat with heat supplied through bio-oil combustion.

- Process heat

In the HO scenario, prunings are removed from orchards and burned in a boiler to provide heat to a processing plant. This study does not propose any specific plant for the use of prunings as a source of heat but relies on the assumption that fossil fuel users in the Hawke's Bay region would be interested in replacing these fuels with woody biomass from orchards. Coal was initially assumed to be displaced, whereas the displacement of natural gas was modelled in the sensitivity analysis.

The heat-only plant would operate at 80% efficiency and 80% availability (7,000 hours per year). Considering that about 18,000 tonnes of fresh prunings would be processed on apple orchards in the HB region (see below) and assuming that about 5% feedstock losses occur during collection, transport, storage and chipping of prunings, the annual feedstock capacity of the biomass conversion plant would be about 17,000 tonnes of fresh prunings. Since the average calorific value of wood chips with 50% moisture content is 8.0 GJ/t (Sims, 2002), the power output of the heat-only plant would be approximately 4.3 MW:

$$PO = \frac{AFC * C * \eta}{AV * 3.6} \qquad [1]$$

where:

PO = power output of heat-only plant;
AFC = annual feedstock capacity of the biomass conversion plant = 17,000;
C = net calorific value of wood chips having 50% moisture content = 8.0 GJ/t;
η = conversion efficiency of the combustion plant = 80%; and
AV = availability = 7,000 hours per year.

In the biochar scenario, the gas delivered by the slow pyrolyser would be captured and combusted to provide heat, mainly to dry the feedstock but some of it would also be used to induce the slow pyrolysis reactions in the reactor. Based on the average yield of pyrolysis gas that has been assumed here (12% of dry mass fed into the reactor), an amount of wood chips would be subtracted from the wood delivered to the plant to complement the heat energy needed to dry the feedstock. The heat generated through the combustion of pyrolysis gas and wood chips would be used internally by the system and therefore cannot be considered to be a delivered function.

The bio-oil would be exported and combusted in a heat boiler. The resulting heat was assumed to displace fossil fuel use in the background system. GHG emissions from the start-up fuel were neglected. Heat production from bio-oil combustion was chosen because it was found to be the most economically competitive bio-oil application in Europe (Brammer *et al*., 2006) and has been proposed as the first stage of a stepwise market introduction approach (Oasmaa *et al*., 2010). In the Netherlands, bio-oil has been co-fired successfully in a large natural gas boiler with minimal retrofitting (Wagenaar *et al*., 2004). Combustion efficiency of fast pyrolysis bio-oil for heating services has been reported to be high for pilot-scale (Khodier *et al*., 2009) and large-scale applications (Solantausta *et al*., 2012). In this study, the conversion efficiency of the bio-oil boiler was assumed to be 85%. Moreover, a range of efficiencies was evaluated in the sensitivity analysis.

Geographical area

This study focused on the area covered by apple orchards in the HB region and respective prunings which would be returned to the same area in the form of biochar.

For the 2011- 2012 season, there were about 5,162 ha of apples harvested in HB (Pipfruit NZ Inc, 2012). The annual prunings production of apple orchards depends – *inter alia* – on climate and soil conditions; type of cultivar; rootstock; tree planting density; and management practices. For Royal Gala, the most widely grown apple variety in NZ, the average annual weight of winter prunings having 50% moisture content is 2,904 (±222) kg per hectare in the HB region (Clothier *et al*., 2012). For Pink Lady, a less prominent variety grown in HB, the annual weight of prunings having 50% moisture content was estimated at about 4,150 (±592) kg per ha (Perie, 2012).

Thus, the yearly wet weight of apple prunings over HB may range between three and four tonnes per ha. The average fresh weight of prunings (3.5 t per ha per year) was taken as the initial assumption and a ±15% variability was considered. Note that this figure is lower than yields observed in other countries, such as Italy and Turkey (Malavasi *et al*., 1987; Ekinci, 2011). To sum up, slightly over 18,000 tonnes of fresh prunings were assumed to be produced annually in apple orchards in the HB region.

Time horizon

The lifetime assumed for the pyrolysis plants was 20 years (Elsayed *et al*., 2003; McCarl *et al*., 2009; Roberts *et al*., 2010; Woolf *et al*., 2010). In terms of carbon fluxes, the time horizon for GHG emissions and carbon sequestration was 100 years. According to Hall and Gifford (2008), about 6% of over-mature apple trees and 3% of shelterbelts are removed from orchards in HB every year. Although wood from over-mature trees are not necessarily processed in the same way as the prunings, the carbon abatement potential of the management of the removed trees for biochar production was included in the sensitivity analysis.

Construction and maintenance of capital equipment

Capital equipment such as tractors, trucks and chipper would probably be used over several years for purposes other than those included in the system boundaries. Therefore their impact per functional unit on this system is small and can be ignored. For the HO scenario, GHG emissions due to the construction of the heat plant have been omitted as well for the same reason and due to lack of data. As a comparison, for the wheat straw case study, GHG emissions due to construction and maintenance of the capital equipment were accounted for as data were available and the results show that these have no major influence on the system (see section 4.4.1). Capital equipment used exclusively for the biochar system (e.g. storage facility, feed hoppers, screw conveyors, dryer, reactor, fans, motors, etc.) would mostly have an economic impact rather than a large CF per tonne of biomass processed. However, it has been roughly estimated and included below.

Based on the carbon balance of the construction of a 20 MW pyrolysis power plant in the UK (Elsayed and Mortimer, 2001) and respective annual feedstock capacity (Elsayed *et al*., 2003), Hammond *et al*. (2011) estimated the GHG emissions linked to the assembly of small-scale, medium-scale and large-scale pyrolysis plants by referring them to a unit of “one tonne of oven dried feedstock input per year”. Following this procedure and considering the time horizon and the moisture content of biomass, the CF per tonne of fresh biomass due to the construction of a pyrolysis plant would be relatively low. For example, in this case (20 years time horizon and 50% moisture content of biomass), the climate-change penalty of the capital investment would be about 4.1 kg CO_2-eq per tonne of fresh biomass:

$$CFC = \frac{CFP * (1 - MC)}{AFC * LP} \qquad [2]$$

where:

CFC = carbon footprint per functional unit due to the construction of the pyrolysis plant;
CFP = greenhouse gas emissions due to the construction of a 20 MW (e) pyrolysis power plant = 19,840 t CO_2-eq (Elsayed and Mortimer, 2001);
AFC = annual feedstock capacity of a 20 MW (e) pyrolysis power plant = 119,774 oven dried tonnes of wood chips per year (Elsayed *et al*., 2003);
LP = lifetime of the slow pyrolysis plant = 20 years; and
MC = moisture content of feedstock modelled in this study = 50%.

Furthermore, GHG emissions arising from the maintenance of the plant were included by assuming that these account for about 2.5% of the climate-change impact produced during its construction (Elsayed *et al*., 2003). Hence, about 2 kg CO_2-eq per t biomass have been attributed to the annual maintenance of the slow pyrolysis plant.

Data quality requirements

The ISO standards (2006a) stated that certain characteristics of data should be specified in order to meet the goal and scope of the study. As far as geographical and time-related coverage are concerned, data for apple orchards in HB were collected from recent available sources. These include published articles, reports for Pipfruit NZ and interviews with researchers working in the field. Furthermore, data for certain processes (e.g. diesel production and combustion of coal and natural gas) were taken from the GaBi 6.0 software’s database.

Since biochar research is relatively recent – biochar slow pyrolysis plants are just emerging and no conclusive biochar results have been found in apple orchards in HB – some data were taken from existing life cycle studies of biochar and tested in the sensitivity analysis. Moreover, this study follows the conservative approach suggested in some LCA standards (ISO, 2006a) and in carbon markets, such as in the clean development mechanism (CDM), i.e. emissions are overestimated and reductions/removals are underestimated.

4.2.2. Life cycle stages

The most important activities at every life cycle stage for each of the scenarios are explained below.

Business-as-usual scenario

The life cycle of the BAU scenario includes only the mulching of prunings.

- Mulching of prunings

In NZ, orchard prunings are generally processed on-site through a mulching-mower driven by a tractor. The prunings retain carbon in the form of mulch for a longer period of time than if they were combusted. However, the mulch decomposes relatively fast (e.g. some months) and therefore the respective carbon is not kept away from the atmosphere for ≥100 years, i.e. the timescale used to evaluate carbon sequestration in the biochar scenario. Therefore, the short-term carbon sequestration provided by mulch has been neglected in this study. Moreover, tree pruning mulch may have a beneficial impact on soil carbon reserves (Youkhana and Idol, 2009) but further investigation in apple orchards in HB is needed to quantify any change of soil carbon stocks due to mulching against removal of prunings and biochar additions.

Average diesel consumption of tractors used during mulching operations in HB have been reported at about 45 l per ha per year (Hume *et al*., 2009). Considering that on average, each ha produces 3.5 (±15%) tonnes of wet prunings per year, an average diesel consumption of 13 l (±15%) per functional unit was calculated.

Heat-only scenario

The life cycle of the HO scenario starts with the collection of biomass followed by loading of biomass into 'C-hook' bins. Then the bins are transported from the orchards to a processing site, where chipping and combustion of biomass take place. From a consequential LCA perspective, the background system includes displacement of mulching of prunings; addition of fertiliser to compensate for nutrient removal; and displacement of fossil fuels for heat production due to heat delivered through biomass combustion.

- Collection of biomass

Spinelli *et al*. (2012) analysed three systems that can collect prunings from orchards. These could replace mulching-mowers since design features allow them to fit under vines and between rows when sitting on a tractor. Milà i Canals (2003) reported diesel consumption of 35-kW and 45-kW tractors used for collecting prunings from one apple orchard in the HB at 3 and 4.5 l per hour respectively, and its use frequency was evaluated at 0.11 hours per ha per year. It should be noted, however, that the only orchardist claiming to collect prunings preferred to save machinery hours for every operation and machinery use figures for that orchard are around the smallest of all analysed sites (Milà i Canals, 2003). If these values were considered, annual average diesel consumption for collecting prunings would be calculated at around 0.1 litres per tonne of biomass.

This value seems unrealistic since diesel consumption for baling straw has been reported to range between 1.3 and 2 litres per tonne of biomass (Dalgaard *et al*., 2001; Cooper *et al*., 2011). Note that energy consumption for agricultural operations is usually given on a fuel use unit and then allocated across the whole area on a per hectare basis and finally divided by the analysed parameter. Since straw yields (weight/area) are higher than apple pruning yields, energy use per biomass unit might be lower for the straw case. Thus, due to lack of reliable data, the highest diesel consumption value provided for baling straw (2 l per tonne of biomass) was used in this study.

- Loading of biomass into bins

One way to transport unprocessed orchard prunings is to load them into 30 m^3 'C-hook' bins, which are then pulled by a truck and trailer unit. According to the life cycle study of using logging residues to produce energy (Sandilands *et al*., 2008), about 0.27 litres of diesel are required to load one tonne of wood onto a truck. This value has been used here.

- Transport of biomass from orchards to a processing site

Different delivery systems of forest arisings have been analysed for NZ's situation (Hall *et al*., 2001). The higher the moisture content and the lower the bulk density of the biomass the more expensive would be to transport it. This would also translate into a higher CF per tonne of biomass transported. Therefore, to obtain conservative estimates, it was initially assumed that prunings would not be air dried through storage or chipped prior to being transported.

The bins could carry about 10 tonnes of fresh prunings. The average transport distance considered in this study is 30 km (60 km roundtrip). Based on the fuel consumption of a truck with a 10 tonne payload capacity (Forgie and Andrew, 2008), it has been estimated that about 2.23 litres of diesel per tonne of biomass would be consumed during the roundtrip transport of prunings – the truck would carry a full load over 30 km on its way to the conversion plant and would return empty.

- Chipping of biomass

Feedstock arriving at the processing site would be chipped prior to combustion. A mobile chipper usually used in orchards has been considered for this task. Blade sharpness, moisture content, type of tree and tree part are some of the factors that determine fuel consumption for wood chipping. For fresh branches of softwood trees, a mobile chipper consumes about 2.2 l of diesel per tonne of fresh biomass (Spinelli *et al*., 2011). Wood chips may be discharged into a hopper or directly into the biomass boiler.

- Combustion of biomass for heat production

The moisture content of the prunings might be reduced by natural air drying occurring along the supply chain. However, it was assumed that a biomass boiler unit would be fed with fresh prunings having moisture content of 50%. The net calorific value of woody biomass with 50% moisture content is about 8 GJ per tonne (Sims, 2002). This value was assumed for the apple prunings. The boiler would operate at 80% efficiency and would need to shut down for about 20% of the time in a year. GHG emissions arising from the combustion of biomass were considered to be "carbon neutral" as future biomass growth would absorb these emissions and therefore have not been accounted for in this study. Furthermore, the timeframe of this faster CO_2 emitting rate – compared to mulch – was not taken into account. Considering feedstock losses and boiler efficiency, about 6.1 GJ of heat would be produced per t biomass:

$$H = (1 - FL) * C * \eta \qquad [3]$$

where:

H = process heat produced from the combustion of wood chips per t biomass;
FL = feedstock losses = 5%;
C = net calorific value of wood chips having 50% moisture content = 8 GJ/t; and
η = conversion efficiency of the biomass boiler = 80%.

- Fertiliser is added

Since prunings would be extracted from apple orchards to exploit their calorific value elsewhere, some fertiliser would have to be applied in orchard soils to compensate for removal of the nutrients formerly added via the prunings. Note that nutrient-dense leaves and small branches would most likely remain in the orchard. Nitrogen (N) content is higher in prunings of apple trees than phosphorous (P) or potassium (K) (Haynes and Goh, 1980; Hume *et al*., 2009). N content is approximately 0.5% of dry matter of pruned wood from apple trees (Green, 2009 cited in Hume *et al*., 2009). Therefore, the most important climate-change impact at this stage would be mainly from N fertiliser application (2.5 kg N per tonne of fresh prunings or 8.75 (±15%) kg N per ha). The most common N fertilisers used in apple orchards in NZ are calcium ammonium nitrate (CAN) and urea (Milà i Canals, 2003). Only the additional application of CAN was considered in this case study.

About 26.5% of CAN is N. Therefore, CAN application would sum around 9.43 kg per tonne of fresh of biomass or 33 (±15%) kg per ha. Cradle-to-regional NZ storage GHG emissions of the imported fertiliser account for 1.88 kg CO_2-eq per kg CAN (Zonderland-Thomassen *et al*., 2011). GHG emissions arising from additional CAN spreading were neglected as it has been reported that spreading of fertilisers and lime has an irrelevant climate-change impact in apple orchards in HB (Hume *et al*., 2009) and this marginal addition of CAN would not alter significantly such an impact. Furthermore, it was assumed that nitrous oxide (N_2O) emissions from soils would not vary significantly from the mulching to the heat scenario because an equivalent amount of N would be applied. To sum up, about 17.7 kg CO_2-eq per t biomass would be emitted at this stage.

- The combustion of fossil fuels for heat production is displaced

Coal was initially selected as the displaced fossil fuel. Therefore, the data process ‘NZ: Thermal energy from hard coal PE’ was selected in GaBi 6.0 to account for GHG emissions that were assumed to be displaced with process heat delivered through the combustion of prunings (6.1 GJ per t biomass). Moreover, the displacement of natural gas instead of coal was included in the sensitivity analysis.

Biochar scenario

The life cycle of an apple orchard biochar closed-loop system was organised in seven stages: 1) collection of biomass; 2) loading of biomass into bins; 3) transport of biomass to the pyrolysis plant; 4) chipping of biomass; 5) drying and slow pyrolysis of biomass;

6) transport of biochar back to the orchards; and 7) application of biochar into soils. As a result of biochar application, it was initially assumed that the use of fertilisers would be decreased. The possible application of additional fertilisers was included in the sensitivity analysis. Furthermore, the combustion of fossil fuels for heat production and the mulching of prunings would also be displaced.

- Collection, loading, transport, and chipping of biomass

These stages are equivalent to the processes explained in detail for the HO scenario (see above).

- Drying and slow pyrolysis of biomass

There are currently no fully operational commercial slow pyrolysis plants for biochar production and detailed data from facilities at the pilot or demonstration stages are often poor and/or kept confidential. Therefore, the model of the slow pyrolysis plant is based on Fantozzi *et al*. (2007) and the internal electricity demand is based on the analysis carried out by engineering students of a 'Process Design' course at Massey University (Kayed *et al*., 2011), which was adapted to meet the requirements of this study.

The slow pyrolysis plant has the capacity to process between 8,000 and 10,000 tonnes of dried biomass per year. The proposed pyrolyser consists of an externally-heated rotating cylindrical reactor operating at 400ºC for 8,000 hours per year. The reactor is oriented at an angle of 5º to the horizontal and rotated to allow gravity to move the prunings down the length of the reactor. The feeding section consists of an airtight biomass hopper rigidly connected to a screw conveyor, which is powered by an electric motor. Based on Kayed *et al*. (2011), the electricity that would be supplied by the grid to run the electric equipment (e.g. screw conveyors, fans, motors, pelletiser) of the slow pyrolysis plant was estimated to be 5.4 kWh per 460 kg of material with 10% moisture content fed into the reactor, which is equivalent to the functional unit. According to the NZ's life cycle electricity dataset (Coelho, 2011), 0.36 kg CO_2-eq are emitted for every kWh of electricity delivered to the consumer. Therefore, electricity consumption of the pyrolysis plant would be responsible for the emission of approximately 1.9 kg CO_2-eq per t biomass.

Once the feedstock had been chipped, it would enter a rotating cylindrical dryer where moisture content of prunings would be brought down from 50 to 10%. For this purpose, about 2.92 MJ/kg dry solid wood (including 10% heat losses) would be required (Jones, 2012). Some of the pyrolysis gas would be combusted to satisfy the dryer needs and another percentage of the gas would be burned in a firebox below the reactor to help to induce the slow pyrolysis reactions Approximately 420 kJ per kg of wood feedstock at 10% moisture content would be needed for the reactor (Fantozzi *et al*., 2007).

Data on the product yields from the slow pyrolysis of apple tree prunings subjected to a temperature of 400^{o}C were not found. Therefore, data for pine wood was considered. Şensöz and Can (2002) reported the product yields resulting from the slow pyrolysis of pine wood at a temperature of 400^{o}C and a heating rate of 7^{o}C/min. In terms of dried wood, the slow pyrolysis plant would produce yield shares of 31% biochar, 12% pyrolysis gas, and 23% bio-oil. Note that yield shares do not add up to 100%. However, the values reported were considered as quantities of products effectively captured and exploited rather than actual product yields. The model is conservative and highly sensitive to these assumptions.

Based on the 50% moisture content of feedstock and the relatively low yield of pyrolysis gas, the amount of heat provided by the combustion of the pyrolysis gas would not suffice to meet the energy needs of the system. Therefore, fresh wood chips have been assumed to be combusted below the dryer to complement the heat energy delivered by the gas. This quantity of wood chips was calculated through iteration. The combustion of about 122 kg of fresh wood chips would supply about 0.88 GJ per functional unit:

$$HRW = \frac{Q * C * \eta}{1000} \qquad [4]$$

where:

HRW = heat energy supplied by the combustion of wood chips to help to meet the energy needs of the system (GJ);

Q = quantity of fresh wood chips combusted = 122 kg;
C = net calorific value of wood chips with 50% moisture content = 8 MJ/ kg; and
η = combustion efficiency of a wood chip burner including 10% heat losses = 90%.

Note that the conversion efficiency of the wood chip burner was assumed to be 80% but a figure of 90% was factored into the equation above since the heat energy required for the system to operate at these conditions includes 10% heat losses (see below). The dryer would require about 1.21 GJ per t biomass, whereas the reactor would require 0.21 GJ per t biomass:

$$ED = (1 - FL - Q) * (1 - MC) * HM \qquad [5]$$

where:

ED = energy needs of the dryer per t biomass (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.122 t;
MC = moisture content of fresh wood = 50%; and
HM = heat energy required to bring moisture content of wood down from 53% to 10% = 2.92 GJ per tonne of dry solid wood (10% heat losses included).

$$ER = (1 - FL - Q) * (1 - MC) * (1 + 0.11) * HR / 0.9 \qquad [6]$$

where:

ER = energy needs of the reactor per t biomass (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.122 t;
MC = moisture content of fresh wood = 50%; and
HR = heat needed for slow pyrolysis reactions to occur = 0.420 GJ / tonne of wood at 10% moisture content.

Note that the energy required to induce the slow pyrolysis reactions was divided by 0.9 to include 10% heat losses and the quantity of wood chips entering the reactor was adjusted to include 10% moisture content as the heat energy needs of the reactor was reported for wood feedstock with this level of moisture (Fantozzi *et al*., 2007). So, total heat energy needs of the system were ~1.42 GJ per t biomass. The combustion of the pyrolysis gas would supply heat energy in the order of 0.54 GJ per t biomass:

$$HS = (1 - FL - Q) * (1 - MC) * SY * CS \qquad [7]$$

where:

HS = heat energy supplied through combustion of pyrolysis gas per t biomass (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.122 t;
MC = moisture content of fresh wood = 50%;
SY = pyrolysis gas yield from slow pyrolysis at 400^{o}C in terms of dried weight of wood fed into the reactor = 12%; and
CS = high heating value of pyrolysis gas = 10.9 GJ/t.

Note that the conversion efficiency of the pyrolysis gas burner was 90% but was not computed in the equation above since the energy demand of the system already includes 10% heat losses. CO_2 emissions arising from the combustion of the pyrolysis gas were considered to be “carbon neutral” and were not taken into account. Furthermore, direct methane and nitrous oxide emissions from the pyrolysis process have been considered to be relatively low (Mortimer *et al*., 2009) so here were considered to be zero.

On-site or at a nearby location (close enough to disregard GHG emissions from transport), the resulting bio-oil would be burned to provide process heat and displace the use of coal in the background system. Some of the properties of bio-oil represent barriers for its storage, handling, and combustion in standard equipment (Czernik and Bridgwater, 2004). Therefore, it was expected that a start-up fuel would be employed to help in achieving steady state conditions. GHG emissions arising from start-up were neglected though. Bio-oil produced from fast pyrolysis of wood has a heating value of 17 MJ/kg with approximately 25% water content (Bridgwater, 2003). The same heating value is assumed for bio-oil produced from the slow pyrolysis process used in this analysis. Taking into account feedstock losses and a bio-oil boiler efficiency of 85%, about 1.37 GJ of process heat would be delivered per t biomass:

$$HB = (1 - FL - Q) * (1 - MC) * BY * CB * \eta \quad [8]$$

where:

HB = heat energy provided per t biomass by bio-oil combustion (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.122 t;
MC = moisture content of feedstock modelled in this study = 50%;
BY = bio-oil yield from slow pyrolysis at 400^{o}C in terms of dried weight of wood fed into the reactor = 23%;
CB = net calorific value of bio-oil with 25% water content = 17 GJ/t; and
η = conversion efficiency of bio-oil boiler = 85%.

The resulting biochar would be compressed by an electric pelletiser to reduce transport costs and facilitate handling and mechanical application into soils. Depending on soil characteristics, the porosity and the particle size of the biochar pellets might affect some biochar-soil dynamics, such as water and nutrient retention, water accessibility by plants, soil macroporosity and possibly GHG emissions from soils. However, it was difficult to assess pelleting conditions. Care might also be needed during pelleting in order to avoid risks of spontaneous combustion of the biochar pellets (Blackwell *et al*., 2009).

Depending on feedstock and slow pyrolysis parameters, a material to bind the biochar fines together might be required. Binders that could be appropriate for biochar-pelleting include clay, gum arabic, cereal flour, starch, sawdust, wood flour, and any combination of these. According to Demirbas (2009), the most common cost-effective binder for charcoal briquetting is starch. Dumroese *et al*. (2011), however, mentioned that attempts to pelletise biochar using wheat starch and polylactic acid failed to produce a cohesive material without the use of wood flour.

Since the chemical and physical properties of the combination of substrate and biochar pellets can be affected by the type of binders and respective addition rates (Dumroese *et al.*, 2011), the use of a specific binder would have to be investigated thoroughly and therefore was not proposed in this study. Furthermore, it is recognised that the life cycle emissions of binder production contributes little to the C footprint of charcoal briquettes (Rousset *et al.*, 2011). Moreover, water would also be added to assist pelleting, and to reduce dustiness and losses.

- Transport of biochar back to the orchards

Biochar pellets would be transported from the pyrolysis facility back to the apple orchards. An average transport distance of 30 km (60 km roundtrip) was assumed. It would be economically pertinent to transport the biochar pellets by the same 30 m^3 bins used for transporting the prunings – this could be done on the way back to the orchards. However, since the production of biomass is seasonal and scattered it might be preferred to store the prunings at the pyrolysis site rather than at the orchards. Therefore, as a conservative measure, the transport of prunings and biochar pellets was assumed to be independent from each other.

Since the bulk density of biochar would be increased through pelleting, the bulk density of the biochar pellets needs to be estimated in order to know how much biochar would be transported in the 30 m^3 'C-hook' bins. The bulk density of fresh biochar produced from the pyrolysis of kiwifruit vines at 550ºC was reported as 410 kg per m^3 (Holmes and Rahman, 2010) but no pellets were considered.

For energy densification purposes, bulk densities of charcoal pellets were calculated to range between 940 and 1,280 kg per m^3 (Deraman *et al.*, 2007) – range varied according to feedstock and pyrolysis parameters. This figure was confirmed by Soto and Nuñez (2008) who found that pellets consisting of about 50% charcoal and 50% sawdust have a density of 1,000– 1,200 kg per m^3.

The only published article found on biochar-specific pellets (Dumroese *et al.*, 2011) showed that biochar pellets containing 43% biochar, 43% wood flour, 7% polylactic acid and 7% starch had a bulk density of 527 kg per m^3. The pyrolysis temperature ranged between 450-500ºC and most of the pellets produced in that trial had a particle size of 2-5 mm. Since wood flour has a low bulk density of typically 190-220 kg per m^3 (Clemons, 2010), it is reasonable to expect in this case a bulk density of biochar pellets higher than the one reported above. However, due to lack of data, the bulk density of 527 kg per m^3 for the biochar pellets was assumed. This means that each 30 m^3 bin could be filled with approximately 16 tons of biochar pellets. However, the payload capacity of the truck, 10 tonnes, is the limiting factor. Therefore, the diesel consumption due to the transport of 2.23 l per tonne of prunings over the specified distance has been used for the transport of one tonne of biochar. In order to refer this to the functional unit, the feedstock-to-biochar ratio, i.e. the amount of prunings collected to produce one tonne of biochar, was estimated to be about 7.8:

$$FTB = \frac{1}{(1 - FL - Q) * (1 - MC) * CY} \qquad [9]$$

where:

FTB = feedstock-to-biochar ratio (dimensionless);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.122 t;
MC = moisture content of feedstock= 50%; and
CY = char yield from slow pyrolysis at 400^{o}C in terms of dried weight of wood fed into the reactor = 31%.

Proportionally, diesel consumed during the transport of biochar from the pyrolysis facility back to the orchards was 0.29 (2.23/7.8) litres per t biomass.

- Application of biochar into soils

Different methods of application and incorporation of biochar into soils have been discussed (Blackwell *et al*., 2009; Graves, 2013). Two methods were explored here: top-dressing and seed drilling. Initially, it was assumed that biochar application would be integrated into existing orchard activities since this option would be the least costly. Therefore, biochar pellets would be applied on the topsoil during regular spreading of fertilisers. A fraction of the biochar would be eventually lost through erosion and another percentage would migrate downwards. Downward migration of biochar is related to the application rate; particle size; type, bulk density and slope of soils; surface erosion; soil biota (e.g. worms); and rainfall – among others (Major *et al*., 2010).

Considering feedstock losses, for each tonne of fresh prunings collected from orchards about 128 kg of biochar are produced. A 1% loss of biochar during transport was also assumed (Hammond *et al*., 2011). Therefore, an annual biochar application rate of about 127 kg of biochar per t biomass collected or 444 (±15%) kg per ha was calculated to be spread on the topsoil. This figure is close to the weight of fertilisers and lime spread regularly on apple orchards in HB (Hume *et al*., 2009). Thus, it was assumed that the application of biochar would require the same amount of tractor fuel consumed during spreading operations, i.e. 1 litre of diesel per ha, or 2 litres of diesel per tonne of biochar, or 0.3 litres of diesel per tonne of biomass.

The method adopted here, however, was the incorporation of the biochar pellets into orchard rows with a seed drill (Graves, 2013). Considering the annual biochar application rate of approximately 444 (±15%) kg per ha estimated above, an average of 18 l of diesel per ha would be consumed by a tractor towing a seed drill that would incorporate biochar into soil depths ranging from 15-40 mm at 150 mm spacing between seed drill openers (Robinson, 2013). Therefore, about 5.1 litres of diesel per t biomass would be consumed during the incorporation of biochar into soils.

Although seed drilling is more expensive than top-dressing, this type of conservation tillage method was selected here because it may reduce biochar losses due to wind and water erosion and place biochar closer to the rhizosphere for soil improvement. Incorporation of biochar into soils may be also important for climate-change mitigation since a fraction of biochar in solid (Major *et al*., 2010) and dissolved form (Jaffé *et al*., 2013) will likely migrate out of the project boundary and may not be credited with long-term C sequestration. Over a 100-years time horizon, a biochar migration factor (BMF) needs to be considered for the solid and dissolved states of biochar. However, since a BMF has not been explored in the literature yet, this was only considered in the sensitivity analysis. It should be noted that the incorporation of biochar during the planting season of apple trees or other rotating crops might prove easier.

An additional 1% loss of biochar material was also assumed during the incorporation of the biochar pellets. So, about 126 kg of biochar per functional unit (or ~440 (±15%) kg per ha per year) were estimated to be incorporated into the soil. Considering the 20 years time horizon, approximately 8.8 (±15%) t of biochar per ha would be incorporated.

The application of biochar would result in C-sequestration. Data on the C content of biochar produced from apple tree prunings was not found, but wood-derived biochar has been considered to be 75% carbon in one CF study of biochar (Hammond *et al*., 2011) and biochar produced from the slow pyrolysis of pine wood at 400^{o}C was reported to be 76.3% C (Enders *et al*., 2012). Data related to the slow pyrolysis of pine wood at 400^{o}C was used. Of this amount, a percentage of carbon would be labile and cannot be assumed to be sequestered within a 100-years timeframe. Roberts *et al*. (2010) considered a carbon stability factor of 80%, whereas Hammond *et al*. (2011) and Ibarrola *et al*. (2012) assumed that 68% of the carbon contained in biochar would remain in the soil after 100 years. In this study, an average biochar carbon stability factor (BCSF) of 74% was assumed. Considering the assumptions mentioned above and the C-to-CO_2 conversion rate (44/12), about 260 kg CO_2 would be sequestered on a per functional unit basis. Since C-sequestration is the most important factor in life cycle studies of biochar systems (Roberts *et al*., 2010; Hammond *et al*., 2011), the significance of the carbon stability factor was explored in the sensitivity analysis.

The literature often suggests that the application of biochar into agricultural soils could result in fertiliser savings, higher crop productivity, changes in soil organic carbon (SOC) stocks, and suppression of N_2O emissions from soils. However, it is highly uncertain how much biochar would have to be applied in order to realise these potential benefits and to what extent. Due to lack of local data, the assumptions made in the three mentioned life cycle studies of biochar about the potential soil-related effects were compared and analysed (Table 22).

Table 22. Comparison of the assumptions made in life cycle studies of biochar systems about the potential soil-related benefits of biochar application into soils

	Roberts *et al*., 2010	Hammond *et al*., 2011	Ibarrola *et al*., 2012
Application rate	5 t C / ha	30 t biochar / ha	30 t biochar/ha
Crop grown	Corn	Wheat	Wheat
Fertiliser savings	7.2% for N, P and K	10% for N; and 5% for P and K	10% for N; and 5% for P and K
Higher crop productivity	Not considered	10%	5%
Decrease in the rate of soil carbon decomposition	Not considered	10%	5%
Suppression of N_2O emissions	50%	25%	15%
Biochar carbon stability factor (percentage of C in biochar that is stable in soils for ≥100 years)	80%	68%	68%

In the case of the lower biochar application rate cited (Roberts *et al*., 2010), it could be hypothesised that about 8.8 tonnes of apple wood-derived biochar incorporated per ha (or ~70 tonnes of prunings) would be required to suppress soil N_2O emissions by 50% and decrease N, P, and K fertiliser use by 7.2%. In this study, these benefits were assumed to be realised at the end of the 20 years cycle. For higher biochar application rates (Hammond *et al*., 2011; Ibarrola *et al*., 2012), a higher amount of feedstock would be needed and therefore potential GHG impacts per tonne of biomass collected would be even less significant.

Average N_2O field emissions from non-organic apple orchards of Royal Gala and Braeburn, the most important grown varieties in HB, were estimated at 102 kg CO_2-eq per ha per year (Deurer *et al*., 2009). Considering the assumptions made by Roberts *et al*. (2010), the annual incorporation of 126 kg of biochar into orchard soils would suppress N_2O emissions by approximately 1 kg CO_2-eq per functional unit.

In addition, the potential influence of biochar additions on crop production was examined. Note that on a hectare basis, an increase in crop production would not reduce

GHG emissions if inputs to the system remained constant. However, the CF of apple production per weight of main product (GHGs per kg of apples) would decrease. A 5-15% increase in crop productivity was evaluated in this study. At the relatively low biochar incorporation rate estimated here (440 kg of biochar per ha per year), it was estimated that the cradle-to-gate CF of one kg of apples would be reduced by ≤1%. Indeed, this could translate into GHG emission reductions if global apple production was displaced as a result of project implementation. However, this is highly unlikely.

- The use of fertiliser is reduced

As implied above, the climate-change impact of one tonne of biomass on fertiliser use could be neglected in LCAs of biochar systems due to the relatively high application rates required to reduce small quantities of fertiliser. However, this potential impact was calculated to illustrate the low level of relevance of fertiliser savings through biochar application. Thus, average yearly fertiliser inputs in HB were considered for Royal Gala and Braeburn (Hume *et al*., 2009). Furthermore, the assumptions made by Roberts *et al*. (2010) and the GHG emission factors cited in the NZ's life cycle fertiliser database (Zonderland-Thomassen *et al*., 2011) were applied. N, P and K fertiliser savings due to biochar application would represent an avoidance of about 0.10 kg CO_2-eq per t biomass.

- The combustion of fossil fuels for heat production is displaced

The bio-oil produced during the slow pyrolysis of wood would displace coal use in the background system. The data process 'NZ: Thermal energy from hard coal PE' was chosen in GaBi 6.0 to take into account GHGs emitted to deliver 1.37 GJ of process heat per t biomass. The displacement of natural gas was also modelled in the sensitivity analysis.

4.2.3. Results

The three systems/scenarios were modelled using the LCA software GaBi 6.0 and evaluated for the climate-change impact category (CML 2001 - Nov. 2010). Data for the production of diesel were taken from the ‘US: Diesel mix at refinery’ process as GHG emissions calculated for the USA are similar to the ones reported for NZ (Barber, 2009).

4.2.3.1. Business-as-usual scenario

Total GHG emissions due to the production of diesel and its combustion in tractors used during mulching operations are approximately 39.6 (±15%) kg CO_2-eq per functional unit (Fig. 11). The ±15% uncertainty is related to the amount of biomass available, ranging from 3 to 4 tonnes of wet prunings per ha, where the average (3.5 t per ha) was taken as the initial assumption.

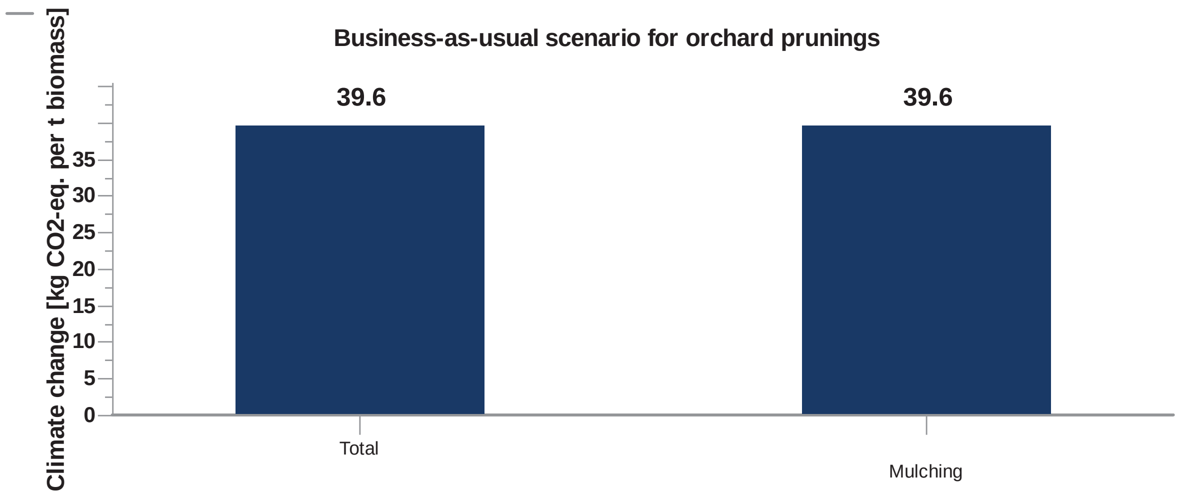

Fig. 11. Breakdown of the climate-change impact per functional unit of the business-as-usual scenario for orchard prunings

4.2.3.2. Heat-only scenario (attributional)

The attributional carbon budget of the HO scenario is about 20.6 kg CO_2-eq per functional unit (Fig. 12). The highest impacts are due to the transport and chipping of biomass. Each of these processes accounts for 6.8 kg CO_2-eq per t biomass. This is followed closely by collection of biomass. Loading of biomass into bins for transport has the lowest impact.

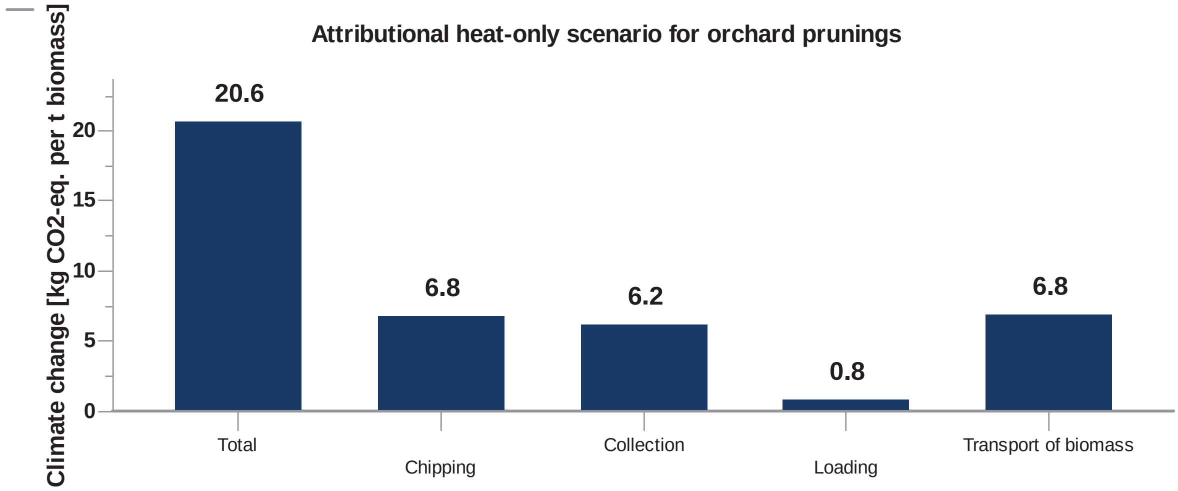

Fig. 12. Breakdown of the climate-change impact per functional unit of the attributional heat-only scenario for orchard prunings

4.2.3.3. Heat-only scenario (consequential)

Following a consequential approach, the HO scenario would have a negative carbon balance of -613.6 kg CO_2-eq per functional unit (Fig. 13). Mulching activities (39.6 kg CO_2-eq per t biomass) formerly undertaken in the BAU scenario (Fig. 11) would be displaced as well as coal combustion for heat production (612.4 kg CO_2-eq per t biomass). Fertiliser (17.7 kg CO_2-eq per t biomass) would have to be consequently applied to compensate for nutrient removal.

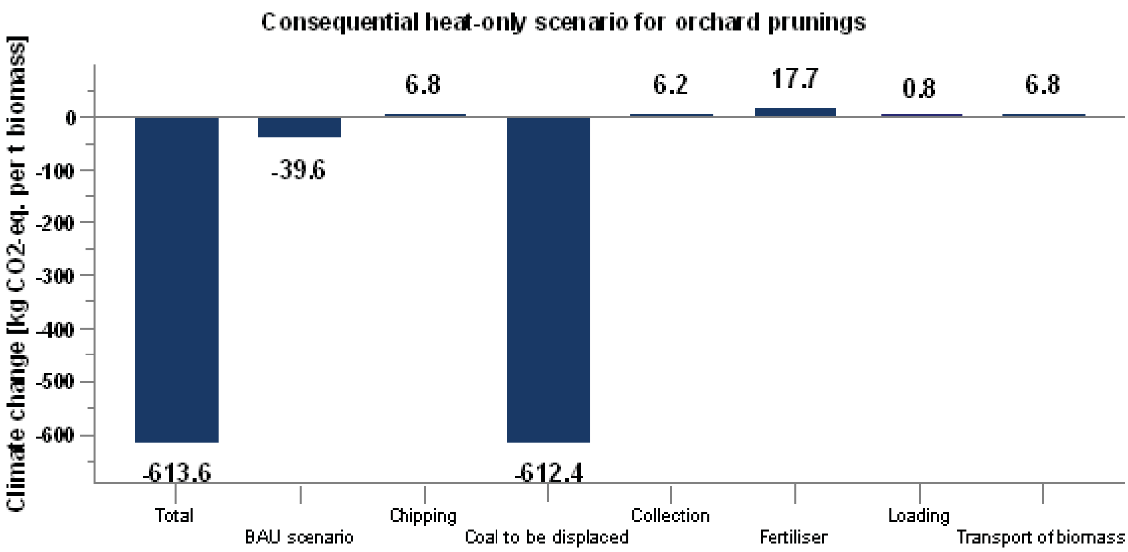

Fig. 13. Breakdown of the climate-change impact per functional unit of the consequential heat-only scenario for orchard prunings

4.2.3.4. Biochar scenario (attributional)

The attributional carbon balance of the biochar scenario is negative at -215.8 kg CO_2-eq per functional unit (Fig. 14). Biomass handling, from collection to chipping of biomass, produces about 20.6 kg CO_2-eq per t biomass. The impact of each of these activities was described above for the HO scenario (Fig. 12). The climate-change impact of the slow pyrolysis plant (8.0 kg CO_2-eq per t biomass) corresponds to the construction, maintenance, and internal electricity use of the plant. These processes account for 4.1, 2.0, and 1.9 kg CO_2-eq per t biomass, respectively. GHG emissions arising from the transport and incorporation of biochar into orchard soils are 0.9 and 15.7 kg CO_2-eq per t biomass, respectively. Suppression of N_2O emissions from soils accounts for 1.0 kg CO_2-eq per t biomass. Carbon sequestration is the most important factor in the biochar scenario (-260 kg CO_2-eq per t biomass).

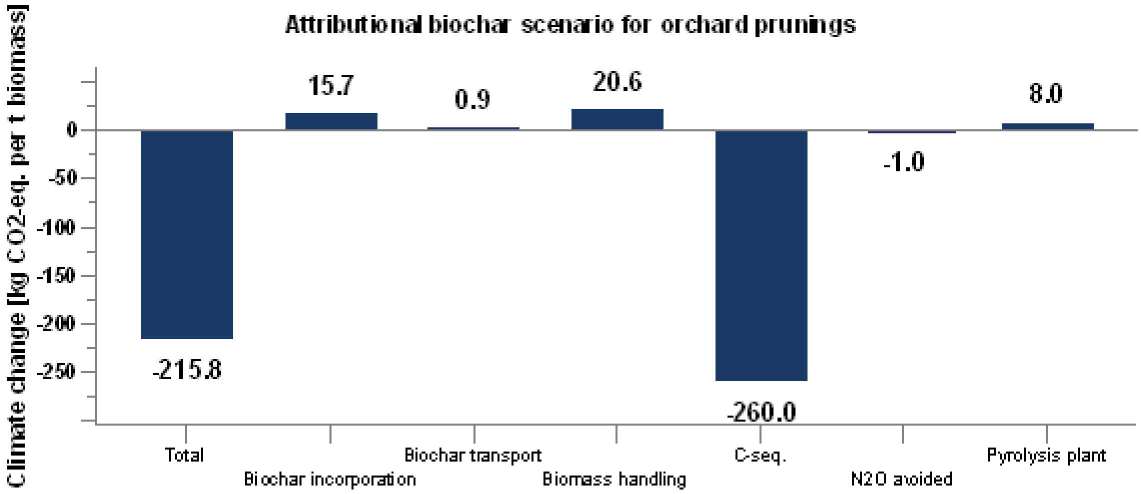

Fig. 14. Breakdown of the climate-change impact per functional unit of the attributional biochar scenario for orchard prunings

4.2.3.5. Biochar scenario (consequential)

The consequential carbon balance of the biochar scenario is -393.0 kg CO_2-eq per functional unit (Fig. 15). Conducting the biochar scenario instead of the BAU scenario would mean that mulching (39.6 kg CO_2-eq per t biomass) would be consequently displaced. The combustion of bio-oil would displace heat energy formerly provided by the combustion of coal and consequently avoid 137.5 kg CO_2-eq per t biomass. It was also assumed that 7.2% of fertilisers formerly used in orchards would be displaced as a consequence of implementing the biochar scenario. This fertiliser displacement (0.1 kg CO_2-eq per t biomass) was added to the suppression of N_2O emissions (0.1 kg CO_2-eq per t biomass) resulting in a combined reduction of 1.1 kg CO_2-eq per t biomass.

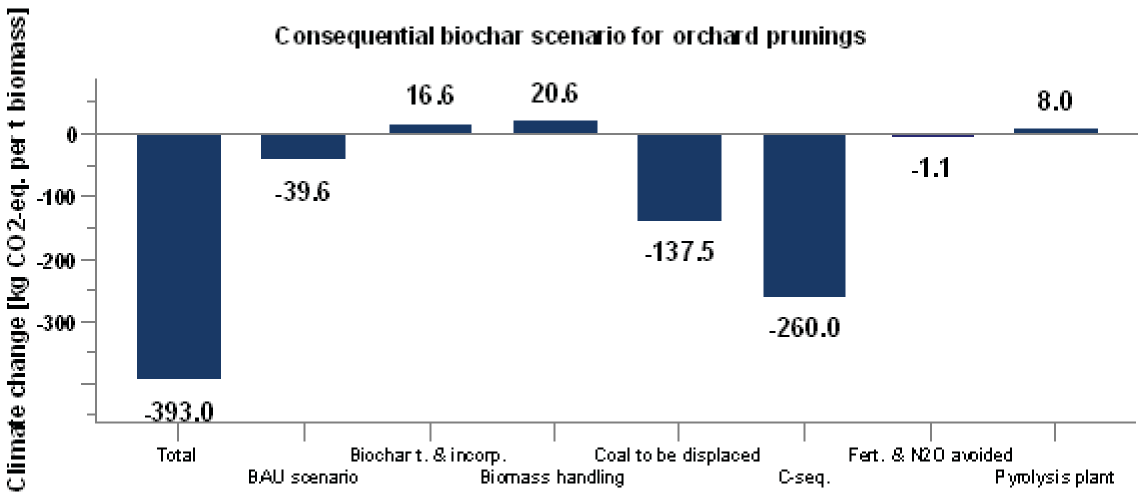

Fig. 15. Breakdown of the climate-change impact per functional unit of the consequential biochar scenario for orchard prunings

4.2.3.6. Comparison of scenarios

In this CF study, the climate-change impact arising from the selection of a management option for the prunings from apple orchards is presented following both the attributional and the consequential approaches (Table 23). Note that under the attributional approach, the alternative scenarios are not comparable because these deliver different functions. When doing the consequential analysis, the scenarios become comparable (Fig. 16).

Table 23. Attributional and consequential climate-change impacts of alternative management options for prunings from apple orchards considering coal as the fuel to be displaced (kg CO_2-eq per t biomass)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with coal displacement (scenarios are comparable)
Business-as-usual scenario	39.6	--	--
Heat-only scenario	20.6	-39.6 (avoided BAU system) -612.4 (displaced coal-based heat generation)	-613.6

		+17.7 (additional fertiliser use)	
Biochar scenario	-215.8	-39.6 (avoided BAU system) -137.5 (displaced coal-based heat generation) -0.1 (reduction in fertiliser use)	-393.0

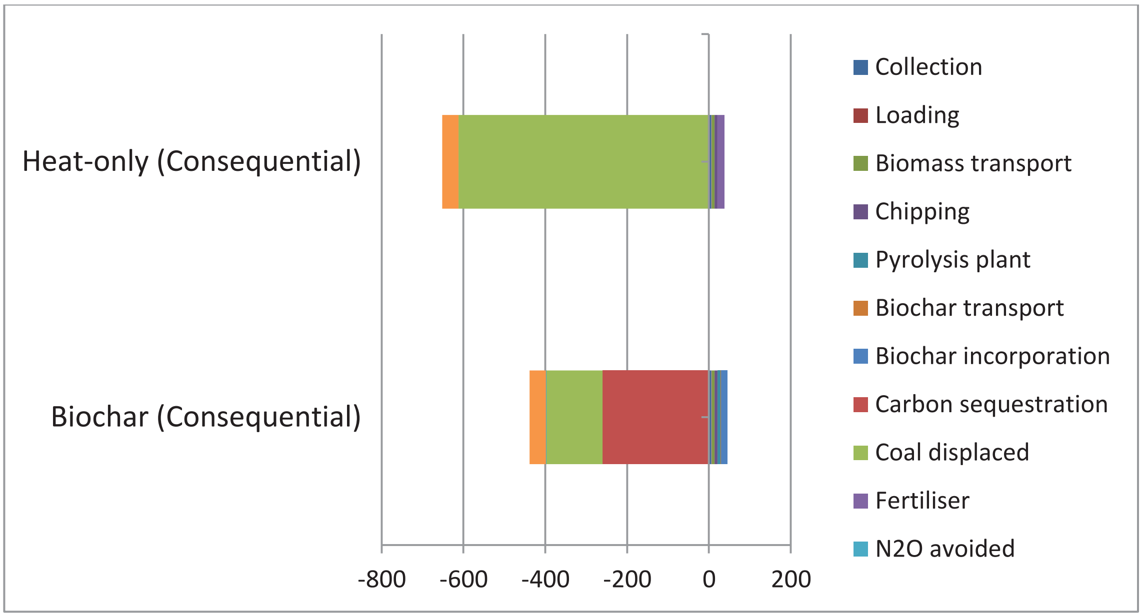

Fig. 16. Comparison of the climate-change impact per functional unit of the alternative scenarios for orchard prunings (kg CO_2-eq per t biomass)

Note that the ±15% uncertainty associated to the amount of prunings available per ha in the BAU scenario has been neglected as it only affects the comparison of scenarios by less than 2% and processes taking place on a ha of land (e.g. biomass collection and incorporation of biochar into soils) would also be affected by the same order of magnitude.

Under an attributional modelling, the biochar scenario would be the only option that is carbon-negative (-215.8 kg CO_2-eq per t biomass). However, the consequential analysis shows that the HO scenario would deliver the highest GHG emission reductions (613.6 kg CO_2-eq per t biomass) if coal was displaced. The biochar scenario would sequester C for about 260 kg CO_2-eq per t biomass irrespective of which approach is followed. In a consequential assessment, the biochar scenario would also avoid approximately 137.5 kg CO_2-eq per t biomass if coal displacement occurred as a result of project implementation. Considering life cycle GHG emissions and fertiliser and N_2O

reductions, the consequential carbon balance of the biochar scenario would be about -393.0 kg CO_2-eq per t biomass.

4.2.4. Sensitivity Analysis

Besides considering the displacement of coal in the consequential analyses discussed above, the influence of natural gas displacement on the consequential carbon balances was examined here. The data process 'NZ: Thermal energy from natural gas PE' was selected in GaBi 6.0 to account for respective GHG emission reductions. The HO scenario would still offer the highest GHG emission reductions if natural gas was assumed to be displaced (Table 24).

Table 24. Attributional and consequential climate-change impacts of alternative management options for prunings from apple orchards considering natural gas as the fuel to be displaced (kg CO_2-eq per functional unit)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with natural gas displacement (scenarios are comparable)
Business-as-usual scenario	39.6	--	--
Heat-only scenario	20.6	-39.6 (avoided BAU system) -374.7 (displaced natural gas-based heat generation) +17.7 (additional fertiliser use)	-376.0
Biochar scenario	-215.8	-39.6 (avoided BAU system) -84.2 (displaced natural gas-based heat generation) -0.1 (reduction in fertiliser use)	-339.7

Furthermore, several ranges of values for the biochar scenario, the main focus of this study, were investigated in the sensitivity analysis considering the consequential displacement of coal or natural gas (Table 25). Then, the most pessimistic and the most optimistic climate-change impact scenarios of the consequential biochar system were calculated by varying all parameters (Table 26).

Table 25. Sensitivity analysis for the consequential biochar system of orchard prunings considering coal or natural gas combustion for heat production as the displaced activity (- indicates a further reduction, whereas + means a further increase in the carbon balance)

Parameter	Original assumption	Range	Impact of variability on the C balance of the system displacing coal as a heat source	Impact of variability on the C balance of the system displacing natural gas as a heat source
Moisture content of mass fed into the dryer (% wet basis)	50%	30 to 60%	-59.4[a] to +28.5%	-59.4[a] to +28.5%
Pyrolysis gas yield (% of dry mass fed into the reactor)	12%	5 to 20%	+4.2 to -5.8%	+4.2 to -5.8%
Bio-oil yield (% of dry mass fed into the reactor)	23%	15 to 30%	+12.0 to -10.8%	+8.5 to -7.7%
Bio-oil boiler efficiency	85%	50 to 90%	+14.3 to -2.2%	+10.1 to -1.5%
Biochar yield (% of dry mass fed into the reactor)	31%	25 to 35%	+12.7 to -8.7%	+14.7 to -10.0%
Biochar losses	2%	0 to 12%	-1.5 to +6.7%	-1.8 to +7.7%
Incorporation of biochar	18 l diesel / ha	12 to 24 l diesel / ha	-1.3 to +1.4%	-1.5 to +1.6%
C content of biochar	76.3%	60 to 90%	+14.0 to -12.0%	+16.2 to -13.8%
Biochar carbon stability factor for ≥100 years	74%	50 to 80%	+21.4 to -5.3%	+24.7 to -6.2%
Transport distance	30 km (one way)	10 to 60 km (one way)	-1.3 to +1.9%	-1.5 to +2.2%
Changes in fertiliser use as percentage of nutrient content of prunings	-7.2%	-15 to +100%	-0.02 to +4.5%	-0.03 to +5.2%
Suppression of N_2O emissions from soils	50%	0 to 100%	+0.25 to -0.25%	+0.3 to -0.3%
Biochar migration factor (% of the solid and dissolved states of biochar that migrate out of the project boundaries within 100 years)	0%	10 to 50%	+6.6 to 33.1%	+7.7 to +38.3%

[a]Air drying of prunings takes place at the pyrolysis facility; -59.9% if air drying of prunings takes place at the orchards

Table 26. Most pessimistic and most optimistic scenarios for the consequential biochar system of orchard prunings considering the displacement of coal or natural gas formerly used to produce heat (see Table 25 for the description of the assumptions made)

	Original scenario	Most pessimistic scenario	Most optimistic scenario
Consequential C balance of the biochar system displacing coal (kg CO_2-eq per functional unit)	-393.0	-35.5	-972.6
Consequential C balance of the biochar system displacing natural gas (kg CO_2-eq per functional unit)	-339.7	-21.5	-837.4

The system is most sensitive to the moisture content of the prunings because it affects both the amount of energy required to dry the feedstock and the amount of product outputs. Bringing down the moisture content of prunings from 50% to 30% through air drying at the pyrolysis facility decreases the consequential C balance of the system by 59.4%. Although air drying of wood at the orchards would also reduce some transport-related GHGs, these are relatively insignificant to justify the handling logistics that would need to be coordinated by a considerable number of orchardists.

The product yields, the carbon content of biochar and the biochar carbon stability factor (BCSF) are central to the performance of the system. However, care should be taken when interpreting the figures estimated in the sensitivity analysis because these variables are interdependently and non-linearly related. Without proper characterisation of the highly-contextual slow pyrolysis parameters it is difficult to estimate accurately the influence of any product yield on the other factors. The assumption here was that for any variation caused in any one product, the other variables remained constant.

A decrease of the biochar yield would mainly result in less amount of biochar-C being sequestered but this could be compensated by its relatively high biochar carbon stability factor (Mašek *et al*., 2013). However, there might be a trade-off between C-sequestration and properties of biochar to improve soil functions. From a soil improvement perspective, it might be preferable to produce a relatively higher amount of biochar with specific characteristics than a very stable biochar.

The variation of the efficiency of bio-oil combustion for heat generation had a relatively small climate-change impact assuming that efficiencies did not drop below 50%. However, the efficiency of the bio-oil boiler is important when comparing the biochar scenario with the HO scenario since this variable would affect the displacement of fossil fuels.

In this study, biochar is transported back to the orchards and incorporated with a seed drill into the inter-row soils of orchards. These processes have a low climate-change impact per tonne of fresh prunings. Transport distances have minimal climate-change impact. Furthermore, it was assumed that 2% of the material would be lost during transport and incorporation. Considering that losses during application of biochar produced from fast pyrolysis have been cited to be as high as 30% for a Canadian field (Hammond *et al*., 2011), the impact of the biochar losses on the system was also tested in the sensitivity analysis. Provided that biochar losses do not exceed 12%, the C balance of the biochar system would be negatively affected by less than 8%.

If it was found that the removal of N provided formerly by mulched prunings was not compensated by the incorporation of biochar into soils, an additional application of fertiliser would be required. The sensitivity analysis shows that this would not have a significant climate-change impact in the short-term. However, a long-term assessment is needed to evaluate possible consequences of erosion and degradation of soils due to the removal of mulch. It was also found that the potential soil-related benefits of biochar assessed in this study (e.g. fertiliser savings, higher crop productivity, and suppression of N_2O emissions from soils) have a negligible impact on the CF of the system.

The impact of biochar on soil albedo could negate some climate-change mitigation potential offered by the system (Meyer *et al*., 2012) but further research is needed to elucidate this impact. In addition, the biochar migration factor (BMF), which describes the percentage of biochar, in solid and dissolved forms, that travels away of the system boundaries over a 100-years time horizon, is a potential drawback. However, due to the current high level of uncertainty involved in assessing a reasonable BMF, the figures estimated in the sensitivity analysis mainly prompt for further research on this variable.

In addition to the prunings, over-mature trees represent a considerable amount of feedstock for biochar production. Removed trees tend to be burned in open fields. According to Hall and Gifford (2008), over-mature trees from apple orchards and shelterbelts in HB represent 40 and 20 tonnes of woody dry matter per hectare, respectively. Assuming that, annually, about 6% of over-mature apple trees and 3% of shelterbelts are removed from orchards in HB, about 18,600 tonnes of total dry wood from trees can be estimated to be produced every year. If the trees, just as the prunings, had 50% moisture content at the time of removal, then it can be assumed that the trees would roughly follow the life cycle pathway modelled for the biochar scenario and have about the same CF per functional unit. Then, the management of over-mature trees in apple orchards for biochar production in HB would represent a regional carbon abatement potential of approximately 14,600 t CO_2-eq per year considering the displacement of coal as a heat source, or about 12,600 t CO_2-eq per year considering the displacement of natural gas as a heat source.

4.2.5. Discussion

To inform policymakers, the climate-change impact of biochar systems should be compared with that of alternative end uses of biomass. In this comparative CF study, therefore, the climate-change mitigation potential of biochar is recognised as one alternative use amongst others, and with consequences for other activities in the economy.

For situations of multi-functionality, as is commonly the case for biochar systems, the consequential LCA approach is pertinent. By including displaced and additional activities required to achieve the same functions, the alternative systems can be compared. Of the analysed management options, the HO scenario would offer the greatest consequential carbon abatement potential (376.0 – 613.6 kg CO_2-eq per t biomass) whereas the consequential climate-change mitigation potential of the biochar scenario would be lower (339.7 – 393.0 kg CO_2-eq per t biomass). Note that the consequential HO scenario offers GHG emission reductions that depend solely on fossil-fuel offsetting (which requires an existing fossil fuel user to convert to biomass as part of the project to claim the offsets), whereas the consequential biochar scenario offers long-term carbon sequestration as well as GHG emission reductions.

If the main objective of biochar systems was climate-change mitigation, the development of slow pyrolysis technologies for biochar production should focus on overcoming technical barriers related to the recovery of co-products for energy generation in order to make the most out of them. Moreover, drying of feedstock to decrease its moisture content is energy intensive, and air drying could prove to be a feasible way to redirect heat energy required internally for this process to export of heat to displace fossil fuel use.

Transport distances make no meaningful contribution to climate change. However, it is acknowledged that logistics and costs of transport could make the investment in a centralised pyrolysis unit unfeasible. Thus, the use of a mobile pyrolysis machine for orchards has been previously suggested (Deurer *et al*., 2009; Holmes and Rahman, 2010). The companies currently offering a continuous mobile unit focus on fast pyrolysis and therefore biochar would not be the main product for such systems. Furthermore, such technologies are still under development. Contrary to a continuous process, a batch unit, such as the one already being employed by Waste Transformation Limited (formerly Kilnz Bioenergy), New Zealand (Knox, 2012) offers a cheaper option. Nevertheless, since co-products are not exploited in the background system, the carbon abatement potential of the biochar would come mostly from long-term C sequestration.

Using LCA figures, the CF of the whole apple supply chain showed that total GHG emissions calculated for Braeburn and Royal Gala integrated apples are about 1.2 kg CO_2-eq per kg of apples produced in NZ and consumed in the United Kingdom (Hume *et al*., 2009). The individual stages of the apple supply chain were divided into orchard operations, packhouse operations, port operations, shipping, repackaging operations in the UK, retailer operations, and consumption.

Considering the consequential C balance of the biochar closed-loop system modelled in this study, GHG emissions attributed to the orchard stage could be neutralised by an average of 38% when displacing coal or by an average of 33% when displacing natural gas. Since, on average, non-organic orchard operations contribute less than 7% to the entire CF of NZ apples consumed in the UK (Hume *et al*., 2009), the consequential biochar system offers the potential to compensate for less than 3% of the total climate-change impact occurring along this entire supply chain. However, it remains to be seen who would claim which carbon credits (see section 4.5.4). Note that the CF of the apples that are consumed domestically would be lower because no shipping, which increases significantly the CF of exported apples, would be involved.

Moreover, over-mature trees periodically removed for redevelopment offer a significant amount of feedstock for biochar-C sequestration and fossil fuel displacement. This means that the climate-change mitigation potential of the management of this woody biomass for biochar production would be about 12,600 – 14,600 t CO_2-eq per year, which is approximately two times higher than that offered by the prunings, on a regional basis.

Finally, although the previous assessment was based on data from apple orchards, an almost identical biochar system could be modelled for other fruit orchards or vineyards. Results would not differ significantly since data would mainly diverge for the soil-related aspects, which, anyway, have a minor and uncertain GHG impact per functional unit.

4.3. Carbon footprint study of using logging residues to produce biochar in a closed-loop system

Logging residues are the largest source of woody end-of-life biomass (ELB) in NZ (see section 3.4). The considerable amount of wood left on tree plantations and the fact that wood-derived biochar has a high carbon (C) content mean that logging residues are a very attractive feedstock for biochar-C sequestration. Moreover, biochar produced from *Pinus radiata*, the dominant forestry species in NZ, was reported to enhance soil sorption of the herbicide terbuthylazine (Wang *et al*., 2010) and the hydrophobic contaminant phenanthrene under NZ's forestry soils (Zhang *et al*., 2010). This could translate into less leaching of pollutants to ground water.

The production of biochar from logging residues is a challenging opportunity for additional C sequestration in NZ's plantations (Wang, 2010). Some of the barriers that the biochar industry faces are related to the limited understanding of the role of biochar in the forestry sector and competition with the procurement of logging residues for energy generation. Therefore, awareness about the potential benefits of producing biochar from logging residues needs to be created within the tree plantation and energy industries whether biochar is incorporated into the same area or elsewhere. The following analysis aims at contributing to that knowledge.

4.3.1. Goal and scope definition

Goal and objectives

The goal of this carbon footprint (CF) study is to compare four different future management options for the wood residues produced during logging operations in NZ. The principal objective is to evaluate alternative conversion pathways of logging residues according to their carbon abatement potential. The main focus is on logging residues left on flat to rolling terrain where felling of trees takes place, i.e. cutover residues, and on residues left on landing sites where the trees are cut into merchantable logs, i.e. landing residues. Logging residues left on steep terrain by cable hauler systems are currently difficult and expensive to recover, and therefore are out of scope in this study. The intended audience is composed of plantation managers, contractors, policy makers and any other stakeholder interested in biochar production and climate-change mitigation in NZ.

Functional unit

The functional unit of this comparative CF study refers to the supply side of the production chain, i.e. 'the management of one tonne of fresh biomass'. Two types of biomass from tree plantations were considered: landing residues and cutover residues left on flat to rolling terrain. Delivery systems of the two types of residues can vary significantly according to their moisture content and material density (Hall *et al*., 2001; Visser *et al*., 2010). It was assumed that logging residues have 53% moisture content (wet basis) along the whole handling process, i.e. from their collection to their delivery to a conversion plant (Sandilands *et al*., 2008).

About 250,000 tonnes of landing residues are collected per annum, largely to be used as fuel in wood processing plants (Hall and Gifford, 2008). At present, practically no residues are recovered from the cutover, i.e. the area where trees are felled. In the future, however, if a considerable demand for logging residues arose, the delivery of landing and cutover residues could be mixed at some point and therefore, it would also be practical to define a functional unit according to the demand side of the production chain, i.e. 'the management of one tonne of biomass delivered to the conversion plant'. Based on this logic, Sandilands *et al*. (2008) presented the results of the Life Cycle Assessment (LCA) of energy generation from logging residues in NZ according to the functional unit of 'one GJ of energy in the energy product'. They assumed that three quarters of this demand would be supplied by cutover residues, whereas the remaining one quarter would be supplied by landing residues. This allocation is based on the assumption that for any given hectare of tree plantations (or cubic metre of logs) about 5% of the volume will remain as landing residues and about 15% of the volume will be cutover residues.

In order to provide a more general and flexible assessment, the two types of residue streams were treated separately in this study. This would allow a broader use of the results in further studies as figures could be easily allocated in relation to the future corresponding shares of landing and cutover residues processed at a conversion plant. Due to transport logistics, distance and load density, cutover residues are likely to be reduced in size before being transported to the plant. Therefore, it makes sense to consider hogging (the most inexpensive and robust comminution option available) at a central processing yard (CPY), within or near the tree plantation, as the place where the two types of residues mix in the production chain. This assumption means that the only difference between the two pathways occurs at the initial stage, i.e. biomass collection. Therefore, the carbon balances of the two production chains modelled in this study do not differ significantly.

System boundaries

The system boundaries extend from the moment when logging residues are left on tree plantations to when they are converted into alternative products. Four different scenarios were compared:

i) business as usual (BAU), in which logging residues are left unprocessed and nutrients are recycled via natural decomposition in plantations (Fig. 17);

ii) heat-only (HO), in which logging residues are removed from plantations and burned to produce process heat at a heat-only plant (Fig. 18);

iii) combined heat and power (CHP), in which residues are removed from plantations and burned to produce process heat and electricity at a combined heat and power plant (Fig. 19); and

iv) biochar, in which residues are removed from plantations and converted by slow pyrolysis into gas, bio-oil and biochar. The pyrolysis gas and bio-oil are combusted to produce process heat, whereas biochar is applied back into tree plantations to maintain soil quality (Fig. 20).

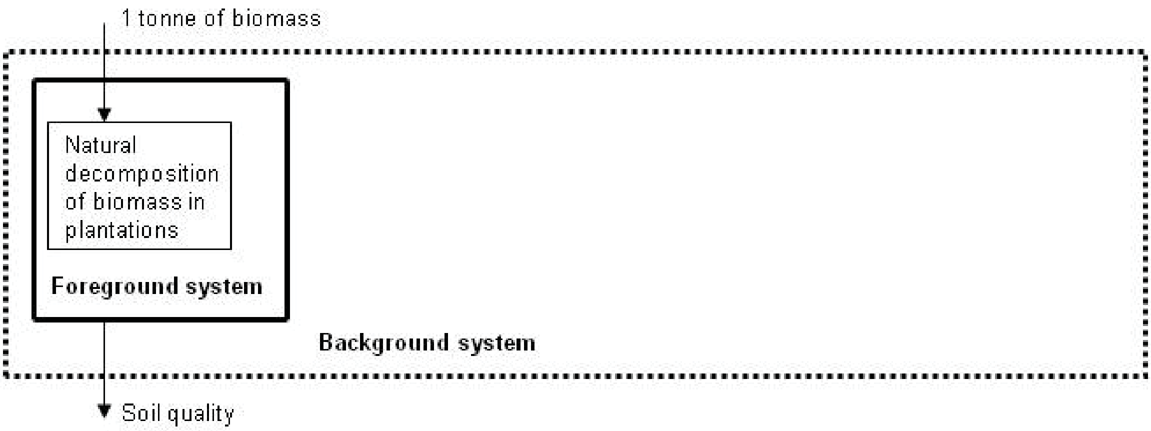

Fig. 17. Business-as-usual scenario for logging residues: logging residues are left unprocessed and nutrients are recycled via natural decomposition in plantations

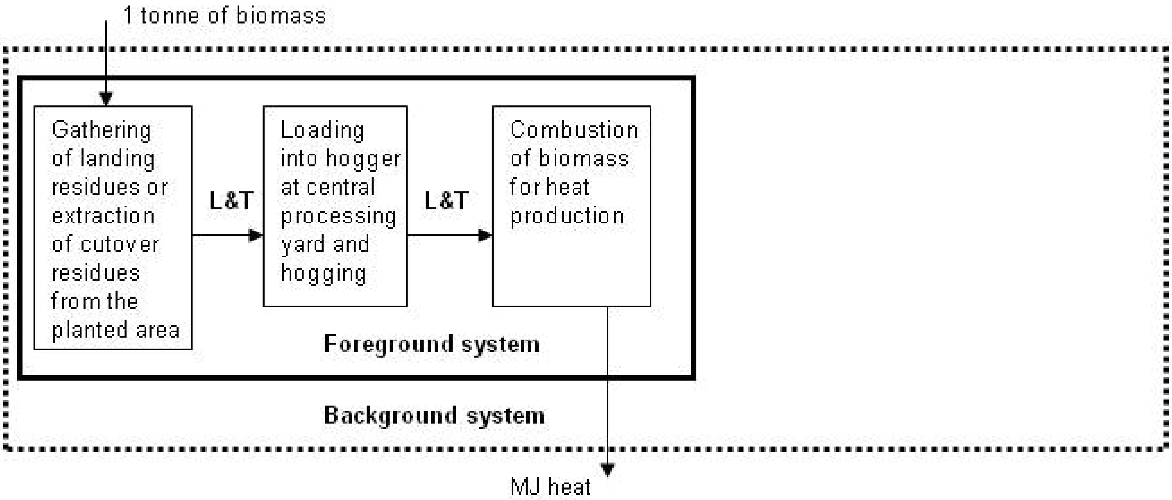

Fig. 18. Heat-only scenario for logging residues (attributional): logging residues are removed from plantations and combusted elsewhere to supply process heat at a heat-only plant (L indicates loading of biomass into trucks and T indicates transport)

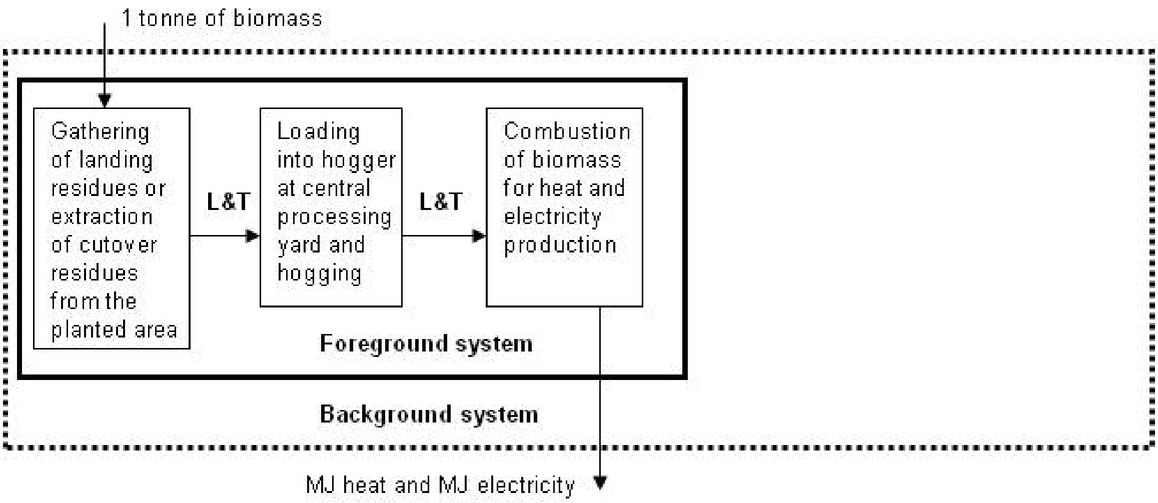

Fig. 19. Combined-heat-and-power scenario for logging residues (attributional): logging residues are removed from plantations and combusted elsewhere to supply heat and electricity at a combined-heat-and-power plant (L indicates loading of biomass into trucks and T indicates transport)

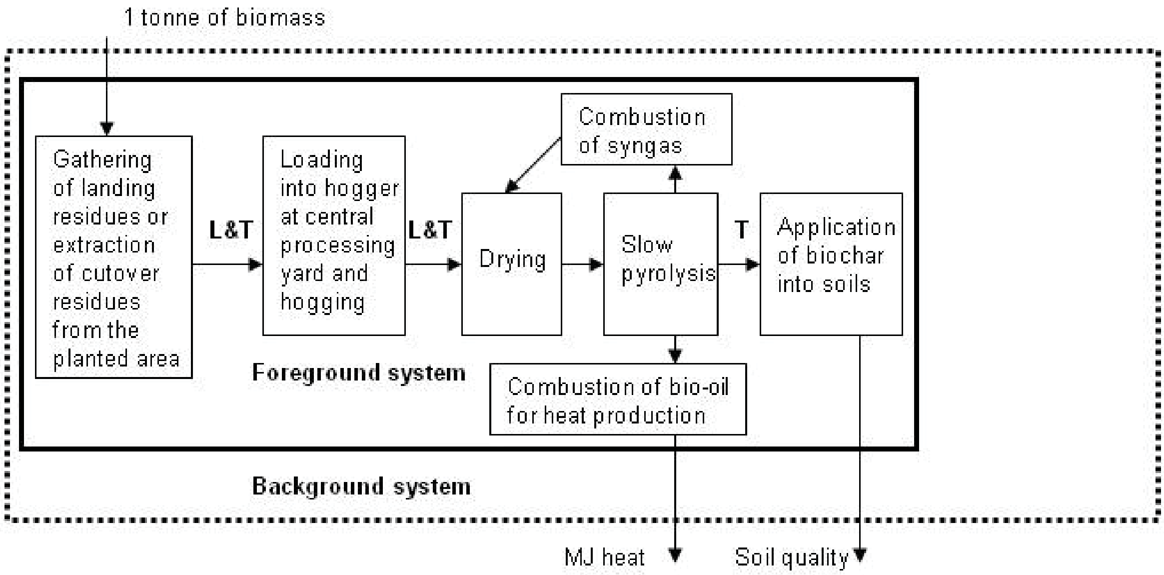

Fig. 20. Biochar scenario for logging residues (attributional): logging residues are removed from plantations and converted by slow pyrolysis into gas, bio-oil, and biochar, which is returned to the same plantation area. The pyrolysis gas is combusted to dry the feedstock; bio-oil is combusted to deliver heat to a processing plant; and biochar is assumed to maintain similar levels of soil quality as in the business-as-usual scenario (L indicates loading of biomass into trucks and T indicates transport)

The life cycles begin with the management of one tonne of fresh logging residues, which were assumed to have a moisture content of 53% (wet basis) and therefore a net calorific value of 7.6 MJ/kg (Hall and Jack, 2008). Logging residues were considered to be outputs of forestry operations that are produced no matter which management practice is undertaken, and so were regarded as free inputs into these systems. Therefore, forestry activities, which include nursery, site preparation, plantation establishment, plantation management, and harvesting, were not accounted for in this study. Moreover, these processes are identical in all of the alternative scenarios.

This study aims at investigating the option for future treatment of biomass that can achieve the largest amount of carbon credits in order to support policymaking. Therefore, alternative scenarios have to be comparable. Since the four scenarios provide different functions (Fig. 17, Fig. 18, Fig. 19, and Fig. 20), the scenarios were expanded to add and/or subtract background processes that would be additionally required and/or displaced in order to deliver equivalent services. In the consequential HO scenario (Fig. 21), indirect consequences include the addition of fertiliser to compensate for removal of nutrients and the subtraction of the combustion of fossil fuels for heat production. In the consequential CHP scenario (Fig. 22), fertiliser was added, whereas the displaced processes were the combustion of fossil fuels for heat production and the delivery of electricity from the grid. In the consequential biochar scenario (Fig. 23), the combustion of fossil fuels for heat production was subtracted and because of the potential of biochar to retain nutrients, it was assumed that similar levels of soil quality as in the BAU scenario would be provided. This assumption was further tested in the sensitivity analysis.

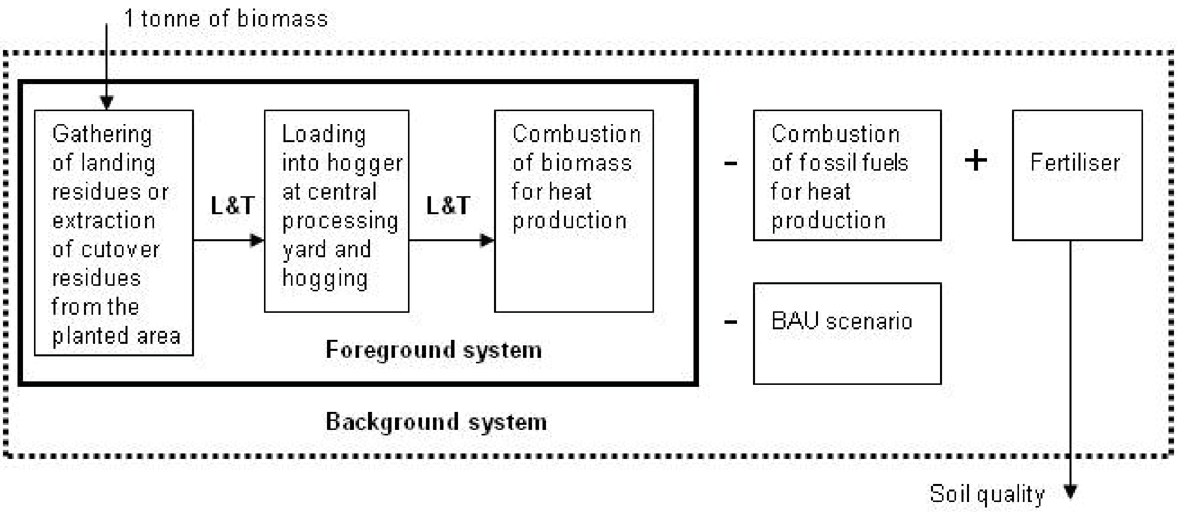

Fig. 21. Heat-only scenario for logging residues (consequential): logging residues are removed from plantations and combusted elsewhere to deliver process heat at a heat-only plant. As a result, the business-as-usual scenario and the combustion of fossil fuels for heat production are displaced whereas some fertiliser is added to replace nutrients (L indicates loading of biomass into trucks and T indicates transport)

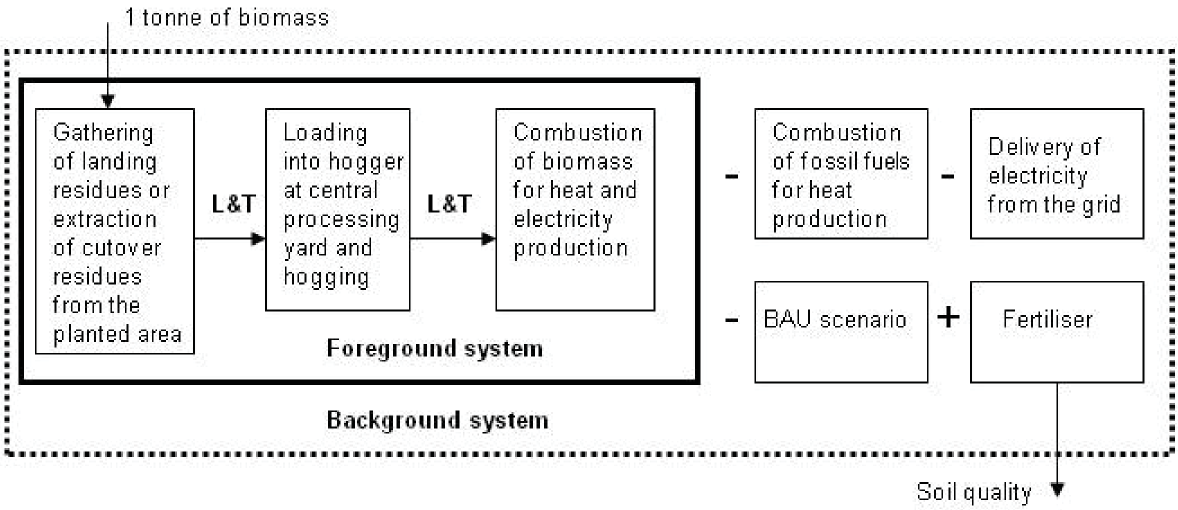

Fig. 22. Combined-heat-and-power scenario for logging residues (consequential): logging residues are removed from plantations and combusted elsewhere to supply heat and electricity at a combined-heat-and-power plant. As a result, the business-as-usual scenario, the combustion of fossil fuels for heat production, and the delivery of electricity from the grid are displaced whereas some fertiliser is added to replace nutrients (L indicates loading of biomass into trucks and T indicates transport)

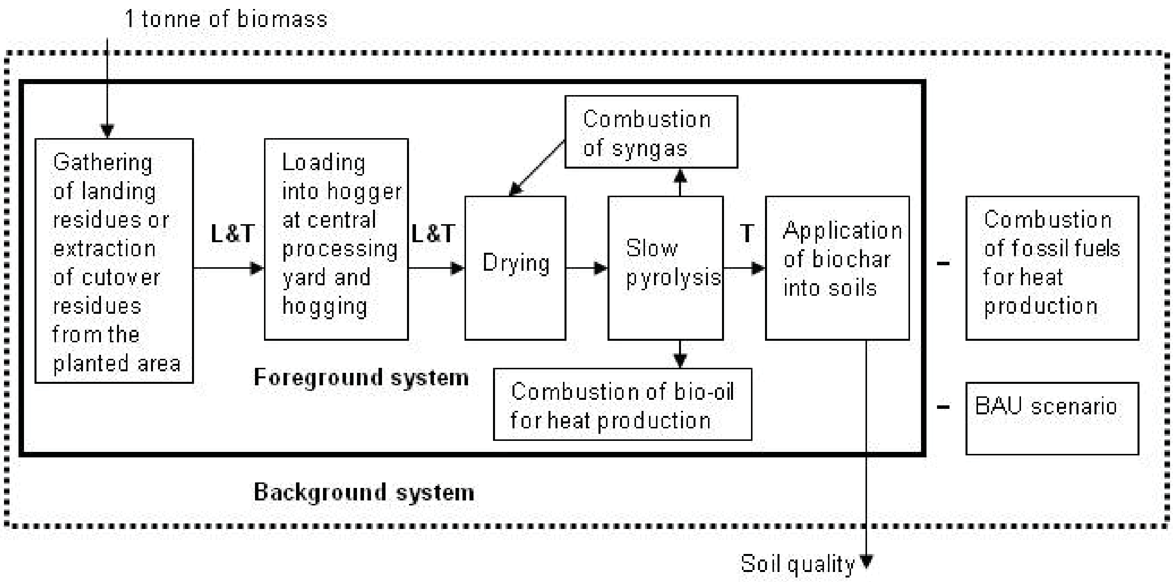

Fig. 23. Biochar scenario for logging residues (consequential): logging residues are removed from plantations and converted by slow pyrolysis into gas, bio-oil, and biochar, which is returned to the same plantation area. The pyrolysis gas is combusted to dry the feedstock; bio-oil is combusted to deliver heat to a processing plant; and biochar is assumed to maintain similar levels of soil quality as in the business-as-usual scenario. As a result, the business-as-usual scenario and the combustion of fossil fuels for heat production are displaced (L indicates loading of biomass into trucks and T indicates transport)

In the BAU scenario, cutover residues are usually left intact at (or near) the stump when log harvesting occurs, whereas landing residues may be pushed to (or over) the edge of the landing site by the logging contractor. This process is currently part of the logging operations and it occurs whether logging residues will be collected or not. Therefore, the BAU scenario was assumed to have a negligible climate-change impact as residues are generally not subjected to any process additional to logging operations and the CO_2 released during their natural decomposition will be absorbed by the next rotation following a "carbon-neutral" cycle. However, it has been suggested that the removal of logging residues or the disturbance of forest floor could affect the long-term storage of carbon and/or nitrogen in NZ's tree plantation soils (Smaill *et al*., 2008; Jones *et al*., 2008). Therefore, nutrient compensation due to the removal of logging residues was included.

The HO scenario was chosen because it was previously found to be the most economic pathway for logging residues to energy production, ahead of five other options in NZ (Hall and Jack, 2008). It also requires the least amount of non-biomass embodied energy and has the lowest GHG emissions of the analysed pathways. The CHP scenario was selected as an alternative pathway since it delivers multiple functions.

Multiple functions

The four alternative scenarios deliver different functions. The BAU scenario maintains soil quality; the HO scenario supplies process heat; the CHP scenario provides process heat and electricity; and the biochar scenario delivers process heat and maintains soil quality. The comparative modelling of these functions is described below.

- Soil quality

In the BAU scenario, logging residues maintain soil quality from nutrient cycling as they release their nutrients back to forest soils during decomposition. Logging residues left on plantations also help to build up soil carbon within the tree plantation area. Taking the BAU scenario as the reference means that all other scenarios have to deliver similar levels of soil quality. Note that 30% of available logging residues were assumed to be retained in the plantations to reduce the risk of soil carbon depletion. Therefore the impact of the withdrawal of 70% of available logging residues on soil carbon levels was considered insignificant. Further research is needed to corroborate this assumption though.

In the HO and CHP scenarios, logging residues are removed from plantations to be burned for energy production elsewhere. It was assumed that an extra amount of fertiliser would have to be applied in order to compensate for this removal of nutrients. Thus fertiliser application was added in the background system of these scenarios to make them comparable.

In the biochar scenario, logging residues are pyrolysed and turned into biochar, which is then applied back into plantation soils. It has been suggested that biochar additions could improve soil quality. The direct nutrient value of biochar is related to the availability of the nutrients in the biochar rather than to the total nutrient content of the biochar (Chan and Xu, 2009). Therefore, the direct nutrient value of wood-derived biochars will be a smaller fraction of their relatively low total nutrient content. Nevertheless, the indirect soil quality value of biochar produced from wood could be high as it offers the potential to retain water and nutrients in soils; reduce leaching; increase pH and cation exchange capacity (CEC); improve fertiliser use efficiency; mitigate soil C loss; and therefore yield higher crop productivity (Lehmann and Joseph, 2009). All these potential benefits, however, need to be assessed for the very specific context of application before large-scale deployment of biochar is undertaken. In this study, the impact of biochar additions on soil quality was assumed to be equivalent to the soil quality function delivered by logging residues in the BAU scenario and this assumption was tested in the sensitivity analysis.

- Process heat

In the HO scenario, logging residues are removed from tree plantations and burned in a fluidised bed biomass combustion plant to produce 20 MW of steam for heat generation only (Sandilands *et al*., 2008). The annual amount of feedstock to be processed depends on the energy conversion efficiency of the plant – among other factors. Sandilands *et al*. (2008) assumed that the HO and CHP plants would operate at 60% combustion efficiency. However, this figure seems low as the conversion efficiency of large-scale wood-fired plants that produce only heat has been cited to be 80% in Norway (Raymer,

2006) and values for Europe generally vary between 70 and 85% (Cherubini *et al*., 2009). In this study, a combustion efficiency of 70% was considered. This figure is higher than previously assumed but it is still conservative. Assuming that the plant operates at 80% availability (7,000 hours per year), the annual feedstock capacity of the 20 MW heat plant would be approximately 95,000 tonnes of wood chips per year:

$$AFC = \frac{PO * AV * 3.6}{C * \eta} \qquad [10]$$

where:

AFC = annual feedstock capacity of the biomass conversion plants (t/year);
PO = power output of heat-only plant = 20 MW;
AV = availability = 7,000 hours per year;
C = net calorific value of wood chips having 53% moisture content = 7.6 GJ/t; and
η = conversion efficiency of the fluidised bed biomass combustion plant = 70%.

Accounting for 5% losses during collection, transport, storage and hogging, about 100,000 tonnes of fresh logging residues would have to be sourced from tree plantations every year. The CHP plant modelled by Sandilands *et al*. (2008) was adapted to match the updated combustion efficiency of 70% and the annual feedstock capacity calculated above. Therefore, the CHP scenario represents the annual combustion of ~95,000 tonnes of logging residues in a fluidised bed biomass combustion plant used to produce 20 MW of steam which is converted into 10 MW of process heat and 3.75 MW of electricity. Consequently, the combustion of fossil fuels for heat production was assumed to be displaced as well as the average grid electricity.

The biochar scenario involves the processing of the same amount of wood, i.e. about 100,000 fresh tonnes of logging residues per year. The pyrolysis gas produced would be combusted to help to induce reactions in the slow pyrolysis reactor but mainly to dry the feedstock. However, this heat was not considered to be a delivered function as this energy is used within the foreground system. The process heat delivered by the biochar scenario corresponds to the combustion of bio-oil. Therefore, to make scenarios comparable, the combustion of fossil fuels for heat production was subtracted.

- Electricity

The CHP scenario encompasses the production of 3.75 MW of electricity in a fluidised bed biomass combined heat and power plant. For the consequential CHP scenario, the marginal displacement of an equivalent amount of coal-based electricity delivered from the grid was substracted. In addition, the average grid mix, instead of coal-based electricity, was considered in the sensitivity analysis.

Geographical area

The geographical scope of this study is restricted to the Central North Island (CNI) region of NZ due to the large volume of logging residues available. Considering the current salvage of logging residues for energy generation, about 1,200,000 dry tonnes of logging residues can be estimated to be available and easily accessible in 2016-2020 (see section 3.3.1). Cutover residues left on steep terrain are not accounted for in this figure. Over 30% of the national wood production is supplied by the CNI region and this is predicted to peak in 2026 to 2030 when more than 70% of logging residues may be concentrated in this area alone (Hall and Gifford, 2008). As calculated above, about 100,000 fresh tonnes of logging residues would have to be collected every year to meet the requirements of the HO, CHP, and biochar scenarios.

Time horizon

A large-scale slow pyrolysis facility was considered for the conversion of logging residues to biochar. The most cited lifetime for large-scale pyrolysis plants is 20 years (Elsayed *et al*., 2003; McCarl *et al*., 2009; Roberts *et al*., 2010; Woolf *et al*., 2010). Hence, the time horizon for the activities described in this study was 20 years. Moreover, a time horizon of 100 years was assumed for carbon sequestration in all scenarios. In the BAU scenario, logging residues left on site would decay over a period of decades and accounting for that carbon sequestered on the forest soil would need time-dependent analysis if compared with energy scenarios (Wihersaari, 2005; Lindholm *et al*., 2011). In contrast, a percentage of the carbon in biochar is argued to remain in the soil for hundreds or thousands of years (Lehmann and Joseph, 2009). In this study, the time horizon of 100 years is used to balance out CO_2-eq emissions against carbon removals. This means that only the recalcitrant fraction of biochar-C can be assumed to be sequestered for 100 years and the carbon temporarily stored in logging residues can be ignored.

Construction and maintenance of capital equipment

GHG emissions associated with the construction and maintenance of the HO and CHP plants and of the machinery used to handle the biomass were neglected. This was mostly due to lack of data but also due to the relatively small impact that these activities have on the whole CF of biomass-to-energy systems. As a comparison, embodied GHGs in capital equipment were calculated for the wheat-straw case study and these can be argued to be negligible (see section 4.4.1).

The CF linked to the establishment of the pyrolysis plant was calculated for the biochar scenario based on Hammond *et al*. (2011). Using the equation developed earlier (see section 4.2.1), the CF due to the construction of the pyrolysis plant would be about 3.9 kg CO_2-eq per functional unit:

$$CFC = \frac{CFP * (1 - MC)}{AFC * LP} \qquad [11]$$

where:

CFC = carbon footprint per functional unit due to the construction of the pyrolysis plant (t CO_2-eq/t);
CFP = greenhouse gas emissions due to the construction of a 20 MW (e) pyrolysis power plant = 19,840 t CO_2-eq (Elsayed and Mortimer, 2001);
AFC = annual feedstock capacity of a 20 MW (e) pyrolysis power plant = 119,774 oven dried tonnes of wood chips per year (Elsayed *et al*., 2003);
LP = lifetime of the slow pyrolysis plant = 20 years; and
MC = moisture content of feedstock modelled in this study = 53%.

Moreover, GHGs due to the maintenance of the slow pyrolysis plant were estimated based on the assumption that 2.5% of GHG emissions arising from the construction of the plant correspond to annual maintenance (Elsayed *et al*., 2003). Therefore, an impact of almost 2 kg CO_2-eq per functional unit is associated with the annual maintenance of the slow pyrolysis plant.

Data quality requirements

According to the ISO (2006a), data quality requirements shall be indicated for the LCA study. Data were collected from recent available sources. Most of the data were particularly drawn from the LCA of energy production from logging residues (Sandilands *et al*., 2008) prepared for the Pathways Analysis report of the Bioenergy Options for New Zealand project (Hall and Jack, 2008). Phone interviews with a forestry researcher (Hall, 2012) and a forest manager (Witehira, 2012) were further conducted in order to confirm figures and inform model development.

Data from specific processes (e.g. diesel production, combustion of fossil fuels, marginal displacement of coal-based grid electricity) were computed using the GaBi 6.0 software's database. Data describing the internal electricity use of the slow pyrolysis plant were obtained from the theoretical analysis undertaken by engineering students of a 'Process Design' course at Massey University (Kayed *et al*., 2011) and adapted to meet the requirements of this study. Data for the average electricity grid mix was based on Coelho (2011).

Since biochar research is relatively recent and long-term local field tests are needed to elucidate the potential soil-related benefits of biochar, hypothetical data from existing CF studies of slow pyrolysis biochar systems (Roberts *et al*., 2010; Hammond *et al*., 2011; Ibarrola *et al*., 2012) were initially evaluated in the first case study presented in this chapter (see *the biochar scenario* in section 4.2.2). It was found, however, that the potential soil-related effects of biochar are highly uncertain and, if corroborated in the field, would represent a negligible climate-change impact per functional unit. This is due to the relatively high biochar application rate needed to observe effects (>10 t of biochar per ha requiring an input of >80 t of fresh biomass collected according to the assumptions made in this case study). Therefore, data associated with the possible soil benefits were omitted from the analysis. This follows the conservative approach suggested in carbon accounting methods such as the clean development mechanism (CDM) of the Kyoto Protocol and some LCA standards (ISO, 2006a).

4.3.2. Life cycle stages

The most relevant processes that take place during the life cycle of each of the alternative scenarios are explained below.

Business-as-usual scenario

The life cycle of the BAU scenario starts with the decision to leave logging residues on forest sites and finishes with the natural decay of wood. In doing so, the nutrients contained in residual wood are recycled in plantation soils.

- Nutrient recycling in plantation soils

There are no processes related to the BAU scenario. Therefore, it was assumed that the climate-change impact per functional unit is zero. Nonetheless, it is well recognised that the abandonment of logging residues on NZ's tree plantations help to maintain soil nutrient pools and carbon stocks (Smaill *et al*., 2008a; Smaill *et al*., 2008b; Jones *et al*., 2008; Jones *et al*., 2011). The amount of recycled nutrients per functional unit was estimated below. The rate of soil organic carbon sequestration, however, is complex as it stems from the time-dependent interaction between climate fluctuations, soil characteristics, tree species and management practices; and the chemical composition of the litter as determined by the main tree crop (Lal, 2005). Due to this complexity, possible changes in soil organic carbon stocks due to the presence (or absence) of logging residues on NZ's forest soils were neglected. Note that a removal rate of 70% of available logging residues was assumed to prevent important soil C loss.

Heat-only scenario

The life cycle of the HO scenario is described by the following pathway: 1) collection of biomass; 2) loading of residues onto trucks; 3) transport of residues to a central processing yard (CPY); 4) loading of residues into a diesel hogger; 5) hogging of residues; 6) loading of wood chips onto trucks; 7) transport of wood chips to the conversion facility; and 8) combustion of biomass for heat production. As a result, the combustion of fossil fuels for heat production is displaced and some fertiliser is added to compensate for nutrient removal.

- Collection of biomass

Two collection methods were considered depending on the type of feedstock. Landing residues are gathered by an excavator type machine fitted with a grapple, which has been modified to handle several small pieces instead of logs. Diesel consumption of the excavator was estimated at 0.47 litres per tonne of landing residues (Sandilands *et al*., 2008). Cutover residues are extracted from the forest to a landing site (or roadside) by a forwarder adapted with a bin or tray. This process requires about 1.6 litres of diesel per tonne of cutover residues (Sandilands *et al*., 2008).

- Loading of residues onto trucks

Landing residues are loaded onto a bin truck at the landing site, whereas cutover residues may be loaded onto a bin truck at the landing site or at the roadside. This process demands approximately 0.27 litres of diesel per tonne of wood for both landing and cutover residues (Sandilands *et al*., 2008).

- Transport of residues to a central processing yard

Logging residues are transported to a CPY for comminution within or near the forest. This intermediate step takes place due to the difficulty of moving large hoggers onto small, wet and rough landing terrains and because of the relatively small amount of residues left at individual landings. A CPY location for comminution will also provide appropriate storage conditions. The transport of logging residues to a CPY needs about 1 litre of diesel per tonne of wood (Sandilands *et al*., 2008).

- Loading of residues into a diesel hogger

Logging residues are loaded into a diesel hogger located at the CPY. Diesel consumed during this process is about 0.27 litres per tonne of logging residues (Sandilands *et al*., 2008).

- Hogging of residues

Wood chips are produced when logging residues are hogged. Although the composition of the two types of ELB is different – cutover residues consist of smaller pieces and may be less prone to dirt contamination than landing residues – a diesel hogger was considered for both landing and cutover residues. A diesel hogger consumes about 1.78 litres per tonne of landing residues (Sandilands *et al*., 2008). Equivalent diesel consumption was assumed for hogging cutover residues. The design of the hogger includes a screen before the chips are discharged at a low speed and therefore the wood has to be blown into a truck from a close distance. This means that the truck must stop next or under the discharge belt for direct loading. In some cases, this is not possible so it was assumed that the wood chips are discharged onto the ground. This can lead to risk of dirt and moisture contamination, material losses and therefore lower production (Visser *et al*., 2010). Note that feedstock losses from collection to conversion were assumed to be 5%.

- Loading of wood chips onto trucks

Loading of wood chips onto trucks at a CPY was assumed to require the same amount of diesel as loading of logging residues, i.e., 0.27 litres per tonne of wood.

- Transport of wood chips to the conversion facility

Wood chips were assumed to be transported by a large high volume truck and trailer unit with a 27 tonne payload capacity. The average transport distance to the conversion facility was 75 km (150 km roundtrip). According to Sandilands *et al*. (2008), this process needs about 3.92 litres of diesel per tonne of residues. This figure may be overestimated as modern trucks are getting more efficient. However, the operation of trucks on forestry roads, which are rough terrain for several km, adds to fuel consumption. Due to the conservative approach stipulated in this study, the higher diesel consumption figure was used.

- Combustion of biomass for heat production

At a fluidised bed biomass combustion plant, wood chips are burned to produce process heat. CO_2 emissions from wood combustion were considered “carbon neutral”, whereas non-CO_2 emissions were neglected. Wood chips were assumed to have a moisture content of 53% and a net calorific value of 7.6 GJ per tonne. The plant was assumed to operate at 70% efficiency. Based on Sandilands *et al*. (2008), the HO plant would require approximately 58 MJ of electricity per tonne of feedstock to produce steam. Considering 5% feedstock losses, about 55 MJ (15.3 kWh) of electricity would be required to produce 5.05 GJ of process heat per functional unit:

$$H = (1 - FL) * C * \eta \qquad [12]$$

where:

H = process heat produced from the combustion of wood chips per functional unit;
FL = feedstock losses = 5%;
C = net calorific value of wood chips having 53% moisture content = 7.6 GJ/t; and
η = conversion efficiency of the fluidised bed biomass combustion plant = 70%.

- Fertiliser is added

In order to calculate how much fertiliser would have to be applied to compensate for the removal of wood, the nutrients recycled in one tonne of logging residues left on site has to be quantified. It should be noted that different tree components have different nutrient content. Branches and particularly foliage have higher nutrient concentrations than stems. Roughly, about 60% of the volume of logging residues is found in approximately 30% of the pieces (Hall, 2012) so most of the 70% of the residues collected would be in relatively large pieces. Since it is in the interest of the residue harvester to collect the larger stem pieces, then it could be assumed that the nutrient-dense branches, bark and needles would be left on the plantation soils. Therefore, in order to calculate nutrient compensation the focus was on stem logging residue as the main component to be collected (Table 27).

Table 27. The weight and nutrient contents of the stem logging residue of a 29-year old stand of Pinus radiata after a clear-felling in the Kaingaroa tree plantation, NZ (adapted from Madgwick and Webber, 1987)

Component	(t/ha)	Nutrients (kg/ha)						
		N	P	K	Ca	Mg	Zn	S
Stem logging residue (dry weight)	40.7	46.2	7.0	64.8	35.3	11.8	0.66	6.5
		Nutrients (kg/t residue at 53% moisture content)						
		N	P	K	Ca	Mg	Zn	S
Stem logging residue (fresh weight at 53% moisture content)	86.6	0.53	0.08	0.75	0.41	0.14	0.01	0.08

Based on Madgwick and Webber (1987), the nutrient content of one tonne of fresh residues to be collected was calculated (Table 27) and the quantities of commercial fertilisers needed to replace these nutrients with their respective climate-change impact were estimated (Table 28).

Table 28. Quantities of commercial fertilisers to be applied to compensate for the removal of nutrients in logging residues and the resulting climate-change impact (kg CO_2-eq per functional unit)

Fertiliser	Nutrient content	Kg of fertilisers needed to compensate for one tonne of residues removed	Life cycle GHG emissions due to fertiliser production	Climate-change impact (kg CO_2-eq per functional unit)
Urea	46% N[a]	1.16	1.786 kg CO_2-eq/ kg urea[d,e]	2.07
Single superphosphate (SP)	9% P[b] 11% S[b]	0.79[c]	0.216 kg CO_2-eq/ kg SP[d]	0.17
Muriate of potash (KCl)	50% K[a]	1.50	0.58 kg CO_2-eq/ kg KCl[d]	0.87
Dolomite	12% Mg[a]	1.14	3.90 kg CO_2-eq/kg Mg[a,f]	0.53
Total	--	3.7	--	3.7

[a]Wells, 2001
[b]Smith *et al.*, 2012
[c]Average between P and S content
[d]Ledgard *et al.*, 2011
[e]This value includes CO_2 emissions from urea application to soils
[f]This value includes CO_2 emissions from dolomite application to soils

The sum of GHG emissions arising from the production and transport from overseas plant to NZ port of P, S, K, Mg and N fertilisers needed to compensate for nutrient removal would be about 3.7 kg CO_2-eq per functional unit. This figure includes CO_2 emissions from soils due to urea and dolomite application but excludes GHG emissions from physical spreading of fertilisers. Coincidentally, the total weight of fertilisers to be applied is approximately 3.7 kg per functional unit.

In order to estimate the GHGs emitted during fertiliser application, the amount of additional fertilisers has to be calculated on a hectare basis. According to the NZFOA (2011) one hectare of 28 year-old *Radiata pine* contains between 650 and 800 m^3 of wood. This yield depends on the type of harvesting system (ground based or hauler) and the quality of the crop. For a ground based system and an average crop quality, the following assumptions were made: 1 m^3 of fresh wood = 1 tonne; total recoverable volume (TRV) = 650 m^3; available amount of landing residues = 5% of TRV; available amount of cutover residues = 15% of TRV; and residues removal rate = 70% of residues per ha. Considering these assumptions, almost 337 kg of total fertiliser material per ha would have to be applied to compensate for the removal of 91 tonnes of logging residues. This corresponds to the application of about one third of total fertilisers generally applied during plantation establishment (Sandilands *et al*., 2011).

Fertiliser application on flat to rolling terrain was assumed to be done by "spot release", which requires the transport of workers to the site (Sandilands *et al*., 2011). The default values given in the guidelines for GHG footprinting of forestry (Sandilands and Nebel, 2010) suggest that GHG emissions arising from the transport of workers for fertiliser application are in the order of 0.06 kg CO_2-eq per ha. It follows that the respective climate-change impact per functional unit in this system would be very small and, therefore, it was omitted from the analysis. In summary, only the climate-change impact of 3.7 kg CO_2-eq per functional unit calculated above was considered at this stage.

It should be pointed out that soil nutrition varies significantly from site to site. In some plantations (e.g. sand dunes) a high residues removal rate would not be prudent, whereas more fertile soils would offer the potential to withdraw more biomass. In Finland, as a comparison, it has been recommended that about 30% of residues be left on plantation soils (Hakkila, 2004). And in a Finnish carbon balance of wood chip production, Wihersaari (2005) considered an outtake of 60% of the residues available and also included nitrogen fertilisation to compensate for nutrient loss.

- The combustion of fossil fuels for heat production is displaced

Heat produced from burning of logging residues was assumed to displace process heat formerly provided through combustion of fossil fuels. Wood processing industries in the CNI region would be the main target users as their facilities demand large amounts of heat, which, is provided, to some extent, by fossil fuels. Hall and Jack (2008) compared the potential of using logging residues for heat generation with the combustion of coal. This comparison was made because some wood processors currently use coal for heat generation and the respective furnace can be fed with wood chips with no major alteration to the system.

Therefore, the use of coal was considered to be displaced. Life cycle emissions from burning coal to produce heat were modelled based on GaBi 6.0 software's data process 'NZ: thermal energy from hard coal PE'. This data set comprises the country specific technology standard plants regarding efficiency and firing technology. Therefore, the heat energy delivered by coal (5.05 GJ per functional unit) would be displaced by the combustion of logging residues. In addition, natural gas was also modelled as a displaced fossil fuel in the sensitivity analysis.

Combined-heat-and-power scenario

The life cycle of the CHP scenario is almost identical to the HO scenario explained above. The difference is that wood chips produced from logging residues are combusted in a CHP plant producing both heat and electricity, instead of just process heat.

- From collection of biomass to transport of wood chips to the conversion facility

Activities occurring at these life cycle stages are equal to the processes described in detail for the HO scenario (see above).

- Combustion of biomass for heat and electricity production

Wood chips are delivered to a fluidised bed biomass CHP plant. The efficiency of combustion to produce steam was assumed to be 70%, whereas the efficiency to produce electricity and process heat from steam was 70% (Sandilands *et al.*, 2008). This results in a net efficiency of 49%. For operation, the CHP plant internally requires about 75 MJ of electricity per tonne of feedstock. Considering 5% feedstock losses, about 71.3 MJ (19.8 kWh) of electricity would be required to produce approximately 3.54 GJ of energy per t biomass:

$$EC = (1 - FL) * C * \eta \qquad [13]$$

where:

EC = energy produced from the combustion of wood chips at a CHP plant per t biomass;
FL = feedstock losses = 5%;
C = net calorific value of wood chips having 53% moisture content = 7.6 GJ/t; and
η = net conversion efficiency of the fluidised bed biomass CHP plant = 49%.

About 73% of the energy output corresponds to process heat, whereas the rest is electricity (Sandilands *et al*., 2008).

- Fertiliser is added

As calculated above for the HO scenario, the total amount of fertilisers to be applied on plantation soils in order to compensate for the removal of nutrients is 3.7 kg per t biomass. Considering default emission factors (Wells, 2001; Ledgard *et al*., 2011), the total production and import of fertilisers has a climate-change impact of about 3.7 kg CO_2-eq per t biomass. GHG emissions produced during the physical application of this fertiliser were neglected as they are insignificant (see above).

- The combustion of fossil fuels for heat production and the delivery of electricity from the grid are displaced

The displacement of coal could happen at dairy or wood processing factories located in the CNI region, but the opportunity could also be harnessed by other industries located within the region. Furthermore, the delivery of coal-based electricity from the grid was assumed to be displaced with electricity generated at the CHP plant. From the amount of energy produced per functional unit, about 2.6 GJ of process heat formerly provided through combustion of coal and 1.0 GJ (278 kWh) of electricity supplied by the grid would be displaced. The GaBi 6.0 software's data process 'NZ: thermal energy from hard coal PE' was used to estimate GHG emission reductions from the displacement of process heat. GHGs avoided due to the marginal displacement of electricity from the grid were modelled using the software's data process 'NZ: Electricity from hard coal PE'. Note that the displacement of the average electricity grid mix was accounted for in the sensitivity analysis.

Biochar scenario

The life cycle of the biochar scenario follows the same handling pathway as the HO and CHP scenarios, from collection of biomass to the delivery of wood chips at a conversion facility. Wood chips are dried and slow pyrolysed to produce gas, bio-oil, and biochar, which is then transported back to the tree plantations to be applied into soils for carbon sequestration, and potentially for improvement of soil functions. Bio-oil is combusted for heat production, which is assumed to displace the use of fossil fuels.

- From collection of biomass to transport of wood chips to the conversion facility

The processes that take place at these life cycle stages are specified in detail above for the HO scenario.

- Drying and slow pyrolysis of biomass

The model of the slow pyrolysis plant was based on Fantozzi *et al*. (2007) and was adapted to meet the requirements of this study. The plant consists of an externally-heated rotating cylindrical reactor operating at $400^{o}C$ for 8,000 hours per year. The reactor is oriented at an angle of 5^{o} to the horizontal and rotated to allow gravity to move the wood chips down the length of the reactor. The feeding section consists of an airtight biomass hopper rigidly connected to a screw conveyor, which is powered by an electric motor. Based on Kayed *et al*. (2011), the electricity required to run the electric equipment (e.g. screw conveyors, fans, motors, pelletiser) was estimated to be 5.4 kWh per 460 kg of feedstock at 10% moisture content for the orchard prunings case study (see section 4.2.1). Proportionally, in this case study, the amount of electricity required to process 420 kg of logging residues with 10% moisture content would be about 4.9 kWh per functional unit.

Once the wood chips had been delivered to the slow pyrolysis plant, they would be introduced into a rotating cylindrical dryer to bring their moisture content down from 53 to 10%. Including 10% heat losses, this process would need heat energy in the order of 3.41 MJ per kg of dry wood (Jones, 2012). Moreover, the rotary kiln requires about 420 kJ per kg of wood at 10% moisture content in order for the slow pyrolysis reactions to take place (Fantozzi *et al*., 2007). The slow pyrolysis plant would produce yield shares – in terms of dried pine wood subjected to a temperature of 400^{o}C and a heating rate of 7^{o}C/min – of 31% biochar, 12% pyrolysis gas and 23% bio-oil (Şensöz and Can, 2002). Note that the reported yields do not add up to 100% and the unreported fraction may have been volatilised. For the purpose of this study, the reported yields were considered as the fractions of products effectively captured and used rather than produced. In views of the lack of maturity of biochar pyrolysis technologies, this is conservative. The model is highly sensitive to these assumptions.

Initial modelling pointed to consider the pyrolysis gas as the energy supplier for the dryer, to bring moisture levels down to 10%, and for the reactor, to help to induce slow pyrolysis reactions. The pyrolysis gas was assumed to have a high heating value of 10.9 MJ per kg (Vamvuka, 2011). It was found, however, that the combustion of the pyrolysis gas alone would not suffice to meet the energy needs of both the dryer and the reactor. Therefore, a certain amount of fresh wood would have to be subtracted from the one tonne of residues collected and then burned to complement the energy supplied by the combustion of pyrolysis gas. Through iteration, the amount of wood chips was calculated for this task. The combustion of 145 kg of fresh wood chips per functional unit would deliver about 1.0 GJ of heat:

$$HRW = \frac{Q * C * \eta}{1000} \qquad [14]$$

where:

HRW = heat energy supplied by the combustion of wood chips to help to meet the energy needs of the system (GJ/t);
Q = quantity of fresh wood chips combusted = 145 kg;
C = net calorific value of wood chips with 53% moisture content = 7.6 MJ/ kg; and
η = combustion efficiency of a wood chip burner including 10% heat losses = 90%.

Note that the combustion efficiency of a wood chip burner was assumed to be 80% but the value of 90% was computed in the equation above as the heat energy needs of the dryer and the reactor include 10% heat losses (see below). In terms of the functional unit, the heat energy needs of the dryer and the reactor were calculated to be ~1.30 and 0.20 GJ, respectively:

$$ED = (1 - FL - Q) * (1 - MC) * HM \qquad [15]$$

where:

ED = energy needs of the dryer per functional unit (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.145 t;
MC = moisture content of fresh wood = 53%; and
HM = heat energy required to bring moisture content of wood down from 53 to 10% = 3.41 GJ per tonne of dry solid wood (10% heat losses included).

$$ER = (1 - FL - Q) * (1 - MC) * (1 + 0.11) * HR / 0.9 \qquad [16]$$

where:

ER = energy needs of the reactor per functional unit (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.145 t;
MC = moisture content of fresh wood = 53%; and

HR = heat needed for slow pyrolysis reactions to occur = 0.420 GJ / tonne of wood at 10% moisture content.

Note that the heat energy needed for the slow pyrolysis reactions to take place in the reactor was divided by 0.9 to account for 10% heat losses and the amount of dried wood was adjusted to include 10% moisture content as the heat needed for the slow pyrolysis reactions to take place was given in terms of weight of feedstock at 10% moisture content (Fantozzi *et al.*, 2007). Total heat energy needs of the dryer and reactor were calculated to be ~1.50 GJ per functional unit. Then the heat energy supplied by the combustion of pyrolysis gas was estimated at ~0.50 GJ per functional unit:

$$HS = (1 - FL - Q) * (1 - MC) * SY * CS \quad [17]$$

where:

HS = heat energy supplied through combustion of pyrolysis gas per functional unit (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.145 t;
MC = moisture content of fresh wood = 53%;
SY = pyrolysis gas yield from slow pyrolysis at 400^{o}C in terms of dried weight of wood fed into the reactor = 12%; and
CS = high heating value of pyrolysis gas = 10.9 GJ/t.

Note that the efficiency of the pyrolysis gas burner was assumed to be 90% but was not computed in the equation above as the energy demand of both the reactor and the dryer include 10% heat losses. Doing the energy balance, the heat energy supplied through burning of 145 kg of fresh wood chips (~1.0 GJ) would complement the energy provided by the combustion of pyrolysis gas (0.50 GJ) to meet the heat energy needs of the system (1.5 GJ). GHG emissions from start-up fuels (e.g. LPG) used to initiate drying and slow pyrolysis processes were neglected due to their minimal impact per functional unit. CO_2 emissions from the combustion of pyrolysis gas were considered carbon neutral and direct methane and nitrous oxide emissions from the pyrolysis process were considered to be zero (Mortimer *et al.*, 2009).

At the slow pyrolysis plant or at a nearby location (close enough to neglect GHG emissions from transport), bio-oil would be combusted to deliver process heat. The pyrolysis of all types of logging residues was assumed to produce bio-oil with similar fuel properties (Das *et al*., 2011). Bio-oil properties represent challenges for its storage, handling and combustion in regular equipment (Czernik and Bridgwater, 2004). A start-up fuel is therefore required to reach stable conditions. GHG emissions from start-up fuels were neglected. According to Bridgwater (2003), fast pyrolysis bio-oil has a calorific value of 17 MJ per kg with about 25% water content. The same heating value was assumed for slow pyrolysis bio-oil. Considering feedstock losses; fresh wood chips combusted to meet the energy needs of the system; and a bio-oil boiler efficiency of 85%, approximately 1.26 GJ of process heat would be delivered per functional unit:

$$HB = (1 - FL - Q) * (1 - MC) * BY * CB * \eta \qquad [18]$$

where:

HB = heat energy provided per functional unit by bio-oil combustion (GJ/t);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.145 t;
MC = moisture content of feedstock = 53%;
BY = bio-oil yield from slow pyrolysis at 400^{o}C in terms of dried weight of wood fed into the reactor = 23%;
CB = net calorific value of bio-oil with 25% water content = 17 GJ/t; and
η = conversion efficiency of bio-oil boiler = 85%.

All biochars produced from different types of logging residues (e.g. tops, branches, and unmerchantable logs), which would be obtained from different harvesting methods have similar properties (Das *et al*., 2011). The resulting biochar would be pelletised by an electric pelletiser at the slow pyrolysis plant to reduce transport costs and dustiness; and facilitate handling and mechanical application into soils (Blackwell *et al*., 2009). Water would be added to assist pelleting and, as explained previously (see section 4.2.2), further research is needed to assess pelleting conditions since these can have an impact on biochar-soil dynamics (Dumroese *et al*., 2011).

- Transport of biochar

Biochar pellets would be transported by road from the slow pyrolysis plant back to the tree plantations in the CNI region. The same average distance of 75 km (150 km roundtrip) has been considered. Although biochar pellets might be transported back to the plantations during the return trip of the truck transporting the feedstock, it was assumed that transport of biochar is done independently. This follows a conservative approach. The bulk density of pelleted biochar was assumed to be 527 kg per m^3 (Dumroese *et al*., 2011). However, the bulk density of biochar was not relevant here as the payload weight of the truck and trailer unit was the limiting factor. Therefore, the same payload of 27 tonnes considered for the transport of wood chips was considered for this process. And the equivalent diesel consumption of 3.92 litres per tonne of residues (Sandilands *et al*., 2008) was assumed for the transport of one tonne of biochar. Then the feedstock-to-biochar ratio, i.e., the tonnage of logging residues collected to produce one tonne of biochar, was calculated to be 8.5:

$$FTB = \frac{1}{(1 - FL - Q) * (1 - MC) * CY} \qquad [19]$$

where:

FTB = feedstock-to-biochar ratio (dimensionless);
FL = feedstock losses = 5%;
Q = quantity of fresh wood chips combusted = 0.145 t;
MC = moisture content of feedstock= 53%; and
CY = char yield from slow pyrolysis at 400°C in terms of dried weight of wood fed into the reactor = 31%.

Hence, the transport of biochar would be responsible for the consumption of ~0.461 (3.92/8.52) litres of diesel per functional unit.

- Application of biochar into soils

At this stage, the processes that can have an impact on the CF of the system include fuel consumption during application of biochar; potential soil quality effects; and biochar-C sequestration in soils. It was assumed that biochar would be applied into the same plantations where the wood originated. After consultation with a forest manager (Witehira, 2012), a four wheel drive tractor (ground spreader) was selected to apply biochar on tree plantations located in flat to rolling terrain. The removal of logging residues would clear some way for the tractor to move around but distribution of biochar would not be even. Based on average values for loading and spreading manure (Dalgaard *et al*., 2001), this process would require an average of 0.6 litres per tonne of biochar. Considering the feedstock-to-biochar ratio and 1% biochar lost during transport (Hammond *et al*., 2011), approximately 0.07 litres of diesel per functional unit would be consumed.

Since carbon sequestration is the main goal of this study, an uneven distribution of biochar should pose no problem. However, incorporation of biochar into soils may be required to reduce the migration of biochar out of the project boundaries and for improvement of soil functions. Therefore, a targeted application of biochar (e.g. spot release) would be more appropriate but also more difficult. Further evaluation on the incorporation of biochar into forest soils is needed since current research on the soil quality effects of biochar in temperate forest soils is limited (McElligott, 2011; Bell and Worrall, 2011).

McElligott (2011) conducted a 30-week laboratory experiment in which biochar produced from the fast pyrolysis of wood was applied to three types of Inland Northwest tree plantation soils in the USA. Biochar was applied at a rate of 25 tonnes per ha using two methods: top-dressing and direct incorporation. This relatively high application rate was selected as previous work suggested effects would not be noticeable at rates of ~2 to 6 tonnes of biochar per ha. The results showed that biochar contributed to short-term soil chemical variations, which depend on soil type and application method. In general, for both application methods, biochar resulted in considerable increments of soil C, organic matter, pH, CEC, and available and exchangeable potassium (K) in all soils. Biochar also decreased N leaching from the Mollisol soil but significantly decreased extractable ammonium in the Andisol soil. The reduction of this soluble inorganic form of N is important as N availability is commonly limited in forest soils and could be disadvantageous for plant growth.

Bell and Worrall (2011) undertook a 28-week experiment, in which lysimeters were placed outdoors under temperate tree plantations and vegetated and bare arable soils typical of North East England. Lump-wood charcoal was incorporated at three different rates: 6.25, 62.5, and 87.5 tonnes per ha. One of the objectives was to assess whether charcoal should be added as a large one-off application or annually at smaller rates. Changes in the net ecosystem respiration, dissolved organic carbon, pH, primary productivity, and nitrate leaching were compared for control and charcoal-amended soils. By and large, the results varied according to the application rate and soil type. For temperate tree plantation soils, the findings support large one-off amendments, whereas the application rate of 6.25 tonnes of lump-wood charcoal per ha augmented the net ecosystem respiration resulting in higher CO_2 losses. This effect, however, was found to be statistically significant only during the first week and was attributed to a possible increase of microbial biomass activity.

A debate on the hypotheses of the observed decomposition (of over 20%) of soil organic C in boreal forests in Northern Sweden after adding charcoal and observing its effects for 10 years (Wardle *et al*., 2008a; Lehmann and Sohi, 2008; Wardle *et al*., 2008b; IBI, 2009) clearly shows that biochar-soil dynamics are not well understood. Therefore, the IBI (2013a) recommends characterising biochar before using it as a soil amendment.

Although the initial aim of this study was to model exclusively the application of biochar into tree plantations, the option to apply biochar elsewhere needs further research. This is due to the lack of long-term local field tests and the difficulty of incorporating biochar into tree plantations. Furthermore, the potential soil-related impacts of biochar can be neglected as they play an uncertain but probably minor role in terms of climate-change mitigation per functional unit. It should be stressed, however, that the appraisal of the soil quality delivered by biochar amendments is fundamental for reasons other than carbon abatement.

During application (top-dressing in this case), it was assumed that 95% of the biochar pellets would eventually migrate downward (Major *et al*., 2010). Therefore, 5% of the biochar material was modelled as losses and cannot be assumed to be incorporated into the soil for carbon sequestration. The average C content of biochar produced by the slow pyrolysis of pine wood at 400^{o}C was 76.3% (Enders *et al*., 2012) and the average biochar-C stability factor, i.e., the fraction of biochar-C that remains stable in soils for ≥100 years, was 74% (average calculated from Roberts *et al*., 2010; Hammond *et al*., 2011; and Ibarrola *et al*., 2012). Considering these assumptions and the C-to-CO_2 conversion rate (44/12), approximately 228 kg CO_2-eq would be sequestered on a per functional unit basis:

$$CSQ = \left(\frac{1}{FTB}\right) * (1 - BL) * CC * BCSF * CCR \quad [20]$$

where:

CSQ= CO_2 sequestered in soils for ≥ 100 years per functional unit (kg CO_2-eq /t);
FTB = feedstock to biochar ratio = 8.52;
BL = biochar losses during transport and application = 6%;
CC = average carbon content of pine wood-derived biochar produced by slow pyrolysis at 400^{o}C = 76.3%;
BCSF= biochar carbon stability factor = 74%; and
CCR = carbon to CO_2 conversion rate = 44/12.

- The combustion of fossil fuels for heat production is displaced

The former use of coal for heat production would be displaced with process heat delivered through combustion of bio-oil (1.26 GJ per functional unit). Note that bio-oil could be transported to the location where heat is in demand. GHG emissions to be displaced were modelled using GaBi 6.0 software's data process 'NZ: thermal energy from hard coal PE'. The displacement of natural gas instead of coal was also considered in the sensitivity analysis.

4.3.3. Results

The results were obtained from the modelling of scenarios using the LCA software GaBi 6.0. The focus is on the climate-change impact category (CML 2001 - Nov. 2010). Data for the production of diesel were taken from the 'US: Diesel mix at refinery' process as GHG emissions calculated for the USA are most similar to the ones documented for NZ (Barber, 2009). The BAU scenario of leaving logging residues on plantation soils was assumed to have a climate-change impact of 0 kg CO_2-eq per functional unit. The results for the HO, CHP and biochar scenarios are presented below.

4.3.3.1. Heat-only scenario (attributional)

The carbon balance of the attributional HO scenario is about 30.0 kg CO_2-eq per functional unit (Fig. 24). The highest impact on the carbon budget (about 40% of the total CF) arises from the transport of wood chips from the CPY to the conversion plant. Loading of wood (0.83 kg CO_2-eq per functional unit) occurs three times: twice into trucks and once into hoggers. Note that this carbon balance is presented for landing residues. The extraction of cutover residues from the forest produces about 3.5 kg CO_2-eq more than gathering of landing residues. Therefore, the CF of the attributional HO scenario describing only cutover residues would be about 33.5 kg CO_2-eq per functional unit.

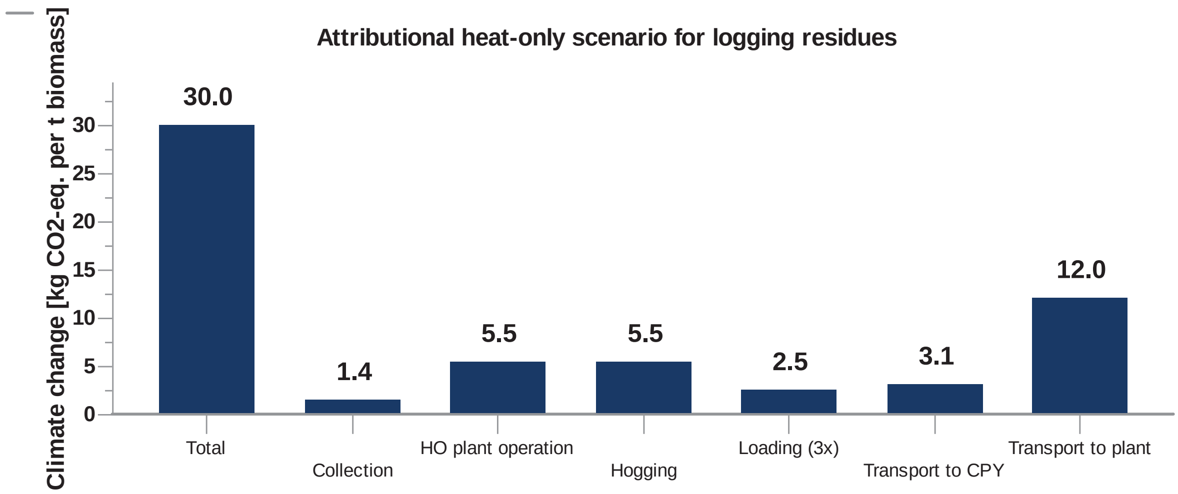

Fig. 24. Breakdown of the climate-change impact per functional unit of the attributional heat-only scenario for logging residues

4.3.3.2. Heat-only scenario (consequential)

The consequential carbon balance of the HO scenario is -473.2 kg CO_2-eq per functional unit (Fig. 25). The combustion of coal that would be displaced as a consequence accounts for -507.0 kg CO_2-eq per t biomass. Addition of fertiliser represents 3.7 kg CO_2-eq per t biomass. Since the BAU scenario was assumed to produce no climate-change impact, the displacement of the BAU scenario did not affect the consequential carbon balance.

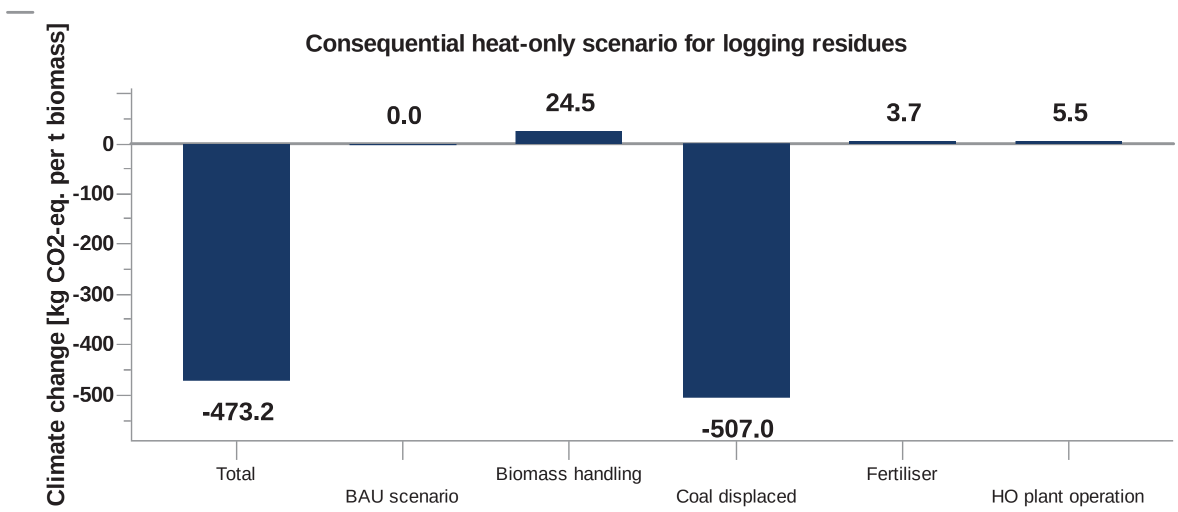

Fig. 25. Breakdown of the climate-change impact per functional unit of the consequential heat-only scenario for logging residues

4.3.3.3. Combined-heat-and-power scenario (attributional)

GHG emissions in the attributional CHP scenario are 31.6 kg CO_2-eq per functional unit (Fig. 26). Since logging residues in the CHP scenario follow the same handling pathway as in the HO scenario, GHGs emitted from the collection to the delivery of logging residues to the conversion plant are similar. The only difference is that the electricity consumed at the CHP plant is slightly higher than at the HO plant. Note that this carbon balance is given for landing residues. The climate-change impact of the management of one tonne of cutover residues in the CHP scenario would be about 35.1 kg CO_2-eq.

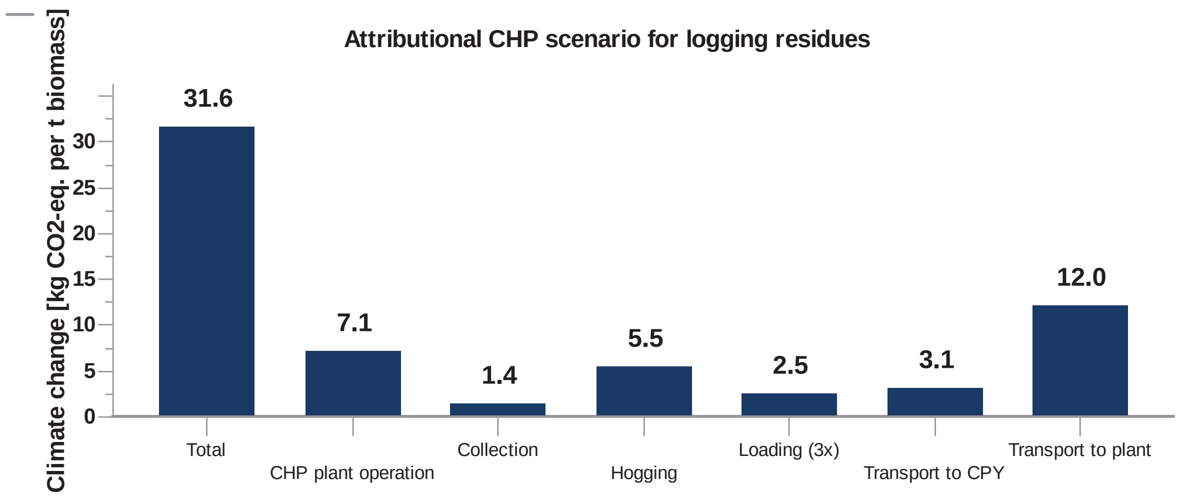

Fig. 26. Breakdown of the climate-change impact per functional unit of the attributional combined-heat-and-power scenario for logging residues

4.3.3.4. Combined-heat-and-power scenario (consequential)

The consequential carbon balance of the CHP scenario is -511.4 kg CO_2-eq per functional unit (Fig. 27). GHG emission reductions due to the displacement of coal combustion formerly used to produce process heat and electricity account for 261.0 and 285.7 kg CO_2-eq per t biomass, respectively. Adding fertiliser increases the CF of the CHP scenario by 3.7 kg CO_2-eq per t biomass.

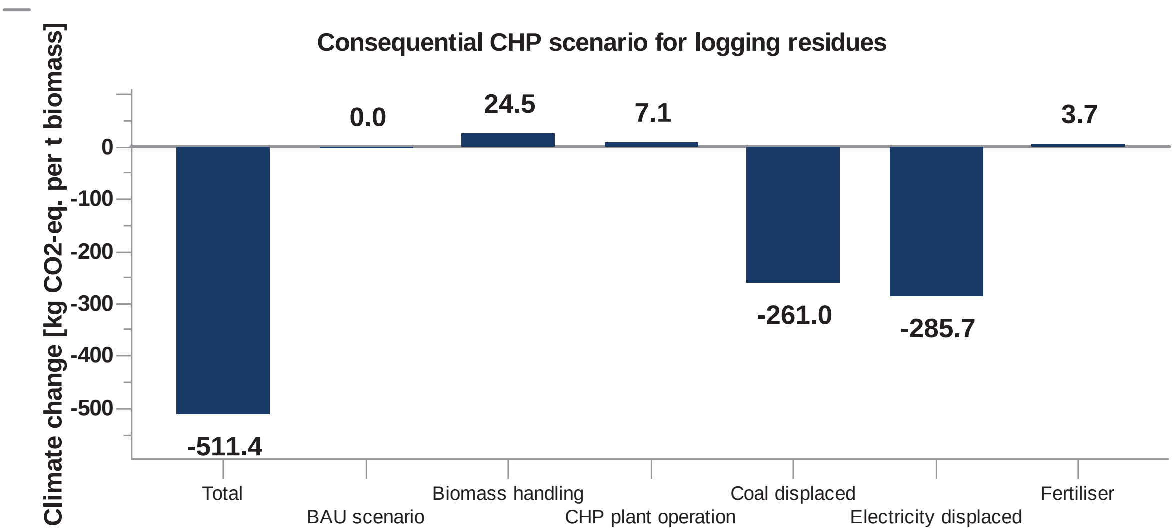

Fig. 27. Breakdown of the climate-change impact per functional unit of the consequential combined-heat-and-power scenario for logging residues

4.3.3.5. Biochar scenario (attributional)

The carbon budget of the attributional biochar scenario is approximately -194.5 kg CO_2-eq per functional unit (Fig. 28). Note that -228.3 kg CO_2-eq per t biomass are due to biochar-C sequestration. As in the HO and CHP scenarios, GHG emissions occurring during handling of logging residues account for 24.5 kg CO_2-eq per functional unit (see Fig. 24 for a detailed breakdown of the different stages). The impact of the pyrolysis plant includes embodied GHGs due to construction (3.9 kg CO_2-eq per t biomass), maintenance (2 kg CO_2-eq per t biomass) and the internal electricity use for plant operation (1.7 kg CO_2-eq per t biomass). The transport and application of biochar represent about 1.4 and 0.2 kg CO_2-eq per t biomass, respectively. The attributional climate-change impact of the biochar system considering cutover residues rather than landing residues would be about -191 kg CO_2-eq per t biomass.

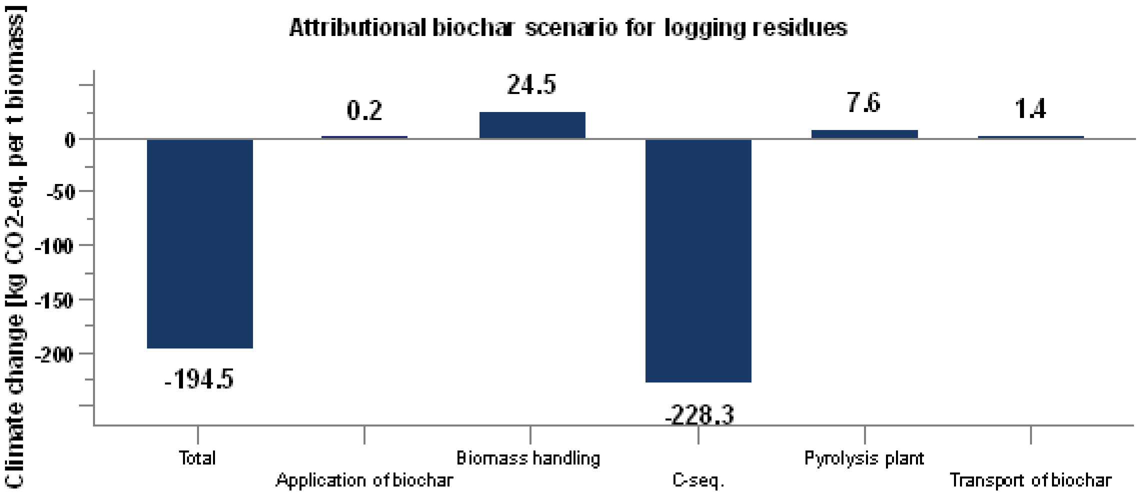

Fig. 28. Breakdown of the climate-change impact per functional unit of the attributional biochar scenario for logging residues

4.3.3.6. Biochar scenario (consequential)

The consequential carbon balance of the biochar scenario is -321.0 kg CO_2-eq per functional unit (Fig. 29). Since the BAU scenario produces no climate-change impact and biochar was assumed to result in similar levels of soil quality as in the BAU scenario, coal displacement is the only consequential activity that has an effect on the carbon balance of this system. This displaced process represents -126.5 kg CO_2-eq per t biomass.

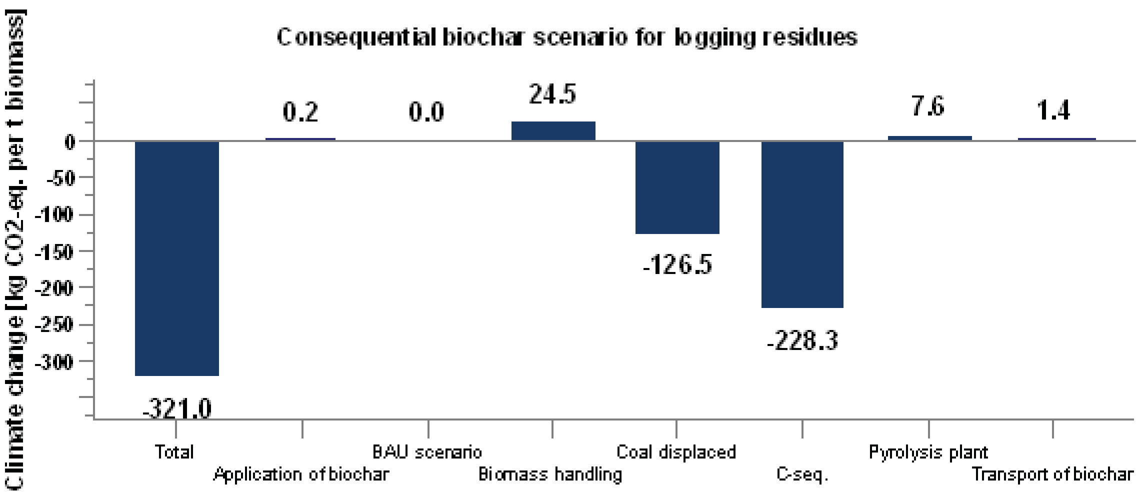

Fig. 29. Breakdown of the climate-change impact per functional unit of the consequential biochar scenario for logging residues

4.3.3.7. Comparison of scenarios

The results are presented for both the attributional and the consequential analyses (Table 29). Note that only the consequential carbon balances presented here are comparable among alternative scenarios (Fig. 30). The BAU scenario was assumed to have a climate-change impact of 0 kg CO_2-eq per functional unit and therefore did not affect the results. Process heat supplied by coal combustion was the activity considered for displacement in all of the alternative uses to BAU, and for the CHP scenario coal-based electricity generation from the grid was assumed to be displaced as well. Furthermore, fertiliser was added in the HO and CHP scenarios.

Table 29. Attributional and consequential climate-change impacts of alternative management options for logging residues considering coal as the source of heat and electricity to be displaced (kg CO_2-eq per functional unit)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with coal displacement (scenarios are comparable)
Heat-only scenario	30.0	-507.0 (displaced coal-based heat generation) +3.7 (additional fertiliser use)	-473.2
Combined-heat-and-power scenario	31.6	-261.0 (displaced coal-based heat generation) -285.7 (displaced delivery of coal-based electricity from the grid) +3.7 (additional fertiliser use)	-511.4
Biochar scenario	-194.5	-126.5 (displaced coal-based heat generation)	-321.0

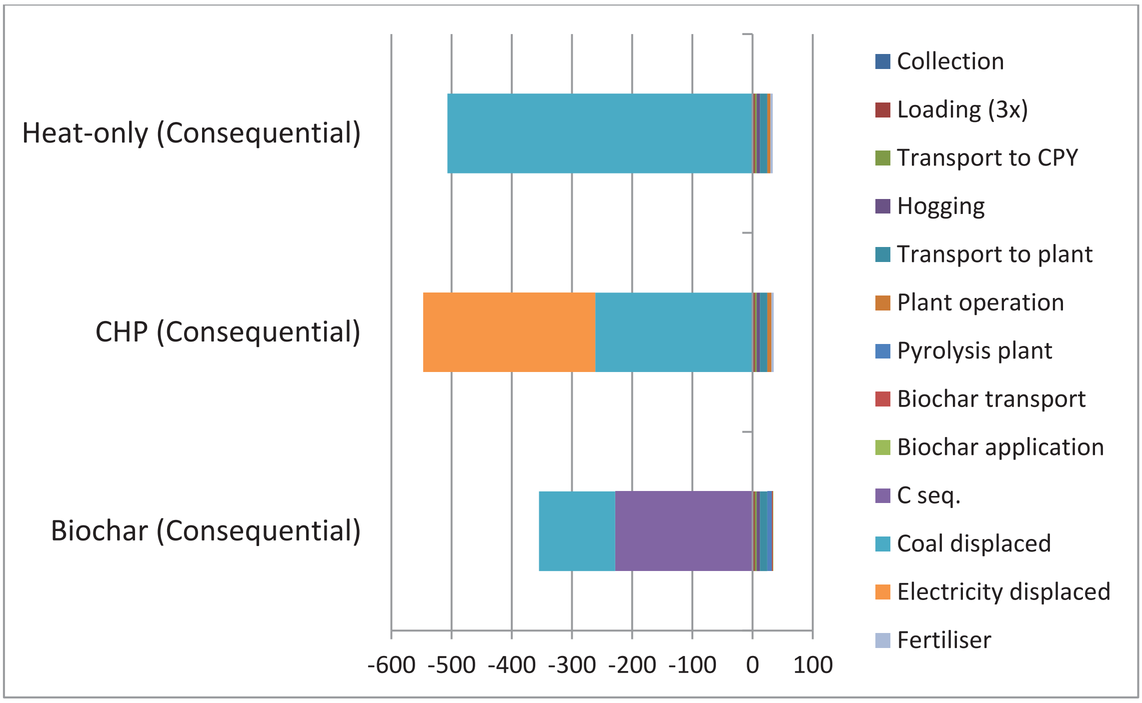

Fig. 30. Comparison of the climate-change impact per functional unit of the alternative scenarios for logging residues (kg CO_2-eq per t biomass)

By comparing consequential carbon balances, the best use of biomass to mitigate climate change can be elucidated. Logging residues that are removed from tree plantations to be combusted at a CHP plant to replace process heat and grid electricity formerly produced by coal combustion would yield the highest GHG emission reductions (511.4 kg CO_2-eq per t biomass). The HO scenario offers the second highest carbon abatement potential of all the alternatives (473.2 kg CO_2-eq per t biomass). The biochar scenario has been ranked third with a total potential for climate-change mitigation of 321.0 kg CO_2-eq per t biomass.

4.3.4. Sensitivity Analysis

Natural gas for heat generation was modelled in the sensitivity analysis as the fossil fuel to be displaced for all scenarios. The data process ‘NZ: Thermal energy from natural gas PE’ was selected in GaBi 6.0 for this task. Furthermore, the displacement of the average electricity grid mix instead of the coal-based electricity was considered for the CHP scenario displacing either coal-based or natural gas-based heat production. Changing these variables, the biochar scenario would offer the second lowest climate-change mitigation potential (Table 30).

Table 30. Attributional and consequential climate-change impacts of alternative management options for logging residues considering natural gas as the source of heat to be displaced for all scenarios and for the CHP scenario, the displacement of both coal-based and average electricity grid mix was modelled. The average electricity grid mix was also considered for the CHP scenario displacing coal-based heat production (kg CO_2-eq per functional unit)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with natural gas displacement (scenarios are comparable)
Heat-only scenario	30.0	-310.2 (displaced natural gas-based heat generation) +3.7 (additional fertiliser use)	-276.5
Combined-heat-and-power scenario displacing natural gas-based heat and coal-based electricity	31.6	-159.7 (displaced natural gas-based heat generation) -285.7 (displaced delivery of coal-based electricity from the grid) +3.7 (additional fertiliser use)	-410.1
Combined-heat-and-power scenario displacing natural gas-based heat and average electricity grid mix	31.6	-159.7 (displaced natural gas-based heat generation) -100.1 (displaced delivery of average electricity grid mix +3.7 (additional fertiliser use)	-224.5
Combined-heat-and-power scenario displacing coal-based heat and average electricity grid mix	31.6	-261.0 (displaced coal-based heat generation) -100.1 (displaced delivery of average electricity grid mix) +3.7 (additional fertiliser use)	-325.8
Biochar scenario	-194.5	-77.4 (displaced natural gas-based heat generation)	-271.9

In the HO scenario, all the energy content of wood is directed towards natural gas displacement. Therefore, the respective GHG emissions reductions are decreased more drastically than the CHP and the biochar scenarios, which, besides displacing the use of fossil fuels for heat generation, involve the displacement of electricity production and the sequestration of carbon, respectively. The consequential CHP scenario displacing coal either as a source of fuel or electricity offers the highest carbon abatement potential. In contrast, the consequential CHP scenario displacing natural gas-based heat production and average electricity grid mix offers the lowest potential. This clearly shows the activities that are assumed to be displaced are key factors when assessing and reporting the consequential climate-change mitigation potential of biomass management systems.

Since biochar is the principal object for assessment in this study, several ranges of values in the biochar scenario were further investigated in a sensitivity analysis. The displaced fossil fuels were considered to be coal or natural gas (Table 31). In addition, the most optimistic and the most pessimistic climate-change impact scenarios of the biochar system were evaluated by modifying all the assumptions made for the different parameters, considering the combustion of coal or natural gas for heat production as displaced activities (Table 32).

Table 31. Sensitivity analysis for the consequential biochar scenario of logging residues considering coal or natural gas combustion for heat production as the displaced activity (- indicates a further reduction, whereas + means a further increase in GHG emissions)

Parameter	Original assumption	Range	Impact of variability on the C balance of the system displacing coal as a heat source	Impact of variability on the C balance of the system displacing natural gas as a heat source
Moisture content of mass fed into the dryer (% wet basis)	53%	30 to 60%	-81.6[a] to +23.4%	-83.2[a] to +23.9%
Pyrolysis gas yield (% of dry mass fed into the reactor)	12%	5 to 20%	+4.9 to -5.6%	+4.9 to -5.8%
Bio-oil yield (% of dry mass fed into the reactor)	23%	15 to 30%	+13.8 to -11.9%	+9.9 to -8.6%
Bio-oil boiler	85%	50 to 90%	+16.3 to -2.2%	+11.7 to -1.6%

efficiency				
Biochar yield (% of dry mass fed into the reactor)	31%	25 to 35%	+13.7 to -9.1%	+16.1 to -10.8%
Biochar losses	6%	0 to 12%	-4.5 to +4.5%	-5.3 to +5.3%
Application of biochar	0.6 l diesel / t biochar	0.4 to 2 l diesel / t biochar	-0.03 to +0.16%	-0.04 to +0.18%
C content of biochar	76.3%	60 to 90%	+15.1 to -12.8%	+17.8 to -15.1%
Biochar carbon stability factor for ≥100 years	74%	50 to 80%	+23.1 to -5.8%	+27.3 to -6.8%
Transport distance	75 km (one way)	30 to 120 km (one way)	-2.5 to +2.5%	-3.0 to +2.9%
Changes in fertiliser use as percentage of nutrient content of logging residues	0%	50 to 100%	+0.6 to +1.1	+0.7 to +1.4%
Biochar migration factor (% of the solid and dissolved states of biochar that migrate out of the project boundaries within 100 years)	0%	10 to 50%	+7.1 to +35.6%	+8.4 to +42.0%

[a]Air drying of logging residues takes place at the central processing yard

Table 32. Most pessimistic and most optimistic scenarios for the consequential biochar system of logging residues considering the displacement of coal or natural gas formerly used to produce heat (see Table 31 for the description of the assumptions made)

	Original scenario	Most pessimistic scenario	Most optimistic scenario
Consequential C balance of the biochar system displacing coal (kg CO_2-eq per functional unit)	-321.0	-25.9	-937.5
Consequential C balance of the biochar system displacing natural gas (kg CO_2-eq per functional unit)	-271.9	-11.9	-802.7

The biochar system is most sensitive to the moisture content of the feedstock, the biochar carbon stability factor and the amount of heat energy delivered through combustion of co-products. The latter has a more significant impact on the climate-change mitigation potential of the system if the displaced fuel was coal rather than natural gas. This is because the displacement of coal represents higher GHG emission reductions than the displacement of natural gas.

For every 6% variation in the biochar carbon stability factor the relative climate-change impact of the biochar system would be affected by 5.8% or 6.8% considering the displacement of coal or natural gas, respectively. The amount of bio-oil produced during slow pyrolysis has to be considered as every 1% decrease in the biochar yield would result in about 1.2% to 1.7% less GHG emission reductions. Since technical challenges to produce energy from bio-oil combustion need to be addressed in the upcoming years it can be argued that the efficiency of the bio-oil boiler was considered to be relatively high (85%). However, the sensitivity analysis shows that for every 5% change in the bio-oil combustion efficiency, the consequential carbon budget of the system would be affected by only 1.6% to 2.2%.

The slow pyrolysis process was initially assumed to convert 31% of the dry weight of pine wood into biochar. Depending on the reference scenario, for every 1% decrease in the biochar yield the climate-change mitigation potential of the system would be reduced by between 2.2% and 2.7%. Another factor that results from the slow pyrolysis process is the carbon content of biochar, which affects the system by approximately 1% for every 1% change. Losses during transport, application, and possibly storage of biochar are to be kept in mind as these translate into less carbon being sequestered within the project boundaries. For every 6% biochar losses, the total carbon balance of the system would be shrunk by 4.5% to 5.3%. The sensitivity analysis confirms that the top-dressing application of biochar on tree plantation soils is not energy intensive in terms of functional unit. However, in order to minimise the migration of biochar out of the project boundaries due to wind and water erosion, the incorporation of biochar into tree plantation soils may be more appropriate during the establishment of plantations.

4.3.5. Discussion

The CHP scenario delivers the highest GHG emission reductions considering the consequential displacement of coal-based grid electricity and either the consequential displacement of coal or natural gas (Table 33). Yet, if natural gas-based heat production and the average electricity grid mix were displaced, the CHP scenario would provide the lowest GHG emission reductions. The HO and biochar scenarios would practically offer similar carbon abatement potentials when natural gas is displaced. However, it is worthwhile to note that in the biochar scenario about 228.3 kg CO_2-eq per t biomass correspond to long-term carbon sequestration regardless of the reference scenario with which it is compared.

Table 33. Consequential climate-change mitigation potential of alternative management options for logging residues (kg CO_2-eq per t biomass)

Management option	Consequential C balance of the system displacing coal as a heat source (kg CO_2-eq per t biomass)	Consequential C balance of the system displacing natural gas as a heat source (kg CO_2-eq per t biomass)
Heat-only scenario	-473.2	-276.5
Combined-heat-and-power scenario displacing coal-based electricity	-511.4	-410.1
Combined-heat-and-power scenario displacing average electricity grid mix	-325.8	-224.5
Biochar scenario	-321.0	-271.9

Note that the climate-change mitigation potential of the management options is presented according to the management of one tonne of landing residues. The carbon balance of the systems would be increased by about 3.5 kg CO_2-eq for the management of one tonne of cutover residues due to the extraction of these residues from the cutover area rather than the gathering of residues at the landing site.

The influence of the consequential biochar system on the average CF of NZ's tree plantation sector was further investigated. Sandilands *et al*. (2011) estimated that the production of 1 m^3 of logs under bark is responsible for: 10 kg CO_2-eq excluding cartage; 17.2 kg CO_2-eq including cartage on an average distance of 52 km from the plantation site to sawmill; or 40.5 kg CO_2-eq including also shipping of export logs to foreign destination ports. These figures include impacts from producing cuttings and seedlings in the nursery; land preparation; plantation establishment and management; harvesting; building of roads and other infrastructure; and the transport of workers to the site. The provision of capital equipment such as trucks and machinery were excluded. Note that Sandilands *et al*. (2011) considered logging residues as a "waste product" free of burden and therefore, there is no double-counting of GHGs involved below.

Considering that the average recovered volumes under bark for radiata pine for the year ended 31 March 2012 was reported to be 541 m^3 per ha (MPI, 2012), the average CF of the tree plantation sector on a ha basis would be: 5,410 kg CO_2-eq excluding cartage; 9,305 kg CO_2-eq including cartage on an average distance of 52 km from the plantation site to sawmill; or 21,910 kg CO_2-eq including also shipping of export logs to foreign destinations.

On a per ha basis, the consequential biochar system could compensate for the CF of export logs by 133% or 113% when displacing coal or natural gas, respectively. This should be attractive enough for the industry to consider the biochar scenario. However, the HO and CHP scenarios offer higher consequential C abatement potential if GHG emission reductions were attributed to the plantation sector and not to the energy users displacing fossil fuels (see section 4.5.4 where the issue of who claims the credits is discussed). Further research is needed to analyse soil-related benefits arising from the application of biochar into NZ's temperate tree plantation soils as these could also play a decisive role in incentivising the widespread use of the technology.

4.4. Carbon footprint study of using wheat straw to produce biochar in a closed-loop system

Cereal straw is an attractive feedstock for biochar production. It has low moisture content and the resulting biochar can be incorporated back into the arable land where it originated relatively easily. Furthermore, biochar produced from cereal straw has the second highest carbon content of all types of NZ's end-of-life biomass (ELB) evaluated in this study (see section 3.4). This means that the respective biochar offers a high C-sequestration potential.

Accessing a sufficient amount of cereal straw for centralised processing (e.g. energy production) is considered to be a challenge (Hall and Gifford, 2008). This is because cereal straw is widely distributed in terms of location and ownership, and its removal from the fields may be perceived to have a negative impact on crop growth due to nutrient depletion and higher risk of soil erosion. However, the production and closed-loop application of straw-derived biochar is different from straw combustion at an energy plant as biochar incorporation can help to improve soil functions (Lehmann and Joseph, 2009). Therefore a biochar system should prove to be a sustainable soil complement to assist productivity.

4.4.1. Goal and scope definition

Goal and objectives

The goal of this carbon footprint (CF) study is to compare four alternative options for the future use of wheat straw in the Canterbury region. The principal objective is to investigate the potential of alternative practices for wheat straw management to mitigate climate change. The results are intended for wheat growers, contractors, energy project developers, policy makers and all stakeholders interested in biochar research and biomass management for climate-change mitigation in NZ.

Functional unit

The functional unit of this comparative CF study is 'the management of one tonne of fresh biomass'. Wheat straw was assumed to have 13% moisture content (wet basis) at the time of baling (Forgie and Andrew, 2008). Furthermore, it was assumed that wheat straw retains 13% moisture content along the supply chain and during storage, from baling to conversion.

System boundaries

The system boundaries extend from the time when wheat straw is left on the fields, after the combine harvester has collected the grain, to when it is converted into alternative products. Current management practices for straw include chopping followed by incorporation into soils; baling for use in construction materials, mushroom growing, mulch, animal bedding or fodder; and burning in open fields for weed and pest control or to reduce tillage passes. However, burning in open fields is less common due to stricter regulations put in place by local governments in NZ (Hall and Jack, 2008) and has become illegal in other countries. Denmark, for example, banned straw burning in fields in 1991 (Skott, 2011). In this study, straw chopping and incorporation was considered as the business-as-usual scenario.

Four different life cycle pathways were modelled using scenarios:

i) business as usual (BAU), in which wheat straw is cut and incorporated into soils to recycle nutrients (Fig. 31);

ii) heat-only (HO), in which wheat straw is removed from fields and burned to produce process heat at a heat-only plant (Fig. 32);

iii) combined heat and power (CHP), in which wheat straw is removed from fields and burned to produce process heat and electricity at a combined heat and power plant (Fig. 33); and

iv) biochar, in which wheat straw is removed from fields and converted by slow pyrolysis into gas, bio-oil and biochar. The pyrolysis gas and bio-oil are combusted to produce process heat, whereas biochar is applied back into wheat-producing land (Fig. 34).

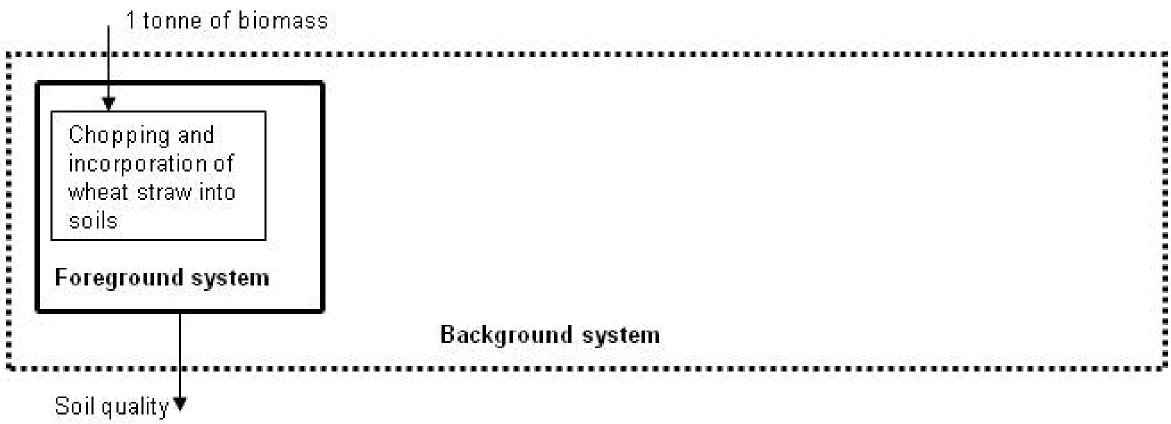

Fig. 31. Business-as-usual scenario for wheat straw: wheat straw is chopped and incorporated into soils to recycle nutrients

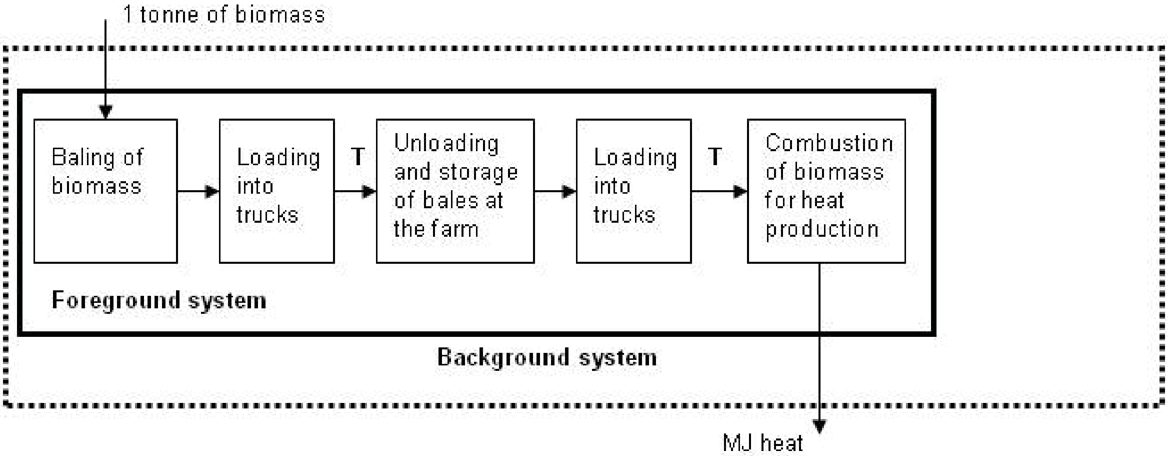

Fig. 32. Heat-only scenario for wheat straw (attributional): wheat straw is removed from fields and combusted to produce process heat at a heat-only plant (T indicates transport)

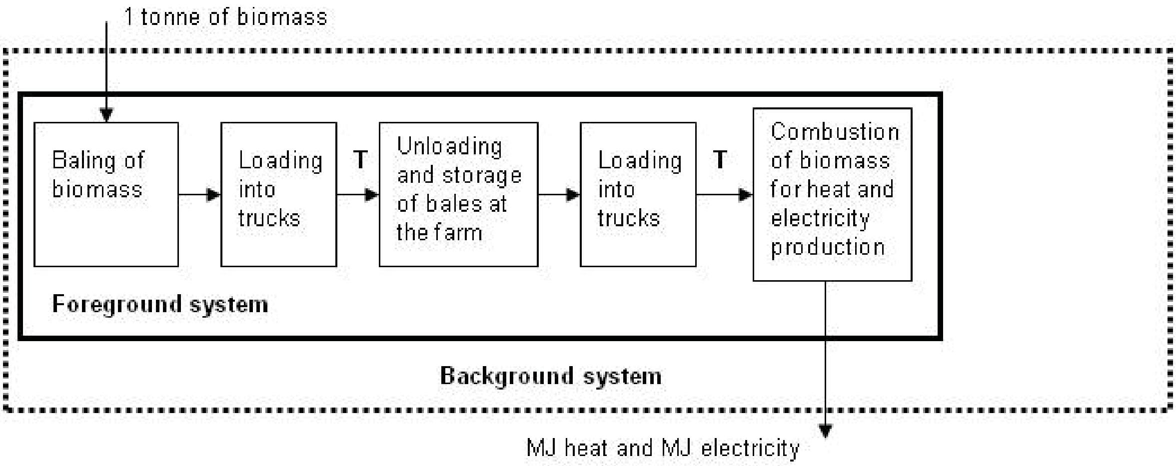

Fig. 33. Combined-heat-and-power scenario for wheat straw (attributional): wheat straw is removed from fields and combusted to produce process heat and electricity at a combined heat and power plant (T indicates transport)

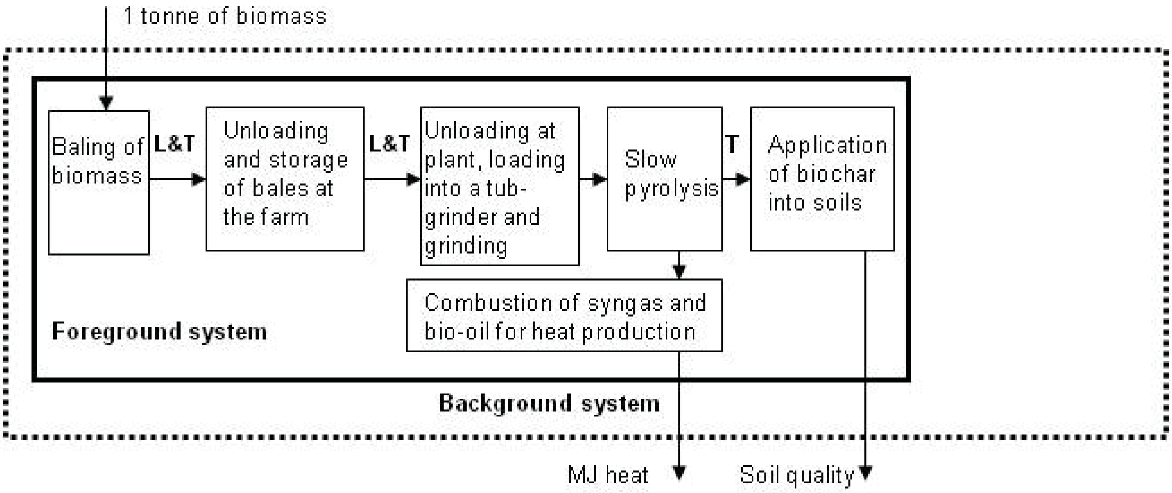

Fig. 34. Biochar scenario for wheat straw (attributional): wheat straw is removed from fields and converted by slow pyrolysis into gas, bio-oil and biochar, which is returned to the same wheat-producing land. The pyrolysis gas and bio-oil are combusted to deliver process heat, whereas biochar is assumed to maintain similar levels of soil quality as in the business-as-usual scenario (L indicates loading of biomass into trucks and T indicates transport)

The beginning of the life cycles is the management of one tonne of wheat straw, which at 13% moisture content was assumed to have an average net calorific value of 14 GJ per tonne. This value derived from Searcy and Flynn (2008) who reported an average net calorific value of 13.9 MJ/kg for straw at 15% moisture content. Straw is an output of wheat production that is created regardless of how it is processed, and so is regarded as a free input into these systems. Therefore, greenhouse gas (GHG) emissions arising from the implementation of the activities needed to produce one tonne of wheat straw were not taken into account. Moreover, these activities are identical in all scenarios.

In order to evaluate the use of straw that can achieve the largest amount of carbon credits, the alternative scenarios have to be comparable. Since the four future biomass management options deliver different functions (Fig. 31, Fig. 32, Fig. 33, and Fig. 34), system expansion/substitution was performed to add and/or subtract background processes additionally required and/or displaced to provide equivalent services. The consequential HO scenario (Fig. 35) includes the addition of fertiliser and the subtraction of fossil fuels combusted for heat production. In the consequential CHP scenario (Fig. 36), fertiliser was added to compensate for nutrient removal, and the combustion of fossil fuels for heat production and the delivery of electricity from the grid were subtracted. In the consequential biochar scenario (Fig. 37), the combustion of fossil fuels for heat production was subtracted, whereas biochar application was assumed to result in similar levels of soil quality as in the BAU scenario. This is mostly due to the capacity of biochar to retain nutrients and enhance nutrient uptake by plants. Furthermore, this assumption was tested in the sensitivity analysis.

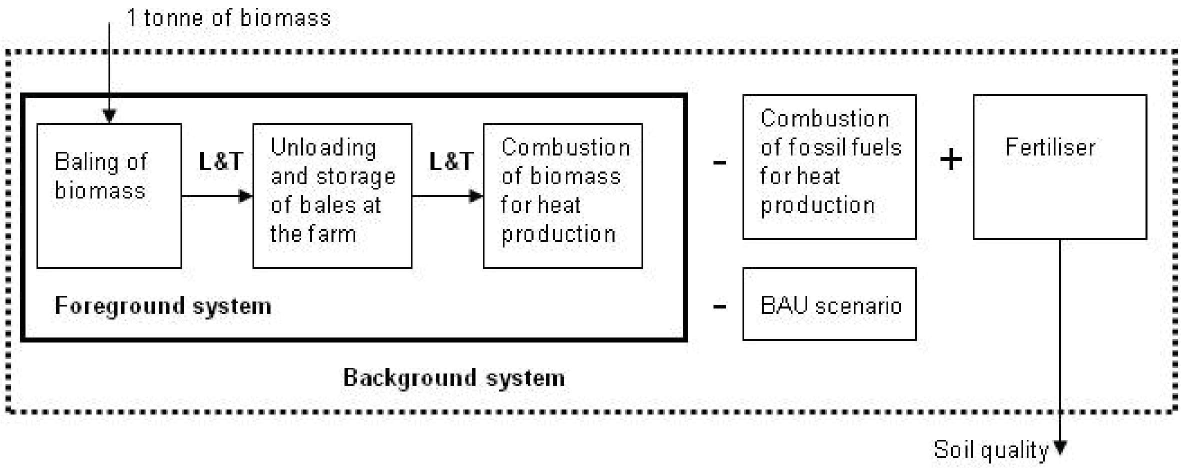

Fig. 35. Heat-only scenario for wheat straw (consequential): wheat straw is removed from fields and combusted to produce process heat at a heat-only plant. As a result, the business-as-usual scenario and the combustion of fossil fuels for heat production are displaced, whereas some fertiliser is added to replace nutrients (L indicates loading of biomass into trucks and T indicates transport)

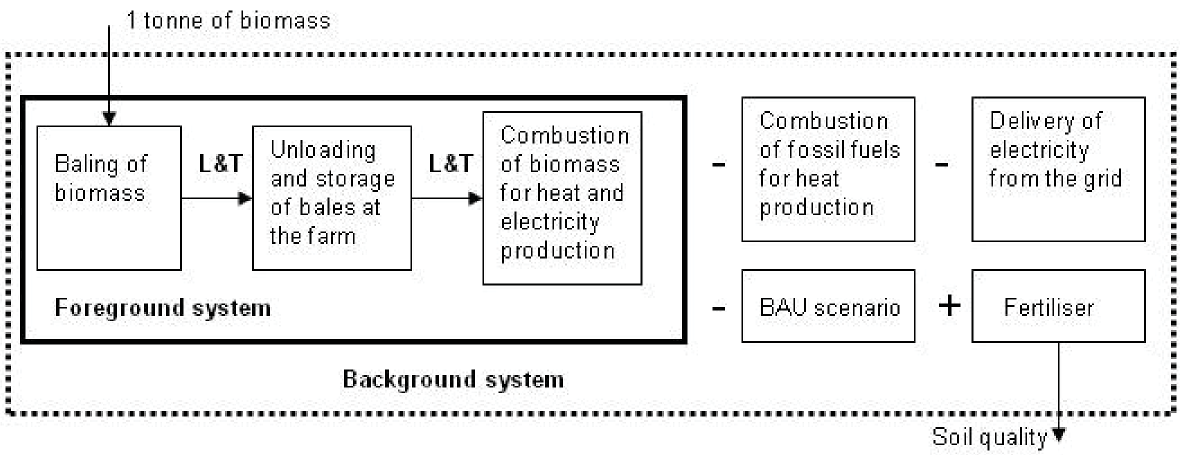

Fig. 36. Combined-heat-and-power scenario for wheat straw (consequential): wheat straw is removed from fields and combusted to produce process heat and electricity at a combined heat and power plant. As a result, the business-as-usual scenario, the combustion of fossil fuels for heat production, and the delivery of electricity from the grid are displaced whereas some fertiliser is added to replace nutrients (L indicates loading of biomass into trucks T indicates transport)

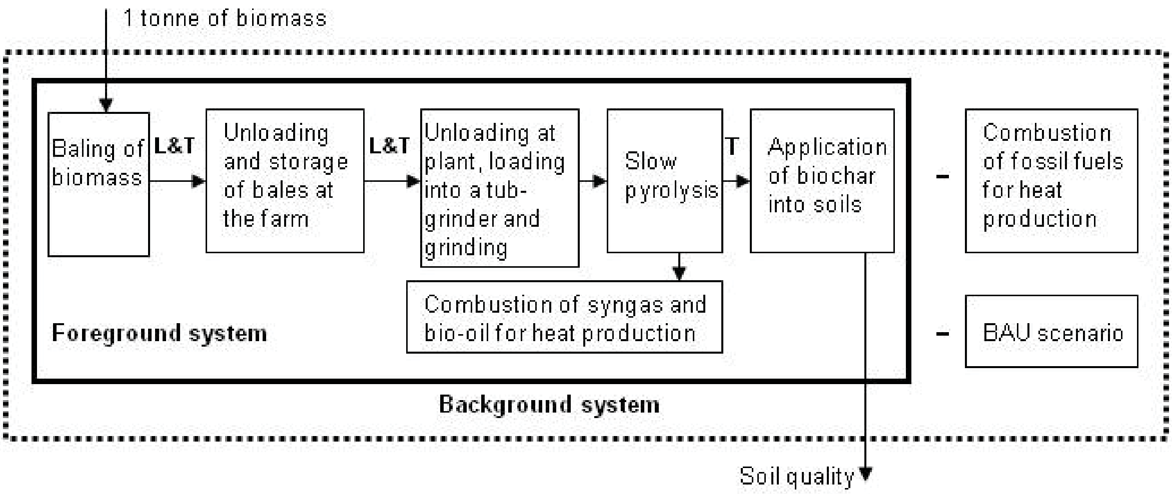

Fig. 37. Biochar scenario for wheat straw (consequential): wheat straw is removed from fields and converted by slow pyrolysis into gas, bio-oil and biochar, which is returned to the same wheat-producing land. The pyrolysis gas and bio-oil are combusted to deliver process heat, whereas biochar is assumed to maintain similar levels of soil quality as in the business-as-usual scenario. As a result, the business-as-usual scenario and the combustion of fossil fuels for heat production are displaced (L indicates loading of biomass into trucks and T indicates transport)

Multiple functions

The four scenarios provide different functions. The BAU scenario maintains soil quality; the HO scenario produces process heat; the CHP scenario delivers process heat and electricity; and the biochar scenario provides process heat and maintains soil quality. The rationale behind the comparative modelling of these functions is explained below.

- Soil quality

In the BAU scenario, the nutrients contained in wheat straw are released back to soils during natural decomposition. This maintains soil quality. Considering this BAU scenario as the reference, the alternative scenarios have to provide similar levels of soil quality. Therefore, the withdrawal of wheat straw from the fields that occurs in the HO and CHP scenarios would lead to the application of additional fertiliser in the background system in order to compensate for nutrient removal. Note that the risk of soil erosion may not be avoided with fertiliser application. In order to prevent major soil dysfunction, it was assumed that only 50% of the available straw would be removed from the fields.

Moreover, the impact of the recovery of 50% of the available wheat straw on soil carbon levels in Canterbury was assumed to be insignificant. This impact is complex to estimate accurately since it depends on several local factors including rainfall, temperature, initial organic matter content, below ground biomass, tillage, fertilisation, straw removal rate, type of rotating crops, type of soil, and microbial activity. In Western Canada, for example, no major reduction in soil carbon has been reported after years of straw removal, and therefore was not considered in life cycle carbon accounting (Searcy and Flynn, 2008). In a recent literature review, Tarkalson *et al*. (2011) pointed at contrasting claims on the effects of straw removal on soil carbon levels under irrigated and rain-fed conditions.

Powlson *et al.* (2011) compared the effects on soil carbon levels resulting from straw incorporation and removal from 23 long-term experiments conducted in mostly cool temperate climate regions in Europe and North America – two studies took place in the subtropical region of Australia though. They concluded that the removal or the incorporation of straw has a small impact on soil carbon content in most situations but warned that it would be unwise to withdraw all the available straw from the fields every year. In a 6-year study in NZ, Curtin and Fraser (2003) found little effect of straw removal on soil carbon and argued that under the relatively warm and mixed farming conditions in Canterbury, where the length of the arable phase of the rotation is less than 6 years, gains in soil carbon as a result of straw incorporation are likely to be small. Hence, it is reasonable to assume that the removal of 50% of the available straw in Canterbury would not have a significant impact on soil carbon levels.

In the biochar scenario, straw-derived biochar was assumed to have low nutrient content but it was assumed that the potential of biochar to ameliorate soil functions (e.g. by retaining nutrients) could result in lower nutrient inputs to the land (Lehmann and Joseph, 2009). Moreover, biochar was assumed to be mixed and charged with the same type of nutrients commonly added to wheat fields. This could be done by mixing the biochar and the fertilisers and allowing the mixture to rest for some days. Currently, there is no evidence about the effects of biochar produced from the slow pyrolysis of wheat straw on soil quality levels in Canterbury’s wheat-growing area. A recent study in Austria, however, suggested that biochar produced from the slow pyrolysis of wheat straw could be considered of low value as a direct nutrient source and of moderate value as a nutrient retainer (Kloss *et al*., 2012). In this study, biochar was assumed to have a neutral effect on soil quality levels relative to the BAU scenario. This assumption follows the approach taken in other case studies (see sections 4.2.1 and 4.3.1) and could be considered to be conservative in views of the potential of biochar management to improve soil functions (Lehmann and Joseph, 2009).

- Process heat

In the HO scenario, wheat straw is taken from the fields to a heat-only plant and burned to fire a boiler, which generates steam for heating or hot water for use in a processing plant (Forgie and Andrew, 2008). In the CHP scenario, straw is burned in a combined heat and power plant to supply process heat and electricity. In the biochar scenario, the co-products, pyrolysis gas and bio-oil, are burned to produce process heat. Therefore, the energy generation facilities would be in close proximity to processing plants that demand a continual source of heat.

In addition, it was assumed that the targeted processing plant would otherwise burn fossil fuels to meet the heat demand. Thus, the scenarios delivering heat would displace this activity and therefore the combustion of fossil fuels for heat production was subtracted in the background system of the HO, CHP and biochar scenarios to account for fossil fuel displacement. Coal was considered as the initial fossil fuel to be displaced and natural gas displacement was included in the sensitivity analysis.

- Electricity

The CHP scenario represents the only life cycle system in which electricity production is a delivered function. If the processing plant using the CHP unit could not take advantage of all the electricity output, then the system would most likely be designed to feed surplus electricity into the national grid. Therefore, it was assumed that a similar amount of electricity from the grid would be demanded in the absence of the CHP scheme and therefore this process was subtracted in the background system. Note that the marginal displacement of coal-based grid electricity was initially assumed in the consequential CHP scenario. In the sensitivity analysis, the displacement of the average electricity grid mix was also considered.

Geographical area

The geographical focus is on Canterbury because it is the largest wheat-producing region in NZ. Timaru, located 160 km southwest of Christchurch on the Canterbury plains, is a possible location for the straw-based energy and biochar plants as the town is surrounded by wheat farms and hosts a number of heat-demanding industries. The area of wheat harvested in Canterbury decreased from 48,000 ha in 2010 to 46,100 ha in 2011 and still accounted for 87.6% of the national harvest (Statistics NZ, 2012). In this study, the average amount of fresh wheat straw available for collection was calculated to be about 4.25 tonnes per ha per year. This is based on average straw production rates of 8.5 tonnes per ha per year and a removal rate of 50% chosen to avoid compromising soil fertility (Hall and Gifford, 2008).

The HO, CHP and biochar scenarios include 5% feedstock losses during collection, transport, storage and shredding of straw. Therefore, to reach the annual feedstock capacity of 40,000 tonnes of wheat straw that has been selected for industrial straw-to-energy production in NZ (Forgie and Andrew, 2008) about 42,000 tonnes of wheat straw would need to be sourced from the fields. This means that approximately 9,900 ha or ~21% of the harvested area in Canterbury in 2011 would be included in the proposed biochar or energy production systems.

Time horizon

The typical lifetime for large slow pyrolysis plants is 20 years (Elsayed *et al*., 2003; McCarl *et al*., 2009; Roberts *et al*., 2010; Woolf *et al*., 2010). This matches the technical lifetime considered for the heat-only and CHP plants modelled in the life cycle study of using straw for energy production in NZ (Forgie and Andrew, 2008). Therefore, the time horizon for the decision to conduct any of the management options was 20 years. Moreover, since the climate change impact of GHG emissions is generally estimated on a 100-years time horizon, only the recalcitrant fraction of biochar-C that can be assumed to remain in soils for 100 years or more is taken into account. This means that the carbon temporarily stored in straw incorporated into soils and the labile fraction of carbon in biochar were not accounted for as long-term carbon sequestration.

Construction and maintenance of capital equipment

GHG emissions due to the construction and maintenance of the capital equipment employed in the BAU scenario (tractor, straw chopper and straw incorporating cultivator) were calculated based on primary energy values estimated for the UK. Primary energy inputs to manufacture the tractor, the straw chopper, and the straw incorporating cultivator account for 93.3, 71.9, and 33.5 MJ per ha per year, respectively (Elsayed *et al*., 2003). The UK figures were then summed and multiplied by the average GHG emission factor suggested for the manufacture, maintenance and international freight of agricultural machinery imported and used in NZ, 0.08 kg CO_2 per MJ (Wells, 2001), and then divided by 8.5 tonnes, the total amount of fresh straw available in one ha of wheat-producing land in Canterbury. This results in a climate-change impact of about 1.9 kg CO_2-eq per functional unit.

Forgie and Andrew (2008) estimated that in order to supply 40,000 tonnes of straw feedstock, the following on-farm equipment would be required: three tractors weighing 7 tonnes each (90 kW); one Hesston 7430 series baler weighing 8 tonnes; and one 10-tonne payload farm truck weighing 5 tonnes. It was also estimated that four similar operations would need to take place simultaneously in order to complete harvest in two months. It was assumed that this equipment would be used for other activities throughout the rest of the year. Then the sum of the weight of all machines was multiplied by NZ's CO_2 emission factor for the manufacture of the machinery (12.8 kg CO_2 per kg), and divided by: 15 years, the working life of on-farm equipment (Wells, 2001); 6 months, to represent the usage of two months per year; and 42,105 tonnes, the amount of biomass handled every year. About 0.5 kg CO_2-eq per functional unit would be emitted during construction, import and maintenance of on-farm equipment.

For long-haul transport to cart the straw and biochar, total equipment that would be used over 20 years time horizon included nine 26-tonne payload trucks weighing 11 tonnes each, and five trailer units weighing 8 tonnes each (Forgie and Andrew, 2008).

Following the same procedure explained above for the on-farm equipment, the respective impact of long-haul transport equipment accounted for about 0.34 kg of CO_2-eq per functional unit.

Note that the CO_2 emission factors for self-propelled vehicles include manufacture, international freight and an allowance for repairs and maintenance (Wells, 2001), but exclude domestic delivery to the farm and storage, which normally have a relatively low climate-change impact on biomass-to-energy systems (Mikkola and Jukka, 2010). The impacts calculated above contribute to the CF of the HO, CHP and biochar scenarios.

It was assumed that the CHP and the heat-only plants would operate throughout the whole year with wheat straw as the only feedstock. Based on material construction data for a 26.7 MW wood-fired CHP plant in Germany (Moerschner and Kazmierczak, 2004), Forgie and Andrew (2008) calculated the amount of construction material, such as concrete, gravel, construction steel and extruded aluminium, that would be required to construct a 33 MW straw-fired CHP plant. The Danish Masnedo plant was the model used in this study (Nielsen *et al*., 2003). The mass of construction materials required were then multiplied by the respective CO_2 emission factors reported for building materials in NZ (Alcorn, 2003) and divided by the lifetime of the plant and by the amount of straw collected per year. The GHG emissions due to the construction of the CHP plant were 7.1 kg CO_2 per functional unit.

Forgie and Andrew (2008) calculated the embodied CO_2 of the heat-only plant by estimating a reduction of the materials used in relation to the manufacture of the CHP plant. Noticeably, the steam turbine, generator and connection to the grid, which would be present in the CHP plant, would not be assembled in the heat-only plant. It was estimated that the heat-only plant would use 5% less concrete; 5% less gravel; 25% less steel; and 5% less extruded aluminium compared to the CHP plant (Forgie and Andrew, 2008). These volumes of materials were multiplied by their CO_2 emission factors (Alcorn, 2003) and divided by 20 years and ~42,000 tonnes of straw collected per year. The impact arising from the construction of the heat-only plant would be about 6 kg CO_2 per functional unit.

The combustion of straw produces ash with relatively high content of alkali metals and chlorine, which can cause corrosion and slagging problems at high temperatures. Therefore, straw-fired plants in Denmark are constructed to allow the replacement of corroded tubes on top of the boiler (Nikolaisen *et al*., 1998). Maintenance and replacement of corroded materials is expensive. Resulting GHG emissions were estimated based on replacing 50% of the steel over the 20 year lifetime of the CHP and heat-only plants (Forgie and Andrew, 2008). These represent approximately 1.82 and 1.36 kg CO_2 per functional unit, respectively.

The embodied GHG emissions of the slow pyrolysis plant were calculated to be 7.2 kg CO_2-eq per functional unit. These were calculated by introducing 13% moisture content and 20 years lifetime of the pyrolysis plant into the equation developed earlier (see section 4.2.1). Data on GHGs emitted during the maintenance of slow pyrolysis plants for biochar production were not found but were evaluated based on the carbon balance estimated for a number of biomass-processing plants in the UK, including a large-scale wood pyrolysis power-only plant. Elsayed *et al*. (2003) assumed that GHGs arising from annual maintenance of all biomass plants account for 2.5% of GHG output from plant construction. Based on this assumption, the climate-change impact due to the maintenance of the slow pyrolysis plant would be about 3.6 kg CO_2-eq per functional unit. This figure is about two times larger than the corresponding impact calculated for the CHP plant, and therefore may be overestimated.

Data quality requirements

Certain characteristics of data should be described for the goal and scope of the study to be met (ISO, 2006a). Most of the data were collected from the LCA study of using straw to produce industrial energy in Canterbury (Forgie and Andrew, 2008), which is based on production processes that are feasible in NZ. Since there are currently no industrial straw-fired plants in the country, data specific to the production of heat and electricity from straw have been adapted from overseas, notably from Denmark (Nielsen *et al*., 2003).

Data for diesel production and marginal displacement of fossil fuels and coal-based grid electricity were drawn from the GaBi 6.0 software's database. Data for the average electricity grid mix in NZ was taken from Coelho (2011). Data for the biochar scenario is based on a considerable number of international studies because no biochar system of this kind exists in NZ. Since a biochar system is extremely dependent on the specific context of production, application, soil and pedoclimatic conditions, data used in this study could be considered merely hypothetical. However, the cautious use of these data can provide insights about the magnitude of the climate-change impacts of each of the life cycle stages and contribute to the development of a general GHG accounting methodology for biochar systems. Therefore, a conservative carbon accounting approach was followed and data uncertainties were further assessed in a sensitivity analysis.

4.4.2. Life cycle stages

The most common processes that would occur along the life cycle stages of the four different scenarios are explained below.

Business-as-usual scenario

Once the wheat straw has been left on the fields by the combine harvester, the BAU scenario involves the chopping and incorporation of straw into soils for nutrient recycling. One key assumption is that the CO_2 released during decomposition of wheat straw on the fields would be absorbed by next year's crop in a so-called "carbon-neutral" cycle. Due to lack of local data, diesel fuel consumption for straw chopping and incorporation into soils was based on dated values for the UK (Elsayed *et al*., 2003) and roughly estimated for the current situation in NZ.

- Chopping of straw

In terms of energy input per ha per year, about 399.8 MJ of diesel are consumed for straw chopping in the UK (Elsayed *et al*., 2003). Considering a consumer energy value for diesel of 37.86 MJ per litre (MED, 2007 cited in Barber, 2009), about 10.56 litres of diesel per ha would be consumed. In order to refer this diesel consumption to the functional unit, the amount of wheat straw processed per ha needs to be estimated. In the UK, the average straw yield was previously cited as 3.78 tonnes per ha (Horne *et al*., 1996; Elsayed *et al*., 2003), which is less than half of what has been recently reported for NZ (Hall and Gifford, 2008). Because of this difference, diesel consumption per ha was divided by the average straw yield between the UK and NZ, which is about 6.14 tonnes of straw per ha. For straw chopping, this translates into 1.72 litres of diesel consumed per t biomass. This average figure falls within the fuel consumption range for chopping straw (0.7-2.1 litres of diesel per tonne of straw) cited for Danish conditions (Dalgaard *et al*., 2001).

- Incorporation of straw into soils

In the UK, straw was estimated to be incorporated through ploughing, which would require 387 MJ per ha (Elsayed *et al*., 2003) or 10.2 litres of diesel per ha. Then, this was divided by 6.14 tonnes of straw per ha, the average straw yield between the UK and NZ, to result in an average diesel consumption for straw incorporation of 1.66 litres per t biomass. Notice that in NZ, if straw is incorporated into soils, it is increasingly with minimum tillage (Safa *et al*., 2010). So, diesel consumption due to the incorporation of straw into soils might be slightly overestimated.

Heat-only scenario

The life cycle pathway of the HO scenario was divided as follows: baling of biomass; loading of bales into farm trucks; transport of biomass within the farm; unloading and storage of bales at the farm; loading of bales into large trucks; transport of biomass to the conversion plant; and combustion of biomass for heat production. Energy inputs required to produce an extra amount of fertilisers that would be applied to compensate for biomass removal were added in the background system, whereas the combustion of fossil fuels to deliver the same amount of heat produced was subtracted.

- Baling of biomass

Forgie and Andrew (2008) estimated that a tractor pulling a Hesston baler 7430 series would be producing 500 kg large square bales at a rate of 23 tonnes per hour and calculated that about 30 MJ or approximately 0.8 litres of diesel per tonne of biomass would be required for this purpose. Since this estimate is about half of what has been reported for baling of straw in Danish agriculture (Dalgaard *et al*., 2001), the average diesel consumption of 1.3 litres per functional unit was assumed here.

- Loading of bales into farm trucks

At the field, a large tractor adapted with a front-end loader for two bales would stack eight bales into a 10-tonne payload farm truck. Forgie and Andrew (2008) estimated that this process would require about 21 MJ or 0.56 litres of diesel per tonne of straw. This figure is slightly higher than the upper range of values reported for Danish agriculture (Dalgaard *et al*., 2001) and was therefore used here.

- Transport of biomass within the farm

A 10-tonne payload farm truck would carry eight bales (4 tonnes) of straw within the farm for storage and would come back empty to the fields. The average one-way transport distance assumed within the farm was 1 km (Forgie and Andrew, 2008). Therefore, diesel consumption was calculated based on a farm truck transporting a 40% full load over 1 km and returning empty. About 0.10 litres of diesel per functional unit would be consumed at this stage. This figure will vary significantly according to terrain, speed, maintenance and driving style.

- Unloading and storage of bales at the farm

A similar tractor employed to load the bales into trucks would be used to unload the bales at the storage site within the farm. The energy required for this activity was calculated as 12.2 MJ per tonne of straw (Forgie and Andrew, 2008), which is equivalent to 0.32 litres of diesel per functional unit.

The square bales would be stored outdoors by wrapping them with plastic. About 186 MJ would be required to manufacture 2 kg of polyethylene sheeting with a lifetime of one year, which is used for the protection of one tonne of wheat straw from the weather (Elsayed *et al*., 2003). Assuming diesel as the primary fuel, about 4.9 litres of diesel per functional unit would be consumed for the manufacture of plastic to store the bales at the farm.

- Loading of bales into large trucks

At the storage site, a large tractor with a front-end loader would load the bales into a 26-tonne payload truck and trailer unit. The truck and trailer unit would carry a total of 24 bales divided in two layers. Similar to the diesel consumed during the loading of bales into farm trucks, about 0.56 litres of diesel per functional unit would be required at this stage.

- Transport of biomass to the conversion plant

Since this study is based on a straw-fired plant with a total feedstock capacity of 40,000 tonnes of straw per year (Nielsen *et al*., 2003), it was estimated that supplying these volumes of biomass to processing plants in Timaru would require an area within an average one-way transport distance of 44 km (Forgie and Andrew, 2008). Due to the volume of bales, the combination of truck and trailer unit would be loaded with only 24 bales (12 tonnes) of straw and would return empty. Based on the diesel consumption of an articulated truck with a maximum payload of 26 tonnes operating at ~50% full load capacity during the one-way trip and coming back empty, Forgie and Andrew (2008) calculated the energy required for the one-way trip as 15.48 MJ per km and for the return trip empty as 7.78 MJ per km. Therefore, fuel consumption due to the transport of biomass to the conversion plant was calculated to be 2.25 litres of diesel per t biomass:

$$DT = \frac{(O * d + R * d)}{C * L} \qquad [21]$$

where:

DT = diesel consumption during long-haul transport of biomass (l/t);
O = diesel energy required for one-way trip = 15.48 MJ/km;
d = one-way distance = 44 km;

R = diesel energy required for return trip = 7.78 MJ/km;
C = calorific value of diesel = 37.86 MJ/litre;
L = load of wheat straw transported = 12 tonnes.

- Combustion of biomass for heat production

At the plant, bales would be unloaded by a crane equipped with grip hooks that can remove 12 bales (one layer) at a time from the truck. Special grabs with microwave moisture-measuring devices are designed to lift pairs of bales and measure their weight and moisture content. The embodied GHG emissions from the construction and operation of these machines were neglected due to lack of data.

The heat-only plant is based on the Sabro district heating plant in Denmark (Forgie and Andrew, 2008). The Sabro plant was constructed in 1991 and therefore represents mature technology. Although shredding of straw bales prior to combustion is common, the boiler plant would be designed for combustion of whole bales. The crane would place the bales in a feeder box and a hydraulic ram stoker would push the bales into a tunnel from where they could be transported by carriers to the burner in the boiler wall (Nikolaisen *et al*., 1998). The volatile gases would be captured and burned. CO_2 emissions from straw combustion are considered "carbon neutral", whereas emissions of other GHGs were dismissed due to their minimal impact. Unburned straw would fall onto the inclined grate and further combusted to produce ash, which would be pushed towards the ash hopper.

The Sabro plant usually operates at 85% efficiency (CADDET, 1997; Forgie and Andrew, 2008). It was assumed that the heat-only plant would require the same amount of diesel per functional unit as the CHP plant (Forgie and Andrew, 2008). This was calculated to be 0.17 litres of diesel per functional unit (see the CHP scenario below). Considering that straw has 13% moisture content and a calorific value of 14 GJ per tonne, about 11.3 GJ of heat would be produced per t biomass:

$$H = (1 - FL) * C * \eta \qquad [22]$$

where:

H = process heat produced from the combustion of wheat straw per functional unit (GJ/t);
FL = feedstock losses = 5%;
C = net calorific value of wheat straw having 13% moisture content = 14 GJ/t; and
η = conversion efficiency of the heat-only plant = 85%.

Besides process heat, fly ash and slag (bottom ash) are co-produced at the plant. In Danish energy schemes, fly ash is sent to landfill because of its high content of heavy metals, whereas slag is taken back to the land and spread as fertiliser. The most important components of slag (weight/weight) have been reported for the Danish Masnedo plant as: 0.8-2.1% phosphorous (P); 0.1-0.7% sulphur (S); 6-13% calcium (Ca); and 0.05-0.08 nitrogen (N) (Nielsen *et al*., 2003). The amounts of fly ash and slag produced from the combustion of one tonne of straw at the Masnedo CHP plant were reported as 8.3 and 54 kg, respectively (Nielsen *et al*., 2003). These outputs were also assumed for the heat-only plant (Forgie and Andrew, 2008). GHG emissions due to the management and transport of fly ash to Timaru's landfill (13-17 km from the processing plants) were neglected.

Assuming 1% loss of slag during transport, the average quantities of nutrients returned to farm via slag on a functional unit basis would be: 0.74 kg P; 0.20 kg S; 4.82 kg Ca; and 0.03 kg N. Forgie and Andrew (2008) assumed that slag would be transported by the same truck carrying the feedstock but on its way back to the farm. Slag would be spread with other fertilisers and no additional application of fertilisers was considered. The assumption was that slag spreading would not exacerbate climate change.

In this study, there is no limitation for the slag co-produced at the plant to be returned to the land. However, it was assumed that nutrient removal would be compensated with chemical fertilisers and not with slag. There are three reasons for this: 1) the use of the Danish average data mentioned above would imply that the resulting slag had slightly more P than the original NZ feedstock, which is not possible; 2) N content of slag is negligible and its application would not affect significantly the spreading of urea, the most polluting fertiliser evaluated in this study (see below); and 3) the additional production and application of chemical fertilisers would give a more conservative CF.

- Fertiliser is added

The nutrient value of one tonne of wheat straw at 13% moisture content has to be estimated in order to apply an equivalent amount of nutrients via fertilisers. NZ's average nutrient content of wheat straw on a weight/weight dry basis is: 0.08% P, 0.13% S, 1.35% K, 0.08% magnesium (Mg), and 0.69% N (Craigie, 2012). Therefore, these percentages were multiplied by 870 kg, i.e. the dry fraction of mass in one tonne of wheat straw at 13% moisture content. Then, the most important nutrients removed per functional unit were calculated to be: 0.70 kg P; 1.13 kg S; 11.75 kg K; 0.70 kg Mg; and 6.0 kg N. This N content of straw correlates well with values reported for French agriculture (Gabrielle and Gagnaire, 2008).

In NZ, the principal phosphate fertiliser used is superphosphate (SP), which has a typical content of 9.0% P and 11.0% S (Smith *et al*., 2012). Considering the P and S content of one tonne of straw estimated above, the average amount of SP to apply would be 9.0 kg per t biomass. GHG emissions based on average values for SP production in NZ, from cradle to production plant, have been reported to be 0.216 kg CO_2-eq per kg SP (Ledgard *et al*., 2011). Therefore, GHG emissions in the order of 1.95 kg CO_2-eq per t biomass would correspond to the production of SP.

Muriate of potash (KCl), which is 50% K (Wells, 2001) would be applied to compensate for K removal. Thus, the K content of one tonne of straw was divided by 0.5 to obtain the amount of KCl to spread. About 23.5 kg of KCl per t biomass would be applied. The GHG emissions of muriate of potash, from cradle to NZ port, were reported as 0.58 kg CO_2-eq per kg KCl (Ledgard *et al*., 2011). Therefore, GHG emissions due to the additional use of muriate of potash account for approximately 13.7 kg CO_2-eq per t biomass.

It was assumed that magnesium in the form of dolomite would be applied. Assuming 90% purity, about 12% of dolomite is Mg and the respective emission factor is 3.9 kg CO_2 per kg Mg (Wells, 2001). This emission factor includes mining, processing and fugitive CO_2 emissions from soil reactions. To compensate for the removal of 0.70 kg Mg, about 5.8 kg of dolomite would be spread and 2.7 kg CO_2-eq per functional unit would be emitted. About 92% of this pollution corresponds to CO_2 emissions from soil reactions, whereas mining and processing contribute to the remaining 8% of the total figure.

In NZ, urea is the predominant form of nitrogen applied to wheat fields (Barber *et al.*, 2011). Urea is about 46% N (Wells, 2001). Since there are about 6 kg N in one tonne of wheat straw collected, about 13 kg of urea would have to be applied to compensate for straw removal. The average estimate of GHG emissions for urea used in NZ, including production; shipping of imported urea to NZ port; and CO_2 emissions from urea application to soils, is about 1.8 kg CO_2-eq per kg urea (Ledgard *et al.*, 2011). Note that CO_2 emissions from soils contribute almost 41% to this emission factor. The total climate-change impact due to N compensation would be approximately 23.3 kg CO_2-eq per t biomass.

The sum of GHG emissions arising from the production and transport from overseas plant to NZ port of P, S, K, Mg and N fertilisers needed to compensate for nutrient removal would be close to 42 kg CO_2-eq per t biomass. Note that this figure includes CO_2 emissions from soils due to urea and dolomite application but does not include diesel consumption due to spreading of fertilisers.

In order to estimate fuel consumption due to fertiliser spreading, the weight of each of the fertilisers calculated above was summed and multiplied by 4.25, i.e. the total tonnage of fresh straw removed per ha. About 221.6 kg of fertilisers per ha would be spread to compensate for nutrient removal. Then the regular weight of fertilisers and lime usually spread on wheat-producing land in Canterbury was needed to come up with a proportional estimate for additional fertiliser spreading. Barber *et al*. (2011) conducted interviews and grower surveys, and provided average data for the Canterbury region.

Based on the amount of nutrients and lime used in Canterbury (Barber *et al*., 2011) and the nutrient content of the commercial fertilisers explained above, the total weight of the regular products used would be about 1,480 kg per ha. Then, diesel consumption per ha due to spreading this amount of fertiliser material needs to be calculated.

Barber *et al*. (2011) gave the total average fuel consumption for wheat growing operations in Canterbury of 75 l of diesel equivalent per ha but did not provide a full breakdown of activities including fertiliser spreading. Safa *et al*. (2010) mentioned that about 13% of the total fuel consumption in Canterbury belongs to fertiliser spreading but reported a total fuel consumption of 65 l of diesel per ha, which is 10 l per ha lower than the one reported more recently (Barber *et al*., 2011). Therefore, as a conservative measure, the high fuel consumption of 75 l of diesel per ha (Barber *et al*., 2011) was multiplied by the 13% total fuel consumption due to fertiliser spreading (Safa *et al*., 2010). This results in diesel consumption of 9.75 l per ha needed to spread about 1,480 kg of fertiliser material. Proportionally, the additional spreading of 221.6 kg of fertiliser material would consume about 1.46 l of diesel per ha or approximately 0.34 l of diesel per t biomass. Note that GHG emissions due to fertiliser spreading might be relatively low but the additional production and transport of fertilisers represents an important climate-change impact that could be overlooked if slag spreading was assumed to compensate for all nutrients removed.

- The combustion of fossil fuels for heat production is displaced

Heat produced during combustion of wheat straw was assumed to displace process heat formerly supplied by burning of fossil fuels. A number of processing plants, including the dairy industry, could use this energy. Considering the scale of the plants (40,000 tonnes of wheat straw per year), two heat-demanding operations in Timaru were selected as plausible locations for the establishment of the biomass conversion plants: 1) the McCain vegetable processing plant, and 2) the Alliance Group freezing works (Forgie and Andrew, 2008). These industries use coal to produce heat and therefore coal combustion was assumed to be displaced (Forgie and Andrew, 2008; Hall and Jack, 2008).

GHG emissions due to the displacement of coal combustion for heat production were then calculated using GaBi 6.0 software’s data process: ‘NZ: thermal energy from hard coal PE’. This process represents NZ’s specific technology standard plants. So, the same amount of thermal energy supplied by straw combustion in the HO scenario, 11.3

GJ (see above) was assumed to be displaced. Moreover, the displacement of natural gas, instead of coal, was included in the sensitivity analysis.

Combined-heat-and-power scenario

The CHP scenario follows the same life cycle pathway as the HO scenario explained above with one exception: wheat straw is burned to produce process heat and electricity rather than process heat only. Therefore, the combustion of fossil fuels for heat production and the delivery of electricity from the national grid were subtracted in the background system. In addition, fertiliser is added to compensate for nutrient removal.

- From baling to transport of biomass to the conversion plant

The life cycle processes during biomass handling that need to take place, from the collection of wheat straw to the delivery of straw bales to the conversion plant, were described in detail for the HO scenario (see above).

- Combustion of biomass for heat and electricity production

The CHP plant is based on the Masnedo heat and power plant in Denmark having a feedstock capacity of about 40,000 tonnes of straw per year. The main production processes taking place at the Masnedo plant, as explained by Nielsen *et al.* (2003) and Forgie and Andrew (2008), are:

1. straw square bales (of about 500 kg each with 10-20% moisture content) are delivered to the plant in trucks and stored, the plant has capacity to store about 1,000 tonnes of straw or over four days of supply;
2. straw is moved by a crane and conveyor lines to two vertical “snails” where the straw is shredded;
3. loose straw is transported to the fire chamber through an air tight channel and burned to heat a boiler to 520^{o}C with a pressure of 90 bar;
4. the walls of the boiler are pipes containing water that becomes steam, which turns the turbine connected to a generator to produce electricity;
5. steam is condensed via heat exchangers and returned to the furnace;

6. condensation heat is transferred to a heat accumulation tank in order to store it while heat is not in demand;
7. heat is distributed to district heating network when needed;
8. the CHP plant is equipped with an electrical filter for reduction of fly ash emission but no SOx and NOx filtration system had been installed; and
9. slag is returned to farm land to be used as fertiliser, whereas fly ash is landfilled due to a high content of heavy metals.

Furthermore, Nielsen *et al*. (2003) provided the average inputs and outputs associated with heat and electricity produced from the combustion of one tonne of wheat straw with moisture content of 10-20% (Table 34).

Table 34. Inventory data for straw-fired at a CHP plant (adapted from Nielsen et al., 2003)

		Unit	**Quantity**
Inputs	Straw	tonne	1.0
	Water	litre	75
	Diesel oil	litre	0.18
	Heat (own production)	MJ	40
	Electricity (own production)	kWh	110
Outputs	**Products**		
	Heat	GJ	8.7
	Electricity	kWh	850
	Air emissions		
	CO_2 (fossil)	g	687
	SO_2	g	680
	NOx	g	1900
	CO	g	910
	HCl	g	670
	N_2O	g	20
	Dioxin	µg	0.32
	Slag and ashes	kg	
	Fly ash (deposit)	kg	8.3
	Slag (returned to farmland)	kg	54

The efficiencies of the Masnedo CHP plant were considered to be 25% for electricity and 63% for heat production (CADDET, 1997; Forgie and Andrew, 2008). Considering that the CHP plant demands about 0.040 GJ of heat per tonne of straw fed to the plant

(Table 34) and that 5% of feedstock is lost during biomass handling, about 8.34 GJ of heat would be produced per t biomass:

$$H_{cogeneration} = (1 - FL) * (C * \eta - H_{plant}) \qquad [23]$$

where:

$H_{cogeneration}$ = process heat produced per functional unit at a CHP plant (GJ/t);
FL = feedstock losses = 5%;
C = net calorific value of wheat straw having 13% moisture content = 14 GJ/t;
η = heat conversion efficiency of CHP plant = 63%; and
H_{plant} = heat demand of CHP plant = 0.040 GJ per tonne of straw fed to the plant.

A similar calculation was done for electricity production. Considering that the CHP plant demands about 0.396 GJ or 110 kWh of electricity per tonne of straw fed to the plant (Table 34), about 2.95 GJ or 819 kWh of electricity would be produced per t biomass:

$$L_{cogeneration} = (1 - FL) * (C * \eta - L_{plant}) \qquad [24]$$

where:

$L_{cogeneration}$ = electricity produced per t biomass at a CHP plant (GJ/t);
FL = feedstock losses = 5%;
C = net calorific value of wheat straw having 13% moisture content = 14 GJ/t;
η = electricity conversion efficiency of CHP plant = 25%; and
L_{plant} = electricity demand of CHP plant = 0.396 GJ per tonne of straw fed to the plant.

In addition, about 0.17 litres of diesel would be consumed per t biomass. Straw combustion would emit CO_2 but a “carbon neutral” cycle was assumed, and therefore respective emissions were not taken into account.

- Fertiliser is added

Similar to the HO scenario, commercial fertilisers would be spread on wheat-producing land to compensate for the extraction of one tonne of wheat straw. GHG emissions due to resorting to P, S, K, Mg and N fertilisers would be in the order of 42 kg CO_2-eq per t biomass (see the HO scenario above). Furthermore, the physical spreading of fertilisers would require approximately 0.34 l of diesel per t biomass (see above).

- The combustion of fossil fuels for heat production and the delivery of electricity from the grid are displaced

Process heat and electricity supplied by the grid formerly produced by combustion of coal would be displaced. The data processes 'NZ: thermal energy from hard coal PE' and 'NZ: Electricity from hard coal PE' were selected from the GaBi 6.0 software's database in order to estimate the respective GHG emission reductions. In addition, the displacement of the combustion of natural gas for heat production was included in the sensitivity analysis as well as the displacement of average electricity grid mix.

Biochar scenario

The biochar scenario encompasses the same life cycle stages specified for biomass handling in the HO and CHP scenarios, i.e. from baling to the delivery of biomass to the pyrolysis facility. As feedstock has 13% moisture content, it was assumed that no drying of straw would be required prior to introducing it into the slow pyrolysis reactor. The feeding system of the slow pyrolysis facility could be constructed in a similar fashion to that of the CHP plant (Nielsen *et al*., 2003). However, it was difficult to extrapolate data specific to the feeding and shredding operations since these processes were not disaggregated from the entire CHP production. Therefore, it was assumed that unloading and loading at the pyrolysis plant would be carried out with a fork-lift truck, whereas shredding of bales would be done with a tub-grinder. The resulting biochar would be brought back to the farms and incorporated into soils to maintain similar levels of soil quality as in the BAU scenario. The pyrolysis gas and bio-oil would be combusted to produce heat, which would displace the combustion of fossil fuels.

- From baling to transport of biomass to the conversion plant

The activities occurring during these life cycle stages were described for the HO scenario (see above).

- Unloading and loading of bales into a tub-grinder, and grinding

In Denmark, straw bales are sometimes unloaded at biomass conversion facilities using fork-lift trucks instead of cranes for storage at the plant (Nikolaisen *et al*., 1998). Since information about fuel consumption of fork-lift trucks is scarce, and the reported data tends to be incomplete or inaccurate (Gaines *et al*., 2008), diesel consumption due to the unloading of bales from the truck and the loading of bales into a tub-grinder was derived from the processes calculated above for the tractor adapted with a front-end loader. Considering 5% feedstock losses, unloading and loading of bales at the plant would consume 0.30 and 0.53 litres of diesel per t biomass, respectively.

Fuel consumption for shredding of wheat straw bales was not found. However, a corn stover supply logistics system including a tub-grinder for comminution of bales with 15% moisture content was used to describe the process. Around 4.5 litres of diesel per t biomass would be consumed during tub-grinding of bales (Morey *et al*., 2010).

- Slow pyrolysis of biomass

The model of the slow pyrolysis plant is based on Fantozzi *et al*. (2007). The plant would operate at 400^{o}C for 8,000 hours per year. Electricity from the grid would be used to run the electric equipment (e.g. screw conveyors, fans, motors, and pelletiser). Using data describing this process for the orchard prunings case study, in which 5.4 kWh are needed per 460 kg of feedstock at 10% moisture content (see section 4.2.1), the proportional amount of electricity required to process 950 kg of straw at the slow pyrolysis plant in this case study would be about 11.2 kWh per t biomass.

The tub-ground straw would be introduced into the slow pyrolysis reactor through a hopper. A start-up fuel, such as LPG, would be burned to power the reactor but GHG emissions from the start-up process were neglected due to their minimal climate-change impact per functional unit. Once running, the reactor would need a relatively low amount of heat to increase the temperature of the feedstock to a certain degree for the slow pyrolysis reactions to take place. Fantozzi *et al*. (2007) reported that for every one tonne of wood feedstock at 10% moisture content, about 0.42 GJ of heat are needed to induce the slow pyrolysis process. The same value has been assumed here for wheat straw. Since 5% of straw collected is lost during handling operations, about 0.950 tonnes of straw at 13% moisture content per functional unit were calculated to be fed into the reactor. This means that the heat of reaction demanded would be about 0.40 GJ per t biomass. Note that straw fed to the plant was assumed to have 13% moisture content and therefore this figure might be slightly underestimated since a bit more heat would be required to evaporate water and decrease moisture content from 13% to 10%.

Pyrolysis gas, bio-oil and biochar yields were based on the examination of wheat straw under slow pyrolysis conditions at temperatures ranging from 300 to 800^{o}C. On a dry and ash-free basis and subject to a 400^{o}C temperature, yields have been reported to be 33% for pyrolysis gas, 15% for bio-oil, and 28% for biochar (Ateş and Işıkdağ, 2008). Since the yields do not add up to 100%, the model is conservative and highly sensitive to these figures. Considering 5% feedstock losses and 13% moisture content, about 0.827 tonnes of dry straw per t biomass would be fed into the reactor. On a functional unit basis, product shares account for 0.273 tonnes of pyrolysis gas; 0.124 tonnes of bio-oil; and 0.231 tonnes of biochar. Therefore the feedstock-to-biochar ratio would be 4.32 (1/0.231).

A certain amount of pyrolysis gas would be combusted to supply the heat of reaction demanded by the reactor, whereas the rest of the gas would be burned to deliver heat energy to the processing plant. So, total heat supplied by the combustion of the pyrolysis gas would be about 2.68 GJ per t biomass:

$$HS = QS * CS * \eta \qquad [25]$$

where:

HS = heat energy supplied per t biomass through combustion of pyrolysis gas (GJ/t);
QS = quantity of pyrolysis gas produced per t biomass = 0.273 tonnes;

CS = calorific value of pyrolysis gas = 10.9 GJ/ t; and

η = conversion efficiency of the pyrolysis gas burner = 90%.

Following a similar procedure, bio-oil combustion would result in about 1.8 GJ of process heat per t biomass:

$$HB = QB * CB * \eta \qquad [26]$$

where:

HB = heat energy provided per t biomass by bio-oil combustion (GJ/t);

QB = quantity of bio-oil produced per t biomass = 0.124 tonnes;

CB = calorific value of bio-oil at 25% moisture content = 17 GJ/t; and

η = conversion efficiency of bio-oil boiler = 85%.

Then, total heat energy delivered by the biochar scenario would be about 4.1 GJ per t biomass:

$$THB = HS - HR + HB \qquad [27]$$

where:

THB = total heat delivered per t biomass by the biochar scenario (GJ/t);

HS = heat energy supplied per t biomass through combustion of pyrolysis gas = 2.68 GJ;

HR = heat energy demanded per t biomass by the reactor for the slow pyrolysis reactions to take place = 0.4 GJ; and

HB = heat energy provided per t biomass by bio-oil combustion = 1.8 GJ;

Since bio-oil combustion can face barriers during start-up, a diesel or LPG burner could be used for this purpose. Respective GHG emissions per functional unit would be low and therefore were neglected. Moreover, similar to the HO and CHP scenarios, CO_2 emissions due to the combustion of the biochar co-products were deemed “carbon neutral”, whereas methane and nitrous oxide emissions from the pyrolysis processes were taken as zero (Mortimer *et al*., 2009).

Biochar would be passed through an electric pelletiser to minimise dustiness, reduce transport costs by densification, and assist handling and application into soils (Blackwell *et al*., 2009). Biochar would also be sprayed with water to facilitate pelleting, reduce losses and minimise potential risks due to breathing of char particles. Note that type of binders and pelleting conditions were not assessed in here as these can play a role in biochar and soil interactions (Dumroese *et al*., 2011). Furthermore, no char losses during pelleting were considered.

- Transport of biochar

The transport of biochar pellets from the pyrolysis facility to the wheat farms and back was assumed to be independent from feedstock transport. Therefore the average roundtrip distance to transport biochar was assumed to be 88 km. Note that the optimal amount of water to add at the plant would have to be examined in order to keep losses and transport costs as low as possible. However, no weight increase due to water spraying of the biochar pellets to be transported was included.

Following the calculation explained above for the transport of biomass to the conversion plant, an articulated truck with a maximum payload of 26 tonnes would consume an average of 0.55 or 0.21 litres of diesel per km when carrying a full load or when travelling empty, respectively (Forgie and Andrew, 2008). Therefore, diesel consumption due to transport of biochar would be about 0.3 litres per t biomass:

$$DTb = \left[\frac{(F * d + R * d)}{L} \right] * WB \qquad [28]$$

where:

DTb = diesel consumption during long-haul transport of biochar (l/t);
F = litres of diesel required for truck with full load during one-way trip = 0.55 l/km;
d = one-way distance = 44 km;
R = litres of diesel required for return trip = 0.21 l/km;

L = load of biochar to be transported = 26 tonnes; and

WB = weight of biochar transported per t biomass = 0.231 tonnes.

- Application of biochar into soils

The factors linked to biochar application into soils that can have an effect on the carbon balance of the system include fuel consumption during physical application of biochar into soils; GHGs avoided due to potential soil-related impacts; and biochar-carbon sequestration in soils.

About 1% of the biochar material was assumed to be lost during transport. Therefore, approximately 0.229 (0.231*0.99) tonnes of biochar per t biomass or almost one (0.229*4.25) tonne of biochar per ha would be applied. At the farm, biochar pellets would be mixed with fertilisers normally spread to grow wheat. This mixture could be allowed to rest for some days before application in order to achieve a more homogeneous adsorption and distribution of nutrients across the biochar surface.

In the HO scenario, it was calculated that about 9.75 l of diesel per ha would be needed to spread about 1,480 kg of lime and fertilisers per year in Canterbury's wheat fields (see above). Assuming diesel consumption for biochar application is proportional to spreading of regular fertilisers, about 1.5 litres of diesel per t biomass would be consumed:

$$BA = \frac{WB*(1-BLT)*FA}{F} \qquad [29]$$

where:

BA = diesel consumption per functional unit during biochar application (l/t);

WB = weight of biochar transported per t biomass = 0.231 tonnes.

BLT = biochar losses during transport = 1%;

FA = diesel consumption per functional unit during regular fertiliser spreading = 9.75 l/ha; and

F = weight of fertiliser material regularly spread annually = 1.48 tonnes per ha.

It was further assumed that incorporation of biochar into the 0-0.3 m soil profile would be through the different tillage operations that are normally conducted in Canterbury's wheat fields (Safa *et al*., 2010). Therefore no additional fuel consumption was allocated to the incorporation of biochar into soils. Furthermore, only an extra 1% loss of biochar pellets during tillage procedures was assumed here.

Biochar addition to soils may be considered to result in a reduction of GHG emissions from soils, fertiliser savings, and changes in soil organic carbon stocks. However the context-specific nature of biochar production and application make these and other soil-related processes highly uncertain. For instance, biochar produced from the pyrolysis of wheat straw at 450^{o}C was reported to decrease the bioavailability of herbicides in amended soils (Nag *et al*., 2011). This could lead to 'under-dosing' of herbicides that could result in a faster development of weed resistance or to higher herbicide application doses that would affect the direct inputs to the land. Due to uncertainties of this nature, speculation around the potential soil-related consequences of biochar application is common in the life cycle studies of biochar systems (Roberts *et al*., 2010; Hammond *et al*., 2011; Ibarrola *et al*., 2012).

In this study, the potential soil-related benefits of biochar systems have been acknowledged and some were hypothesised in the apple orchard case study (see section 4.2.3). The results show, however, that in terms of climate-change mitigation per functional unit, these factors, if proven to be positive, could still be neglected due to the high inputs of biomass needed to achieve relatively little benefits. The exclusion of these potential impacts might also help to keep carbon accounting, validation, monitoring, reporting, and verification methods as simple as possible.

Biochar-C sequestration in soils also involves a degree of uncertainty but to a lesser extent than soil-related benefits. The main factors affecting the carbon sequestration potential of biochar are: the char yield; the C content of respective biochar, which varies with type of feedstock and production parameters; and the biochar-C stability factor (BCSF), i.e. the fraction of carbon in biochar that can be assumed to remain in soils for at least 100 years. These variables in this study were originally assumed to be: 28% for the biochar yield (Ateş and Işıkdağ, 2008); 71% for the carbon content of wheat straw-derived biochar (Mahinpey *et al*., 2009); and 74% for the BCSF (average calculated from Roberts *et al*., 2010; Hammond *et al*., 2011; and Ibarrola *et al*., 2012). Including feedstock and biochar losses during transport and application, carbon sequestration accounted for about 437.0 kg CO_2-eq per t biomass:

$$CSQ = BP * (1 - BL) * CC * BCSF * CCR \quad [30]$$

where:

CSQ = CO_2 sequestered in soils for ≥ 100 years per t biomass (kg CO_2-eq/t);
BP = biochar produced per t biomass = 0.231 tonnes;
BL = biochar losses during transport and application = 2%;
CC = carbon content of wheat straw-derived biochar = 71%;
BCSF = biochar-carbon stability factor = 74%; and
CCR = carbon to CO_2 conversion rate = 44/12.

- The combustion of fossil fuels for heat production is displaced

The combustion of coal to produce heat at a processing plant in Timaru was assumed to be displaced with heat supplied by burning the pyrolysis gas and bio-oil produced from the slow pyrolysis of wheat straw. GaBi 6.0 software's data process 'NZ: thermal energy from hard coal PE' was chosen to calculate the respective GHG emission reductions. The amount of energy produced in the biochar scenario (4.1 GJ per functional unit) would be equivalent to the total process heat displaced.

4.4.3. Results

The carbon balances of the four different scenarios were calculated by focusing on the climate-change impact category (CML 2001- Nov. 2010) of the LCA software GaBi 6.0. Diesel production was modelled using the data process 'US: Diesel mix at refinery' since respective GHG emissions are similar to the ones reported for NZ (Barber, 2009).

4.4.3.1. Business-as-usual scenario

Total GHG emissions arising from the BAU scenario account for about 12.3 kg CO_2-eq per functional unit (Fig. 38). The climate-change impact arising from chopping and incorporation activities is 5.3 and 5.1 kg CO_2-eq per t biomass, respectively. GHGs due to construction and maintenance of the capital equipment contribute about 15% to the total CF of the BAU scenario.

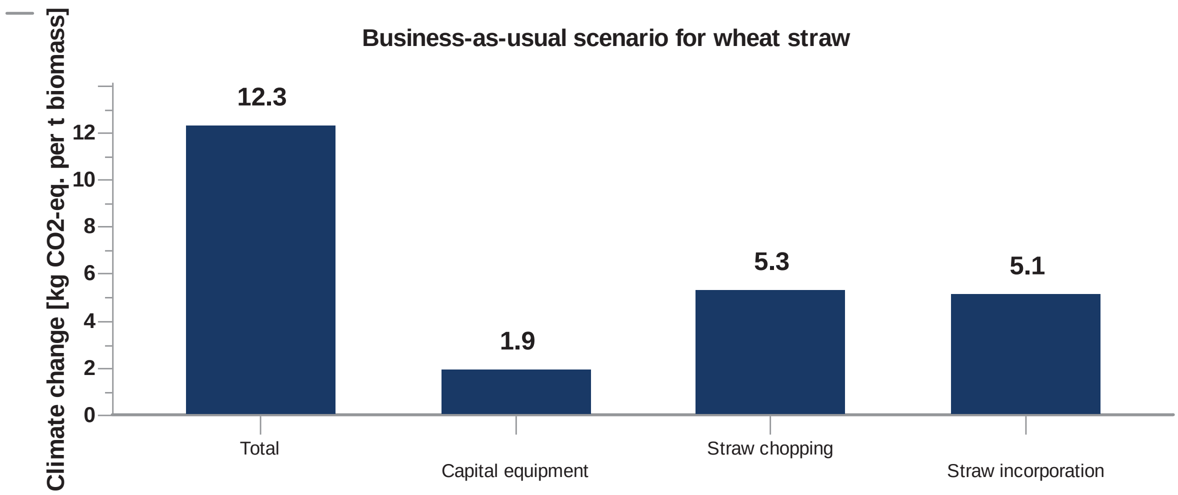

Fig. 38. Breakdown of the climate-change impact per functional unit of the business-as-usual scenario for wheat straw

4.4.3.2. Heat-only scenario (attributional)

The attributional carbon budget of the HO scenario is approximately 39.5 kg CO_2-eq per functional unit (Fig. 39). Biomass handling represents 30.7 kg CO_2-eq per t biomass. This impact is divided in increasing order of magnitude and in kg CO_2-eq per functional unit as follows: 0.3 due to transporting bales within the farm; 1.0 due to unloading bales at the farm; 1.7 due to loading bales into trucks (this process happens twice); 4.0 due to baling straw; 6.9 due to transporting bales to the conversion plant; and 15.1 due to manufacturing plastic sheeting for storing bales at the farm. The sum of embodied GHG emissions in capital equipment (construction and maintenance of machinery and heat-only plant) and plant operation is about 8.7 kg CO_2-eq per t biomass.

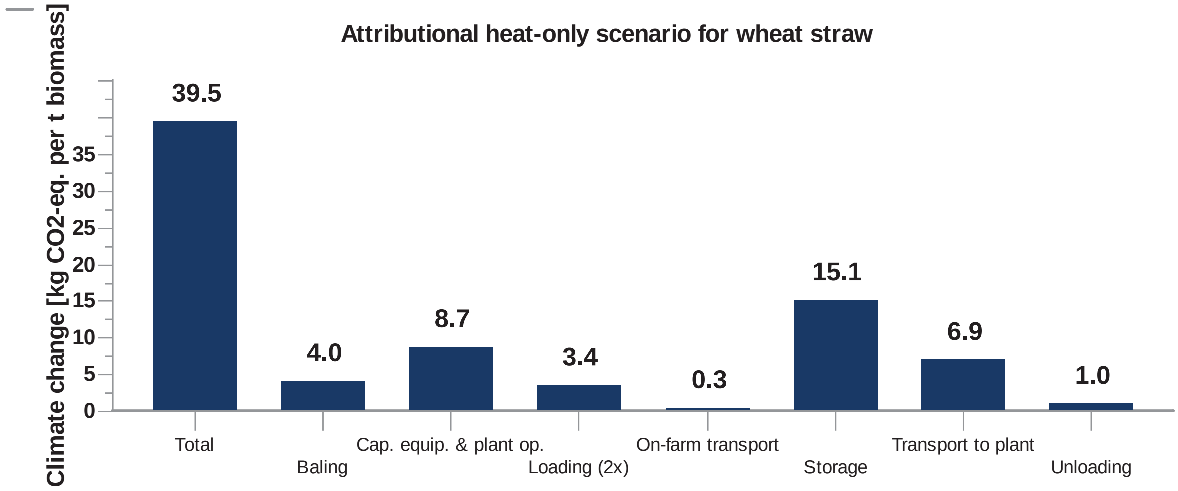

Fig. 39. Breakdown of the climate-change impact per functional unit of the attributional heat-only scenario for wheat straw

4.4.3.3. Heat-only scenario (consequential)

The heat-only scenario has a consequential C balance of -1,064.2 kg CO_2-eq per functional unit (Fig. 40). Biomass handling (30.7 kg CO_2-eq per t biomass) was broken down above (Fig. 39). Capital equipment and plant operation have a combined impact of 8.7 kg CO_2-eq per t biomass. Addition of fertiliser accounts for 43.0 kg CO_2-eq per t biomass. About 69% of this impact is created from the production and transport of commercial fertilisers to NZ port; 29% correspond to CO_2 emissions from soils; and the remaining 2% is due to fertiliser spreading. As a result of project implementation, the BAU scenario (Fig. 38) would be displaced. GHG emission reductions due to the displacement of coal use for heat production represent 1,134.4 kg CO_2-eq per t biomass.

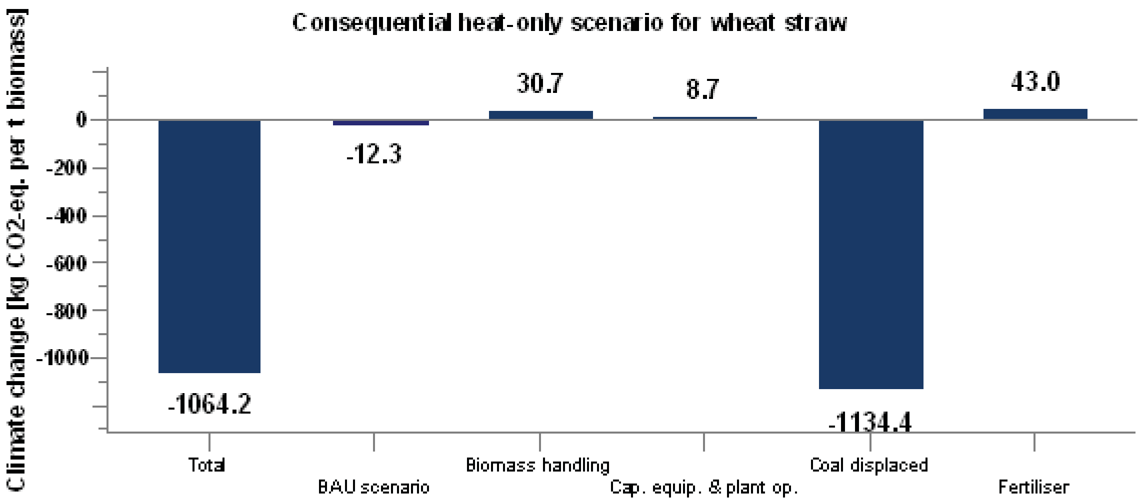

Fig. 40. Breakdown of the climate-change impact per functional unit of the consequential heat-only scenario for wheat straw

4.4.3.4. Combined-heat-and-power scenario (attributional)

The attributional CHP scenario would have a CF of 41.0 kg CO_2-eq per functional unit (Fig. 41). Impacts produced during biomass handling and fertiliser-related processes were described above for the HO scenario. These account for 30.7 and 43.0 kg CO_2-eq per t biomass, respectively. The construction and maintenance of the machinery and the CHP plant would have a combined impact of about 9.7 kg CO_2-eq per t biomass, whereas diesel consumed to operate the plant would release an overall output of about 0.6 kg CO_2-eq per t biomass. Therefore, the total impact for capital equipment and plant operation is about 10.3 kg CO_2-eq per t biomass. The slight difference of 1.5 kg CO_2-eq per t biomass between the attributional carbon balances of the HO and CHP scenarios is due to the increased construction and maintenance of the CHP plant.

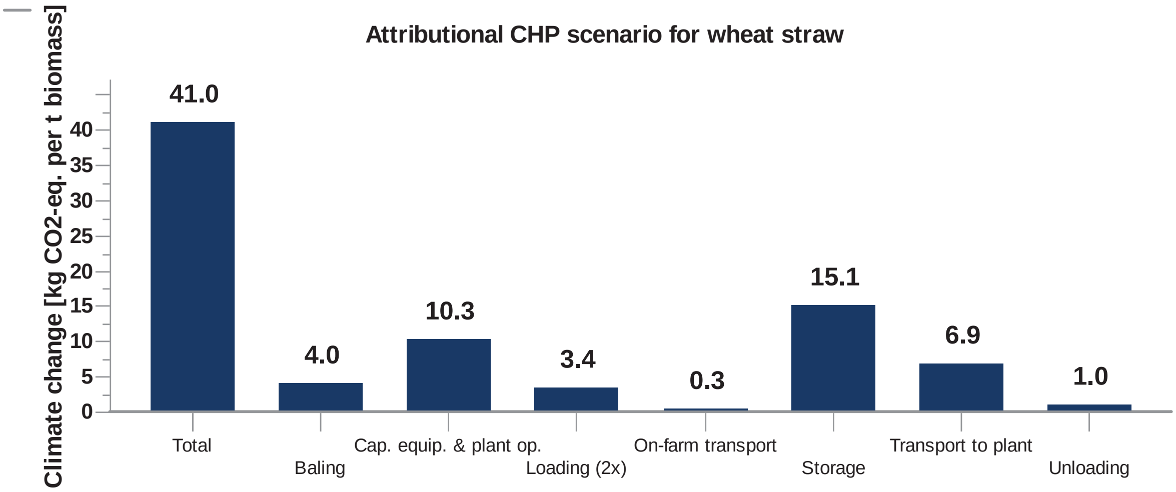

Fig. 41. Breakdown of the climate-change impact per functional unit of the attributional combined-heat-and-power scenario for wheat straw

4.4.3.5. Combined-heat-and-power scenario (consequential)

About 1,608.4 kg CO_2-eq per functional unit would be avoided in the consequential CHP scenario (Fig. 42). This assumes that the BAU scenario, the combustion of coal for production of process heat and the delivery of coal-based electricity from the grid would be displaced at about 12.3, 837.2 and 842.9 kg CO_2-eq per t biomass, respectively.

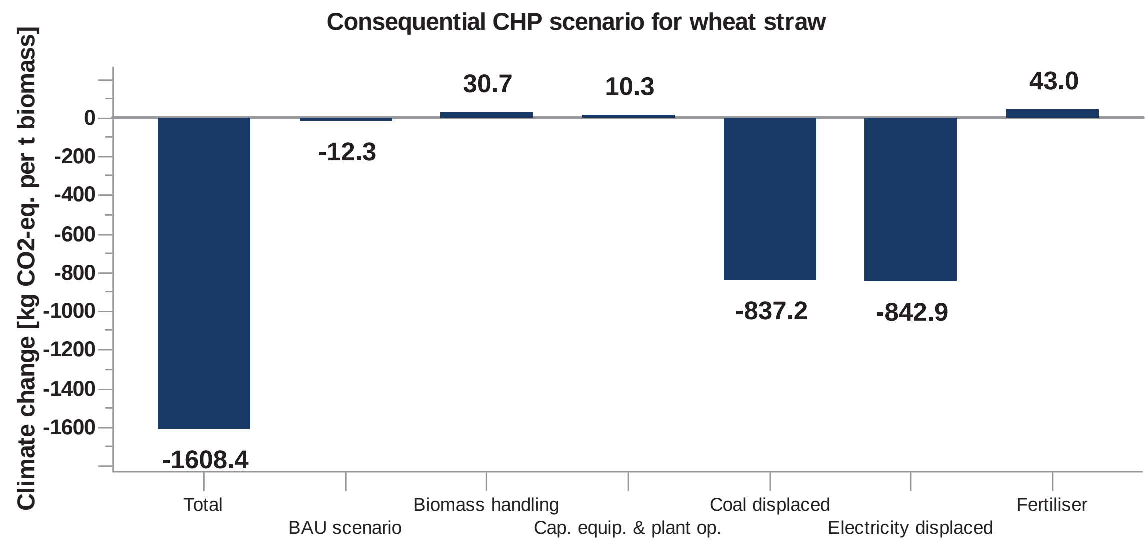

Fig. 42. Breakdown of the climate-change impact per functional unit of the consequential combined-heat-and-power scenario for wheat straw

4.4.3.6. Biochar scenario (attributional)

The attributional carbon balance of the biochar scenario is negative at about -368.7 kg CO_2-eq per functional unit (Fig. 43). Long-term carbon sequestration accounts for -437.0 kg CO_2-eq per functional unit. Biomass handling (30.7 kg CO_2-eq per t biomass) includes processes described above for the attributional HO scenario (Fig. 39) plus unloading, loading of bales into a tub-grinder, and grinding, which have a combined impact of about 16.4 kg CO_2-eq per t biomass. GHGs arising from the manufacture and maintenance of capital equipment (the machinery and the slow pyrolysis plant) account for about 11.7 kg CO_2-eq per t biomass, whereas the electricity required to run the slow pyrolysis plant would generate about 4.0 kg CO_2-eq per t biomass. In total, these two factors have a combined impact of 15.7 kg CO_2-eq per t biomass. At the end of the supply chain, biochar transport and application represent 0.9 and 4.6 kg CO_2-eq per t biomass, respectively.

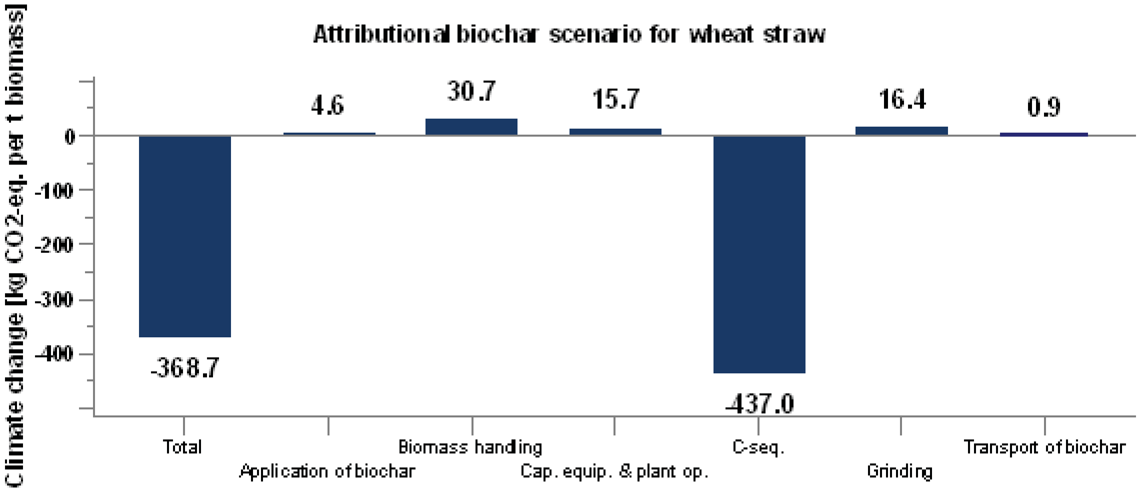

Fig. 43. Breakdown of the climate-change impact per functional unit of the attributional biochar scenario for wheat straw

4.4.3.7. Biochar scenario (consequential)

The consequential carbon budget of the biochar scenario is -792.6 kg CO_2-eq per functional unit (Fig. 44). GHG emissions corresponding to the combustion of coal, which are assumed to be displaced as a consequence of the implementation of the biochar scenario, are in the order of 411.6 kg CO_2-eq per t biomass.

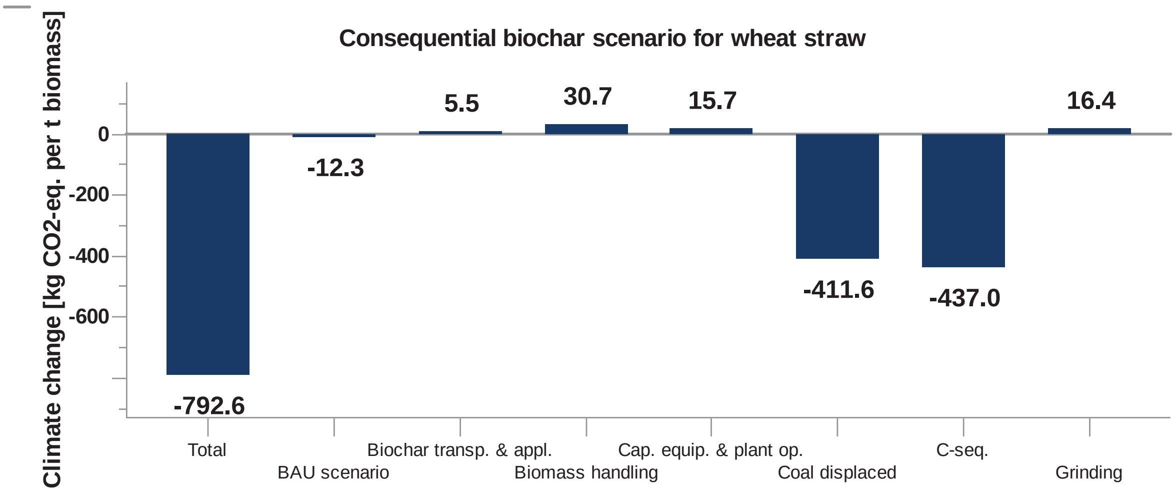

Fig. 44. Breakdown of the climate-change impact per functional unit of the consequential biochar scenario for wheat straw

4.4.3.8. Comparison of scenarios

Four different life cycle pathways for the management of wheat straw are presented according to their attributional and consequential climate-change impacts (Table 35). In order to make scenarios comparable, a consequential approach was followed (Fig. 45). Therefore, the alternative scenarios to BAU were expanded to include the displacement of the combustion of coal for heat production. Furthermore, the delivery of coal-based electricity from the grid was also subtracted in the CHP scenario and fertiliser was added in the HO and CHP scenarios.

Table 35. Attributional and consequential climate-change impacts of alternative management options for wheat straw considering coal as the source of heat and electricity to be displaced (kg CO_2-eq per functional unit)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with coal displacement (scenarios become comparable)
Business-as-usual scenario	12.3	--	--
Heat-only scenario	39.5	-12.3 (avoided BAU system) -1,134.4 (displaced coal-based heat generation) +43.0 (additional fertiliser use)	-1,064.2
Combined-heat-and-power scenario	41.0	-12.3 (avoided BAU system) -837.2 (displaced coal-based heat generation) -842.9 (displaced delivery of coal-based electricity from the grid) +43.0 (additional fertiliser use)	-1,608.4
Biochar scenario	-368.7	-12.3 (avoided BAU system) -411.6 (displaced coal-based heat generation)	-792.6

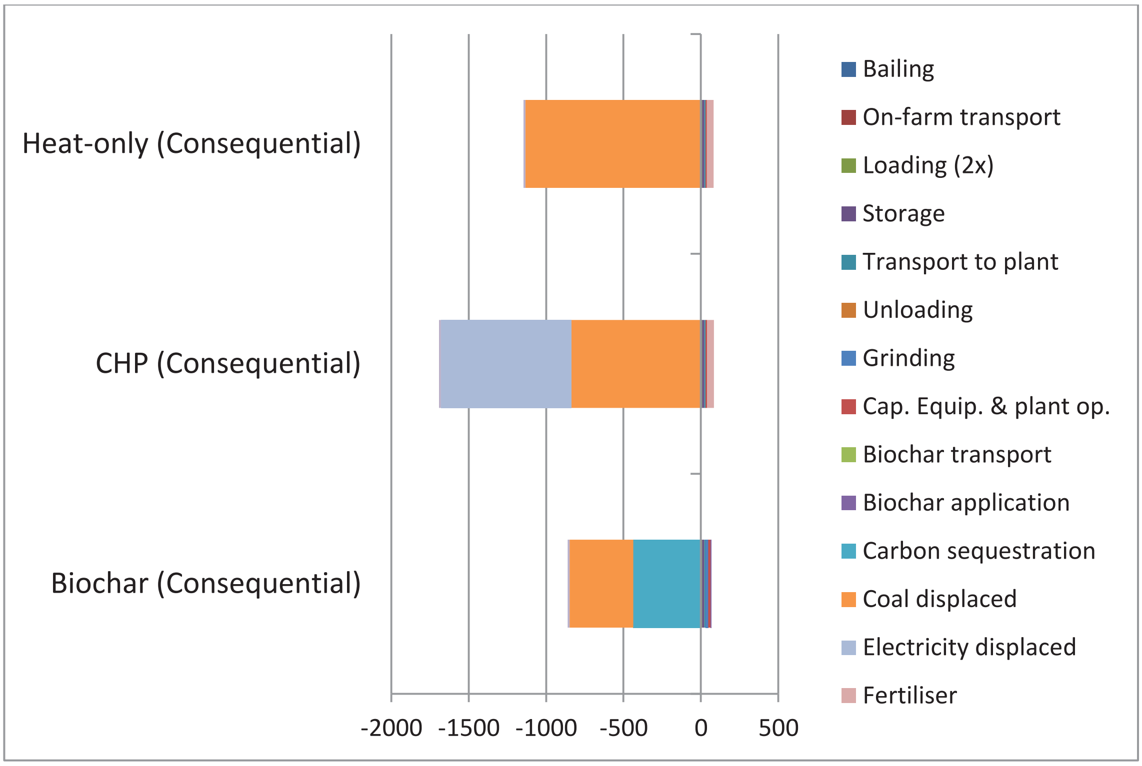

Fig. 45. Comparison of the climate-change impact per functional unit of the alternative scenarios for wheat straw (kg CO_2-eq per t biomass)

Following an attributional approach, the biochar scenario is the only carbon-negative pathway at around -368.7 kg CO_2-eq per t biomass. However, when compared, all of the alternative scenarios to BAU can be argued to mitigate climate change. The highest GHG emission reductions would result from the CHP scenario (-1,608.4 kg CO_2-eq per t biomass) followed by the HO scenario (-1,064.2 kg CO_2-eq per t biomass), and the biochar scenario would come last (-792.6 kg CO_2-eq per t biomass).

4.4.4. Sensitivity Analysis

The data process 'NZ: Thermal energy from natural gas PE' was used in the sensitivity analysis to model respective GHG emission reductions in all scenarios. For the CHP scenario displacing either coal-based or natural gas-based heat generation, the displacement of the average electricity grid mix was also modelled in the sensitivity analysis (Table 36).

Table 36. Attributional and consequential climate-change impacts of alternative management options for wheat straw considering natural gas as the source of heat to be displaced for all scenarios and for the CHP scenario, the displacement of both coal-based and average electricity grid mix was modelled. The average electricity grid mix was also considered for the CHP scenario displacing coal-based heat production (kg CO_2-eq per functional unit)

	Attributional carbon balance (scenarios are not comparable)	Displaced and additional activities when expanding the system for consequential assessment	Consequential carbon balance with natural gas displacement (scenarios become comparable)
Business-as-usual scenario	12.3	--	--
Heat-only scenario	39.5	-12.3 (avoided BAU system) -694.1 (displaced natural gas-based heat generation) +43.0 (additional fertiliser use)	-623.9
CHP scenario displacing natural gas-based heat and coal-based electricity	41.0	-12.3 (avoided BAU system) -512.3 (displaced natural gas-based heat generation) -842.9 (displaced delivery of coal-based electricity from the grid) +43.0 (additional fertiliser use)	-1,283.5
CHP scenario displacing natural gas-based heat and average electricity grid mix	41.0	-12.3 (avoided BAU system) -512.3 (displaced natural gas-based heat generation) -294.8 (displaced delivery of average electricity grid mix) +43.0 (additional fertiliser use)	-735.4
CHP scenario displacing coal-based heat and average electricity grid mix	41.0	-12.3 (avoided BAU system) -837.2 (displaced coal-based heat generation) -294.8 (displaced delivery of average electricity grid mix) +43.0 (additional fertiliser use)	-1,060.3
Biochar scenario	-368.7	-12.3 (avoided BAU system) -251.8 (displaced natural gas-based heat generation)	-632.8

In the HO scenario, the energy content of wheat straw is released to maximise heat production for the displacement of fossil fuels, whereas the CHP and biochar scenarios also entail electricity displacement and carbon sequestration, respectively. Therefore, if natural gas, instead of coal, were to be displaced, the HO scenario would be the most affected in terms of avoided GHGs. In contrast, the biochar scenario, which delivers the least amount of heat energy, suffers the slightest decrease in GHG emission reductions. Moreover, the carbon sequestration potential of the biochar scenario is not influenced by the definition of the fossil fuel to be displaced.

For the biochar scenario, the main focus of this study, several ranges of values were calculated in the sensitivity analysis, and their influence on the final results was considered for both the scenario where coal is displaced and the scenario where natural gas is displaced (Table 37). Moreover, all the variables in the biochar scenario were modified to estimate the most optimistic and the most pessimistic climate-change impact scenarios for the displacement of coal or natural gas (Table 38).

Table 37. Sensitivity analysis for the wheat straw to biochar system considering coal or natural gas combustion for heat production as the displaced activity (- indicates a further reduction, whereas + means a further increase in GHG emissions)

Parameter	Original assumption	Range	Impact of variability on the C balance of the system displacing coal as a heat source	Impact of variability on the C balance of the system displacing natural gas as a heat source
Pyrolysis gas yield (% of dry mass fed into the reactor)	33%	30 to 35%	+3.8 to -1.6%	+2.9 to -1.3%
Bio-oil yield (% of dry mass fed into the reactor)	15%	10 to 30%	+8.0 to -22.3%	+6.1 to -17.1%
Bio-oil boiler efficiency	85%	50 to 90%	+9.8 to -0.9%	+7.5 to -0.7%
Biochar yield (% of dry mass fed into the reactor)	28%	25 to 35%	+5.9 to -13.6%	+7.3 to -17.0%
Biochar losses	2%	0 to 10%	-1.1 to +4.6%	-1.4 to +5.7%
Application of biochar	1.5 l diesel / t biomass	1 to 2 l diesel / t biomass	-0.2 to +0.2%	-0.2 to +0.2%
C content of biochar	71%	60 to 75%	+8.6 to -3.2%	+10.8 to -3.9%

Biochar carbon stability factor for ≥100 years	74%	50 to 80%	+17.9 to -4.4%	+22.4 to -5.5%
Transport distance	44 km (one way)	20 to 80 km (one way)	-0.1 to +0.1%	-0.1 to 1.0%
Additional application of fertilisers as percentage of nutrient content of wheat straw	0%	50 to 100%	+2.7 to +5.4%	+3.4 to +6.8%
Biochar migration factor (% of the solid and dissolved form of biochar that migrates out of the project boundaries within 100 years)	0%	10 to 50%	+5.6 to 27.6%	+6.9 to +34.6%

Table 38. Most pessimistic and most optimistic scenarios for the consequential biochar system of wheat straw considering the displacement of coal or natural gas formerly used to produce heat (see Table 37 for the description of the assumptions made)

	Original scenario	Most pessimistic scenario	Most optimistic scenario
Consequential C balance of the biochar system displacing coal (kg CO_2-eq per functional unit)	-792.6	-204.7	-1,207.3
Consequential C balance of the biochar system displacing natural gas (kg CO_2-eq per functional unit)	-632.8	-125.6	-964.6

From the sensitivity analysis, the 'hot spots' of the biochar system can be identified. The biochar carbon stability factor (BCSF) and the biochar migration factor (BMF) are the most important variables followed by the amount of heat that can be exploited from the burning of co-products. The biochar scenario displacing coal combustion would undergo a relatively severe drop in GHG emission reductions if the amount of heat exploited outside the system boundaries was negatively affected.

For every 6% decrease in the BCSF, about 4.4% or 5.5% less carbon would be sequestered when displacing coal or natural gas, respectively. The BMF, which was originally neglected in this study since it has not been explicitly mentioned in the literature, can, for every 10% variation, affect the carbon balance of the system displacing coal or natural gas by 5.6% or 6.9%, respectively. Yields of co-products are central to the carbon balance of this slow pyrolysis system. The pyrolysis gas yield affects the carbon balance of the system by about 1.6% or 1.3% for every 2% variation, considering the displacement of coal or natural gas, respectively. For every 5% variation in the bio-oil yield, a change of about 8.0% or 6.1% would be observed in the carbon balance of the system displacing coal or natural gas, respectively. Transport and application of biochar produce a very low climate-change impact per t biomass, so the carbon balance of the system is practically not sensitive to these factors. Further research is needed to estimate a possible reduction in the C balance of the system due to the impact of biochar on soil albedo.

4.4.5. Discussion

According to the assumptions made in this study, the HO, CHP and biochar scenarios all reduce GHG emissions compared with the business-as-usual scenario (Table 39).

Table 39. Consequential climate-change mitigation potential of alternative management options for wheat straw (kg CO_2-eq per t biomass)

Management option	Consequential C balance of the system displacing coal as a heat source (kg CO_2-eq per t biomass)	Consequential C balance of the system displacing natural gas as a heat source (kg CO_2-eq per t biomass)
Heat-only scenario	-1,064.2	-623.9
Combined-heat-and-power scenario displacing coal-based electricity	-1,608.4	-1,283.5
Combined-heat-and-power scenario displacing average electricity grid mix	-1,060.3	-735.4
Biochar scenario	-792.6	-632.8

If coal combustion for both heat and electricity production was displaced, then the best use of biomass to mitigate climate change would be described by the CHP scenario. Relative to such a reference, the HO and CHP scenarios offer the potential to reduce about 1,064.2 and 1,608.4 kg CO_2-eq per t biomass, respectively. The biochar scenario could mitigate 792.6 kg CO_2-eq per t biomass, including 437.0 kg CO_2-eq per t biomass due to long-term C sequestration.

If combustion of natural gas, instead of coal, was assumed to be displaced, the CHP scenario would still be the best use of biomass for climate-change mitigation (-1,283.5 kg CO_2-eq per t biomass). Given this reference, the HO and biochar scenarios would offer carbon abatement in the order of 623.9 and 632.8 kg CO_2-eq per t biomass, respectively. Note that the difference among the alternative carbon balances is more important when coal combustion is assumed to be displaced rather than natural gas. Therefore, the definition of the reference scenario is a key variable. It is important to note that the biochar scenario would result in long-term carbon sequestration irrespective of the other activities assumed to be displaced.

Wheat growing in Canterbury has an average CF of 3,060 kg CO_2-eq per ha per year, including straw processing (Barber *et al*., 2011). Therefore, the consequential carbon balance of the biochar system would lead to the compensation of about 110% or 88% of the annual GHG emissions from wheat production when displacing coal or natural gas, respectively. However, it is unlikely that all GHG emission reductions would be claimed by the farm since energy users would also be affected when displacing fossil fuels (see section 4.5.4). Further research is needed to elucidate the effects of biochar produced from the slow pyrolysis of wheat straw at different production parameters on the soil quality levels of Canterbury's wheat-producing fields and on the ecotoxicological risks that are potentially involved when applying biochar into soils.

4.5. Overview of the three case studies and further interpretation of the results

The goals of this study were:

1) to compare future alternative management scenarios for three types of end-of-life biomass (ELB) streams available in NZ; and
2) to elucidate the best option for future treatment of biomass to mitigate climate change in order to support policy decision processes.

Using a life cycle approach, the three types of ELB that were evaluated in case studies were:

- prunings from apple orchards in the Hawke's Bay region (see section 4.2);
- logging residues in the Central North Island region (see section 4.3); and
- wheat straw in the Canterbury region (see section 4.4).

In each of these case studies, the climate-change impact arising from the management of one tonne of fresh biomass was analysed using scenarios (Table 40). Current management practices for the three types of ELB were considered as the business-as-usual (BAU) scenarios. Following a consequential LCA approach, the alternative systems to BAU were expanded to add and/or subtract processes that would be operated and/or displaced for the alternative scenarios to deliver equivalent services.

Table 40. Climate-change impact of alternative biomass management options evaluated in case studies and scenarios (kg CO_2-eq per t biomass)

Management option	Orchard prunings		Logging residues		Wheat straw	
Business-as-usual scenario	39.6		0		12.3	
	Consequential C balance of the system displacing coal as a heat source			Consequential C balance of the system displacing natural gas as a heat source		
	Orchard prunings	Logging residues	Wheat straw	Orchard prunings	Logging residues	Wheat straw
Heat-only scenario	-613.6	-473.2	-1,064	-376.0	-276.5	-623.9
Combined-heat-and-power scenario displacing coal-based electricity grid	Not considered	-511.4	-1,608	Not considered	-410.1	-1,283
Combined-heat-and-power scenario displacing average electricity grid mix	Not considered	-325.8	-1,060	Not considered	-224.5	-735.4
Biochar scenario	-393.0	-321.0	-792.6	-339.7	-271.9	-632.8

The alternative scenarios to BAU involved the collection, transport, and conversion of biomass at a central plant to produce: 1) heat-only (HO), for the three case studies; or 2) combined heat and power (CHP), for the logging residues and wheat straw case studies; or 3) biochar and heat, for the three case studies. Note that biochar is modelled as being produced from slow pyrolysis at 400^{o}C, and transported and applied back into the land where the biomass originated. In the case of heat production, coal and natural gas were modelled as the fossil fuels to be displaced. In the case of power generation, both coal-based and average electricity grid mix were modelled to be displaced as a consequence of project implementation. Refer to sections 4.2.1, 4.3.1 and 4.4.1 for a complete description of the goal and scope definition phase for each case study.

The HO scenario (-276.5 to -1,064.2 kg CO_2-eq per t biomass); the CHP scenario (-410.1 to -1,608.4 kg CO_2-eq per t biomass); and the biochar scenario (-271.9 to -792.6 kg CO_2-eq per t biomass) all contribute to climate-change mitigation relative to a reference scenario in which fossil fuels are assumed to be kept in the ground as a result of project implementation. Note that in the case of processing logging residues, the HO scenario (-473.2 kg CO_2-eq per t biomass) would provide the highest GHG emission reductions if the average electricity grid mix was displaced in the CHP scenario (-325.8 kg CO_2-eq per t biomass) rather than coal-based power generation. For the CHP scenario using wheat straw, this was not the case due to the relatively high efficiency of the CHP straw-fired plant upon which this study was based (Forgie and Andrew, 2008).

The results show that, on a per-tonne-of-biomass basis, the management of wheat straw for energy or biochar production in the Canterbury region offers the highest potential to mitigate climate change (Table 40). This is due to the relatively low moisture content of straw, which results in the production of more biochar and more co-products that can be exploited for heat production and fossil-fuel displacement. Refer to sections 4.2.5, 4.3.5, and 4.4.5 for a complete discussion of the results of each case study. The sections 4.5.1 – 4.5.6 presented below provide further interpretation of the overall results.

4.5.1. Soil-related benefits can be neglected for carbon-accounting purposes

For carbon footprint (CF) studies using the functional unit of 'the management of one tonne of fresh biomass', 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. The rationale behind this argument is not only due to the high inputs of fresh biomass required to apply >10 tons of biochar per ha and achieve relatively little soil-related benefits, but also 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.

4.5.2. Assumptions about displaced activities play a key role in determining the overall C balances of the systems

The consequential climate-change impact of the alternative scenarios was calculated in relation to the BAU scenario continuing to the same point in the future. The consequential approach was followed because modelling the scenarios, as they occur, in a so-called attributional LCA approach without system expansion/substitution would result in different services delivered and because the study aims at supporting a decision. Therefore, the attributional scenarios modelled here could not be compared against each other, but scenarios could be compared in terms of their consequential climate-change mitigation potential, which depends on the assumptions made about the displaced activities. The reference constructed for comparison is also known as the baseline scenario.

All of the alternative scenarios to BAU evaluated in the three case studies offer a potential to mitigate climate change according to the activities assumed to be displaced. If coal formerly combusted to produce heat and electricity was assumed to be displaced then the combustion of biomass to supply heat and power to a processing plant would be the most promising option. The use of marginal versus average data (as illustrated clearly by the CHP scenario) has a significant impact on the results. Note that the C sequestration potential of biochar is not affected by the assumptions made about the displaced activities.

4.5.3. The scale is important

The results of the CF studies are given per functional unit, which is extremely small relative to the country scale. Since the goals of the study are around policy support then one needs to look at the national level. On a regional basis, the prospective carbon abatement for converting logging residues in the Central North Island (CNI) region into energy and/or biochar outperforms the climate-change mitigation potential offered by the other two case studies in their respective regions. This is due to the considerably larger quantities of logging residues that are available in the CNI region, which can potentially be converted into more energy and/or biochar than the orchard prunings and the wheat straw in the respective regions. Therefore, from a policy perspective, it may be preferable to incentivise the recovery of logging residues for energy and/or biochar production since these management options will result in the highest mitigation of national GHG emissions as long as, in the case of exclusively bioenergy purposes, the biomass offset fossil fuel combustion.

4.5.4. Claiming the carbon credits from biochar systems may need negotiation

The influence of the consequential climate-change mitigation potential of the three biochar systems on the CF of the main products (apples, logs and wheat grain) was calculated earlier (see sections 4.2.5, 4.3.5, and 4.4.5). Note that these previous calculations implied that all the potential carbon credits achieved by the biochar systems would be allocated to the land managers that provide the biomass feedstock. However, allocating all GHG project emissions, fossil-fuel GHG emission reductions, and biochar long-term CO_2 removals to the land managers is unlikely as the processing plants using the energy products would also be responsible for some of the GHGs emitted during the supply chain. GHG emission reductions that could be argued to be achieved by displacing fossil-fuel use at a processing plant would most likely be monitored, reported and claimed by the former fossil fuel users rather than the land managers if these were not using fossil fuels in such a way in the reference scenario.

In the case of the HO and CHP systems, it would make sense for the managers of the processing plants to claim all the reductions but also the GHGs emitted during the supply chain. In those cases, the land managers may receive a monetary compensation for the removal of biomass from the land. In contrast, the biochar system may be more appealing to the interest of the land managers in order for them to clearly claim carbon sequestration resulting from the addition of biochar into the land they manage. Thus, the implementation of the biochar system may offer a plausible way to reach a carbon-credit agreement between the managers of the processing plants and the land managers. Life cycle GHG emissions due to project implementation (excluding biochar transport and application) and GHGs avoided due to the displacement of fossil fuels with bio-oil combustion could be attributed to the CF of the processing plants, whereas GHG emissions due to the transport, application and C sequestration of biochar could be accounted for by the land managers. In short, the land managers could clearly claim carbon credits due to biochar-C sequestration but could not claim fossil fuel substitution if they did not replace fossil fuels with renewable energy from pyrolysis.

4.5.5. Biochar long-term CO_2 removals are different from avoided fossil-fuel GHG emissions

From a NZ government perspective, if the policy priority is the treatment of biomass that will result in a reduction in national GHG emissions, the selection of the HO and CHP systems for greater GHG reductions can be misleading. Accelerating the release of all C in the biomass through combustion may reduce national GHG emissions only if two criteria are met: 1) the biomass must be produced sustainably and 2) fossil fuels are displaced, i.e. kept in the ground, as a result of project implementation. Since energy demand is not currently capped and the price of carbon traded is extremely low it remains to be seen if supplying energy from biomass combustion would actually curtail the extraction of fossil fuels. Moreover, energy demand is expected to continue growing with an increasing population.

The long-term removal of CO_2 is absent from the C balances calculated for the HO and CHP scenarios since these are only related to fossil-fuel offsetting. Retarding significantly (>100 years) the release of a fraction of C in the biomass through biochar application into soils would reduce atmospheric CO_2 concentration levels irrespective of any of the other activities assumed to be displaced.

4.5.6. Summary

The environmental impacts of biochar systems need to be evaluated from a life cycle perspective. Numerous methodological factors in Life Cycle Assessment (LCA) need to be clarified since there are no current Product Category Rules (PCRs) for biochar. When comparing different uses of biomass, a system expansion/substitution approach is appropriate for making equivalent the different services delivered by alternative scenarios. This can lead to the 'consequential LCA' modelling but the meaning of this term and its respective requirements still need consensus. Existing life cycle studies of biochar have followed an approach that falls in between the attributional-consequential spectrum.

In this study, the results were presented following both the attributional and the consequential approach. Using attributional modelling, the biochar scenarios for the three case studies result in net carbon sequestration whereas the HO and CHP scenarios result in net GHG emissions. Using consequential modelling, the CHP scenario provides the greatest carbon abatement, provided that coal is kept in the ground as a consequence. Of the three case studies, wheat straw management offers the greatest climate-change mitigation potential per t biomass due to its low moisture content. However, at a national scale, the management of logging residues would deliver the greatest reduction in national GHG emissions due to the large volumes of logging residues available in the Central North Island region. It follows that different modelling approaches and interpretation of the results have implications for the carbon financing of biochar systems, and this is discussed in the following chapter.

CHAPTER 5: SUSTAINABLE BIOCHAR SYSTEMS FOR CARBON FINANCE

Modern biochar systems can be designed to mitigate climate change, promote sustainable development and increase the adaptability of the land to a specific use. However, there are many criticisms of the industry that have grown around climate change mitigation and sustainable development, of which biochar may be a small part. First, these criticisms will be explored briefly in order to inform the ensuing discussion on monetising the sequestration of carbon in biochar systems.

Several criticisms faced by the industry relate to the implementation of large scale biochar systems through carbon markets. Principal among these is the concept of biochar as a 'carbon offset', which does not reduce atmospheric CO_2 concentration levels – at best, it compensates for GHGs emitted elsewhere and at a different time. Note that the production and use of biochar offers potential in areas other than current climate policy, such as soil remediation, crop productivity enhancement, waterways protection, waste management, and renewable energy generation (Lehmann and Joseph, 2009).

Within climate policy, there are very few economic strategies in place to incentivise biogenic carbon sequestration. Most importantly, biochar-carbon sequestration is currently excluded from the carbon markets. Nevertheless, this has not stopped the recent proliferation of over 110 biochar producing companies and organisations, offering a wide range of biochar products (Knight, 2012). However, it should be stressed that, while biochar is a highly-variable innovative concept, its broad diffusion and successful adoption is challenging even if carbon markets were adapted to provide a funding mechanism.

Carbon markets in general are questionably effective in trying to reduce GHG emissions and promote sustainable development (Bachram, 2004; Lohmann, 2006; Gilbertson and Reyes, 2009; Hansen, 2009; Böhm and Dabhi, 2009; Spash, 2010; Bertram and Terry, 2010; Newell and Paterson, 2010; McNish, 2012; Spash and Lo, 2012; Aldy and Stavins, 2012). Whether carbon markets could be improved and further regulated (or not) will be debated over several years. Carbon markets can be considered "*in vivo*" experiments (Callon, 2009) that are currently being introduced in Australia, Brazil, California, China, Mexico, Quebec and South Korea (Kossoy and Guigon, 2012). In the meantime, CO_2 keeps on accumulating in the atmosphere; major polluters make windfall profits from trading hot air; and many financial institutions have closed down their carbon trading desks (Böhm, 2013).

In spite of the inherent uncertainties in carbon markets and their ability to achieve economic, environmental and social integrity (Lohmann, 2010; Whitington, 2012), it is generally agreed that some sort of direct and/or indirect sources of putting a price on carbon are needed to encourage climate-change mitigation. Options include taxation, 'cap-and-then-fund' schemes (such as the Clean Development Fund; see section 2.4.2), and carbon labelling of products. Regardless, in view of the lack of short-term alternatives readily available for carbon finance, the current trend is to direct attention towards carbon trading.

The importance of being aware about the social and ethical implications of how carbon markets are constructed and who is benefitting or negatively affected in the process cannot be over-emphasised (Randalls, 2011; Paterson and Stripple, 2012; Descheneau, 2012; Bond, 2012; Childs, 2012; Böhm *et al*., 2012). The making and expansion of these complex and confusing mechanisms relies on the discourse and enthusiasm of participants for the novelty of carbon market products (e.g. registries, software, risk hedging, standards, rating agencies, allowances, offsets, futures, options, swaps, etc.), which in fact have no use value. According to Descheneau and Paterson (2011), it is "the romance, not the finance which makes carbon markets go round". In view of the many criticisms of carbon trading, proponents tend to react by labelling the critiques as "anti-capitalist ideology" and opponents as "die-hard left-wing ideologues" (Michaelowa, 2011) without actually engaging with their arguments (Böhm and Dabhi, 2011).

The trading of carbon, an intangible commodity, can therefore have an impact on the psyche of groups and individuals up to the point of confusion and distraction from taking more effective measures (Smith, 2007; Nerlich and Koteyko, 2010; Paterson and Stripple, 2010), such as reducing the demand for fossil fuel-based goods and services. Carbon markets have also provoked "strong, if diverse and confused, movements of societal self-defence" all over the world (Lohmann, 2010), especially where local communities have been disadvantaged by carbon offsetting programmes (Böhm and Dabhi, 2009; Lejano *et al*., 2010). It should be noted that, in terms of social development, successful emission reduction projects may exist but opinions tend to be divided on the subject as carbon abatement in developing countries is just one life cycle stage in the supply chain of an offset.

The main aim of this chapter is to focus on the technical aspects that should be considered for biochar systems to be eligible for carbon finance, whether this is channelled through carbon markets or not. Each of these aspects is discussed in the following sections:

- life cycle sustainability issues concerning biochar production and application into soils;
- a number of methodological issues and challenges relating to carbon financing;
- biochar long-term CO_2 removals are compared with other bio-carbon sequestration options;
- public perception issues about the possible inclusion of biochar in carbon markets; and
- the chapter is summarised and a way forward is proposed.

5.1. Sustainability issues

Biochar systems must demonstrate that they are sustainable, which means that the activity associated with sourcing biomass, its handling and conversion to biochar, then transport and soil application must not negatively affect aspects such as emissions to air, land or water, biodiversity, indigenous rights, or land-use change. How biochar systems are to demonstrate sustainability is complex and is mostly still in the planning stage because systemic use of biochar is not widespread. Japan was the first country to publicly recognise and authorise charcoal as a specific material for soil amendment in 1984 (Ogawa and Okimori, 2010). More recently (April 2013), Switzerland became the first European country to conditionally approve the use of certified biochar in agriculture (Schmidt, 2013). Initially, the Swiss certified biochar must be produced from the pyrolysis of untreated wood and its production and use must comply with national regulations. Experience from these two countries should provide valuable lessons for developing sustainability standards for biochar systems.

The International Biochar Initiative (IBI) released version 1.1 of the *"standardized product definition and product testing guidelines for biochar that is used in soil"* (IBI, 2013a). Known as the "*IBI Biochar Standards*", the document aims at assisting biochar producers in assessing the basic physico-chemical properties of biochar products and respective concentrations of toxicants in order to provide certainty to users about the safety and efficacy of the materials to be used as soil amendments. Although the scope of the standards is limited to characterising the biochar products, the IBI places some restrictions on the types of feedstock to be used. For example, to qualify as biochar under these standards, the feedstock may not contain more than 2% of contaminants (on a dry weight basis). Municipal solid waste containing hazardous materials or wastes may not be eligible for the IBI's certification. The standards include recommendations for safe production processes but do not prescribe production and material handling parameters, nor do these provide thresholds or terms for defining the sustainability of the feedstocks or biochar products. The disclaimer states that "further documentation and guidance is necessary to identify appropriate sustainability practices and/or safe and effective production processes" (IBI, 2013a).

The sustainability implications of biochar application into soils heavily depend on the scale and location of the project; type of technology and feedstock employed; the implementing actors; and the drivers for implementation. Similar to the evaluation of the environmental impacts produced along the supply chain, the sustainability of biochar systems should be assessed from a life cycle perspective (Cowie *et al*., 2012a), from the sourcing of the biomass feedstock to the application of biochar into soils and subsequent effects. Indirect consequences (such as land, crop, and fuel displacement; and changes in water and fertiliser use) may also occur as leakage. Identification of hazards to minimise potential risks associated with biochar production and use is desired during the early stage of design, both project-wise and policy-wise (Downie *et al*., 2012). The remainder of this section discusses sustainability using this life cycle approach.

5.1.1. Sourcing of the biomass feedstock

If produced from biomass arising from clearing indigenous forest, biochar would not provide net emission reductions from a life cycle perspective (Lehmann, 2009) and would also pose a risk to biodiversity conservation (Glaser, 2007). In NZ, by and large, sustainable biochar production may be derived from four different sources of biomass:

1. agricultural and tree plantation residues (e.g. cereal straw and corn stover; woody biomass residues from orchards, vineyards, and tree plantations);
2. farm manures (e.g. dairy, piggery, poultry);
3. biomass sent to landfills (e.g. municipal solid waste, wood processing residues, sewage sludge); and
4. purpose-grown crops (e.g. biomass plantations for energy and/or biochar production and respective residues).

Generally, it is believed that end-of-life biomass (ELB) feedstocks (listed above under 1-3) would pose less sustainability risks than purpose-grown crops. This reasoning rests on the assumption that no extra land and no additional activities up to the point of harvest would be required for feedstock production. In terms of size and type of feedstock, a recent agricultural survey that aimed at eliciting stakeholder views on the use of biochar in NZ (Quade, 2010) showed that respondents prefer the idea of a small-scale on-farm system where ELB is sourced on-site rather than the further expansion of exotic plantations. Participants noted that the presence of chlorine and copper contaminants in orchard and vineyard prunings arising from sprayed agri-chemicals would need to be analysed before using respective biochars as soil amendments. Furthermore, feedstock collection incurs expenses and environmental impacts that need to be considered.

Different ELB can deliver alternative services, such as energy generation, roughage production and soil amendment. Moreover, the use of charcoal to replace the coal employed in iron- and steel-making processes has been recommended by Coal Action Network Aotearoa (Fitzsimons, 2013). Therefore, LCA studies of biochar systems require comparative analysis of alternative management pathways for biomass through case studies in order to elucidate the most sustainable use of biomass in each situation.

Biochar has been endorsed by Stavi and Lal (2013) as one (of only two) of the most promising conservation practices that can efficiently deliver carbon sequestration under a wide array of physical and biotic conditions, but the removal of some agricultural residues from the land (e.g. wheat straw and corn stalks) to use as feedstocks has been discouraged. This is because they deliver a considerable number of ecosystem benefits if left on the soil (Lal and Pimentel, 2009). Nevertheless, under present NZ conditions, the withdrawal of about 50% of available residues of this type of biomass has been argued to not compromise soil functions (Hall and Gifford, 2008). Moreover, if biochar produced from these feedstocks is incorporated back into the original land, the potential negative effects of removing the biomass from soils could be counterbalanced (Blanco-Canqui, 2013). If biochar amendment does not compensate for nutrient removal, then the carbon leakage arising from the production and application of additional fertilisers need to be accounted for in the carbon footprint (CF) of the system.

Manures, often used as organic fertilisers, are a by-product of animal farming that cost practically nothing to produce, but collection and storage costs can be significant, especially for extremely wet feedstocks. In NZ, most of the manure is produced by grazing animals and is therefore difficult to collect. In contrast, a portion of the pig, poultry and dairy manure (when already collected at the milking shed), could be sourced for biochar production provided that sustainability guidelines were complied with throughout the life cycle of the system and possible carbon leakage, arising from the diversion of organic matter that would have been added to soil, is taken into account.

The biomass that is sent to landfills could be diverted to pyrolysis plants for the production of biochar for zero or even negative costs and could result in a number of waste management benefits. Since this material may have high moisture content and/or be contaminated, appropriate treatment for biochar production would be required. The NZ standard for composts, soil conditioners and mulches – NZS4454:2005 – although not mandatory, specifies physical, chemical and biological requirements for soil amendment products derived from organic materials, and provides a benchmark for producers to carry out safe, hygienic, efficient and environmentally responsible operations (WasteMINZ, 2009). Similar regulation could be introduced for biochar systems.

Crops and/or residues sourced from purpose-grown plantations to produce biochar require more stringent sustainability assessment than ELB streams since this possibility involves tradeoffs in land use for food, feed, fibre, timber, fertiliser, energy, biodiversity, and recreation. Lehmann (2009) mentioned that dedicated plantations for the exclusive production of biochar are not likely to be economically profitable, whereas Roberts *et al*. (2010) found that accounting for land-use change caused by deliberate growth of switchgrass for feedstock production could render a biochar project carbon-positive. Instead, biochar has been promoted as a by-product that could trigger carbon-negative biofuel production, “possibly first in Africa” (Mathews, 2008a). This prospect has invoked reaction by researchers that have connected large-scale biochar deployment and dispossession, via carbon markets, of land, resources, labour values and livelihoods from African communities, especially where the creation of anthropogenic dark earths originates from indigenous cultures (Leach *et al*., 2012).

Since global land grabbing is happening anyway, a growing body of research has emerged in order to understand the embedded power relations and the impacts of such a land rush on both industrialised and industrialising nations (Polack *et al*., 2013; Wolford *et al*., 2013). The production and use of biochar may be argued to be positioned at the periphery of the land grab, i.e. biochar as a by-product. Hence, current market logic points at building a sustainability certification process for sourcing of the biomass feedstock based on existing voluntary frameworks. For example, the Forest Stewardship Council (FSC) and the Round Table on Sustainable Biomaterials (RSB), which attempt to reach compromise among a number of stakeholders, have been proposed to be adjusted to encompass the cradle-to-grave sustainability of biochar systems (Cowie *et al*., 2012a).

The experience of voluntary certification in assuring the sustainability of forest management would then be useful for the biochar industry (Bloomfield, 2012; Johansson, 2012; Visseren-Hamakers and Pattberg, 2013). Forest certification can be considered as “work in progress” (Teitelbaum and Wyatt, 2013), which has been successful in “raising awareness and disseminating knowledge” about forest sustainability (Rametsteiner and Simula, 2003) while “promoting a longer term, cumulative change” in practice (Dare *et al*., 2011). A similar attitude of striving for continual improvement was recently displayed by the World Wildlife Fund (WWF) in a statement on the review of the principles and criteria of the Roundtable for Sustainable Palm Oil production. WWF (2013) stated that “it is, unfortunately, no longer possible for producers or users of palm oil to ensure that they are acting responsibly” but still deemed the review as a “step in the right direction”.

In NZ, forest certification and forest governance in general need to include the interests of Māori (Gouldin, 2006; Levack, 2006). Rotarangi (2012) found that, under certain leasing conditions, increasing the sense of Māori participation, representation, deliberation, and empowerment in land management decisions was central to success. In the case of indigenous forests, Memon and Wilson (2007) argued that seeking to accommodate interests of multiple stakeholders may be counterproductive and concluded that the state still plays a big role within NZ’s forest industry.

It is therefore through the long-term process of settling differences among stakeholders (including the state, civil society groups and indigenous peoples) that sustainability certification schemes may become legitimised (Paterson, 2009; Partzsch, 2011; Bloomfield, 2012) – notwithstanding possible dropouts during the consensus-building exercise (Johansson, 2012). In regards to the Marine Stewardship Council (MSC), Bush *et al*. (2013) argued that one way to reconcile credibility, accessibility and continuous improvement is to consider degrees of sustainability instead of a strict differentiation between sustainable and non-sustainable practices. It remains to be seen if Biofuelwatch, the most outspoken watchdog group opposing large-scale biochar deployment (see below), will consent to meet with proponents in order to establish labels for biochar systems based on degrees of sustainability (e.g. biochar-plus and biochar-minus).

5.1.2. Handling of the biomass feedstock

Once the biomass feedstock has been collected, biomass losses, environmental impacts, costs, and social opportunities (e.g. employment) arising from handling the feedstock need to be taken into account. The handling process generally involves transport, storage, drying, and comminution of the feedstock. The higher the moisture content the more energy input, cost and GHG emissions result. Biomass that is sent to landfills as municipal solid waste (excluding the plastics components) may require extensive drying and sorting to obtain a high-quality feedstock for biochar production. When the landfill is not properly managed, methane emissions and volume of the material that would have been sent to landfill would be reduced. This opportunity would be less attractive when the waste already provides a service (Ibarrola *et al*., 2012). Manures also need appropriate handling and a number of farms (or co-operatives) would possibly need to work together as a co-operative to supply enough feedstock to a biochar plant.

The scale and location of the pyrolysis plant is therefore important since large-scale centralised systems would require moving large volumes of biomass from different sources. In NZ, many ELB streams are highly distributed in terms of availability, ownership and location (see section 3.4) and it would be logistically challenging to handle large quantities of biomass to run a centralised biochar system. In contrast, small-scale (static or mobile) pyrolysis units for production and use of biochar on-site are appropriate for handling vastly dispersed feedstocks. One exception is logging residues (and, if developed, biomass from new plantations). These can be available in large quantities and could be delivered to a large-scale biochar facility by a relatively small number of logging managers that already have experience in hauling large volumes of wood. However, the incorporation of the resulting biochar into tree plantation soils would also make the use of mobile units attractive for the logging industry. Handling of the feedstock and biochar would then become more practical.

5.1.3. Conversion of the biomass feedstock into biochar

Depending on the situation, some pyrolysis technologies are more appropriate than others to produce biochar (Brown, 2009). Pratt and Moran (2010) found that, by 2030, improved stoves and small-scale kilns for biochar production in developing countries may be more effective, in terms of costs and C abatement, than large-scale (slow and fast) pyrolysis plants in developed countries. Biochar-making stoves can also improve the living conditions of users since these are more efficient and less smoky than traditional biomass stoves.

To increase the sustainability of using small-scale kilns to produce biochar, these should at least flare the methane released during pyrolysis in order to convert it into CO_2, which, on a 100-year timescale, has a 25-times lower global warming potential than methane. Furthermore, there are concerns for occupational health and safety when fixed or mobile pyrolysis units are employed to produce biochar, especially in tree plantations, as fire hazards can be pronounced (Quade, 2010).

In the case of large-scale pyrolysis plants, the co-products (pyrolysis gas and bio-oil) would likely be captured and used to produce heat and/or electricity. In the traditional process of charcoal manufacture, these co-products are not captured which is why the conversion efficiency is around a very low 10%. If the gas and liquid energy carriers from pyrolysis are used on-site or sold and exported, then these could substitute for the use of fossil fuels and further decrease the C balance of the system. However, the handling, storage, refining and combustion of bio-oil needs special care as it is unstable and highly acidic (Czernik and Bridgwater, 2004). In order to run the processes, some consumption of fossil fuels and/or electricity may be required. So, respective GHG emissions must be accounted for together with all the environmental impacts arising from the production of biochar, such as those resulting from the construction of capital equipment.

Large-scale biochar production will need to comply with regulatory and environmental standards to mitigate risks posed by industrial processes (Downie *et al*., 2012). When regulation is already in place, the production of biochar will be subjected to the requirements set by local jurisdictions in terms of planning, consenting, licensing, decommissioning, etc. Conversely, in countries where regulation is absent or not fully enforced, biochar production should face the same challenges as other processing industries.

Depending on feedstock and production parameters, biochar production may lead to the release of acid gases, polycyclic aromatic hydrocarbons (PAHs), heavy metals, polychlorinated biphenyls, dioxins, and dibenzofurans – among other toxic substances

(IBI, 2013a; Downie *et al*., 2012). The potential emissions to the environment and concentrations of toxicants in the biochar need to be measured and controlled in order to set a minimum level of health protection. Air pollutants are already regulated in certain countries including New Zealand (MfE, 2011), so the same regulations will apply to the influence of biochar production on air quality.

In contrast, the incorporation of biochar into soils is relatively new and various analytical methods to assess the potential toxicity of biochars are just starting to emerge. The "*Biochar Standards*" (IBI, 2013a) recommend some testing protocols to evaluate the concentrations of toxicants in biochars, although other methods are being proposed for the same purposes. For example, Hilber *et al*. (2012) developed a simple and robust extraction method (using toluene) to quantify total concentrations of PAHs in biochars. This method, they claimed, could be used specifically to minimise PAH content of biochars and set standards for registration and legalisation of biochar products. Yet, the IBI (2013a) suggests applying a thermal extraction method elaborated by the US Environmental Protection Agency in 1996 to quantify PAHs in soils, sludges and solid wastes. The divergence of methods is also evident for determining the stability of biochar (see below).

The biochar manufacturers and users need to be accountable and so methodologies for accounting and validation, monitoring, reporting, and verification (VMRV) of biochar production should be developed. For this purpose, characterisation of biochar production should provide reliable and conservative estimates of important factors such as product yields, carbon content of products, biochar carbon stability factor and possible soil-climate-crop interactions with biochars produced under specific conditions.

5.1.4. Handling and application of biochar into soils

Biochars are highly reactive. Due to their pyrophoric and chemisorption properties, biochars can spontaneously combust or become hydrophilic thereby affecting their capacity to adsorb water (Antal and Grønli, 2003). Char fines that are closely pressed together and chars with a high content of volatiles are more prone to self ignition than large pieces of lump char. The charcoal business, which follows provisions to store, transport and handle the material, is a valuable reference for the biochar industry. For example, the Regulation of Dangerous Goods in Germany stipulates a storage period, between production and shipment, of four days for lump charcoal and eight days for finely powdered or granulated charcoal (Antal and Grønli, 2003). Downie *et al*. (2012) also recommended establishing generic material safety data sheet (MSDS) information to handle and work safely with biochar products.

The small particle size (dustiness) and density of biochar are important characteristics that require consideration for health and safety during handling and application of biochar into soils. Biochar dust can be easily blown away by the wind and pose pollution and health risks to neighbours. In order to minimise this risk during storage and transport, biochar could be covered with sheets or sprayed with water or liquid manure (Blackwell *et al*., 2009). Biochar produced from rice husk is of particular concern since it can contain crystalline silica and breathing exposure to the product could cause silicosis, scleroderma, lupus, arthritis, tuberculosis, kidney disorders, and cancer (Shackley *et al*., 2012). Hence, the handling of this type of biochar should follow health and safety precautions that go beyond using masks and gloves. Crystalline silica is argued to be usually formed at high temperatures above 550^{o}C (Blackwell *et al*., 2009) or even above 800^{o}C (Shackley *et al*., 2012), so this should not be a major problem for controlled slow pyrolysis technologies. Note that rice is not cultivated in NZ.

Through pelleting or granulation, increasing the packing density of the biochar material should reduce transport costs, losses, and corresponding GHG emissions. It may also facilitate the handling and incorporation of biochar into soils. This is important given that as much as 30% of biochar was cited to be blown away by the wind during application (Hammond *et al*., 2011). Compaction of biochar should then be conducted carefully to avoid increasing risks of auto-ignition. The impact of binders on biochar-soil dynamics and corresponding GHG emissions need to be considered too.

Various methods of biochar application into soils exist for different situations. These include uniform topsoil mixing; injection with manures and slurries; deep-banded application in rows; top-dressing; “ecological delivery” via animal feed; and no-till methods (Blackwell *et al*., 2009; Graves, 2013). However, parameters affecting the application of biochar, such as post-production treatments (e.g. aging and/or inoculation with microorganisms); biochar application rate; frequency of application; particle size; soil depth; and tillage are still unclear. Since erosion of biochar can be significant for the purposes of proving its 100-year permanence in soils (see below), incorporation of biochar closer to the rhizosphere could be preferable over topsoil dressing. However, this may not be practical in some landscapes, such as in tree plantations or in steep terrain. Furthermore, disturbance to the topsoil structure can increase GHG emissions and may represent another trade-off for biochar systems if the biochar needs to be protected from erosion. The use of reduced tillage or no-till methods could resolve this issue. GHGs emitted, but mainly costs and benefits are likely to guide decisions on how to apply biochar into soils.

5.1.5. Effects of biochar application into soils

The effects of biochar application into soils can be divided into basically three aspects: 1) soil remediation; 2) crop productivity; and 3) climate-change related impacts. Through reduced leaching, biochar application into soils could also lead to the protection of waterways.

Soil remediation

A plethora of studies show that, under certain conditions, different biochar-application arrangements can adsorb (and sometimes immobilise, and even help to degrade if degradable) PAHs, hydrocarbons, herbicides, nitrates, pesticides, hormones, heavy metals, and other contaminants (Wang *et al*., 2010; Zhang *et al*., 2010; Uchimiya *et al*., 2010; Cao and Harris, 2010; Sarmah *et al*., 2010; Bushnaf *et al*., 2011; Beesley and Marmiroli, 2011; Park *et al*., 2011; Chen *et al*., 2012; Xu *et al*., 2013; Ventura *et al*., 2013). Thus, different authorities dealing with the control of these substances could incorporate biochar into their regulations. For example, the accumulation of the biotoxic metal cadmium in NZ soils due to the historical application of phosphate fertilisers is of concern (McDowell *et al*., 2013). Biochar application for cadmium immobilization could then be considered in the Tiered Fertiliser Management System for protection of human and animal health, ecological receptors, groundwater, food safety standards, and trade (Cavanagh, 2012).

The circumstances for soil remediation with biochar additions, however, are extremely variable and need further long-term customised field research. For example, Nag *et al*. (2011) found a decreased efficacy of herbicides in biochar-amended soils, which could change the dosing of herbicides and have negative environmental consequences. Quilliam *et al*. (2013) studied the effect of fresh biochar on soil PAH degradation and questioned the role of biochar as a source or as a sink for PAHs in agricultural soils, whereas Fischer and Glaser (2012) implied that biochar and compost could act in synergy in order to immobilise pesticides, heavy metals, and persistent organic pollutants, such as PAHs. Therefore, soil background contamination should be considered along with the toxicant concentrations of biochars, evaluated and perhaps treated at the production stage.

Crop productivity

The use of biochar generally influences the physical structure, chemistry and biology of soils, which in turn can affect the production of crops. Biochar research usually highlights the potential of biochar to improve soil functions, and therefore can cut down the amount of fertilisers and hence the respective GHG emissions (Lehmann and Joseph, 2009). However, the mechanisms that induce these possible benefits remain poorly understood (Atkinson *et al*., 2010).

Short-term laboratory experiments in NZ show that biochar can have negative, neutral or positive effects on one or more of the soil variables under analysis (Free *et al*., 2010; Sivakumaran *et al*., 2010b; Anderson *et al*., 2011; Taghizadeh-Toosi *et al*., 2012a; Taghizadeh-Toosi *et al*., 2012b). A global meta-analysis of the effects of biochar application on crop productivity included 16 short-term (≤2 years) studies that met the imposed criteria (Jeffery *et al*., 2011). Yield or above-ground biomass changes were evaluated. The results ranged from -28% to +39% of the control with an average value of +10%. This suggested that the main mechanisms for yield increase may have been a liming effect; an improved water holding capacity of the soil; and an improved crop nutrient availability. The greatest positive changes were observed in acidic and neutral pH soils, and in soils with a coarse or medium texture.

More recently, another review of the effects of biochar on plant productivity and nutrient cycling identified 371 short-term (≤3 years) experiments and found that, despite soil and climate variability, the average impact of biochar on various ecosystem properties was neutral to positive (Biederman and Harpole, 2013). On average, it was found that biochar application into soils increased aboveground productivity; soil microbial biomass; rhizobia nodulation; plant K tissue concentrations; soil P; soil K; total soil N; and total soil C, compared with control parameters. However, it is uncertain how long these effects will last since biochar additions in a 3 year field trial suggest that the possible benefits of biochar on crop productivity can be transitory (Jones *et al*., 2012; Quilliam *et al*., 2012). If this was confirmed in other situations, the possible impacts of additional future application of biochar would have to be further examined.

To obtain sustainability certification of biochar systems, Verheijen *et al*. (2012) recommended to extend research and development systematically to cover all soil functions and threats to soil at several spatiotemporal scales, and proposed to apply an A to N procedure where "biochar with properties A, B, C (including concentrations of contaminants), which makes it appropriate for ecotopes with properties D, E, F to grow crop types G and H, but not crop I, at biochar application rates of J (Mg ha^{-1} per year) every K years, to L (Mg ha-1 per year) every M years, up to a maximum biochar loading capacity of N (g kg-1)." If this procedure is pursued, then accounting and VMRV methods should be developed accordingly. To achieve such a programme, considerable coordination and long-term planning and funding may be required.

Climate-change related impacts

Short-term laboratory experiments simulating biochar application into soils have found that biochar may reduce, increase or do nothing to N_2O, CH_4, and CO_2 emissions from soils (Spokas and Reicosky, 2009; Luo *et al*., 2011; Zheng *et al*., 2012; Wang *et al*., 2012b; Yu *et al*., 2013) and that the effects can change over time due to weathering (Spokas, 2013). The longest biochar field study found in the peer-reviewed literature indicated that, after a 10 year observation, over 20% of mass and humic C was lost in a charcoal amended organic soil (Wardle *et al*., 2008a). However, the reasons for this happening are not clear (Lehmann and Sohi, 2008; Wardle *et al*., 2008b; IBI, 2009).

In NZ, there is a special focus on suppression of N_2O soil emissions from grasslands. Contrasting results from short-term laboratory experiments (Clough *et al*., 2010; Taghizadeh-Toosi *et al*., 2011; Taghizadeh-Toosi *et al*., 2012b) point towards undertaking long-term field investigations. More recently, efforts have been made to understand how various biochar application arrangements affect N cycling and thus N_2O emissions (Clough *et al*., 2013; Cayuela *et al*., 2013). Further research is required.

Even when incorporated, the large-scale application of biochar into soils can reduce albedo (Genesio *et al*., 2012; Meyer *et al*., 2012) and therefore aggravate climate change. The inclusion of the effects of biomass systems on albedo in LCA has only recently gained importance (Schwaiger and Bird, 2010; Muñoz *et al*., 2010; Bright *et al*., 2012; Cherubini *et al*., 2012). Meyer *et al*. (2012) normalised albedo forcing impacts to CO_2-eq emissions, reductions and removals of biochar systems, and estimated that a concentration of biochar in soils of 30-32 t/ha – highly unlikely to be applied at once under real conditions – decreased the C balance of the system by 13-22% over a 100 year timeframe.

A possibility that has not been explored yet in the literature is the positive feedback that a reduction of surface albedo may have on local rainfall. Based on Laval (1986) and Fuller and Ottke (2002), Laine (2012) suggested using fossil hydrocarbon coke as "agrichar" to decrease surface albedo and increase rainfall above large deserted areas such as in Africa. The author speculated that using coke largely available from the production of heavy crude oil in Venezuela and tar sands in Canada could substitute biochar application in places where there is a dearth of biomass.

Since the impact of biochar on the albedo of the soil surface may be highly irregular (e.g. due to intermittent vegetation cover), it seems that the lifetime of biochar systems should assume a 100 year cycle. This proposition is important since over a 20-year timescale the potential climate-change impacts arising from changes in fertiliser use, soil GHG emissions, and soil organic C stocks are small per tonne of biomass collected, and could therefore be neglected in LCA studies. This is because a high biochar application rate (≥10 ton of biochar per ha) is usually considered for achieving relatively little benefits (see *the biochar scenario* in section 4.2.2).

Long-term carbon sequestration is the most important climate-change related impact of biochar systems. The characterisation of biochar products aims at classifying biochars in terms of their stability (IBI, 2013a). When mixing feedstocks for producing biochars with higher agronomic potential (Ro *et al*., 2010), it is unknown how blending would affect the stability of the biochar products. To realise the carbon value of biochar, project developers must prove that a certain fraction of the carbon in biochar is retained in soils for at least 100 years. Research into the cycling of char specifically incorporated into soils as biochar is recent so the cycling of black carbon, which consists of a continuum of carbonaceous materials that range from slightly charred biomass to charcoal to soot (Glaser, 2007; Zimmerman, 2010), may provide some insights for biochar systems. Although black carbon found in soils originates from the incomplete combustion of biomass and fossil fuels, and from coal dust (Rodionov *et al*., 2010), biochar may follow a similar pathway.

Black carbon produced during vegetation fires has been part of the global carbon cycle over geological time scales but has not been well understood (Kuhlbusch and Crutzen, 1995; Zimmerman, 2010; Rodionov *et al*., 2010). Natural cycling of black carbon has been linked to what has been called the "missing sink" of carbon (1.4 - 1.6 Gt C per year) in global cycling models (Shrestha *et al*., 2010). Estimates suggest that ~5-40% of the organic carbon found in soils and sediments may be in black carbon form (Zimmerman, 2010). In Terra Preta (TP) soils, char has been found to account for ~25% up to 88% of total organic carbon (Solomon *et al*., 2007; Mao *et al*., 2012) suggesting that char is highly recalcitrant in soils with turnover times ranging from hundreds to thousands of years. In a recent meta-analysis, however, Singh *et al.* (2012) found an average turnover time of 88 years for black carbon in soils and warned against using charcoal as a C sequestration technique to offset fossil fuel emissions. Further research is needed to determine the turnover rates of different biochars produced and applied to soil under various conditions.

Black carbon can be redistributed globally through the atmosphere, rivers and oceans, and it has been recently found that it can be dissolved in water as well (Masiello and

Louchouarn, 2013). Jaffé *et al*. (2013) quantified the average annual global flux of soluble charcoal from land to oceans as >10% of the global riverine flux of dissolved organic carbon, and mentioned that biochar application to soils may enhance the translocation and export of dissolved black carbon into marine systems and have negative environmental consequences on microbial loop dynamics and aquatic food webs. Black carbon in solid form can also move horizontally and vertically through the landscape. Biochar losses, possibly through surface runoff, have been found to be up to 53% of the total biochar applied to soils at depths of 0-0.15m and with a slope of ≤2% (Major *et al*., 2010). Therefore, biochar application into flat land may delay more effectively the movement of black carbon out of the project boundary than applying biochar into steep soils. This is a limiting factor for biochar application in New Zealand since over two-thirds of the country is on hilly and mountainous terrain. However, while the soil benefits would be lost if the biochar is eroded, the carbon may be further stabilised as the biochar is deposited in lakes or oceans where there is lack of oxygen.

Fine char particles seem to decompose more rapidly in soils than coarse fractions of char (Rumpel *et al*., 2007; Zimmerman, 2010; McBeath *et al*., 2013), whereas large char particles are believed to be more prone to erosion (Foereid *et al*., 2011). Physical protection of biochar, for example with mulch on top of it, may help to lower decomposition and migration rates. The analysis above shows that large-scale application of biochar into soils requires evaluation of the fate of both the solid and dissolved states of carbon in biochar over a 100 year time horizon.

While biochar is purported to be a ‘sustainable’ tool to increase food production for the growing world population, the discussion above shows that it is clear that biochar is not always sustainable or advantageous. What is also clear is that more research is needed, including social and ethical implications since population growth will invariably put more pressure on climate change and resources in general.

5.2. Methodological issues

There are currently no approved official GHG accounting methodologies for biochar projects. An attempt to include biochar in carbon markets was the development of the

“*Biochar Protocol*” (Driver and Gaunt, 2010), which was co-funded by ConocoPhillips Canada, the largest investor in the extraction of tar sands. The “*Biochar Protocol*” was relevant to the North American context and specifically targeted the Verified Carbon Standard (VCS) and the Alberta Offset System (AOS) in that Canadian Province. A previous unsuccessful effort in seeking approval of a generalised biochar methodology under the VCS was made by Carbon Gold (2009).

The “*Biochar Protocol*” has now evolved into the “*Biochar Carbon Offset Protocol*” with new funding partners excluding ConocoPhillips. This exclusion is probably due to avoid perception of a link between tar sands and biochar (Biofuelwatch, 2011). The “*Biochar Carbon Offset Protocol*” was recently submitted for validation to the American Carbon Registry, a voluntary carbon offsetting program (IBI, 2013b). Due to the context-specific features of biochar systems, a significant number of biochar methodologies could be derived from the general set of methods proposed.

5.2.1. Carbon accounting

In comparison with a reference scenario, the implementation of biochar systems can reduce GHG emissions in a number of ways (Gaunt and Cowie, 2009; Roberts *et al*., 2010; Woolf *et al*., 2010; Hammond *et al*., 2011; Ibarrola *et al*., 2012). Most importantly, biochar systems offer the potential to remove atmospheric CO_2 and sequester C in soils on a long-term basis. However, the biochar carbon stability factor (BCSF), possibly the most important C accounting parameter of biochar systems, varies with type of feedstock, production parameters and post-production conditions, such as weathering.

Numerous analytical methods are being studied in order to predict the stability of biochars in soils for at least 100 years (Spokas, 2010; Calvelo Pereira *et al*., 2011; Harvey *et al*., 2012; Crombie *et al*., 2012; Cross and Sohi, 2013). Approaches for estimating climate change mitigation of biochar systems should be simple, reliable and inexpensive. The IBI (2013a) recommended testing the basic characteristics of biochars, which include their elemental composition, and suggested using the molar ratio of hydrogen (H) to organic carbon (C_{org}) as an indicator of biochar carbon stability. Lower values of $H:C_{org}$ ratio (0.1-0.7) are correlated with greater carbon stability as indicated by incubation studies. Nonetheless, further research is needed to start assigning BCSFs to different biochars produced and applied under a diverse matrix of conditions.

As explained above, biochars can move out of the soil profile in solid and dissolved form. Therefore, a biochar migration factor (BMF), which has not been explicitly mentioned in the literature to date, should be investigated in order to estimate the presence of biochar in the soils over a century. If most of the biochar was found to migrate out of the project boundaries within the 100-years period, then this would negate the so-called 'permanence' of soil carbon sequestration. Given that long-term ocean carbon sequestration is yet to be explored, biochar could in the meantime become a technology offering temporary C sequestration similarly to reforestation but its mitigation value would be less attractive.

Because of this uncertainty in whether or not biochar stays within the project boundary, carbon accounting methodologies for biochar systems need to follow a life cycle approach and using LCA methodology is appropriate for this purpose. The most important methodological choices and assumptions in LCA of biochar systems are: definition of the goal, scope and decision-context of the study; definition of the functional unit; recognition of multiple functions; selection of system boundaries and allocation approach; choice of impact categories; inclusion of indirect consequences; and definition of the reference scenario. Each of these variables was explained in detail in section 2.3.5. The definition of the reference scenario is further elaborated below from a carbon market perspective.

Definition of the reference scenario from a carbon market perspective

The definition of the reference (baseline) scenario is highlighted as a very important factor in carbon accounting of biochar systems (see section 2.3.5). In offsetting mechanisms this step refers to the highly controversial and abstract concept of 'additionality' (Trexler *et al*., 2006), that is, project developers need to define a reference scenario and show that this will continue into the future provided that biochar systems were not subsidised with C revenues. Since carbon offsetting mechanisms still seem to be a political option, developers must prove the additionality of a project by demonstrating that a biochar system would not be implemented if carbon finance was not granted.

Experience with carbon markets shows that the definition of the reference scenario relies, in many occasions, on the ingenuity of project developers in coming up with a persuasive story. In order to explain the subjectivity involved in the definition of the reference scenario required to prove the additionality of a project, a quote from the carbon markets literature is illustrative: "If you are a good storyteller you get your project approved. If you are not a good storyteller you don't get your project through" (Schneider, 2007 cited in Gilbertson and Reyes, 2009). Further, paradoxical stories are common, to attract investment on the one hand, while aiming at complying with additionality requirements on the other. For example, Lohmann (2009b) recalled a remark made by James Cameron, a carbon broker at Climate Change Capital, who noted that many carbon project proponents "tell their financial backers that the projects are going to make lots of money", while they claim to officials of the Clean Development Mechanism (CDM) "that they wouldn't be financially viable" without C finance.

Due to the economic value of the combination of services provided (McCarl *et al*., 2009), a biochar system may be non-additional in some cases (Baum and Weitner, 2006; Bruges, 2009; Sohi *et al*., 2009). Different authors, however, make the case that carbon funding may be needed in various other situations (Roberts *et al*., 2010; Pratt and Moran, 2010; Galinato *et al*., 2011). So, in view of the determination of biochar proponents in targeting carbon offsetting mechanisms despite the current knowledge and methodological pitfalls (Maraseni *et al*., 2010), it is anticipated that 'good stories' may be approved first. The dissemination of small biochar-making stoves in Africa, for example, may be one way forward for biochar carbon offsets (Whitman and Lehmann, 2009).

In the situations where the level of energy supplied by the biochar system turns out to be higher than the amount of energy formerly consumed in economically poor areas such as in Africa, an approach has been developed in carbon offsetting mechanisms to claim more hypothetical GHG emission reductions than actually achieved. The concept of "suppressed demand" (Winkler and Thorne, 2002; Gavaldão *et al*., 2012) consists of defining a reference scenario that theoretically delivers equivalent amounts of services than the ones supplied by the carbon project. The result of boosting the reference scenario is that more carbon credits, i.e. more money, can be claimed. This tweak in C accounting does not necessarily result in GHG emission reductions but may promote investment in less industrialised countries.

Sectors covered by emissions trading schemes do not have to prove the additionality of their domestic projects because these are already considered additional to business as usual. For biochar systems to obtain C finance under the New Zealand's emissions trading scheme (ETS), first, GHG accounting methodologies for biochar systems need to be developed. Second, the agriculture sector has to enter the NZ ETS and carbon sequestration in the form of biochar is included in soil carbon accounting. Alternatively, the stabilisation of C in biochar may be credited when the feedstock is pyrolysed and it is demonstrated that the biochar will not be combusted. Furthermore, if the point of obligation for agriculture was set at the individual farm level, rather than at the processor level (Adams and Turner, 2012), soil carbon and biochar carbon sequestration could be traced and reported by individual farms leading the way in biochar application.

If these criteria were met in the NZ ETS, then, the definition of the reference scenario would not pose a major problem since the change in GHG emissions arising from undertaking a project should be reflected within the same sector or within the sector where the rate of GHG emissions was affected. For example, if a farm were to produce biochar and to export the bio-oil co-product to a processing plant, then the possible changes in fertiliser use due to on-farm biochar application should be observed and reported by the farm (or by the importer or manufacturer of fertilisers if the point of obligation is placed at the processor level), and the possible changes in the use of energy displaced (e.g. coal, natural gas, electricity) due to bio-oil combustion should be observed and reported by the processing plant. In the same way, the waste sector should observe fluctuations in the rate of GHGs emitted if the ELB managed is diverted to pyrolysis. International carbon leakage, however, may arise if the biomass feedstocks or the biochars are imported into or exported out of NZ.

5.2.2. Validation, monitoring, reporting and verification (VMRV)

VMRV methods for biochar systems are presently lacking. Monitoring and reporting activities are usually carried out by project developers, whereas validation and verification must be performed independently by third-party entities. Processes associated with the machinery used in the life cycle supply chain of biochar products, from sourcing of the biomass feedstock to the application of biochar into soils should be straightforward to include in VMRV methods (Gaunt and Cowie, 2009). VMRV issues concerning alterations in fertiliser and energy use could also be readily addressed. From an LCA perspective, suppression of GHG emissions from soils is not a hotspot in the life cycle of biochar systems, so these may be initially neglected for C accounting purposes. This is conservative.

The main challenge is to develop and apply VMRV methods to the soil carbon sequestration in the form of biochar over a century. This issue points at long-term monitoring, reporting and verification of changes in soil organic carbon (SOC) stocks resulting from biochar application. Therefore, SOC should be estimated for the site before biochar is applied in order to set a baseline value, which should then be validated. Due to its spatial and temporal variability, accounting of SOC needs a considerable soil sampling work over spatiotemporal scales (Sanderman and Baldock, 2010; Chappell and Viscarra Rossel, 2013), and also requires dealing with uncertainties about soil erosion (Page *et al*., 2004; Van Oost *et al*., 2012; Webb *et al*., 2012; Sanderman and Chappell, 2013).

Due to the expense of sampling at sufficient intensity to detect acceptable levels of minimum change in SOC stocks on a yearly basis, a period interval of 10 years has been proposed for monitoring SOC in the European Union (Saby *et al*., 2008). NZ is currently estimating the SOC pools by a separate and independent system within the Land Use and Carbon Accounting System (LUCAS) managed by the Ministry for the Environment (Beets *et al*., 2011). SOC monitoring is the domain of soil scientists who should collaborate with LCA practitioners and support project developers of biochar systems.

Validation and verification of projects in carbon markets has been done by auditors that are appointed and paid by project developers themselves. Furthermore, the auditors may not count with the expertise and/or time required for the job and may therefore rely on the information provided by developers. These are some causes of conflicts of interest, corruption and fraud in carbon markets (Lund, 2010; Cloatre and Wright, 2012; Hickmann, 2013). Therefore, in order to increase the integrity and credibility of these activities, the auditors should be selected and paid by an independent body that administers the program rather than by the developers of biochar projects. Auditors of biochar systems would also be more reliable if they were versed in biochar research.

In summary, the above discussion on the methodology for the inclusion of biochar into carbon markets illustrates that it is complex, principally, because many factors contribute to uncertainty about the stability of biochar over long time-frames. To navigate through this complexity, an LCA approach is needed so that biomass systems are comparable to the background systems of not doing biochar, or to other uses for the biomass, and which include complementary services. The issues are many and extend beyond accounting for the temporal mass and energy flows; to the problems of measurement (e.g. of soil carbon over large acreages); to business conflicts, such as additionality (which requires the project to prove itself unprofitable without carbon finance while on the other hand selling its profitability to attract investors); and to the necessity to have independent third-party auditors to validate long-term claims of sequestration which, if not independent, exposes the validation process to corruption. Furthermore, the pathway to biochar obtaining carbon finance has significant hurdles; in NZ, this requires agriculture to be included in the NZ ETS and soil carbon and biochar to be permitted as a form of amending the soil carbon pool.

Since the performance of biochar systems under real conditions is largely unknown, transaction costs and risks related to the cycle of a carbon project, from the initial stakeholder consultation to the issuance of credits, are likely to be high enough that they hinder the delivery of expected carbon credits (Cormier and Bellasen, 2013). A conservative approach for C accounting of biochar systems may reduce risks encountered at the validation, monitoring, reporting and verification stages. Moreover, buffer funds, such as those created for other highly uncertain biomass-based carbon sequestration technologies (see sections 5.3.1 and 5.3.4), may help in mitigating risks associated with biochar systems.

5.3. Options for bio-carbon sequestration

The annual increase of atmospheric CO_2 has more than doubled from less than 1 part per million (ppm) per year in the early 1960s to about 2 ppm per year in the last decade (Hansen *et al*., 2013). Daily mean concentration of atmospheric CO_2 measured at Mauna Loa Observatory surpassed 400 ppm in May 2013 (Williamson, 2013a). From one climate science perspective, “if humanity wishes to preserve a planet similar to that on which civilization developed and to which life on Earth is adapted”, atmospheric CO_2 must be shrunk to ≤350 ppm (Hansen *et al*., 2008). Therefore, burning biomass for energy to substitute for the use of fossil fuels, although potentially beneficial, is not enough alone to bring atmospheric CO_2 levels down. The deployment of negative emission technologies (NETs) is indispensable. Even most of the model scenarios that aim at reaching the political target of 450 ppm include the use of NETs, such as capturing and storing CO_2 from bioenergy production (Kriegler *et al*., 2013; Davis *et al*., 2013).

McGlashan *et al.* (2012) evaluated five types of NETs (including biochar) based on their potential to deliver a 0.1 ppm CO_2 reduction per annum and found that the degree of scale-up needed for NETs to have a substantial impact on atmospheric CO_2 is probably unrealistic in <20 years. Notwithstanding current technical challenges, future incentives for the dissemination of NETs should treat and value atmospheric CO_2 removals different than GHG emission reductions since these two approaches to climate-change mitigation are not equivalent. Mathews (2008b), for instance, argued that the creation of carbon credit multipliers could prioritise carbon sequestration.

A number of NETs have been assessed and compared elsewhere (Workman *et al*., 2011; McGlashan *et al*., 2012; McLaren, 2012). Although some NETs may compete with or complement each other, the focus of this section is on the main technical differences and similarities in C accounting and crediting between biochar and other techniques for bio-carbon sequestration. These include the enhanced version of reduced emissions from deforestation and forest degradation (REDD+); afforestation and reforestation (A/R); harvested wood products (HWP); bioenergy with carbon capture and storage (BECCS); anaerobic storage of forestry and agricultural residues; and soil carbon management including biochar application.

5.3.1. Reduced emissions from deforestation and forest degradation

Standing natural forests have already removed CO_2 from the atmosphere and currently fix carbon in their biomass in a balanced form with dead trees replaced by new growth. Therefore, avoiding deforestation keeps the forest C stocks locked up but does not result in significant amounts of additional CO_2 removals. Since 2007, however, initiatives that promote reduced emissions from deforestation and forest degradation (REDD) have incorporated conservation practices; sustainable management of forests; and enhancement of forest carbon stocks in developing countries, into a framework known as REDD+ (Thompson *et al*., 2011). Further inclusion of other land-use changes, such as agriculture, into this political structure is articulated as REDD++ (Stephan, 2012).

The EU ETS and the CDM, the world's largest compliance carbon markets, do not currently allow the trading of REDD credits. According to Stephan (2012), this exclusion over a decade is mainly due to on-going repetitive discourses that stress two issues:

1) responsibility and burden sharing, since avoiding deforestation in developing countries may generate large amounts of relatively cheap credits and therefore could allow polluters to pay a trifling sum to meet GHG targets without taking effective domestic actions, while postponing innovation and further promoting carbon lock-in; and
2) large technical uncertainties involved in the accounting, validation, monitoring, reporting and verification of REDD credits required to realise the commodification of avoiding deforestation.

The discourse has recently shifted sharply and parties to the UNFCCC are now deliberating on whether REDD+ should be included in a new protocol that is expected to enter into force in 2020 (Fosci, 2013). However, performance-based payments, as broadly agreed for REDD projects, have not been yet determined to be channelled through a fund or thorough the incorporation of REDD into the carbon markets. Furthermore, addressing the risks of non-permanence and leakage through baseline definition are major requirements for “the carbonification of forests” (Stephan, 2012). To resolve the issue of non-permanence in afforestation and reforestation (A/R) projects, a separate category that treated A/R credits as temporary was created in the CDM but this has failed to attract investments in such projects.

In contrast, the voluntary carbon markets, which represented less than 0.1% share of the global carbon markets in 2011, have been very active in transacting REDD credits. Despite dropping 59% by volume between 2010 and 2011, voluntary REDD credits yielded the highest value of any project category (Peters-Stanley and Hamilton, 2012). This decrease in volume was attributed to the availability of cheap credits and political and technical issues. In order to minimise risks to safeguard the delivery of a well-calculated amount of REDD credits and appease critics, some voluntary standards (e.g. VCS, Plan Vivo, CarbonFix) have exploited a series of stratagems employed by insurance companies. The creation of a buffer system in particular allows project developers to deposit a certain share of credits into a fund, which could be used as a back-up if the project failed to achieve expectations or was destroyed by fire, pests or storms.

The buffer approach performed in voluntary carbon markets then leapfrogs the debate on the non-permanence aspects of carbon sequestration in forests and clears the way for making REDD credits tradable with fossil fuel-derived GHG emissions. It is uncertain if this strategy will be mimicked by the UNFCCC. Since qualification, commensuration and legitimisation of REDD are complex and obscure processes (Stephan, 2012), the commodification of avoiding deforestation is prone to fraud. Therefore, Gupta *et al*. (2012) proposed to hold to account those engaged in carbon accounting of REDD+ projects by scrutinising “who counts, how and for whom”, and with what consequences. In the future, carbon accountability of C accountants may be pursued in other carbon project categories as well.

5.3.2. *Afforestation and reforestation (A/R)*

Carbon credits generated by A/R projects are excluded from the EU ETS, whereas these could only contribute up to 1% of the total CDM portfolio of Annex I countries (Nijnik and Halder, 2013). A/R is barred from the EU ETS because of the same reasons that excluded REDD projects (see above). In the CDM, the possible non-permanence of A/R credits due to fires, pests, storms or market forces, is addressed by issuing temporary credits. These credits are not compatible with other types of carbon commodities and have to be renewed or replaced with non-forestry credits once these expire.

Although forestry credits are easier to communicate than other types of credits, their temporality makes it difficult to find credit buyers in the compliance markets. Moreover, the carbon sink potential offered by A/R projects is believed to be often diminished as the trees mature. The trading of A/R credits in the voluntary carbon markets, however, is useful for companies that try to ‘green’ their image. Due to common knowledge, consumers easily understand that trees provide co-benefits, such as biodiversity protection and improvement of indigenous peoples’ livelihoods. Therefore, it is argued that the need to constantly repeat the story behind the A/R carbon offsets is at least as important as the actual credits (Stephan, 2012).

Under the NZ ETS, carbon sequestration through domestic A/R projects is credited for as long as the trees remain standing. An economic model suggested that the NZ ETS may encourage increased afforestation and rotation age, and discourage thinning and deforestation (Adams and Turner, 2012). Since significant logging is expected by 2020 due to a planting boom in the 1990s, afforestation, mostly on least productive land, has been anticipated to occur to offset harvest. Due to the fluctuations of carbon prices in the NZ ETS, the time of harvest and replanting is a strategic decision for land owners that are expected to avoid large penalties.

5.3.3. *Harvested wood products (HWP)*

HWP generally include wood-based materials used for construction timber, furniture, plywood, paper, and paper-derived products. Each of these HWP categories has a different useful half-life and therefore a specific temporary C-sequestration contribution that was not fully recognised in the past. Different approaches for accounting and reporting of carbon stored in HWP have been discussed elsewhere (Winjum *et al*., 1998; Flugsrud *et al*., 2001; IPCC, 2003; Dias *et al*., 2009; Suadicani, 2010; Manley and Maclaren, 2010; Frieden *et al*., 2012). The approach adopted in the Kyoto Protocol is explained below.

During the Kyoto Protocol first commitment period (2008-2012), the carbon stored in HWP was assumed to be released to the atmosphere at the time of harvest. This C accounting assumption is known as "instant oxidation" and has been observed in the NZ ETS as well (Manley and Maclaren, 2010). However, the new post-2012 accounting rules for land-use, land-use change and forestry (LULUCF) require that countries engaged in a second commitment period report any changes to HWP carbon pools. Under the new rules decided in Durban, increases in the HWP carbon pool are considered as annual negative emissions, just as increases of standing biomass.

The number of HWP credits depends on the half-life of the end product. The standard half-lives agreed in Durban for sawn wood, wood panels, and paper are 35, 25, and 2 years, respectively. Countries are left to decide to account for i) only domestic HWP; or ii) only exported HWP; or iii) both domestic and exported HWP. Standard half-lives may be used for all situations, whereas domestically-determined values may be employed for domestic HWP, and values may be obtained from the importing country for exported HWP. Imported wood or wood products are excluded from national accounting as well as HWP sent to landfills.

The EU ETS does not cover the land-use sectors, so accounting and reporting of European HWP carbon pools is carried out only at the national level. Therefore, this situation incentivises entities capped under the EU ETS to use wood for energy. In contrast, owners of tree plantations in NZ will benefit from the inclusion of HWP into the NZ ETS since this covers the forest sector, which will see a major drop of C stocks in the 2020s. It is likely that NZ's land owners will choose to account for and report both domestic and exported HWP because about 69% of wood produced in NZ is exported. The crediting of these temporary units will buy plantation owners some time and NZ's post-2020 carbon liabilities will be significantly reduced (Manley and Maclaren, 2010).

Some voluntary carbon markets credit C sequestration in HWP but since this is not a stand-alone activity it is aggregated into the estimated CF of a forestry project. For example, the VCS (2012) suggests estimating and reporting long-term carbon storage in HWP for forestry projects based on Winjum *et al*. (1998). It is also important to point out that crediting for C stored in HWP does not necessarily result in additional CO_2 removals since wood is mostly grown to produce HWP anyway. However, pricing the C stored in HWP can place this temporary C-sequestration technique above other competing uses of biomass by providing a surplus revenue stream to HWP, which are already in high demand.

5.3.4. Bioenergy with carbon capture and storage (BECCS)

The combustion of biomass for energy production emits CO_2, which could then be compressed, transported, injected and stored underground in an energy-intensive process. The potential for BECCS to withdraw CO_2 from the atmosphere then relies on constant flows of sustainable biomass converted to useful bioenergy by combustion to release the CO_2 and on new installations that ensure long-term sequestration of CO_2 in suitable geological formations, such as in inaccessible coal seams, saline aquifers, and depleted oil wells. Therefore, there are temporal and spatial constraints that need to be considered for matching CO_2 sources with sinks (Tan *et al*., 2013).

Current capture processes can remove between 85% and 95% of the CO_2 emitted by large-scale industries (Scott *et al*., 2013). Physical seepage of CO_2 arising from capture, transport, maintenance, injection, and storage could pose a health hazard to humans and animals, and contaminate water basins (Rusin and Stolecka, 2013) and surface vegetation (Al-Traboulsi *et al*., 2013). The risk of CO_2 seepage can also increase following the events of earthquakes or volcanic eruptions. Issues of limited availability of suitable sites; non-permanence; high operational costs; safety; and legal liability make CCS a controversial technology. The only current project examples involve CO_2

injection from natural gas fields (such as in saline aquifers at Sleipner, Norway) and when used for enhanced oil recovery. No thermal power plant CCS system yet exists.

The development of BECCS is dependent on the expansion of fossil fuel use with CCS technology. One of the features of CCS is that the captured CO_2 can be injected into oil wells to increase oil production through enhanced oil recovery (EOR). Hence, incentivising BECCS technology implies incentivising climate change through the growth of fossil fuel commerce. However, CCS may be perceived as a better option than otherwise venting the CO_2 to the atmosphere. Pacifying critics and guiding public perceptions about further subsidising the fossil fuel industry with CCS is then central to the future of the technology. For example, in Norway, a front runner in research and development of CCS technology, Røyrvik *et al.* (2012) argued that “the discourse is not really about CCS but politics in the form of narratives on promises, alliances and emotions caused by political actions”, which frame the technology “as either something good or something bad”.

After many years of political discourse and negotiations, CCS was officially included in the CDM in 2011. To address non-permanence, project proponents must place five percent of the credits granted in a reserve fund. If monitoring showed that no CO_2 seepage had taken place 20 years after the end of the last crediting period for the project, then the carbon credits held in the reserve fund could be awarded to the project developers. Moreover, liability for CCS projects would be automatically transferred to the host country in case a participant was not able to continue with the project. Günel (2012) described how power relations within the political process contributed to the decisions on the requirements for CCS to be accepted in the CDM.

In the EU, CCS projects are mainly promoted through the €1 bn European Energy Programme for Recovery and through the New Entrants’ Reserve (NER300) fund of the EU ETS, which was initially expected to raise €5-6 billion from the sale of 300 million CO_2 allowances in the carbon markets (Nykvist, 2013). Initially, about 2/3 of the NER300 fund was reserved for CCS projects, whereas the rest was planned to be directed to renewable energy technologies. The NER300 figure was later adjusted down to €4.5 bn due to low prices of the EU ETS credits (Lohwasser and Madlener, 2013) and at current carbon prices it is anticipated to gather only around €2 bn (NER300, 2013), representing a 65% drop from initial expectations.

The NER300 funds are distributed through two rounds of calls for proposals and disbursed based on performance, i.e. the amount of CO_2 stored for CCS projects. The first round, covering 200 million allowances raised approximately €1.5 billion, and 3 (out of 13 submitted) CCS and 16 (out of 66 proposed) renewable energy demonstration projects were selected for co-funding. In the second round, the NER300 programme will be funded with the proceeds from the auctioning of the remaining 100 million allowances as well as about €288 million that was not allocated in the first round. Reduced funding from constrained national budgets and low carbon prices, and a lack of sufficiently serious and coordinated political efforts suggest that the industry should accept regulation (e.g. through reform of the EU ETS and implementation of emissions performance standards) for CCS to play a role in the GHG emission reduction targets of the EU (Scott, 2013).

In comparison with other countries, NZ has only four or five large-scale point sources of CO_2 and has a relatively large renewable electricity production at around 70% of total generation. Although the potential for NZ-based CCS to be applied to the few industrial plants, such as those producing steel, cement and fertilisers, is relatively low, there is a need for regulation and legal policy since there is interest in advancing this technology in NZ (Richardson, 2013). Note that C sequestration through CCS or BECCS can be negated due to possible rebound effects of the energy product, i.e. additional demand of energy if saved. Moreover, since the main objective of BECCS seems to be electricity production, C sequestration is a by-product that may need a different level of incentives than other biomass-based NETs.

5.3.5. Anaerobic storage of forestry and agricultural residues

The literature usually divides this kind of long-term bio-carbon sequestration into: wood harvest and storage (WHS) and crop residue oceanic permanent sequestration (CROPS). WHS consists of harvesting dead wood or selective cutting of less productive trees and then storing the wood in: 1) trenches on carefully selected sites (e.g. 5-25 m below ground) within the tree plantations; 2) above-ground shelters; and 3) suitable landfills (Zeng, 2008). The largely anaerobic storage conditions would supposedly delay wood decomposition for 100-1000 years. It has been argued that this strategy offers a continuous stream of stored C that is low-tech, distributed, safe, reversible, and relatively easy for monitoring and verifying the amount of stored C (Zeng, 2008; Zeng *et al*, 2013). However, to date no WHS operation has been conducted specifically for long-term C storage.

CROPS consists of collecting and baling agricultural residues, such as corn, wheat and soybean residues; transporting bales by trucks and barges to deep ocean sites; ballasting bales as required with stones; and sinking bales for deposition on ocean sediments at depths greater than 1000-1500 m (Metzger and Benford, 2001). Storing bales below the thermocline, under cold and anaerobic conditions, would preserve carbon “for thousands of years” since at those depths there is an apparent limited marine mechanism that could breakdown lignocelullose compared to that of the terrestrial lignin peroxidises (Strand and Benford, 2009). Keil *et al*. (2010) conducted a 2-year incubation experiment in which they simulated CROPS and compared the remineralisation rate of residues (≤8%) with that of marine plankton (19%), and extended the model to suggest that about 75% of the crop residues would remain in the sediment during a 100-year period. CROPS is believed to alter the deep ocean sediment communities but the intensity and extent of this impact is unknown and needed for regulation.

Improved land and crop management, and job creation are requirements for the desired functioning of WHS and CROPS rather than co-benefits. Compared to other biomass-based NETs, which can deliver energy, waste management, biodiversity, water and/or soil improvements, these technologies do not necessarily provide extra services and primarily rely on a sufficient price of carbon for their implementation. However, these techniques seem to be simple and arguably efficient in sequestering carbon. Further research is needed for WHS and CROPS to be seriously considered for long-term or temporary carbon financing.

5.3.6. Soil carbon management including biochar application

Globally, the present quantity of organic C (including charcoal) stored in soils (~2,344 Gt C) is about 2.8 times higher than the atmospheric C concentration (~848 Gt C) and ~4.2 times larger than the terrestrial biotic C pool (~560 Gt C) (Stockmann *et al*., 2013). Therefore, small alterations to the size of the SOC stocks can have a substantial impact on climate change. Kirschbaum (2010), for example, argued that a mere change of 10% to the total SOC would be equivalent to all human-made CO_2 emissions over 30 years. Hence, there is interest in soil C-cycling manipulation through changes in land use and agricultural practices.

By and large, land use changes from cropland to pasture and from cropland to permanent forest may lead to the highest increases in SOC stocks (Stockmann *et al*., 2013). Increasing soil carbon of course has to be balanced with the respective land use especially if this is livestock production since it has a relatively high carbon footprint. Agricultural methods reputed to boost SOC include cover cropping; perennial pastures; crop residue management; reduced and no tillage; aerobic rice cultivation; integrated nutrient management including use of compost and manure; controlled grazing; complex crop rotations; agroforestry; and biochar application (Stavi and Lal, 2013). There are advantages and limitations arising from the implementation of most of these practices (Lal, 2009). According to Sanderman and Baldock (2010), the few available studies that report a relative increase in SOC stocks with the adoption of improved agricultural practices suggest that these are often due to mitigation of SOC losses rather than extra sequestration of SOC. For NZ's already C-rich soils, Parsons *et al* (2009) argued that this difference between sequestering versus maintaining sequestered soil carbon must be recognised in carbon pricing mechanisms.

The EU ETS and the CDM preclude soil carbon sequestration as individual activities, whereas the Alberta Offset System (AOS) in Canada is the only compliance carbon market that has allowed fully the trading of soil carbon credits created within the state (Swallow and Goddard, 2013). In the process of including SOC in the CDM, there are two A/R large-scale methodologies that restrict the extent of soil disturbance to ≤10% of the project area to prevent high SOC losses and leave the accounting for changes in

SOC stocks as part of the A/R activities as optional. Methodology AR-ACM0003 version 01.0.0 covers A/R of lands except wetlands (UNFCCC, 2012b), whereas AR-AM0014 version 02.0.0 covers A/R of degraded mangrove lands (UNFCCC, 2012c). This optional characteristic may incentivise project developers to build expertise on SOC accounting and monitoring without penalties – it is highly unlikely that project participants would voluntarily report SOC losses.

In Australia, soil carbon sequestration could generate C credits under the Carbon Farming Initiative (Cowie *et al*., 2012b). In fact, a soil carbon methodology was recently submitted to the Australian government for revision (Nason, 2013). The most relevant aspects of this advancement are: i) the estimated cost (AU$30 per ha) needed to define the soil C baseline value is expected to decrease over time; ii) credits are not granted for maintaining SOC but only for increasing it; and iii) the 100-year permanence requirement is perceived as a potential drawback.

Similar to other technically- and socially-challenging bio-carbon projects, the VCS of the voluntary carbon markets has already commodified soil carbon. Through the World Bank Bio-Carbon Fund, which also manages buffer funds to insure against risks and secure investment, the first soil carbon voluntary offset project is called the Western Kenya Carbon Farming Initiative (Swallow and Goddard, 2013). The respective soil carbon methodology is being tested for replication in other poor areas of the world.

Although biochar application is part of soil carbon management it is often treated separately since it involves incorporating significant (and sometimes externally-sourced) highly-recalcitrant C into the land. Relative to most of the other biomass uses intended for atmospheric CO_2 removals, biochar-carbon sequestration does not need to be continuous on a yearly basis and can result from one-off applications. It can be decentralised and does not rely on geological formations for storage. Biochar systems can also deliver GHG emission reductions through fossil fuel substitution and can offer other non-carbon co-benefits. Compared with the C temporarily stored in living trees or in harvested wood products, a fraction of the carbon sequestered in the form of biochar may last for hundreds or thousands of years. However, biomass converted into biochar and applied into soils yields less CO_2 removals than biomass introduced into BECCS systems, which is also likely to be easier for complying with accounting and VMRV methods.

Monetisation of biochar through crediting for carbon sequestration should not follow exactly the same rules and principles as those activities that reduce the future rate of GHG emissions. In fact, it needs a different system to capture the long-term removals of atmospheric CO_2 as distinct from avoiding GHG emissions to the atmosphere. This distinction could prioritise bio-carbon sequestration by pricing CO_2 ppm removals higher than CO_2-eq emission reductions. A range of ways of implementing this could be explored; for example, similar to other bio-carbon sequestration technologies, buffer funds for biochar systems could be established in order to bypass the requirement of proving its 100-years permanence, so that after 100 years, the buffer is released. Alternatively, temporary crediting could be imposed on biochar-C sequestration, but this option could depreciate the value of biochar credits.

Another way forward is to create carbon credit multipliers to promote carbon sequestration (Mathews, 2008b). In this sense, biochar long-term CO_2 removals could obtain 1.5, 2 or 3 times the number of credits given to GHG emission reductions achieved through other means (e.g. bioenergy production that failed to capture and sequester CO_2). However, conceiving soils exclusively for their capacity to store carbon may obscure their ability to perform vital functions. Future pricing of biochar-carbon sequestration will most likely be influenced by how the public perceives the financial means pursued to achieve the goal of biochar application.

5.4. Public perception issues

The diffusion of the relatively-new term ‘biochar’ is growing. However, the addition of charcoal to soils is an archaic soil management tool. Old charcoal-amended soils or Anthropogenic Dark Earths (ADE) have been found not only in what is presently known as Brazil (Sombroek, 1966), from where the term ‘Terra Preta’ originated, but also in other regions of the world (Blackmore *et al*., 1990; Paz-Rivera and Putz, 2009; Downie *et al*., 2011; Sheil *et al*., 2012).

Leach *et al*. (2012) suggested that the traditional making and cultivation of ADE soils found in Liberia, Sierra Leone, Guinea and Ghana is “embedded in social-ecological relations and histories”, which have not been explored for modern biochar application. In Liberia, for example, it has been estimated that up to 30% of the country’s annual cocoa production may stem from ADE soils that were built over centuries (if not millennia) of human-nature interrelationships. Biochar application for C sequestration cannot really be subjected to the same environments that allowed the formation of ADE. Therefore, the public will perceive the promotion of biochar technology as part of the current political economy that needs to consider environmental, social and cultural implications. In this sense, attitudes toward carbon finance for biochar systems are divided and play a major role in how the discourse and future of biochar is constructed.

Monetisation of biochar carbon does not need to be realised through carbon trading because alternatives exist (Böhm and Dabhi, 2009), but proponents for the proliferation of biochar systems are focused on the inclusion of biochar in carbon markets since these are currently the preferred method of most governments and corporations to address climate change.

5.4.1. Proponents for the inclusion of biochar in carbon markets

Due to the potential of biochar systems to withdraw atmospheric CO_2 and lock carbon in soils for more than 100 years, a number of biochar researchers and several pyrolysis companies advocate for the monetisation of biochar-carbon. Many of these believe that the large-scale deployment of biochar systems is of paramount importance in order to reach the 350 ppm target. Because of a lack of political will that is required to recognise and promote alternative carbon-financing mechanisms, proponents of biochar are arguably pushed to concentrate their efforts on the inclusion of biochar in carbon markets.

The majority of these advocates perceive the commodification of biochar-carbon as mainly a technical matter and are not necessarily well acquainted with the social, ethical, and psychological relations described in the social literature of carbon markets. As explained above, methodological issues for carbon finance of biochar systems could be addressed in a number of ways, such as using conservative estimates; issuing temporary credits; establishing buffer funds; creating carbon credit multipliers; and inventing a new carbon unit (e.g. removals of ppm CO_2). But since future generations are at stake (Hansen, 2009) – a case for intergenerational justice – the question follows: Does the end (biochar application) justify the means (carbon markets)?

5.4.2. Opponents to the immediate inclusion of biochar in carbon markets

Broadly, there are two fronts of opposition to the immediate inclusion of biochar in carbon markets. These are based on: 1) the different areas of uncertainty in biochar knowledge that could not only offset its climate-change mitigation potential but could also result in negative environmental consequences; and 2) the mounting evidence showing that carbon markets are obscure, complex and confusing mechanisms that are subjected to corporate manipulation for profit maximisation, resulting in exploitation of the less financially powerful, including the natural environment (Lohmann, 2006; Spash, 2010; Lohmann, 2010; Bertram and Terry, 2010; Bond, 2012). Note that the arguments against treating NETs as carbon offsets are not specific to biochar application (McLaren, 2012) and that carbon taxes can also be watered down through corporate power (Spash and Lo, 2012).

On the first front of opposition, which does not necessarily imply agreement with the second front, a number of scientists have recommended to proceed with caution and have discouraged the large-scale deployment of biochar systems until the uncertainties are resolved (Wardle *et al*., 2008b; Trumper *et al*., 2009; Royal Society, 2009; Sutherland *et al*., 2010; Singh *et al*., 2012; Merfield, 2012). Leach *et al*. (2012) further warned that “the imperative to ‘take biochar to market’ is driving and disciplining the science conducted around it”, and thus questions are driven more by the carbon markets than by the interest in improving soil functions.

On the second front, which draws inspiration and arguments from the first front, civil and non-governmental groups (led by Biofuelwatch) have voiced their preoccupations about the potential negative consequences arising from the inclusion of biochar in carbon markets (African Biodiversity Network *et al*., 2009; ETC Group, 2010; Ndameu and Biofuelwatch, 2011; Ernsting *et al*., 2011). Concerns include sourcing of large volumes of biomass feedstock; land grabbing in countries least responsible for contributing to climate change; misleading farmers based on false expectations; and letting polluters off the hook by allowing offsets to replace domestic pollution control measures. The Biofuelwatch Declaration, signed by approximately 150 organisations to "keep biochar and soils out of carbon trading", framed the biochar technology as "a new big threat to people, land, and ecosystems" (Rainforest Rescue, 2009). This discourse permeated into the media.

Monbiot (2009) communicated his disbelief in considering biochar as a miracle cure to tackle climate change. He recalled that some governments demanded that biochar is made eligible for carbon credits and considered their proposal to boil down to: "we must destroy the biosphere in order to save it". A debate followed (Lovelock, 2009; Goodall, 2009; Kharecha and Hansen, 2009; Read, 2009), in which respondents of Monbiot's scepticism showed their support to continue with biochar research rather than ruling it out because of knowledge gaps. None of them mentioned anything explicit about carbon trading. However, James Hansen firmly rejected carbon trading because it obstructs forces to bring atmospheric CO_2 levels down to 350 ppm (Hansen, 2009).

Along these lines, Bill McKibben, the high profile leader of the international grassroots campaign that aims to mobilise a global climate movement (350.org), sent a video message to the IBI at its conference in Rio de Janeiro, Brazil, in September 2010. McKibben (2010) warned that if biochar projects are used to offset or "green-wash the work of big fossil fuel companies, then they are dangerous and counterproductive and do more harm than they ever could possibly do good." McKibben also called for extreme caution in other areas such as scale and industrialisation of biochar; ELB regarded as waste; and social justice and development implications, such as land grabbing.

The proposal to make biochar eligible in carbon markets has resulted in strong opposition to biochar by a number of researchers and civil society groups concerned about land grabbing and carbon offsetting. Others have argued that the application of biochar into soils could bring effectively CO_2 levels down but not if linked to carbon markets (Bruges, 2009). Based on Spash (2010), it can be concluded that when biochar proponents are motivated by other objectives, such as soil improvement, then carbon markets become “a means to an end rather than the primary motive and may then be treated as such with resulting lack of knowledge, care and attention”. There is an obvious need for biochar advocates to analyse the social and ethical implications of carbon trading when promoting the inclusion of biochar in carbon markets.

5.5. A summary for action: proposing a way forward

To attract carbon finance, biochar must prove that it is sustainable and benefits the soil and environment. Evidence must be gathered using methodology based on the principles of LCA to properly measure the climate-change mitigation potential of biochar systems. More research is required both to determine the long-term soil-biochar interactions and to determine the sequestration potential over a 100-year time horizon. For example, recent evidence suggests higher than expected fluxes of black carbon out of system boundaries. For credits to be issued to biochar projects, the sequestration must be independently validated and verified, which will require a minimum of two degrees separation to avoid the corruption that has blighted some CDM projects. Even with this methodology and rigorous accounting and VMRV procedures, acceptance requires the precursors of agriculture being included in international C reckoning, along with soil carbon as a measurable pool, plus biochar being recognised as a soil carbon modifier. Alternatively, the pyrolysis of biomass could be recognised as the project activity that leads to the sequestration of carbon in biochar and project developers must demonstrate that the biochar is not combusted to claim the credits.

There is some way to go before any of these happen, principally because there are no cost effective methods of measuring accurately soil carbon over large acreages. The terms of entry to the carbon markets also need to be decided. Various approaches discussed here include using conservative sequestration estimates, issuing temporary credits, establishing buffer funds, creating carbon credit multipliers, and inventing a new carbon unit (i.e. for long-term atmospheric CO_2 removals as opposed to avoiding GHG emissions). However, no approach has consensus because focus has now moved to the (dis)functionality of the carbon markets which are increasingly criticised as obscure, complex and manipulated, resulting in exploitation of people and the environment. It is these latter consequences that proponents of biochar generally do not consider. Opponents also cite that biochar (and indeed all offsets) may be used to compensate for yet higher fossil fuel emissions, and therefore abet the increasing atmospheric CO_2 levels. It is the weighing of these arguments that lead to the following conclusions.

CHAPTER 6: CONCLUSIONS AND RECOMMENDATIONS

The conclusions and the recommendations drawn from this work focus mostly on the implications for biochar systems obtaining carbon finance.

6.1. Conclusions

Ten different end-of-life biomass (ELB) streams were identified with potential for climate-change mitigation in New Zealand. Purpose-grown plantations were not considered due to possible land-use conflicts. Based on several assumptions, if 80% of these ELB sources were converted into biochar through slow pyrolysis and the biochars were incorporated into soils, about 1.7 Mt CO_2 could potentially be sequestered every year in NZ. This equates to 2.4% of the country's total GHG emissions for 2011 (MfE, 2013). At a national level, this potential may seem low but can be significant for the carbon footprint (CF) of agricultural products; for example, a closed-loop biochar system using logging residues significantly reduces the CF of export logs. Therefore, the results of the CF studies of biochar systems can be interpreted as part of the CF arising from land use rather than purely biomass management.

The most promising feedstock for large-scale biochar production and carbon sequestration was found to be logging residues because of numerous reasons, such as the fact that they are concentrated in large quantities particularly in the Central North Island region and wood-derived biochar has a high C content. Biochar production is also promising as a waste management strategy (although some feedstocks have a relatively low C sequestration potential) as a means for diverting biomass from landfills provided that respective biochars are shown to have no or very low content of contaminants. Most of the other available ELB streams in NZ are widely distributed in terms of availability, ownership and location, so the installation of a centralised biochar plant may be challenging. For these dispersed feedstocks, such as orchard prunings, small-scale mobile pyrolysis reactors may be most applicable and further studies are recommended.

Three different types of feedstock were evaluated in detail using LCA methodology: 1) prunings from apple orchards in Hawke's Bay; 2) logging residues in Central North Island; and 3) wheat straw in Canterbury. The focus of the CF studies was on the climate-change impact category of LCA methodology and the goals were: i) to compare future alternative management scenarios for these three feedstocks; and ii) to determine the use of biomass to mitigate climate change in order to support policymaking in NZ. The functional unit chosen was 'the management of one tonne of fresh biomass'.

The alternative management options considered were a business-as-usual (BAU) scenario; a heat-only (HO) scenario; a combined-heat-and-power (CHP) scenario; and a biochar scenario. These scenarios were modelled following the attributional LCA approach without system expansion/substitution and the biochar scenario was the only option to mitigate climate change due to carbon sequestration that occurs regardless of what happens in the background system. However, attributional scenarios without system expansion/substitution cannot be compared with each other because these deliver different functions. Furthermore, one goal of this study was to support policymaking. Therefore, the consequential LCA approach was adopted to adjust the services delivered to make the scenarios comparable and also address the policy-relevant question about the use of biomass that can achieve the highest amount of carbon credits.

Under the consequential modelling, the HO scenario (-276.5 to -1,064.2 kg CO_2-eq per t biomass); the CHP scenario (-410.1 to -1,608.4 kg CO_2-eq per t biomass); and the biochar scenario (-271.9 to -792.6 kg CO_2-eq per t biomass) all offer potential to mitigate climate change relative to a reference scenario in which fossil fuels are assumed to be kept in the ground as a consequence of project implementation. Using marginal versus average data has been a point of debate in LCA and this factor affects the results significantly. In addition, note that biogenic CO_2 emissions arising from the natural or the accelerated decomposition of biomass products through combustion were not accounted for since these were assumed to be absorbed by the next crop rotation in a so-called 'carbon-neutral' cycle.

In terms of the functional unit (one tonne of biomass), the management of wheat straw offers the highest carbon abatement of the three ELB feedstocks evaluated. This is due to the low moisture content of straw which results in relatively high quantities of pyrolysis products. The management of biomass with high moisture content can face high energy costs due to drying prior to pyrolysis and decrease its potential to mitigate climate change. In terms of national carbon abatement, the management of logging residues offers the highest potential of all ELB streams available in NZ since significant volumes of this material are left on plantation soils in Central North Island and wood-derived biochar has high carbon content.

Potential soil-related GHG benefits offered by biochar application, such as fertiliser savings, suppression of N_2O emissions from soils, changes in soil organic carbon (SOC) stocks, and higher crop productivity can be initially neglected in CF studies of biochar systems. This is because the conditions for the realisation of these benefits are uncertain and current experiments suggest that a high biochar application rate (e.g. ≥10 t/ha) would be required to produce a relatively small impact, that is, there is a small benefit per unit feedstock biomass. In addition, validation, monitoring, reporting and verification methods that would be needed especially to confirm changes in N_2O emissions and SOC stocks might be too complicated for current broad implementation.

An important conclusion is that sequestering carbon on a long-term basis in the form of biochar must be considered differently to avoiding fossil-fuel GHG emissions through production of bioenergy without carbon capture and storage. The assumption that bioenergy will avoid a portion of fossil fuels formerly used to meet the global energy demand being extracted from the ground, and therefore respective GHG emissions will also be avoided, is questionable. Constrained by the current political economy, it is uncertain if the production of bioenergy could slow down the increasing energy demand that drives the extraction and combustion of fossil fuels. Furthermore, although it is usually considered that the next biomass rotation will absorb the respective biogenic CO_2 emissions, the combustion of biomass without CCS instantly increases atmospheric CO_2 levels. Temporal calculations of the effect of CO_2 release (immediately by combustion or slowly by natural processes) were out of scope.

The production of biochar and its incorporation into soils provides long-term atmospheric CO_2 removals regardless of how the economy is manipulated. This means that biochar is one of the few accessible technologies, apart from temporary carbon storage in trees, that provides a pathway to future global ‘negative emissions’ that are currently being reported as necessary by the UNFCCC and IEA if climate change is to be kept within acceptable levels. To take advantage of this fundamental difference between the value of sequestration versus emissions offsetting, biochar must be recognised separately in carbon-pricing mechanisms.

Sources of carbon finance for biochar systems include taxation to raise revenue to fund biochar activities directly, “cap-and-then-fund” mechanisms, ecolabelling of products, and carbon trading. In recent years, the focus has been on the latter since carbon markets are the preferred climate policy tool of most governments and corporations. However, carbon markets are obscure systems, which can be questioned from various angles, ranging from linguistics and ethics to strong criticisms of conventional economics and forms of governance. As suggested by Spash (2010), current carbon markets are fundamentally flawed because they promise a painless way to deal with climate change without making radical transformations to the growth economy that causes this problem in the first place. The market need for validation, monitoring, reporting and verification makes the schemes complex. The carbon markets can confuse and distract people from taking more effective measures such as reducing the demand for fossil fuel-based products. Also markets are poor at incorporating social, psychological and ethical considerations.

6.2. Recommendations

The CF studies evaluated here used conservative estimates of mitigation potential. More precise data is needed. The use of marginal or average data should also be explicitly described. Characterisation of the pyrolysis process is required to improve data quality regarding product yields. Long-term customised field research is also needed both to determine the possible biochar-soil interactions and the carbon sequestration potential over the 100-year time horizon. This would provide more certainty to the biochar carbon stability factor (BCSF) used in LCAs. Since a portion of biochar may move out of the project boundaries within this timeframe (e.g. by erosion), a conservative biochar migration factor (BMF) also needs to be applied in future LCAs of biochar systems. Research into the BMF may also reveal application practices, where it may be preferable to incorporate the biochar into the soil in order to protect it from erosion. Research is also needed on the impact of biochar application/incorporation into soils on surface albedo. In addition, the economics of biochar production and application into soils should be investigated from a life cycle perspective.

LCA studies of biochar from other types of ELB streams available in NZ (e.g. farm manures, putrescible waste and sewage sludge) should be conducted although biochars produced from these feedstocks would have relatively low carbon content and therefore low potential for climate-change mitigation. However, pyrolysis is an attractive waste management technology and respective biochars usually offer high agronomic potential. In those cases, carbon sequestration would be a bonus, or could even become a priority (see below). Moreover, different biochar system configurations (e.g. fast pyrolysis, mobile units) could be additionally modelled in order to identify the most appropriate technology for the situations under analysis.

One way to clearly differentiate bioenergy applications without CCS from biochar systems is to account for time and rate of biogenic carbon fluxes occurring over 100 years as the recently-developed dynamic LCA approach proposes (Levasseur *et al*., 2010). Accounting for time and rate of biogenic CO_2 emissions will show that bioenergy without CCS increases and decreases atmospheric CO_2 at certain periods of time. For example, if all logging residues were combusted at any given year, it would pose a risk in meeting national GHG targets because it would take several years (20-25 years) for future plantations to compensate for most of the CO_2 emitted. Note that in the case of biochar systems, CO_2 emissions from bio-oil and pyrolysis gas combustion would also be accounted for immediately in a dynamic LCA but a fraction of the carbon in the form of biochar will remain sequestered over 100 years. If biogenic CO_2 emissions were accounted for in dynamic LCA studies of biomass management pathways, the pyrolysis of short-lived ELB streams such as paper and putrescible waste could be prioritised since respective biochars would withdraw CO_2 from of the atmosphere at a faster pace than longer-lived ELB streams such as logging residues.

Biochar and temporary carbon storage in trees are feasible routes to a global carbon negative emissions profile. Several approaches for financially incentivising biochar carbon sequestration were explored. These include using conservative carbon-accounting estimates, issuing temporary credits, establishing buffer funds, creating carbon credit multipliers, and inventing a new unit such as ppm CO_2 reductions for recognising long-term atmospheric CO_2 removals as opposed to avoiding GHG emissions. However, further work is needed to reach consensus on the conditions for any of these approaches to be agreed and adopted internationally. There is also a need to seek alternative ways of finance to carbon markets.

This work contributes to further and wider discussion on the importance of incentivising the implementation of biochar systems to reduce atmospheric CO_2 concentrations. This carbon-negative technology is attractive in areas other than current climate policy, because of the potential of biochar systems to deliver various services. The diffusion and adoption of biochar technology does not need to derive solely from carbon finance; indeed, a focus exclusively on climate change mitigation may conceal its potential to become an important ingredient in land management and soil improvement. Therefore, long-term field research should be commissioned to demonstrate to farmers and stakeholders these other benefits and that it is not a temporary hype as labelled by its critics. Biochar could then be advanced on a local and regional basis in the same way as other agricultural practices are promoted. While biochar technology is currently facing numerous barriers for broad implementation, its future may be more promising.

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