BECCS
Bioenergy with Carbon Capture and Storage
Sustainability, Challenges, and Potential

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Submitted in total fulfilment of the requirements of the degree of Doctor of Philosophy

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April 2018
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Abstract

Bioenergy with carbon capture and storage (BECCS), as a negative emission technology, has been assigned a key role for achieving ambitious mitigation targets in several climate models.

BECCS is a multifaceted complex system which consists of a range of variables such as type of biomass resource, conversion technology, carbon dioxide capture process and storage options. Each of the pathways to connect these options has its own environmental, economic and social impacts and needs to be assessed through a holistic sustainability framework. Too often, however, the sole focus of assessment models is on techno-economics of BECCS to produce energy and deliver negative emissions. This study proposes an integrated adaptive management approach to model technical, economic, environmental, social and political aspects of BECCS systems. The adaptive management system employs a multi-criteria decision-making tool to rank BECCS systems against a set of key sustainability criteria. The aim of such adaptive management systems is to facilitate the decision-making process by evaluating the sustainability of the BECCS systems and introducing a systematic methodology to analyse the synergies and trade-offs between different criteria and mitigation scenarios.

The technical and economic impact values of the BECCS systems were calculated using a techno-economic assessment model based on published technical data. Environmental impacts were calculated using SimaPro software. With only very limited practical experience on BECCS deployment available, scoring qualitative social and policy criteria was based on prior literature associated with bioenergy systems and carbon capture and storage systems separately, for which social and policy experience is available.

It is crucial to adapt the objectives and criteria of an adaptive management system used for implementing a BECCS system according to the regional parameters. To this end, opportunities for application of BECCS in the Australian power sector, using the adaptive management principals, were investigated. Having significant resources of organic waste for bioenergy production and the accumulated practical knowledge through ongoing carbon capture and storage projects, makes Australia a good candidate for deployment of BECCS. A feasibility assessment conducted in this study found that, based on the quantity of biomass resources available, BECCS options in Australia have the potential to remove a total of 25 Mt CO₂ per
year from the atmosphere as negative emissions and supply up to 13.7 TWh of renewable power.

Co-firing in existing power plants equipped with carbon capture and storage could be an effective near-term mitigation option and provides a bridging technology to deliver secure energy for a growing population while cost-effectively lowering CO₂ emissions. In this study, the global technical potential and challenges of co-firing biomass with carbon capture and storage to achieve zero or negative emissions was assessed. The results show that direct co-firing of up to 20 per cent biomass in a modern pulverised coal combustion plant equipped with CO₂ capture and storage could deliver negative emissions of up to -26 kg CO₂ per MWh.

One of the main challenges to deploy BECCS at the level required in the stringent emission scenarios is expanding sustainable bioenergy production. Intensification of bioenergy production could result in severe pressure on natural resources, especially land and water, and competition between food, feed, and energy. Thus, it potentially could lead to controversial economic, ethical, and environmental issues. To avoid the social uncertainties and environmental impacts resulting from dedicated energy crop production, this study focuses on BECCS potential restricted to the presumption of no land-use expansion and no increase in water consumption. Hence, it is recommended that any projection of the potential of BECCS to deliver negative emissions in the future should be limited to bioenergy using organic residue and waste.

A sensitivity analysis using scenarios with different sustainability paradigms for mitigation was conducted. Selection of a scenario determines the objective of the decision-making analysis. The scenarios examined in this study reflect the effect of socio-economic, technical and environmental drivers on the sustainability ranking of BECCS systems. Water-use, and land-use change, levelised cost of electricity production, and global technical capacity to deliver negative emission were used to demonstrate the trade-off between environmental, economic and technical performance of the BECCS systems using municipal solid waste and residues from agricultural and forestry sectors. The priority was assigned to the technical potential of BECCS to deliver negative emission with minimum environmental ramifications and economic cost. The results endorse BECCS systems using municipal solid wastes under all scenarios and trade-offs.
The results of this study show that sustainable agricultural residues have the potential to deliver global negative emissions of 1.7 Gt CO₂/year, forestry residues can deliver global negative emissions of 1.1 Gt CO₂/year, and organic municipal solid waste can provide global negative emissions of 2 Gt CO₂/year. Therefore, globally, the total sustainable BECCS potential is 4.8 Gt CO₂/year, with the total energy produced (through sustainable BECCS pathways) is around 50 EJ. It should be noted that 4.8 Gt CO₂ is considered to be the maximum amount of negative emission that could be delivered under the holistic sustainability criteria used here. This compares with a value of 20 Gt CO₂/year negative emissions used in many global models. Even the annual 4.8 Gt contribution may be constrained further by technical, economic and policy challenge, although advances in biotechnology for example, might conceivably lead to new opportunities for BECCS. However, it would be dangerous to base a global mitigation strategy on as-yet-undiscovered biotechnologies.

Lack of integrated governmental and public support has made investment in BECCS a high political and financial risk. Therefore, a more certain way forward to underpin BECCS deployment, is to ensure that there is strong social support and integrated policy schemes that recognise, support and reward negative emission, for without negative emissions delivered through BECCS and perhaps other technologies, there is little prospect of the global targets agreed to at Paris, being met.
Declaration

This is to certify that

- the thesis comprises only the original work by the author towards the degree of Doctor of Philosophy,
- due acknowledgement has been made in the text to all other material used,
- the thesis is less than 100,000 words in length, exclusive of tables, figures, bibliographies and appendices.

Nasim Pours, April 2018
I would like to express my deep gratitude to Prof. Paul Webley and Prof. Peter Cook, my supervisors, for their excellent guidance, encouragement, support and unwavering optimism, without which this thesis would not have been possible.

I would like to thank Prof. Geoff Stevens, my committee chair, for his advice and assistance in keeping my progress on schedule.

My grateful thanks are also extended to staff at the Department of Chemical Engineering and the Peter Cook Centre for CCS Research.

I also wish to thank my friends and officemates for their support and encouragement throughout my study.

This journey would not have been possible without the support of my family. I am especially grateful to my parents, for their devotion and for teaching me the zeal for learning. My heartfelt thanks go to my siblings, Paria and Aryan, for their love and constant inspiration.
Publications

Journal Papers


Conference Papers


Preface

The project reported in this thesis was carried out by the candidate at the Department of Chemical Engineering, University of Melbourne, Australia, under the supervision of Prof. Paul A. Webley and Prof. Peter J. Cook. The project was funded by the University of Melbourne and the Peter Cook Centre for CCS Research.
To my parents, Fariba and Namdar
# Contents

## INTRODUCTION

1.1 Research Background  
1.2 Challenges Facing BECCS Deployment  
1.3 Knowledge Gaps  
1.3.1 Sustainability Framework for BECCS  
1.3.2 Baseline Bioenergy Surveys  
1.3.3 Optimized Source and Sink Matching  
1.3.4 Regional Studies of BECCS Feasibility  
1.3.5 Social Assessment  
1.3.6 Policy Tool Analysis  
1.4 Structure of This Thesis  

## CURRENT STATUS OF BIOENERGY WITH CCS (BECCS)

2.1 Climate Models and Need for Negative Emissions  
2.1.1 Negative Emission Technologies (NETs)  
2.2 Bioenergy with Carbon Capture and Storage (BECCS)  
2.2.1 BECCS Potential to Deliver Negative Emission  
2.2.2 Bioenergy for BECCS  
2.2.3 Carbon Capture and Storage (CCS)  
2.2.4 Global Status of BECCS Projects  
2.3 Economic Implications of BECCS  
2.3.1 Role of BECCS in Achieving an Economic Cost for a 2°C Target  
2.3.2 Emission Overshoot  
2.3.3 Carbon Price  
2.3.4 International Market for Biomass and Bioenergy  
2.3.5 BECCS and Carbon Lock-In  
2.4 Environmental Impacts of BECCS  
2.4.1 Land-use  
2.4.2 Water Consumption  
2.4.3 Efficient Biomass Production Methods  
2.5 Social Impact of BECCS
2.5.1 Food security and affordability | 37
2.5.2 Land tenure | 38
2.6 Conclusions | 39

**Co-firing coal with biomass in a BECCS context** | 40
3.1 Introduction | 41
3.1.1 Co-firing options | 42
3.1.2 Biomass for co-firing | 43
3.1.3 Examples of co-firing reported in the literature | 45
3.2 Methodology | 47
3.2.1 MEA CO₂ capture technology | 47
3.2.2 Fuels | 48
3.2.3 Impact on plant efficiency | 49
3.2.4 CO₂ emissions | 51
3.2.5 Slagging | 51
3.2.6 Fouling | 52
3.3 Results | 53
3.3.1 CO₂ emissions profile | 53
3.3.2 Impact of co-firing on efficiency | 54
3.3.3 Slagging & Fouling | 55
3.3.4 Impact on CO₂ capture process | 57
3.3.5 Impact on ESP collector | 58
3.4 Discussion | 59
3.5 Conclusions | 62

**Municipal solid waste as a resource for BECCS** | 63
4.1 Introduction | 64
4.1.1 MSW incineration | 65
4.1.2 LFG combustion | 66
4.2 Methodology | 68
4.2.1 MSW combustion plant | 68
4.2.2 LFG combustion plant | 68
4.2.3 Carbon capture and storage system | 70
4.2.4 Cost of electricity production | 72
<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>4.2.5 Environmental Impact Assessment</td>
<td>75</td>
</tr>
<tr>
<td><strong>4.3 Results</strong></td>
<td>80</td>
</tr>
<tr>
<td>4.3.1 Cost of Electricity Production</td>
<td>80</td>
</tr>
<tr>
<td>4.3.2 Environmental Impact Assessment</td>
<td>86</td>
</tr>
<tr>
<td><strong>4.4 Discussion</strong></td>
<td>91</td>
</tr>
<tr>
<td><strong>4.5 Conclusions</strong></td>
<td>93</td>
</tr>
<tr>
<td><strong>Agricultural and Forestry Residues as a Resource for BECCS</strong></td>
<td>94</td>
</tr>
<tr>
<td>5.1 Introduction</td>
<td>95</td>
</tr>
<tr>
<td>5.1.1 Organic Residues: A Resource for BECCS</td>
<td>96</td>
</tr>
<tr>
<td>5.1.2 Potential of Organic Residues</td>
<td>97</td>
</tr>
<tr>
<td><strong>5.2 Methodology</strong></td>
<td>99</td>
</tr>
<tr>
<td>5.2.1 Modelling BECCS Power Plants</td>
<td>100</td>
</tr>
<tr>
<td>5.2.2 Cost of Electricity Production</td>
<td>100</td>
</tr>
<tr>
<td>5.2.3 Environmental Impact Assessment</td>
<td>102</td>
</tr>
<tr>
<td><strong>5.3 Results</strong></td>
<td>104</td>
</tr>
<tr>
<td>5.3.1 Techno-Economic Assessments</td>
<td>104</td>
</tr>
<tr>
<td>5.3.2 Environmental Impact Assessment</td>
<td>107</td>
</tr>
<tr>
<td><strong>5.4 Discussion</strong></td>
<td>112</td>
</tr>
<tr>
<td>5.4.1 Sustainable Residue Removal</td>
<td>112</td>
</tr>
<tr>
<td>5.4.2 Availability of Residues for BECCS</td>
<td>114</td>
</tr>
<tr>
<td>5.4.3 Feasibility of Implementing BECCS Using Residues</td>
<td>116</td>
</tr>
<tr>
<td>5.4.4 BECCS and Food Security</td>
<td>117</td>
</tr>
<tr>
<td>5.4.4 Social Benefits of BECCS</td>
<td>117</td>
</tr>
<tr>
<td><strong>5.5 Conclusions</strong></td>
<td>119</td>
</tr>
<tr>
<td><strong>Opportunities for Application of BECCS in the Australian Power Sector</strong></td>
<td></td>
</tr>
<tr>
<td><strong>6.1 Introduction</strong></td>
<td>121</td>
</tr>
<tr>
<td>6.1.1 The Bioenergy Sector in Australia</td>
<td>121</td>
</tr>
<tr>
<td>6.1.2 CCS in Australia</td>
<td>123</td>
</tr>
<tr>
<td><strong>6.2 Modelling BECCS Systems</strong></td>
<td>126</td>
</tr>
<tr>
<td>6.2.1 Source and Sink Matching</td>
<td>126</td>
</tr>
<tr>
<td>6.2.3 BECCS Power Plants</td>
<td>128</td>
</tr>
<tr>
<td>Section</td>
<td>Title</td>
</tr>
<tr>
<td>---------</td>
<td>----------------------------------------------------------------------</td>
</tr>
<tr>
<td>6.2.4</td>
<td>COST OF ELECTRICITY PRODUCTION</td>
</tr>
<tr>
<td>6.2.5</td>
<td>ENVIRONMENTAL IMPACT ASSESSMENT</td>
</tr>
<tr>
<td>6.3</td>
<td>RESULTS</td>
</tr>
<tr>
<td>6.3.1</td>
<td>COST OF ELECTRICITY PRODUCTION</td>
</tr>
<tr>
<td>6.3.2</td>
<td>ENVIRONMENTAL IMPACT ASSESSMENT</td>
</tr>
<tr>
<td>6.3.3</td>
<td>TRADE-OFF ANALYSIS</td>
</tr>
<tr>
<td>6.3.4</td>
<td>EFFECT OF DIFFERENT BECCS CONFIGURATIONS ON ITS OVERALL PERFORMANCE</td>
</tr>
<tr>
<td>6.4</td>
<td>DISCUSSION</td>
</tr>
<tr>
<td>6.4.1</td>
<td>TECHNICAL POTENTIAL FOR CO₂ MITIGATION USING BECCS IN A AUSTRALIAN CONTEXT</td>
</tr>
<tr>
<td>6.4.2</td>
<td>POLICY MECHANISMS FOR BECCS</td>
</tr>
<tr>
<td>6.4.3</td>
<td>FEASIBILITY OF BECCS DEPLOYMENT IN AUSTRALIA</td>
</tr>
<tr>
<td>6.5</td>
<td>CONCLUSIONS</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Section</th>
<th>Title</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>7.1</td>
<td>INTRODUCTION</td>
<td>156</td>
</tr>
<tr>
<td>7.1.1</td>
<td>MULTI-CRITERIA DECISION ANALYSIS (MCDA)</td>
<td>157</td>
</tr>
<tr>
<td>7.1.2</td>
<td>APPLICATIONS OF MCDA IN PLANNING SUSTAINABLE ENERGY SYSTEMS</td>
<td>157</td>
</tr>
<tr>
<td>7.1.3</td>
<td>MAIN CRITERIA FOR MCDA</td>
<td>159</td>
</tr>
<tr>
<td>7.2</td>
<td>METHODOLOGY</td>
<td>161</td>
</tr>
<tr>
<td>7.2.1</td>
<td>ANALYTIC HIERARCHY PROCESS (AHP)</td>
<td>161</td>
</tr>
<tr>
<td>7.2.2</td>
<td>SUSTAINABILITY CRITERIA AND SUB-CRITERIA</td>
<td>164</td>
</tr>
<tr>
<td>7.2.3</td>
<td>MODELLING BECSS ALTERNATIVES</td>
<td>167</td>
</tr>
<tr>
<td>7.2.4</td>
<td>IMPACT EVALUATION</td>
<td>168</td>
</tr>
<tr>
<td>7.2.5</td>
<td>SCORING AND WEIGHTING THE CRITERIA</td>
<td>173</td>
</tr>
<tr>
<td>7.3</td>
<td>RESULTS</td>
<td>175</td>
</tr>
<tr>
<td>7.3.1</td>
<td>SENSITIVITY ANALYSIS</td>
<td>176</td>
</tr>
<tr>
<td>7.3.2</td>
<td>TRADE-OFF ANALYSIS</td>
<td>179</td>
</tr>
<tr>
<td>7.4</td>
<td>DISCUSSION</td>
<td>183</td>
</tr>
<tr>
<td>7.5</td>
<td>CONCLUSIONS</td>
<td>186</td>
</tr>
</tbody>
</table>
CONCLUSIONS 187

8.1 IS CO-FIRING COAL WITH BIOMASS A VIABLE OPTION IN A BECCS CONTEXT? 188

8.2 WHAT ARE THE OPPORTUNITIES FOR APPLICATION OF BECCS IN THE AUSTRALIAN POWER SECTOR? 189

8.3 HOW IMPORTANT IS ORGANIC RESIDUE/WASTE AS A RESOURCE FOR BECCS? 189

8.4 COULD MUNICIPAL SOLID WASTE BE A VALUABLE RESOURCE FOR BECCS? 190

8.5 WHAT IS THE POTENTIAL FOR AGRICULTURAL AND FORESTRY RESIDUES TO BE A RESOURCE FOR BECCS? 190

8.6 HOW SUSTAINABLE IS BECCS? 191

8.7 IS AN ADAPTIVE MANAGEMENT SYSTEM USEFUL WHEN ASSESSING THE OPPORTUNITIES FOR SUSTAINABLE DEPLOYMENT OF BECCS? 192

8.8 HOW FEASIBLE IS LARGE-SCALE BECCS DEPLOYMENT GLOBALLY? 192

BIBLIOGRAPHY 195
List of Figures

Figure 2-1- Emission pathways for different CO$_2$-eq concentration scenarios 9
Figure 2-2- Bioenergy with carbon capture and storage (BECCS) 12
Figure 2-3- Global Technical Bioenergy Potential by main resource category for the year 2050 17
Figure 2-4- Global map of CCS projects 21
Figure 3-1- CO$_2$ emissions from a co-firing at different biomass-to-coal ratios 54
Figure 3-2- Efficiency penalty due to co-firing biomass at different biomass-to-coal ratios 55
Figure 3-3- Change in slagging propensity at different biomass-to-coal ratios 56
Figure 3-4- Fouling index of co-firing PC plant at different biomass-to-coal ratios 57
Figure 4-1- System boundary for MSW-CCS inventories 76
Figure 4-2- System boundary for LFG-CCS inventories 77
Figure 4-3- CO$_2$ emission from LFG-GT, MSW-CFB and Coal-PC with and without CCS 80
Figure 4-4- LCOE with and without CCS and cost of CO$_2$ avoided under business-as-usual scenario 81
Figure 4-5- Impact of carbon price on LCOE 82
Figure 4-6- Impact of REC on LCOE 84
Figure 4-7- Impact of negative emission refunding scheme on LCOE 85
Figure 4-8- Comparative impact assessment of the waste-based systems per 1 kg waste used 87
Figure 4-9- Apportioned CO$_2$ emission of sub-process in MSW-CCS and LFG-CCS per functional unit 89
Figure 5-1- System boundary for residue-based BECCS models inventories 103
Figure 5-2- Impact of REC on LCOE of BECCS, Coal-CCS and coal w/o CCS 105
Figure 5-3- Impact of Carbon Pricing on LCOE of BECCS, Coal-CCS and coal w/o CCS 106
Figure 5-4- Impact of negative carbon credit on the LCOE of BECCS, Coal-CCS and coal w/o CCS 107
Figure 5-5- Apportioned CO$_2$-eq emission of the BECCS cradle to grave LCA processes 111
Figure 6-1- Distribution of bioenergy electricity and heat generation facilities in Australia 127
Figure 6-2- Australia’s basins for CO$_2$ storage potential and CCS projects 127
Figure 6-3- System boundary for BG-CCS and FR-CCS inventories 133
Figure 6-4- System boundary for MSW-CCS inventories 134
Figure 6-5- System boundary for LFG-CCS inventories 134
Figure 6-6- Impact of REC on LCOE 138
Figure 6-7- Impact of carbon price on LCOE 139
Figure 6-8- Impact of negative carbon credit on LCOE 141
Figure 6-9- GWP, water consumption and LCOE trade-off 144
Figure 6-10- Centralised biomass hub system and centralised CO\textsubscript{2} storage hub system 145
Figure 6-11- Apportioned CO\textsubscript{2} emission of sub-processes of BG-CCS configurations 146
Figure 7-1- Schematic of the AMS for BECCS systems 164
Figure 7-2- Schematic of the main processes throughout the lifecycle of BECCS systems 167
Figure 7-3- Overall Sustainability Index of BECCS alternatives under baseline scenario 175
Figure 7-4- Overall Sustainability Index (SI) of BECCS alternatives under scenario No.1 177
Figure 7-5- Overall Sustainability Index (SI) of BECCS alternatives under scenario No.2 178
Figure 7-6- Overall sustainability Index (SI) of BECCS alternatives under scenario No.3 179
Figure 7-7- LCOE and negative emission trade-off 180
Figure 7-8- Sustainable BECCS potential to contribute to the 2°C target 181
Figure 7-9- Water use, land-use change and negative emission trade-off for BECCS alternatives 182
List of Tables

Table 3-1- Proximate and ultimate analysis of the fuels 48
Table 3-2- Ash composition of the fuels 49
Table 4-1 Techno-economic input data 74
Table 4-2- Elemental compositions of MSW and black coal 74
Table 4-3- Composition of MSW 75
Table 4-4- Degradable organic content of the waste contents 79
Table 4-5- Actual impact assessment of the waste-based systems with and without CCS per functional unit 88
Table 5-1- Elemental compositions of the organic residues and coal 99
Table 5-2- Techno-economic data of the BECCS plants 101
Table 5-3- CO₂ emission, LCOE with and without CCS and cost of avoided emission 104
Table 5-4- Lifecycle impact assessment of the BECCS systems per each KWh of electricity generated 109
Table 5-5- Availability of Sustainable crop residues 114
Table 5-6- Potential of agricultural residues to deliver negative emission and energy through BECCS 115
Table 5-7- Crop production, Land: Land Harvested (Mha), Yield: crop yield (t/ha), Production: total production (Mt) 116
Table 6-1- Elemental compositions of the fuels 130
Table 6-2- Techno-economic input data 131
Table 6-3- Composition of Australian MSW 132
Table 6-4- CO₂ emission, LCOE with and without CCS and cost of avoided emission 137
Table 6-5- Lifecycle impact assessment of the BECCS systems per 1 KWh of electricity generated 142
Table 6-6- Annual net CO₂ emission and electricity generation 147
Table 7-1- values of RI for small problems 163
Table 7-2- Criteria and sub-criteria system for BECCS sustainability evaluation 165
Table 7-3- Impact evaluation of the BECCS alternatives for each sub-criterion 168
Table 7-4- Weights and consistency ratio of the criteria and their corresponding sub-criteria 174
Table 7-5- weightings of sustainability criteria and their consistency ratio under four scenarios 176
Chapter 1

Introduction
1.1 Research background

Bioenergy with carbon capture and storage (BECCS) is a negative emission technology that offers permanent net removal of carbon dioxide (CO$_2$) from the atmosphere. In most of climate models this negative emission is seen as essential to keep global warming well below +2 °C by 2100. The majority of integrated assessment models have incorporated a BECCS-derived negative emission potential up to 20 Gt CO$_2$/year. The challenge of scaling up bioenergy for BECCS to the level required to achieve 2 °C scenarios is producing sustainable biomass while maintaining a balance with food and fibre production. In addition, any expansion of bioenergy production must be in accordance with economic development, social benefits and policy/regulatory regimes.

1.2 Challenges facing BECCS deployment

Historically, unsustainable biomass harvest for fuel has led to loss of natural forests, degradation of productive lands, increased greenhouse gas emissions, loss of biodiversity and carbon stock and depletion of water resources. Intensification of energy crops production for BECCS could result in severe competition between food, feed, and energy, thus leading to a range of economic, ethical, and environmental issues. Another challenge is the lack of integrated governmental and public support, which in turn has made investment in BECCS subject to economic, regulatory and sovereign risk.

It is therefore very important to integrate the economic, ecological, political and social impacts of BECCS technologies into a sustainability framework. Such a framework can be used to provide a comprehensive adaptive management system that in turn enables decision-makers to plan BECCS options in a transparent and timely manner.

1.3 Knowledge Gaps

Provided that the sustainability criteria of large-scale deployment are met, BECCS has the potential to be a transitional pathway from the present carbon-intensive energy system towards a carbon-free future. To be able to exploit the advantages of BECCS, a greater level of knowledge on a number of key topics is essential;

1.3.1 Sustainability framework for BECCS— BECCS is a complex multi-faceted system and it is very important to integrate the economic, ecological, political and social impacts of BECCS technologies into a sustainability framework. Such a framework can be used to provide
a comprehensive adaptive management system that in turn enables decision-makers to plan BECCS options in a transparent and timely manner. The first step should be to identify the main impact categories and their corresponding criteria. The criteria should be comprehensive and account for regional circumstances. A decision-making tool can then be used to evaluate the overall performance of each BECCS option against the defined sustainability criteria. Large-scale BECCS deployment should be planned within such a sustainability framework. Otherwise, large-scale BECCS deployment could lead to adverse outcomes that outweigh its inherent benefits.

1.3.2 Baseline bioenergy surveys— most of estimations of global bioenergy potentials are based on general assumptions, with no regard to vast differences that exist in regional access to bioenergy resources. Surveys of land availability, annual yield and type of biomass, level of technical and economic development, supply-chain management analysis, lifecycle environmental assessment, socio-cultural evaluation and knowledge of local climate variations of each region are essential for determining bioenergy potential. The aim of such surveys should be to provide an accurate map of sustainable biomass resources at the regional, national and global levels.

1.3.3 Optimized source and sink matching— provided the necessary data is available (and it has to be recognised that this is not yet the case in most parts of the world) this could be addressed via a generic decision-making model that finds the most optimal match between several nodes of BECCS i.e. biomass resources (source), biomass pre-treatment and energy conversion plants, and CO₂ capture, CO₂ transport and geological storage sites (sinks). The aim of such a model would be to ascertain optimal BECCS pathways that deliver negative emission and energy at the lowest economic, environmental and social cost.

1.3.4 Regional studies of BECCS feasibility— A National Determined Contributions submitted by a country to the Paris Agreement is an indication of that country’s ambition to contribute to control of global warming. There is a considerable gap in the potential for BECCS deployment between developed and developing countries. Currently the majority of BECCS projects are underway or being developed in developed countries, whereas, developing countries, many of which have the greatest potential for deployment of BECCS, lag far behind. In the light of existing global agreements on meaningful emission cuts, it is important to conduct country-by-country assessments on the potential role that BECC could play in the achievement of national and global mitigation targets. The feasibility assessment should
account for the development of low carbon national energy systems, access to biomass resources and CCS technologies, economic development, political structures and an agreed national mitigation agenda.

1.3.5 Social assessment— Large-scale BECCS has yet not been practiced and there is a lack of information on its public acceptability. In addition, BECCS is a combination of two core technologies, bioenergy and CCS, both of which are poorly understood and subject to antipathy by the public. Social assessment of BECCS is therefore an essential prelude to its deployment. This will require active engagement with media, trusted NGOs and local stakeholders.

1.3.6 Policy tool analysis— in most climate models BECCS is regarded as a bridging technology, from the current-carbon intensive energy system to a fully carbon-free energy system. The timeframe used in these models usually encompass the interval 2020-2100. However, BECCS is a complex energy technology and strong policy support is essential to accelerate its deployment and exploit its full potential. At present, none of major national mitigation policy schemes include BECCS. To recognise and reward negative emission is a key requirement to underpin BECCS deployment. Extensive research to identify and evaluate the possible national and international policy mechanisms to fast-track BECCS deployment is needed.

1.4 Structure of this thesis

With these challenges in mind, this PhD project aims to answer a number of major questions, such as;

− What are appropriate sustainability criteria for BECCS deployment?
− What is the sustainable potential of BECCS?
− What is the potential role of BECCS in decarbonising the power sector?
− How technically feasible is BECCS?
− How environmentally acceptable is BECCS?
− How economically viable is BECCS?
These questions are addressed in this thesis, which is structured as follows:

**Chapter 2— Current status of Bioenergy and CCS (BECCS):** this chapter presents a comprehensive literature review of over 200 recent publications on global status of BECCS, its technical, economic, environmental, social and political challenges and opportunities.

**Chapter 3— Co-firing coal with biomass in a BECCS context:** global technical potential and challenges of co-firing biomass with CCS to achieve zero or negative emission are assessed in chapter 3. The effect of co-firing different second and third generation biomass resources on CO₂ emission, boiler efficiency, extent of slagging, fouling, plant emission profile and the performance of the CO₂ capture process are evaluated. Direct co-firing at a biomass-to-coal ratio up to 20% (on a mass basis) in a pulverised coal combustion plant with conventional monoethanolamine CO₂ capture technology is modelled.

**Chapter 4— Municipal solid waste as a resource for BECCS:** this chapter provides an estimate of the potential for using municipal solid waste as a resource for BECCS. Techno-economic and environmental impact assessments evaluate the feasibility of two specific BECCS options; municipal solid waste incineration with carbon capture and storage, and landfill gas combusted in a gas turbine with carbon capture and storage. The levelised cost of electricity and the cost of avoided emissions are used as economic indices. Lifecycle environmental impact categories considered are global warming potential, abiotic depletion, ozone layer depletion, eutrophication, acidification, particulate matter, human toxicity (cancer), human toxicity (non-cancer), freshwater eco-toxicity and water availability.

**Chapter 5— Agricultural and forestry residues as a resource for BECCS:** the potential and limitations of using residues from forestry and major agricultural crop as feedstock for BECCS is investigated in chapter 5. The techno-economic and environmental assessment (similar to chapter 4) is conducted to assess the sustainability of residue-based BECCS and its potential to produce renewable energy and negative emission.

**Chapter 6— Opportunities for application of BECCS in the Australian power sector:** This chapter assesses the potential contribution of BECCS to achieving long term decarbonising using the Australian energy sector as an example. The availability of sustainable bioenergy resources and the economic viability and environmental impacts of BECCS are evaluated in the Australian context. The opportunities for establishing BECCS and associated CCS systems and their potential/challenges are discussed in this chapter.
Chapter 7—An Adaptive Management System for Sustainable Deployment of BECCS: This chapter proposes an integrated adaptive management system to model technical, economic, environmental, social and political aspects of a BECCS system. The adaptive management system employs a multi-criteria decision-making tool, using an analytical hierarchy process, to assess several BECCS alternatives and rank them against a set of key sustainability criteria. The biomass resources for the BECCS alternatives in this study are organic waste from municipalities and residues from the agricultural and forestry sector. The aim of such an adaptive management system is to facilitate the decision-making process when evaluating the sustainability of the BECCS alternatives; It also introduces a systematic methodology to analyse the synergies and trade-offs between different criteria and mitigation scenarios.

Chapter 8—Conclusion: results of chapters 3-7 are used to present a picture of potential of BECCS under defined sustainability criteria. Technical, economic, social and political challenges and possible solutions to achieve the maximum potential of sustainable BECCS systems are considered. The derived “sustainable” BECCS potential is then compared to the quantities of CO$_2$ removal envisaged in a number widely cited of integrated assessment models.
Chapter 2

Current Status of Bioenergy with CCS (BECCS)
2.1 Climate models and need for negative emissions

The majority of climate models show that in a baseline scenario with no attempt to control the anthropogenic greenhouse gas (GHG) emission, atmospheric carbon dioxide equivalent (CO$_2$-eq) concentration would reach levels between 750 and more than 1300 ppm by 2100 [1]. In these models this level of CO$_2$ concentration corresponds to a more than 5 °C increase in the global temperature [1, 2]. The ramifications of such a temperature rise could include severe desertification, droughts, changing patterns of precipitation and temperature, collapse of forests and biodiversity, extreme weather events and rise in global sea levels [2].

A growing global consensus on the need to mitigate anthropogenic GHG emissions led to a historical agreement at the 2015 United Nations Conference of Parties (COP 21) in Paris, known as the Paris Agreement [3]. The Paris Agreement sets out a global action plan to put the world on track to avoid dangerous climate change by limiting global warming to well below 2 °C [3]. The 2 °C scenario (2DS) limits the global budget to 600–1200 Gt CO$_2$-eq from 2015 to 2100 [4-9]. Current emission trends are likely to overshoot this budget; therefore, urgent emission reduction measures and tight emission budgets are essential.

Under the evolving Paris Agreement, each country is expected to develop its own mitigation strategy, called the Intended Nationally Determined Contributions (INDC). Submission of INDC is legally binding, however achieving the targets is not. The Paris Agreement places the onus on countries to meet the mid-term mitigation targets and increase their reduction target every five years in order to balance the source and sink of anthropogenic GHG emission in the second half of the 21st century [3, 10]. The pathway to accomplish this target is not explicitly stated in the Agreement. Hence, the choice of national mitigation strategies depends on national energy system configurations, natural resources, infrastructure, access to mitigating technologies and economic and political structures. The most popular mitigation pathways proposed, are increased use of renewables, improving energy efficiency, and modifying consumption patterns. In addition to zeroing the GHG emissions, most integrated assessment models (IAMs) suggest that large-scale removal of CO$_2$ from the atmosphere is essential to achieve the 2 °C target [1, 11-15]. These climate models show that by removing CO$_2$, global warming can be held well below 2 °C above pre-industrial levels during the 21st century and return to below 1.5 °C by 2100.
Several negative emission technologies (NETs) have been proposed to remove CO$_2$ from the atmosphere. There is a wide range of estimates of the extent of NETs required to achieve the 2 °C target. According to the IPCC Fifth Assessment Report (AR5) [1], over 100 of the 116 scenarios associated with concentrations between 430–480 ppm CO$_2$ (<+2 °C target) rely on removal of around 5–20 Gt CO$_2$-eq annually, starting from mid-century—see Figure 2-1.

Azar et al. [16], found that the 2DS requires reduction in atmospheric CO$_2$ concentration by 0.5–1 ppm a year, which would be delivered by a NETs contribution of 8–16 Gt CO$_2$/year. The Paris Agreement calls for net zero emissions in the second half of the 21st century. However, considering the current INDCs the near-term global emission is likely to increase [8]. This is shown in Figure 2-1; as seen, the majority of 2DS trajectories net emission is likely to peak during 2020-2030 before declining. A carbon-cycle and climate model developed by Rogelj et al. [17] showed that pathways with NETs make the emission reduction after this peak period achievable at a higher rate.

In IAMs each atmospheric CO$_2$ concentration level implies an economic cost. As the emission target becomes more stringent, a higher economic cost is likely to be imposed. One way to illustrate this is to model the CO$_2$ price required for each of these concentration targets. Most IAMs focus on the least-cost pathways to stabilise GHG emission [17] and they attribute an essential role to NETs; Mainly because NETs makes achieving representative concentration...
pathway corresponding to 2DS (RCP 2.6) by the end of century attainable at a substantially lower cost [6]. In another study by Lemoine et al. [15], the effect of different policy options on optimal emission paths and policy costs was assessed. The results showed that making negative emission strategies available, enables an 80% reduction in the cost of keeping CO$_2$ concentrations in the year 2100, near their current level (~ 400 ppm).

2.1.1 Negative emission technologies (NETs)

Several negative emission technologies have been proposed to remove CO$_2$ from the atmosphere. Below are brief descriptions of the major NETs discussed in the literature.

Bioenergy with carbon capture and storage (BECCS) — the CO$_2$ absorbed from the atmosphere during biomass photosynthesis is captured and geologically stored when biomass is burnt or gasified for energy production or converts to biofuel [6, 7, 11, 12, 18-20]

Direct air capture of CO$_2$ from ambient (DAC) — uses chemicals such as amines to capture CO$_2$ from the air and store it geologically or mineralise it for solid storage [6, 7, 11, 12, 18-20]

Afforestation/reforestation and forest management (AR) — planting trees which capture CO$_2$ from the atmosphere to grow and store it as organic carbon [6, 7, 11, 12, 18-20]

Biochar — fixing organic carbon by combusting biomass in a low oxygen environment in a slow pyrolysis process [6, 11, 12, 18-20]

Ocean fertilisation (OF) — adding nutrient such as iron, phosphate or nitrogen to the surface water so to increase ocean CO$_2$ uptake through phytoplankton photosynthesis, and eventually increase the biotic ocean carbon [6, 11, 12, 18-20]

Enhanced weathering of minerals (EW) — adding silicate minerals to soil in order to increase CO$_2$ absorption from the atmosphere by accelerated natural carbonation process [6, 11, 12, 18-20]

Soil carbon sequestration (SCS) — enhancing the sequestration of carbon in soil by soil management techniques such as no tillage, which reduces the loss of carbon through oxidation [6, 7, 11, 12, 18-20]
2.1.1.1 NETs technical potential

A wide range of estimations of the NETs potential to remove atmospheric CO$_2$ can be found in the literature. Most of the studies count a higher negative emission potential for BECCS, AR and DAC than other NET options [7, 11, 18, 19]. According McLaren [19], BECCS with up to 10 Gt CO$_2$, AR with up to 3 Gt CO$_2$, DAC with more than 10 Gt CO$_2$, biochar with up to 3 Gt CO$_2$ and ocean fertilisation with around 1 Gt CO$_2$ offer the highest negative emission potential. A study by Smith et al. [11] showed a higher technical potential up to 12 Gt CO$_2$ per year in 2100 for BECCS, AR and DAC. An estimation by Fuss et al. [7] confirmed this range, with BECCS, AR and DAC each having the highest negative emission potential of up to 12.1 Gt CO$_2$/year in 2100. Koelbl et al. [9] estimated that BECCS could offer 10 Gt CO$_2$/year negative emission in 2050 and around 20 Gt CO$_2$/year in 2100, which is multiple the potential of AR with only up to 4 Gt CO$_2$/year. A more conservative estimation suggests less removal capacity for BECCS, 3.3–7.5 Gt CO$_2$/year [19].

The cumulative technical potential of the NETs is higher than the negative emission required for 2DS. However, Gasser et al. [21] found that in the absence of other mitigation strategies, the need for negative emissions would rise beyond its technical capacity. They emphasised that NETs are unlikely to be the “panacea” and achieving 2 °C target demands significant contribution of conventional mitigation methods as well.

Despite the promise of large scale CO$_2$ removal, negative emission options still encounter uncertainties regarding their socio-political responses, techno-economic barriers and their impact on natural carbon cycle and the environment [6, 7]. Among several NET options only BECCS and AR have been included in most IAMs [11]. AR is the only NET that has been implemented at scale, though far less than its estimated negative emission potential [7].

2.2 Bioenergy with carbon capture and storage (BECCS)

Using biomass for energy production is seen as carbon neutral in that the carbon released to the atmosphere during energy conversion was first taken from the atmosphere during photosynthesis. However, in the case of BECCS this CO$_2$ is not released back into the atmosphere but is captured, transported and permanently stored in a suitable geological formation. In effect, a negative flow of CO$_2$ from the atmosphere to the subsurface is established - see Figure 2-2. BECCS is a bridging technology between bioenergy and CCS technologies. There are a variety of options for BECCS to produce electricity or biofuel. This
study focuses on BECCS through bioelectricity. BECCS uses different biomass resources such as energy crops, agricultural and forestry residue or organic waste. The energy conversion could be through combustion, gasification or fermentation. For each of these conversion technologies a compatible CO₂ capture process (drying, pre or post or oxy-fuel combustion) needs to be applied.

The application of bioenergy combined with carbon capture and storage (CCS) as a negative emission technology was first identified by Herzog in 1996 [22]. A range of terms were used for this combination until 2003, when it was labelled as BECCS by Kraxner et al. [23]. Over the last decade there has been a noticeable increase in BECCS recognition and also in its advancement. Kemper [5] considered a decade ago, that BECCS was merely a costly sub-category of CCS which would only be feasible at small scale. Now, BECCS has found a more coherent narrative with dedicated chapters in reports by the IPCC, IEAGHG, the Bio-CCS Joint Task Force (ZEP / EBTP) [1, 24, 25] and close to 300 scientific journal publications [26].

![Figure 2-2: Bioenergy with Carbon Capture and Storage (BECCS) [18]]
BECCS is the most common NET included in IAMs and is used widely in emission scenarios [8]. Unlike other NETs, BECCS offers a two-fold advantage of delivering negative emission and providing carbon-free energy. Smith et al. [11] estimated the net energy supplied by BECCS could be 170 EJ/year in 2100, compared to other NETs which are net consumers of energy.

Another advantage of BECCS is the possibility of irreversible CO$_2$ storage. Inherent susceptibility of terrestrial carbon stocks to disturbance such as wildfires (350 Mha burnt per year globally [7]) makes sequestration by other land-based NETs such as AR and SCS less reliable [6].

Compared to other NETs, BECCS has the most immediate potential and there have been several pilot-scale demonstrations [18]. However, successfully commercializing and deploying large-scale CCS underlies BECCS deployment at scale.

The role of BECCS to achieve ambitious emission targets is a controversial subject. For instance, van Vuuren et al. [27] found that excluding BECCS could make the 2SD target out of reach. On another study, Edmonds et al. [28] applied a multi-aspect model to assess the role of BECCS in limiting RCP2.6 by 2100. They showed that whilst BECCS is not essential to reach 2SD it would reduce cost of achieving it [28].
2.2.1 BECCS potential to deliver negative emission

A wide range of views on the negative emission potential of BECCS can be found in the literature, ranging from 1000 EJ/year to 100 EJ/year and with a CO₂ removal capacity of 0–20 Gt CO₂/year [5]. The majority of IAM models have considered BECCS in the portfolio of essential mitigation technologies, with a removal potential of 2–10 Gt CO₂/year [6]. This value is comparable with the CO₂ removed by the natural carbon cycle, such as the ocean (9.2 ± 1.8 Gt CO₂) and terrestrial carbon sinks (10.3 ± 2.9 Gt CO₂) [6]. According to Gasser et al. [21], a “best-case” to achieve the RCP 2.6 target in addition to conventional mitigation would require BECCS with annual negative emissions of 1.8–11 Gt CO₂. In recent IEA global models, BECCS could potentially deliver negative 14 Gt CO₂ between 2015–2050, of which 11 Gt CO₂ is captured from biofuels with CCS and 3 Gt CO₂ from dedicated and co-firing BECCS for power [29]. A review study by Kemper [30] found the global technical potential of BECCS, through biomass gasification and direct combustion, to be around 10 Gt CO₂/year in 2050. Woolf et al. [31] estimated a lower global net negative emission of 3.3–7.5 Gt CO₂/year. Ricci and Selosse [32] used the multiregional TIAM-FR optimization model to assess the global and regional potential of BECCS. Their study showed that by 2050, BECCS and CCS could generate 23% to 30% of total global electricity, equivalent to 5.7–7.6 Gt CO₂ captured and stored. Most of this projected capacity lies in developing countries, especially China, India and Brazil. In a complementary study, Ricci and Selosse showed that a near-term widespread adoption of CCS with 15% BECCS, would be the preferable pathway to achieving stringent emission targets [33]. In a study by Koornneef et al. [34], the economic potential of BECCS would be up to 3.5 Gt CO₂-eq/year negative emissions from the power sector and 3.1 Gt CO₂-eq/year in transportation. However, these potentials are not for the whole sectors but for the "best" routes, i.e. BIGCC-CCS and FT biodiesel in 2050. An assessment of the assumptions underpinning the feasibility of BECCS in IAM scenarios by Vaughan and Gough [35] showed that assumptions regarding technical aspects of BECCS are realistic. However, their results warned that the socio–political assumptions and projections of the future large-scale deployment of bioenergy are unrealistic [35].
2.2.2 Bioenergy for BECCS

The technical potential of BECCS strongly depends on regional biomass resources and access to geological storage sites. So far, limited access to adequate biomass feedstock and high transportation costs has constrained the capacity of the current bioelectricity plants to 1–100 MW [36]. Bioelectricity with 525 TWh (≈ 1.9 EJ) constitutes only 2.3% of global power sector [37]. Bioelectricity is currently generated in 62 countries around the world. The U.S (69.1 TWh), Germany (49.1 TWh), and Brazil (32.9 TWh), have the highest share of bioelectricity in their power sector [38, 39]. Hence, to ensure the economic viability and technical feasibility of widespread deployment of BECCS, a significant upscaling of bioelectricity plants with high efficiency is crucial [40]. The share of bioenergy in the world’s electricity generation is expected to increase to 4.1% by 2035 [41, 42]. Based on the IEA bioenergy roadmap, global bioelectricity generation will grow to 3,100 TWh by 2050 [41]. That would require 510 GW of bioelectricity capacity. IEA forecasts that 10% of this capacity will be equipped with CCS to produce negative CO$_2$ emission [41].

In the 1.5 °C and 2 °C scenarios modelled by Schaeffer et al. [43] approximately 150 EJ/year of biomass in primary energy supply will be required by 2050, of which 50% would be used in BECCS. In these scenarios, bioenergy used for BECCS by 2100 will increase to 150–200 EJ/year. Scenarios in the Special Report on Renewable Energy Sources and Climate Change Mitigation (SRREN) [44] showed a total potential of 200–400 EJ/year from mid-to-end of century [16]. They concluded this much bioenergy if utilised with BECCS, could remove up to 9.2 Gt CO$_2$/ year.

The challenge of scaling up bioenergy at the level required in a 2 °C scenario is to produce sufficient sustainable biomass while maintaining the balance with food and fibre production. In addition, bioenergy expansion must act in accord with technical and economic development, social benefits and policy/regulatory regimes [45]. There are various estimates of the technical potential of bioenergy depending on the assumptions used in the models. Kemper [30] addressed the lack of standard methodology and the effect of climate change as the main reasons for the large variations in estimating the global potential of bioenergy. Sustainability and socio-economic constraints are the main sources of uncertainty.

The technical potential is a general term used to express how much bioenergy would be accessible considering the availability of productive land area (ha), annual yield
(tonne/year/ha) and energy content (MJ/tonne) of the biomass. The degree of sustainability of biomass production is usually defined using the following indices [34];

- **Strict**: land occupied by expanding nature reserves and all lands with severe and mild risk of water scarcity is excluded; no land use change is acceptable.

- **Mild**: new nature reserves and water-scarce areas are excluded.

- **No criteria**: no sustainability criteria are taken into account and all the productive land is used for bioenergy production.

Smith and Bustamante [46] illustrated the degree of uncertainty in the literature regarding global bioenergy potential in 2050 (Figure 2-3). The total potential with a high degree of agreement is around 100 EJ/year, with dedicated energy crops and forestry/agricultural residues estimated to have the highest potential (25–675 EJ/year). An estimation by Creutzig et al. [47] confirmed a sustainable technical potential of bioenergy of up to 100 EJ/year in 2050. In a study by Koljonen et al. [48], under scenarios with sustainability constraints, global potential of bioenergy is between 200 and 500 EJ/year by 2050, which will cover up to 50% of projected world primary energy demand. Van Vuuren et al. [27] showed that the sustainable potential of bioenergy in 2050 may vary between 0 and 200 EJ/year with a median value of around 150 EJ/year. In a study by Haberl et al. [49], bioenergy potential varies from 30–1000 EJ/ year by 2050 which sustainability constraints reduces it to 160-270 EJ/year. Kemper’s review suggested a range of 80–190 EJ/year as the most likely range for global bioenergy by 2050 [30]. Chum et al. [50] found that if dedicated biomass plantations were established in the available lands, 26 to 116 EJ/year could be produced. Azar et al. [51] assumed bioenergy availability of 200 EJ/year. Long et al. [52] suggested that global bioenergy potential varies between 100 to 400 EJ/year. Updated figures from AR5 [1] suggested that bioenergy could play a significant role in the energy system, providing 10–245 EJ/year in 2050, and 105–325 EJ/year in 2100. Selosse et al. [33] found that in the absence of BECCS, bioenergy still will have to make up a considerable share of power portfolio to meet the low emission targets. According Koelbl et al. [9] with no BECCS in the technological portfolio, biomass will dominantly be used in the biofuel sector.
2.2.2.1 Net primary production

Estimations of total bioenergy potential could be put into the context of net primary production (NPP). NPP is the total solar energy that the earth stores in plants in the form of carbon uptake through photosynthesis [11]. The NPP of the earth at present is about 2,200 EJ/year [49, 53]. The current human exploitation of NPP is around 300 EJ/year, of which approximately 230 EJ/year is used for food, animal feed, fibre and energy and the 70 EJ/year is lost during harvest or burnt in anthropogenic field fires [53]. Haberl et al. [49, 53] showed that considering the maximum biospheric capacity of NPP, the upper biophysical limit for primary bioenergy is about 190 EJ/year in 2050. It is also estimated that the maximum potential for bioenergy from organic residues across municipality, agricultural and forestry sectors is limited to 60 EJ/year [49, 53-55]. Therefore, any projection of bioenergy potential higher than 250 EJ/year (20–30% of global primary energy demand) exceeds the biophysical limits [53]. It is worth noting that is the natural upper limit of harvestable bioenergy, is further constrained by technical, economic, environmental and social complications.
Photosynthesis is an inefficient process with less than 5% efficiency at today’s CO₂ concentration of 400 ppm [56]. For instance, fast-growing sugarcane on highly fertile land in Brazil, converts only around 0.5% of incoming solar radiation into sugar. For maize grown in Iowa, the energy conversion rate is around 0.3% into biomass, of which around 50% is carbon on dry basis [57]. Every year around 0.02% of solar energy received on earth leads to fixing approximately 53.6 Gt C/year in plants, of which approximately 56% (30 Gt C/year) is in the aboveground portion of plants [53, 58, 59]. The assigned 1.4–5.5 Gt C/year removed by BECCS in IPCC IAMs would be 4.5%–18% of the total NPP [11]. In other words, the upper limit of CO₂ removal capacity envisaged in these models requires BECCS deployment at a scale exceeding the total current human uptake of NPP.

### 2.2.2.2 Carbon neutrality

The fundamental assumption in counting the potential of BECCS to deliver negative emission is that biomass is generally a carbon neutral energy resource; the same CO₂ produced from biomass combustion will be absorbed by plants through photosynthesis [46, 50, 60, 61]. This has recently been challenged especially for long rotation biomass resources (e.g. from forestry) [62], which the rate of harvest might exceed the time required for the plant to be replenished [63]. In general, the rate of plants decomposition is often so high that most of NPP instead of assimilating in natural carbon sinks escapes to the atmosphere [64].

Zanchi et al. [65] argued that the concept of sustainable management does not always correspond to a concept of carbon neutrality. Biomass extracted from forests in which harvest is less than the net annual increment can still result in more GHG emissions than an alternative energy source within near-to-medium time horizons. Spatial and dynamic boundaries, functional units, reference systems, and the selection of methods for considering energy and material flows across system boundaries are some of the main parameters for evaluating net CO₂ emission of bioenergy systems [50].

McKechnie et al. [66] integrated lifecycle assessment (LCA) and forest carbon analysis to assess the total GHG emissions of forest bioenergy over time. Forest biomass has a much slower growth rate, typically with a 60 to 100 years carbon cycle [66]. Thus, removal of forest biomass if not balanced with its growth rate affects the organic carbon stock and net positive CO₂ emission. According to their study, annual or semi-annual harvesting of agricultural crops for bioenergy allows sufficient time for replenishment of the organic carbon. Johnson [67]
proposed a system to count ‘carbon-stock change’ which accounts for all the carbon stocked and lost through soil carbon, land-use change, and emissions and storage in biomass.

It is worth mentioning that in BECCS system using enhanced oil recovery (EOR) storage option the net negativity is more subject to scrutiny as it could result in more non-biogenic CO₂ emission.

2.2.3 Carbon capture and storage (CCS)

2.2.3.1 CCS technologies

In carbon capture and storage systems the CO₂ produced from combustion is captured from the flue gas, compressed and permanently stored in suitable geological formations. The selection of CO₂ capture technology depends on the type of plant and fuel [68-70]. Post-combustion capture via chemical absorption using monoethanolamine (MEA) is the most mature and widely used technique in the power generation sector [71]. Energy required for solvent regeneration in an MEA installation is estimated to be in the range of 10–15% [72, 73]. A study by Bui et al. [74] suggested that using advanced solvents combined with heat recovery from the boiler exhaust gas could reduce this efficiency penalty to zero.

Several means for CO₂ transportation such as road and rail networks, pipelines trunks and shipping facilities have been considered [68, 69, 75, 76]. According to Dennis et al. [68] parameters such as volumes of CO₂, distance between CO₂ source and storage and typology of transporting infrastructure available determine the best option for CO₂ transport.

Main options for geological formation are saline aquifers, depleted oil and gas reservoirs (EOR), un-mineable coal beds and basalts [68-70]. Among these, EOR is more mature but storage in saline aquifer is becoming a more favoured option in recent projects [68].

2.2.3.2 CCS role in achieving 2 °C target

A study by Cuellar-Franca and Azapagic [71] showed that adding CCS can reduce the CO₂-eq emission from power plants by 63–82% per unit of electricity generated, depending on the CO₂ capture option used [71]. CCS can deeply reduce emissions from the current fossil fuel dominant energy sector without risking the energy security [13]. Deep mitigation from the power sector with more than 1,950 GW of coal-fired plant would require large-scale retrofitting for CCS integration [29].
More recently and after the Paris Agreement, the role of CCS to achieve “well below 2 °C” target has become prominent [14, 29]. In 2DS pathways CCS delivers 94 Gt CO₂ emissions reductions by 2050 [29]. In a drastic upscaling, CCS could store 800 Gt CO₂ by 2100 [77]. This includes emissions from both fossil fuel-based power and BECCS.

In addition to the power sector, CCS provides the possibility to decarbonise emission-intensive industries such as cement, steel, fertilisers and gas processing [29, 77]. According to McCulloch [29] under 2DS, CCS captures around 29 Gt CO₂ from industrial sources. For most of these industries no other effective way to drastically cut their emission has been devised yet [13, 77].

According to Gasser et al. [21], a “best-case” to achieve the 2DS target would need 183–916 Gt CO₂ storage capacity. In a special IPCC report on CCS [70], the global storage capacity of geological formation is up to 2,017 Gt CO₂. A higher range of the global storage capacity between 100 and 10,000 Gt CO₂, with outliers up to 200,000 Gt CO₂ could be found in the literature [1, 32, 68, 78]. In its 2017 report, the global CCS institute estimated the total capacity of CO₂ storage in geological formation of 3900 Gt [77]. Considering these estimations, the quantity of CO₂ stored required to achieve stringent scenarios is well below the global storage capacity.

2.2.3.3 Global status of CCS

Despite its potential as a large-scale mitigation option CCS has not yet contributed to significantly reducing emissions globally at scale. To date, twenty one CCS projects with the overall capacity of 37 Mt CO₂/year are operating worldwide [75, 77]- see Figure 2-4. So far around 220 Mt CO₂ has been stored underground [77]. This is far less than the capacity required to make a significant contribution to the 2 °C scenario emission trajectory [29, 75]. To acquire the CCS rate set forth in climate models, many more CCS facilities are required. Large-scale implementation of CCS with permanent dedicated CO₂ storage began with Sleipner project in Norway in 1996 [79]. The largest CCS plant to date is Century liquefied natural gas processing plant in the U.S which is expected to transport approximately 8.4 Mt CO₂ per year in via a pipeline system to oil fields for enhanced oil recovery [29, 75, 77].

2.2.3.4 CCS challenges

CCS currently increases the levelised cost of electricity production by 45–70% [77]. However, it is a heteronomous technology involving several capture, transportation and storage options.
The cost of the CCS chain depends on several factors. Typically, the capture process for coal-fired power generation can constitute more than 50% to the total cost of a CCS project [77]. According to the CO2CRC [80] the capital cost of CCS is expected to decrease by 30%–50% by 2030 [81]. CO₂ transport and storage hubs can further reduce the cost of CCS by cutting the cost of infrastructure by sharing facilities [82].

Uncertain policy support and economic infeasibility have hindered the large-scale development of CCS [29]. Inclusion of CCS in the Clean Development Mechanism (CDM) could have a profound impact on its cost and proliferation [77]. Under current circumstances, deployment of large-scale CCS is not expected until well into the 2020s [71].

![Global map of CCS projects](image)

**Figure 2-4- Global map of CCS projects [77]**

### 2.2.4 Global status of BECCS projects

Globally there have been twenty BECCS projects, mostly located in North America, Europe and Scandinavia [30, 83, 84]. Currently five of these projects are operating, capturing CO₂ from ethanol production plants with a total capacity range of 0.1–1 Mt CO₂/year negative emission [85]. Five projects have been cancelled mostly due to lack of economic viability and the remainder are either completed or under evaluations/planning [30]. The BECCS projects under planning, use CCS coupled with a variety of bioenergy technologies such as waste-to-energy (in Norway and The Netherlands), ethanol plants (France, Brazil and Sweden), biomass
combustion/co-firing (Japan), pulp and paper (2 projects in Sweden), biomass gasification (the U.S) and a biogas plant (Sweden) [5, 30].

The large-scale deployment of BECCS took its first step in 2017, with the commencement of the Illinois Industrial CCS Project (IICCSP). The project has received 140 million US dollar in capital support from the U.S Department of Energy and will also be able to access CO₂ storage credits of USD 20/t CO₂ [30]. IICCSP was established in 2011 [86]. In this project the CO₂ released during the fermentation process to produce ethanol at the Archer Daniels Midland (ADM) ethanol plant in Decatur, Illinois, is captured, transported and stored in a deep saline formation in the Mount Simon Sandstone [86]. During its operational period from November 2011 to November 2014, the IL-ICCS project injected 1 Mt CO₂ into the subsurface. Since 2017 the IIICCSP project has increased the CO₂ injection rate up to 1 Mt CO₂/year [86].

Recently a waste-to-energy agency in the Oslo municipality (EGE) conducted a feasibility study to assess the opportunities for CO₂ capture from a waste incineration plant at Klemetsrud [87]. Technical assessments show that the project could capture up to 315,000 tonnes of CO₂ annually at a 90% CO₂ capture rate by 2020 [87]. Around 50%-60% of this CO₂ is biogenic. Klemetsrud provides both heat and electricity and currently produces 20% of the Oslo’s total emission. The feasibility study showed that absorption technologies including advanced amine solvent and chilled ammonia for CO₂ capture were technically viable. CO₂ transportation via tanker truck to Oslo Harbour for further shipment was deemed to be a better solution than pipeline. The plant will operate throughout the year. The project is part the roadmap of the Oslo climate target for 2030 and 2050 [87]. The project fits into the EU’s circular economy to reduce landfill of waste to a maximum of 10%. Energy recovery is seen to be the best way to deal with the portion of the municipal waste which cannot be recycled or reused, but including CCS is the only way to decarbonise this sector. Another waste-to-energy power plant with CCS is ARV- Duiven in Duiven, the Netherlands. ARV- Duiven power plant with 70 MW capacity incinerates MSW to produce around 126 GWh electricity. From 2018, ARV- Duiven is planning to capture up to 50 Kt CO₂ per annum using the MEA capture process [88].

Co-firing biomass with CCS has been deployed in two power plants. One is the Maasvlakte power plant 3 (MPP3) in the Netherlands which became operational in 2015. With a capacity of 1070 MW, MPP3 is capable of accepting up to 30% biomass; its CCS component is subject to a commercial decision [89]. The other project is the Mikawa power plant in Omuta in Japan
with 49 MW capacity, that has been recently retrofitted to co-fire biomass with CCS. This project is sponsored by Japan’s Ministry of the Environment [77].
2.3 Economic implications of BECCS

One of the important parameters, which constrains achieving the potential for removing CO₂ of the NETs is their economic performance. With no large-scale NET project, estimation of their cost merely relies on hypothetical situations in the future. Most studies agree that among three NETs with highest technical potential (namely BECCS, AR and DAC), DAC with up to $400–$600 /t CO₂ is the most costly option and BECC with $150–$250 /t CO₂ and AR with $10–$65 /t CO₂ have a lower cost [11, 18, 19, 90]. The scope of this study is limited to power generation BECCS options, but it is worth mentioning that a lower value of BECCS cost could be achieved at bioethanol refinery with less than $25/t CO₂ [91].

The high cost of these NETs, highlights the necessity of an effective mitigation mechanism such as carbon price. According to McGlashan [18] an effective carbon price in their definition should be more than to mitigation cost of the cheapest large-scale NET option. McLaren [19] foresaw that the suitable technical and economic prerequisites for NETs deployment at the level of 10–20 Gt CO₂ per year could become obtainable by 2030–2050. According to McGlashan et al. [18] the cost of a large BECCS power plant is $59–111/t CO₂. In a study by McLaren [19] a range of $70–$250/t CO₂ is suggested for BECCS [19]. This study indicated that in the absence of rapid development, the cost of BECCS is more likely to be $150/t CO₂ [19].

Although the future technical potential of BECCS is estimated to be considerable, not all of it will be economically viable. One of the major contributions of BECCS would be in the power generation sector. The current electricity market is deeply dependent on fossil fuels and there is no united price for CO₂ emission. As a result, the high production cost of BECCS, considering its high investment and O&M costs, would not be competitive in the power market. Based on a review by Kemper [30], the levelised cost of electricity production (LCOE) through BECCS lies between $70–$230 /MWh.

2.3.1 Role of BECCS in achieving an economic cost for a 2 °C target

To have a clearer picture of how including BECCS in the portfolio of mitigation technologies influences the cost of achieving different emission targets, detailed models to analyse the impact of energy demand, technical potential and negative emission of BECCS, land-use change emission, availability of other abatement options and discount rate are required. Three of the most frequently mentioned models covering these criteria are listed below;
TIMER/IMAGE: assumes long-term dynamics of the energy production and allocates a large market share to low costs primary energy carriers and a small share to high cost ones [92, 93].

MESSAGE: calculates the cost minimal supply under constrained resource availability, given technologies and demand for energy; energy demand, supply, and emission patterns are taken into account [94, 95].

GET: chooses the primary energy source, conversion technology and transportation, which meets a price elastic energy demand at the lowest aggregate cost subject to carbon constraints (tax or emission cap) [96, 97].

All these are linear programming models. Azar et al. [16] compared the cost at the net present value of future annual cost to meet the stabilization target of these three models- using a discount rate of 5% in 2100. Furthermore, it is assumed 200–400 EJ/year bioenergy is used from mid-to-end century. Their study showed that in all the three models BECCS makes attaining low stabilization emission targets possible with a lower cost than can be achieved with no BECCS scenarios. The impact of BECCS on mitigation costs to achieve a level of 450 ppm in 2100 varies between 1-6 trillion US dollars across the models. The difference mainly lies in their initial assumptions. In the GET model, BECCS makes very low target emission like 300 ppm achievable by the end of century. The models show that the influence of BECCS only becomes significant in low emission targets (< 450ppm), and as the target exceeds 500 ppm the effectiveness of BECCS tends to decrease. In these models BECCS reduces the cost of achieving 2SD target by 200%–1200%. However, unlike IAM models in AR5, BECCS is not essential for this target in any of these three models. These models give an indication of how BECCS could lower the cost of ambitious mitigating targets but do not indicate the timeline for BECCS large-scale implementation.

Climate sensitivity and discount rates assumed in such climate models significantly affect the cost of mitigation. The climate sensitivity is the ultimate change in global mean temperature in response to a change in radiative forcing [98]. Across the numerical global climate models (GCMs) a wide range of 0 °C to 10 °C for doubling of atmospheric carbon dioxide concentration has been reported [98, 99]. Probability distribution models show a climate sensitivity between 2 °C and 4.5 °C as more probable [51, 99].

Discount rate determines the well-being (including of environmental services) of the future generations compared to the current generation. Zero discount rate means same level of welfare.
for all generations. Whereas, a positive discount rate projects a reduced welfare for future generations. Choosing the discount rate in climate models in addition to being an economic parameter is an ethical choice. Stern [2] challenges imposing high discount rates in models which outcomes could have decisive long-term effect on policy making and mitigation pathways. Hence, he recommends annual discount rate of 1% in the Stern Review’s model [100]. Nordhaus suggested higher discount rates and argued that Stern’s 1% is unrealistic and is based on vague assumptions [101]. A discount rate below 3% is seen as a low risk assumption [99]. Any positive discount rate adds to the economic value of BECCS, as it postpones the costs of mitigation, presenting a “discount” opportunity [16, 30]. In an assessment by Azar et al. [51] the discount opportunity of BECCS for achieving 2SD is around 0.2% of GDP (gross domestic product).

2.3.2 Emission Overshoot

One of the controversies regarding NETs is the “moral hazard” of delaying radical mitigation programmes with the excuse of “undoing” it later [18]. Most of the IAMs rely on availability of large-scale NETs, mainly BECCS and AR, both in the overshoot scenarios and in the scenarios without overshoot when negative emission is to compensate for residual emissions from sectors where mitigation is more expensive [1, 35, 102, 103]. Anderson and Peters [8] reliance on BECCS in the majority of IAMs is due to the high discount rates assumed in these models. More delay in radical mitigation means more reliance on NETs in the future [9]. Lemoine et al. [15] argued that the prospect of large-scale deployment of NETs diverts R&D focus and funding from “carbon-free” technologies into “emission intensity” technologies [15]. That escalates the risk of surpassing the natural capacity to limit catastrophic climate changes. Anderson and Peters [8] concluded that “negative-emission technologies are not an insurance policy, but rather an unjust and high-stakes gamble”. Stern argued that delay in urgent actions for drastic cut in GHG emission significantly escalates the cost of mitigation in the future. To avoid that, Stern recommends 30% to 70% mitigation before 2050 [100, 104].

Kartha and Dooley [54] identified unfeasibility, under-delivery of negative emission, unacceptable environmental and social consequences and ineffectiveness in controlling emission as the main risks associated with large-scale negative emission technologies, such as BECCS. However, their study suggested pathways with reliance on smaller level of BECCS as more viable.
In the majority of other IAM models, overshooting is allowed. In other words, the total emission must be stabilized at the target defined, but it could exceed the limit for a period of time and be compensated by negative emission afterwards. So, the least cost trajectory after introducing BECCS is pushed towards less abatement in the near-term and more in the future and that allows for emission overshoot in near-term. Azar et al. [51] used GET to model the effect of BECC on the cost of mitigation in overshoot and ceiling (no overshoot is permitted) scenarios. The model assumes climate sensitivity of 3 °C per doubling of CO₂ ppm, bioenergy availability of 200 EJ/year and storage capacity of 2000 Gt CO₂. Abatement cost is defined as total discounted cost of temperature-target scenarios compared to unconstrained “business-as-usual” reference case. In these scenarios the cost of achieving a 2 °C target with BECCS is up to 1.3 % of GDP [51]. In the overshoot scenario, the impact of BECCS becomes more prominent for a 2 °C target, saving up to 1% of GDP [51]. For the ceiling targets BECCS has only a marginal impact at more stringent temperature targets. Because in the ceiling scenario the temperature is restricted and there is no value in negative emissions as the target is met. However, BECCS is still in the portfolio to compensate for emission from the fossil fuel-based sector and its role becomes crucial when such emissions are very difficult to eliminate.

The Stern Review’s model estimated the overall costs and risks of climate change in a business-as-usual scenario between 5% to 20% of global GDP each year [100]. Muratori et al. [102] warned about the possible effect of BECCS deployed at scale on global GDP, due to altering the financial flow from energy and agricultural trade patterns.

2.3.3 Carbon Price

As the main incentive for BECCS is its capacity to produce negative emissions, the avoided cost of CO₂ emission determines its economic potential compared with other technologies on the market. Therefore, one of the decisive factors to estimate the cost of BECCS is the carbon price. A carbon price has been identified as the most effective policy mechanism to support BECCS deployment [102]. Stern recommended a high carbon tax of $300/t CO₂ for ambitious climate scenarios [100, 104].

Many IAMs show that inclusion of CCS and BECCS in the mitigation portfolio reduces the CO₂ price needed to attain stringent emission targets [16, 28, 30, 34, 102, 105, 106].

Muratori et al. [102] used the Global Change Assessment Model (GCAM) to explore the global and regional economic impacts of BECCS [102]. According to this study, to achieve 2SD
target, a carbon tax is an effective policy mechanism, under which mitigation would be an economic burden and BECCS would entail a net subsidy [102]. If as portrayed in the Paris Agreement, zero net emission in the second half the 21st century would mean subsidies to BECCS that would equate revenues from emission.

Luckow and Wise [105] found that without CCS to stabilize CO\textsubscript{2} concentration at 450 ppm by 2100, the carbon price would be more than 700 $/t CO\textsubscript{2}, but reduces to around 350 $/t CO\textsubscript{2} when CCS is in the portfolio. They reported that at carbon prices above $150/t CO\textsubscript{2}, over 90% of biomass in the energy system is used in combination with CCS [105]. The model used in this study is the global change assessment model (GCAM) which determines the least-cost pathways for allowable emissions. According to Haszeldine, in the presence of large-scale CCS deployment the cost of CO\textsubscript{2} abatement to achieve net zero emission by mid-century will reduce from $500 to $50 per tonne [107]. In a study by Koornneef et al. [34], with a CO\textsubscript{2} price of 50 Euro/t CO\textsubscript{2}, the economic potential of BECCS would be up to 20 EJ/year for power generation and about 26 EJ/year for biofuel in the transportation sector. They used a discount rate of 5% in these models which due to the high capital and operating and maintenance costs of BECCS, has an important impact on the outcomes. Rhodes and Keith [106] found a much lower carbon price of $34 /t CO\textsubscript{2} for the economic competitiveness of BECCS in the electricity market [106].

The majority of scenarios in AR5 assume a carbon tax of around $100 /t CO\textsubscript{2}. According to most models [16, 28, 105, 106] this would be sufficient to facilitate large-scale BECCS deployment.

To put that in a cost-and-benefit context, Ackerman and Stanton [99] estimated the social cost of carbon up to $1,500/t CO\textsubscript{2} in 2050. That is much higher than the cost of mitigation required in most ambitious mitigation targets including BECCS at scale. This implies mitigation costs are much lower than the cost of damage due to each tonne of CO\textsubscript{2}.

These models assume a globally integrated policy and price on CO\textsubscript{2}. Although some developed countries have levied a tax on CO\textsubscript{2} emission, there is no integrated price for CO\textsubscript{2}. The European emission trading system (EU ETS) [108] includes 31 European countries and covers a wide range of emitting sectors. The current carbon price under EU ETS fluctuates around 8.5 Euro per tonne CO\textsubscript{2} [109], which is far less that the level required for sharp abatements. Lack of effective and wide-spread carbon price would hinder BECCS large-scale deployment and limit it to niche applications.
2.3.4 International market for biomass and bioenergy

Unlike other forms of energy, especially fossil fuels, there is no international market for biomass. One reason is that at present most of the biomass (around 75%) is being used in traditional ways (cooking, heating) in rural areas [110]. Consequently, biomass trade is mostly local. Another reason why the international market for biomass is difficult to develop is the characteristics of biomass. Biomass has lower energy density and bulk density than conventional fossil fuels. This profoundly affects the cost of its storage and transportation. Methods like pelletisation and torrefaction have been devised to enhance the energy density of biomass [111, 112]. The international market for bioenergy is limited to biofuel (ethanol and biodiesel) and has grown from almost zero in 2000 to 120–130 PJ in 2009 [113]. The U.S and Brazil dominate the ethanol market with a 90% share in production, while the EU covers 60% of biodiesel production [50]. Both in the EU and the U.S, supportive policies have made the domestic market more profitable. International trade is more economically viable for the exporting countries with lower production cost such as Argentina, Malaysia, and Indonesia for biodiesel and Brazil for ethanol [110]. The main policies to create a bioenergy market can be through targeting domestic consumption, production and import/export by introducing promotion or mandates like taxes, subsidies and tariffs [114].

2.3.5 BECCS and carbon lock-in

In recent decades, measures such as large-scale deployment of renewables, technological advancement and socio-political changes are undermining the interrelation between economic growth and fossil fuel exploitation [115]. However, having a plentiful supply of accessible fossil fuels as the main energy resource, has kept the price of electricity and fuel low in most countries. This has created inertia in the energy market and a reluctance to invest in cleaner but more expensive technologies. Historically, this inertia has fundamentally influenced the energy system configuration and infrastructures, interrelatedness of technologies, legitimation and learning effects of users, producers and regulators [116]. This profound role of fossil fuel in human development is known as a carbon lock-in [116, 117]. There are concerns among experts about how CCS may influence carbon lock-in. Vergragt et al. [116, 118] suggested that large-scale CCS might lead to perpetuating fossil-fuel base power industry by 2100 and consequently delay development of other mitigation pathways. This effect of CCS could potentially increase cost of fundamental changes in the energy sector.
By displacement of existing fossil fuel energy sources and delivering net removal of CO$_2$ from the atmosphere BECCS could be a way to weaken this effect of CCS [14, 119]. This transition could occur both in the electricity and transportation sectors. For instance, dedicated firing or high percentage biomass co-firing with coal would be a way to combine fossil fuel CCS and BECCS and thus enhance the infusion of renewables in the energy sector [116]. Producing biofuel for transportation is another way to expanding CCS utilization to non-fossil fuel resources. BECCS can be a source of hydrogen production. In addition to electricity generation, hydrogen could fuel the electric transportation which is another way of promoting the changes towards a fossil-free transportation sector [120, 121]. BECCS has the potential to be a sustainable bridging technology, by providing an immediate negative carbon pathway while other carbon-free energy technologies are developing.
2.4 Environmental impacts of BECCS

There is great deal of uncertainty regarding the possible environmental impacts of large-scale NETs. The major impacts found in the literature are land use change, water use, energy input, effect on albedo and natural carbon cycle [7, 11, 12, 54, 122, 123].

Sustainability of biomass feedstock production is one of the main sources of uncertainty in estimating the global technical potential of BECCS. Climate dynamics, economic and technological development, human population growth (and its effect and demand for food, fodder and fuel) and natural carbon cycle are some of critical uncertainties influencing the future potential of BECCS [5, 14, 30, 46, 124-126].

Historically, unsustainable biomass harvest and forest clearing in some areas, has led to loss of a considerable proportion of natural forests and degradation of productive lands [46, 50, 127], increased GHG emissions, loss of biodiversity and carbon stock [128-134] and depletion of water resources [128-130, 135]. Intensification of energy crops production could result in severe competition between food, feed, and energy feedstock supplies, leading to controversial economic, ethical, and environmental issues [5, 125, 136].

If strategies for large-scale BECCS deployment do not meet strict sustainability criteria, its negative externalities could be very problematic. Therefore, expanding bioenergy production must be carefully considered against the background of sustainability [126].

2.4.1 Land-use

Increasing demand for biomass and limited arable lands may cause expansion of agricultural lands, loss of natural forests and ecological reserves. Land-use change includes; direct (LUC)- where the land encroaches on neighbouring forests or reserves, or indirect (iLUC)- where the land previously cultivated for food production is used for energy crops and some other natural forests is substituted for food agriculture [45, 46, 50, 127].

Unsustainable expansion of bioenergy production will exacerbate the GHG emissions from agricultural and forestry systems. Emissions from agriculture, forestry and other land-use change (AFLOU) account for approximately 10 to 12 Gt CO$_2$-eq/year of anthropogenic GHG emissions; with 5–5.8 Gt CO$_2$-eq/year from agricultural production and 4.3–5.5 Gt CO$_2$-eq/year from land use and land-use change activities [46]. According to Le Quéré et al. [137] emissions
related to land-use change represent about 9% of the total emissions in the last decade. These emissions were principally associated with deforestation and expanding agricultural land use.

GHG emission in agriculture is due to land-use change, fertilizers, livestock and fossil fuelled machinery used. With present practices, the agriculture sector is one of the major contributors to global warming. Agricultural activities account for approximately 58% of the total N2O and 47% of the total CH4 emission [136].

Net forest emission is the difference between GHG emissions gained and emitted from forest land. CO2 emission lost is due to oxidisation of carbon stock in biomass as a result of conversion of forest land to pasture or agricultural land. CO2 gained occurs by fixation of carbon through increasing the forest land. The net emission from forest land conversion in 2011 was 3.74 Gt CO2-eq which was almost 70% of the emission from the agricultural sector. Burning forest is another source of CH4 and N2O emission from combustion of biomass and organic soil. In 2011 the total GHG emissions from burning forests was 290 Mt CO2-eq [138].

Tubiello et al. [139] found that land-use emissions have remained stable at about 4.8 Gt CO2-eq/year, whereas emissions from agriculture has kept growing by approximately 1% , to 5.4 Gt CO2-eq/year.

2.4.1.1 Land available for bioenergy expansion

The area of land needed for bioenergy production depends on the productivity of the land, efficiency of production practices and the type of biomass. The estimates in the literature of the area available for bioenergy production varies from 60 to 3700 Mha [49]. Some scenarios of future bioenergy supply have suggested that land devoted to energy crops could equal the current land used for crop cultivation [140]. Souza et al. [141] estimated the global potential of modern bioenergy in the range of 80-200 EJ would require 200–500 Mha land. The European Energy Agency estimated the potential for sustainable bioenergy production of approximately 340 EJ/ year in 2050 [7], which according to Spiertz et al. the area of land needed to produce energy crops for this much bioenergy, would be 250 Mha [142]. Azar et al. [16] showed that with an annual yield of 10 t/ha over about 500 Mha area, the total potential of 200–400 EJ/ year from mid-to-end of century is achievable. This much bioenergy if utilised through BECCS, could remove up to 9.2 Gt CO2 / year. According to Fajardy et al. [124], in order to deliver around 12.1 Gt CO2 / year through BECCS, between 363 and 2400 Mha marginal lands will be required. In another study, Smith et al. [11] estimated a smaller land
area of 380–700 Mha would be needed to deliver 12.1 Gt CO$_2$-eq/year negative emission. In this study, they found that depending on the source of biomass, land-use intensity of BECCS varies from 0.03 to 0.46 Mha/Gt CO$_2$-eq per annum. A model developed by the U.S National Research Council (NRC) [143] suggested that to provide 100 EJ/year (with mean annual yield of 10 t/ha), up to 500 Mha land is required.

Deng et al. [144] estimated the global bioenergy potential by 2070 of between 40–190 EJ/year with 35–160EJ from energy crops grown in abandoned cropland and grassland. They found that to produce this amount of bioenergy 370–1320 Mha (3- 10% of global land area) is required. Current agricultural lands are 12% of total land [144]. Long et al. [145] estimated that capacity of energy crop production in “sustainable land-use” is only about 16 EJ.

Kemper [30] concluded that lack of sufficient data, inconsistency in models, assumptions, definition and interlinked economic and ecological dynamics, has led to this wide range of estimations of land-use change due to bioenergy expansion.

2.4.1.2 Challenges of land-use expansion

From the total 13 Gha land on the planet, the land area suitable for crop production is 4.5 Gha of which only 2.7 Gha is available for agricultural [146]. Current land used for crop cultivation is approximately 1500–1600 Mha, or 12% of the total land [140]. Of the land available for expanding bioenergy crops, 800 Mha of forest, 200 Mha of protected land and 60 Mha of human settlements must be excluded [136].

Another crucial limiting factor is land required to supply food and fibre for the growing population. According to the Food and Agriculture Organization (FAO) [147] with an average cereal yield of 3.6 t/ha, an additional 72 Mha will be required to meet the global food demand by 2050. In this estimate, population growth, diet change and improvements in agricultural productivity have been considered. A study by Tilman et al. [148] showed a much greater land expansion, of 890 Mha, to meet the food demand in 2050. The limitations on land-use is especially crucial in Asia due to intensified food production for the rapidly growing population [142]. It must be noted that because of accelerating land degradation and urbanization each year around 5–7 Mha of agricultural land are lost globally [149].

Most assessments for sustainable bioenergy crop production use abandoned agricultural land, degraded land, marginal land and waste land unused agricultural lands, which is estimated to
be between 320 and 580 Mha of low productivity land [136]. Chum et al. [50] argued that although using marginal lands for bioenergy production is often seen as an option to avoid land use change, its low productivity, long distance to bioenergy plants, loss of biodiversity and the competitive use of land by the local communities, could be a challenge. In addition, remoteness of marginal lands and their long distance from the centralised bioenergy facilities may result in logistic and economic challenges [150]. McLaren et al. [19] concluded that due to uncertainties and environmental impacts, future bioenergy potential should be restricted to zero land-use expansion and increases in productivity.

2.4.2 Water consumption

Total withdrawal of freshwater from aquifers, streams and lakes by humans is around 3,853 km$^3$/year [151]. Around 70% of this amount is used for agriculture and in some fast-growing economies this percentages is up to 90% [149, 151]. To obtain the potential of BECCS at scale, a considerable proportion of the available water resources might need to be dedicated to biomass production. Although there are some plants that grow with low water consumption and high resistance to drought, in most regions biomass production could take a significant share of the available fresh water [135]. In particular, expansion of bioenergy from energy crops may intensify pressure on water resources [30]. One of the expected impacts of climate change is changing the precipitation patterns, loss in soil moisture and water scarcity [152]. That will be especially a challenge in regions like sub-Saharan Africa, Middle East and western America, Mexico and Australia, which are already facing a water scarcity [128, 130, 152]. According to Creutzig et al. [47], water scarcity in areas such as Middle East, parts of Asia and western USA reduce the technical potential of bioenergy by 17% by mid-century.

Fajardy et al. [124] estimated that in order to deliver around 12.1 Gt CO$_2$/year, between 3.6 and 15.7 Tm$^3$ water will be required. Smith et al. [11] found that the water needed to deliver the same negative emission through BECCS in 2100 would be approximately 720 km$^3$.

A portion of the water consumption associated with BECCS is due to the CCS process. For instance, the water used for cooling in a MEA–based CO$_2$ capture unit is around 106 m$^3$/t CO$_2$ [153].

Water contamination is another ecological concern. Using fertilisers and pesticides for biomass production could be a source of water pollution. In addition, CO$_2$ leakage and its subsequent
environmental impacts on acidification of underground water is the major concern for geological storage [129, 130, 135].

2.4.3 Efficient biomass production methods

To meet the degree of biomass production needed for food and bioenergy demand and bearing in mind the limited natural resources and probable adverse impacts of climate change, a fundamental enhancement of the agriculture system seems essential. One way to do this would be to increase the efficiency without increasing environmental damage [152], which could potentially reduce emissions from this sector by 770 Mt CO$_2$-eq/year by 2030 [154].

Several options to increase the efficiency of agricultural production while mitigating its correspondent emissions have been proposed. The green economy and climate-smart agriculture are two of the main concepts contributing to this goal. Green economy “combines the concepts of economic efficiency and production efficiency in agriculture given the increasing scarcity of the natural resources” [152, 155]. Climate-smart agriculture (CSA) is a way to adapt agricultural practices under climate change in order to assure secure crop production [138, 147, 152]. Conservation agriculture (CA) is a CSA method which promotes practices to improve the mitigation and adaptation of agriculture to climate change through minimal mechanical soil disturbance (i.e. no tillage and direct seeding), crop rotation, restoring soil cover and bio-control of pest and weed [147, 156, 157].

Sustainable crop production intensification (SCPI) [152] is another method which employs efficient and smart use of crops, water and nutrients. One example is planting legume crops which helps to biologically fix the nitrogen in soil [152, 158]. This method increases production, improves the annual yield and reduces N$_2$O emission [152, 158].

Increasing soil carbon stock is a method to enhance the resilience and productivity of agricultural land [159]. According to Meybeck and Gitz [152] enhancing soil carbon reduces soil erosion and improves water retention. Perennial crops such as sugarcane are favoured on sites susceptible to soil erosion. Perennial plant is a crop that remains productive for more than two years. They offer multiple environmental services such as restoring degraded lands, soil conditioning and water retention [45, 47, 160]. In addition, nitrate leaching is lower on land cropped with perennial crops than annual crops [161, 162].
Agroforestry is another solution which offers environmental and economic benefits to the agricultural sector and adds to its resilience [152]. Planting trees and shrubs on parts of degraded agricultural lands enhances the soil conditions by infiltrating water and keeping soil moisture, lowering the water table and preventing soil erosion by acting as a barrier in case of flooding. The other benefit is growing trees as a source of wood for energy use, which means more income, increased carbon sinks and improved agro-biodiversity [147, 156, 157]. Agroforestry has been practiced on about 46% of all agricultural lands, especially in Southeast Asia and central and South America [147, 156, 157]. Under the Clean Development Mechanism (CDM) of the Kyoto Protocol, agroforestry through reforesting and afforesting is a way of generating carbon credits. Ethiopia and Namibia are two African countries that have agroforestry projects under CDM [147]. Another example of agroforestry is growing eucalyptus mallee tree in the southern regions of Australian known as the sheep-wheat belt [163], where the level of precipitation is low, at 300–700 mm/year. Growing mallee in these wheat belts has been shown to improve soil salinity and provides an extra source of income.

Landscape planning is a way to diversify the land-use with crop rotation and planting multi-purpose crops [50, 152, 164]. It helps to enhance the soil productivity and biodiversity. Contract multifunctional farming can provide an economic benefit, especially for small-scale farmers to sell part of their yield for bioenergy while sustainably intensifying their production [164, 165]. These methods could be merged into an integrated production system of food-fodder-bioenergy, which applies the no-waste concept [164].

Excessive use of fertilisers for energy crop production leads to N$_2$O emission, which has 300 times the global warming potential (GWP) potency of CO$_2$. Clarke et al. [166] estimated that additional nitrogen fertiliser to deliver BECCS at scale is around 73.5 Mt N/year in 2100 [30]. A simple and traditional solution is to minimise use of synthetic fertilisers by utilising livestock manure (which is otherwise a source of CH$_4$ and N$_2$O emission) as organic fertiliser, so to make the waste of one system an input to another system [152].

One of the most highly controversial methods is Genetic Modification (GM) [152]. GM is a double-edged sword which promises to intensify production with higher efficiency use of resources. However, it is argued that there are unknown risks of how it might affect the natural ecosystem [103].
2.5 Social impact of BECCS

One of the main challenges of expanding bioenergy from dedicated energy crop is to maintain food security and affordability. Land-use change and its likely impacts on social equity and land ownership is another challenge of bioenergy.

2.5.1 Food security and affordability

The current food production system is very inefficient. According to McKenzie and Williams [167] around 40% of the food produced is lost throughout its production to consumption. With current food production system, 32 countries are facing food crisis, with approximately 870 million people estimated to be undernourished and 1 billion malnourished [152]. Despite the current inefficiencies to cover food demand the food production has to grow by 60% to feed around 9 billion people in 2050 [125, 154, 168, 169]. Reducing food loss and change in diet are most likely to change the demand for food by 30–50% [49, 144]. However, in the absence of required advancements in food production and in light of limited land and water resources, this put even more constraints on expanding lands for energy crop cultivation; especially when energy crops substitute food crops or food crops are used for bioenergy production.

The impact of this change will be more severe in developing and underdeveloped countries where price of food constitutes a higher share of the income. A study by Popp et al. [170] showed that large-scale bioenergy deployment (up to 300 EJ from energy crops) could potentially increase the food price by 82% in Africa, 73% in Latin America and 52% in Asia Pacific by the end of century. This could be a source of inequity in access to food across the globe. This effect has already been observed in 2007–2008, when alongside other factors high bioethanol demand in fuel market in north America led to sharp increase in food price followed by food riots in many developing countries [171].

According to the IPCC AR5 [1] around 100 EJ/year can be supplied from agricultural and forestry residues, dung and organic waste. Lotze-Campen et al. [172] showed that deployment of this bioenergy will most likely increase the food price by 5%, whereas the direct effect of climate change on food price will be around + 25%. A study by Muratori et al. [102] confirmed that inclusion of BECCS in portfolio of mitigation technologies in a 2SD scenario would lessen the demand for biomass and thus the subsequent impact on food price.
Ferroukhi et al. [173] showed that bioenergy “as long as it is sustainably produced and managed” can offer social benefits to rural area by improving energy and food security and poverty alleviation through creating new market for biomass and wastes and also job creation.

2.5.2 Land tenure

Although it might not be a problem in most developed countries, land ownership in many regions with highly productive lands such as South American and Southeast Asian countries, is not always clear. Decision-making processes on whether to use land for food or energy could lead to local conflicts and inequity.

A negative socio-economic effect which may be associated with indirect land-use change (iLUC) and a shift of agricultural production towards marginal lands is the large-scale privatization of lands by the most capitalized producers which may lead to the displacement of smallholder farmers and of whole rural communities [174]. Land management and new efficient production methods such as integrated food-energy systems, crop production on abandoned or degraded lands and producing bioenergy from wastes or residues are ways to help solve this problem [173].
2.6 Conclusions

Using organic waste from municipal, agricultural and forestry sectors for BECCS is a way to avoid the ecological and social challenges of dedicated energy crops. Organic residues and waste are perceived to have less environmental impacts as they are inevitable by-products of high value food and fodder production. The inherent uncertainties in accessible potential of BECCS in future and decentralised biomass resources and CO$_2$ storage sites, dictate that any decision-making process must be based on an adaptive management model. This study proposes an adaptive management model using a sustainability framework for BECCS deployment. This model is to assist decision-makers and educating the public on comparative sustainability of different BECCS routes. A bottom-up assessment of bioenergy availability from organic waste and residues under very strict sustainability criteria is conducted. In a global context, the potential contribution of the sustainable bioenergy to supply energy and deliver negative emission through BECCS is evaluated. Furthermore, the sustainability framework is applied to assess the role of BECCS in decarbonising the Australian energy system.
Chapter 3

Co-firing Coal with Biomass in a BECCS Context
3.1 Introduction

Currently, coal-based power generates 9,690 TWh or 41% of total electricity in the world [175], which with emitting 10 Gt CO$_2$/year is responsible for 31% of global CO$_2$ [115, 176]. In the developing countries, fossil fuels are continuing to provide a major share of power generation. Replacing a share of the fossil fuel input with biomass is a way to reduce emissions from the power sector [177]. Co-firing biomass with coal is an option that offers near-term CO$_2$ mitigation with minimum modifications and moderate investment [176, 178-180]. Global primary energy demand is a determinant of growth in gross domestic production and is expected to increase by 2.4% per annum [181]. The majority of this growth will occur in developing countries such as China and India where low efficiency coal-fired power plants supply, respectively up to 80% and 70% of their power [182]. The low cost of retrofitting the existing coal-fired plants for co-firing is especially an advantage for abating emissions in developing countries with considerable coal-based power such as China, where 90% of coal-fired power plants are under 20 years old [183]. Refurbishment of existing coal-fired power plants helps to weaken the dependency on fossil fuels and inclusion of renewable sources in the power market eventually reduces GHG emission [179, 184]. It is estimated that replacing 10% of coal power capacity with biomass would contribute to the reduction of CO$_2$ emission by 0.5–0.6 Gt per year [185-188]. China and the U.S are the two largest CO$_2$ emitters in the world. According to McGlynn et al. [183], replacing 25% of coal with sustainable biomass at existing power plants in these two countries could reduce emissions by 1 Gt CO$_2$ per year. This gives an important indication of the opportunities to reduce emissions without phasing out a large number of existing coal-fired power plants.

The possibility of using a wide range of biomass types for co-firing with coal enhances the fuel flexibility and security of the power plant [40, 179, 189]. In addition, co-firing some types of biomass could result in reducing NOx and SOx emission and avoid trace elements such mercury and cadmium [179, 189-194]. Using waste biomass for co-firing also avoids their decomposition which is otherwise a source of GHG emissions [191]. In addition, for local biomass producers, selling their organic waste to the nearby co-firing plants provides economic benefits [179]. In countries with renewable tax credits or a carbon pricing scheme in effect, co-firing could offer economic incentives for coal-fired power suppliers.

Co-firing is a mature technology with numerous examples around the world. There have been more than 200 co-firing installations worldwide with a total capacity of 1–10 GW [179, 182,
Europe with approximately a hundred co-firing plants is the pioneer of co-firing biomass in the world. Drax power station commissioned in the 1970s and 1980s in the UK is one of the pioneers of co-firing technology in Europe. Starting from only 3% co-firing in 2003, Drax power plant has upgraded four of its six generating units to accommodate 100% biomass firing using compressed wood pellets [196]. This is equivalent to around 2640 MW, or 66% of the generating capacity of the station, which could supply power to four million households [197, 198]. In the U.S about 560 coal-fired power plants supply around 39% of total electricity and around 40 of the plants use biomass for co-firing [178, 180]. Australia and some of Asian countries such as Japan, China, and South Korea have already adopted co-firing technologies [178]. Schemes to introduce mandatory percentages of renewables in the EU and several states in the U.S have been the main incentive to promote using biomass in the power sector [178, 183]. In Australia, introduction of a Federal Mandatory Renewable Energy Target (MRET) and later implementing a Renewable Energy Target (RET) scheme with the possibility to trade renewable energy certificates, encouraged biomass co-firing in Australia. The majority of these co-firing activities are small-scale pilot plants in Queensland and New South Wales [199-201].

3.1.1 Co-firing options

Depending on the co-firing ratio and the boiler design, three technologies have been investigated:

Direct co-firing: direct co-firing is the least expensive and most commonly utilised technology that allows for small co-firing ratios. In this method, coal and biomass are fired in the same boiler. The milling process could be together or separate. Direct co-firing is applicable to pulverised fuel boiler (PC), circulating fluidised bed (CFB) and bubbling fluidised bed (BFB) technologies. Higher rates of up to 20% co-firing are possible when cyclone boilers are used [183, 185, 202-206].

Indirect co-firing: gasification of biomass and using the product gas in the coal furnace makes it possible to introduce higher proportions of biomass. In indirect co-firing, a large range of biomass types (especially low-rank biomass with high contaminations) can be used; the product gas must be cleaned before entering the furnace to enhance its combustion performance [183, 185, 202-207].

Parallel co-firing: in this technology, biomass is combusted in a separate furnace and the steam produced is then integrated into a coal power plant system. With parallel co-firing, the highest
proportion of biomass is possible. However, the need for a separate furnace and other related equipment makes it the most costly co-firing option [183, 185, 202-207].

Choice of the co-firing option depends on the environmental concerns, applicability, operational experience, efficiency, economics and biomass availability.

3.1.2 Biomass for co-firing

Most of the co-firing installations employ direct co-firing at biomass to coal ratios of less than 3% [179], but in recent years, it has been increased up to 20% [205]. With today’s technical advancements, up to 15% co-firing without major modifications of the boiler is possible [189]. A variety of biomass feedstock such as agricultural residues, forestry and wood processing residues, organic waste and energy crops could be utilised for co-firing [183]. So far, woody biomass has made up to 80% of biomass used in co-firing plants [178, 186]. Biomass is often classified based on its properties; one frequently used method is to categorise biomass into woody biomass, herbaceous biomass, wastes and derivate, and aquatic biomass types [190]. Alternatively, biomass is classified into first, second and third generations. The first-generation of biomass is derived from food sources such as corn and sugarcane. The second-generation biomass is from non-food resources such as wood, agricultural and forestry residue and organic waste. The third-generation uses algae as a biomass feedstock to produce energy. In comparison with the first generation of biomass, second and third generations are perceived to be a more sustainable feedstock for bioenergy production [50]. However, these resources are widely dispersed around the globe and their selection for large-scale energy production is strictly dependent upon regional availability and abundance, technical feasibility and economic viability.

Due to the high volatile and moisture content of biomass, co-firing biomass could result in some technical implications such as decreasing the efficiency of the boiler [73, 204, 208, 209] and damage to the furnace such as slagging and fouling [189, 192, 210, 211]. The moisture content in the fuel affects the calorific value and the composition characteristics of its combustion by delaying the ignition process [203, 206]. Volatile matter in a fuel affects its reaction rate and combustion temperature. After de-volatilization, biomass leaves a highly porous char, which its rate of reaction is than coal char. Rapid escape of volatile gas through the biomass char can lead to explosions in the furnace and causes flame instability [203, 206].
Pelletisation (mechanical drying) combined with torrefaction (thermal drying) is a way to enhance biomass combustion characteristics [112, 191, 194, 212-215].
3.1.3 Examples of co-firing reported in the literature

Different technical, environmental and economic aspects of co-firing biomass with coal have been extensively studied in the literature [176-180, 183, 184, 187-192, 194, 195, 201-210, 216-224]. On the other hand, research on co-firing biomass with coal combined with CCS is limited. A study by Bhave et al. [225] evaluated the techno-economic impact of biomass co-firing in a 50 MW coal-fired power plant with amine-based CO₂ capture in the context of BECCS. Their study showed that the co-firing ratio is the principal factor affecting levelised cost of electric production. They also reported that co-firing with CCS results in a relatively moderate “cost of avoided emission” of $100/t CO₂. Gladysz and Ziebigk [226] presented a system approach to environmental and techno-economic analysis of biomass co-firing and dedicated biomass in an oxy-fuel combustion unit. Their results show that co-firing could reduce the cost of electricity production. In addition, they showed that 20% co-firing may lead to zero emissions while 30% or more leads to negative CO₂ emissions of the plant. Akgul et al. [193] applied a mixed integer nonlinear programming (MINLP) model to examine the potential for reducing GHG emission from existing power generation plants in the UK through co-firing biomass with CCS. They showed that its high cost makes it economically viable only under a high carbon price (£120–£175/t CO₂). Khorshidi et al. [220] studied different co-firing configurations in a 500 MW sub-critical coal-fired power plant with MEA post-combustion capture. They showed that both the emission intensity and the cost of electricity of co-firing with CCS would be marginally lower than a coal-fired plant with CO₂ capture. Alia et al. [189] studied co-firing of up to 40% to 100% in a pulverised supercritical power plant with post-combustion CO₂ capture and found decreasing power output with increasing biomass fraction. A study by Fogarasi et al. [186] on the technical and economic aspects of biomass co-firing with CO₂ capture showed CO₂ emissions and net electrical efficiency changed only 1% between the case studies with 100% coal and 100% biomass. Hetland et al. [40] addressed the technical potential of direct-fired and co-fired agricultural residue systems, combined with post-combustion capture. Their study showed that by introducing CCS with typically 90% capture rate the power generation efficiency will drop by roughly 10% with the addition of CCS.

There are few studies of other forms of co-firing biomass with CCS. For instance, Khorshidi et al. [216] showed the effect of co-firing gasified biomass with natural gas in a large combined cycle plant with CCS. Their results showed reduction in the efficiency of the plant by 4.5 % and the possibility of delivering negative emissions of up to –0.3 t CO₂/MWh.
Other than lab-scale and conceptual studies, co-firing biomass with CCS has been deployed in two power plants. One is the Maasvlakte power plant 3 (MPP3) in the Netherlands which became operational in 2015. With a capacity of 1070 MW, MPP3 is capable of accepting up to 30% biomass; its CCS component is subject to commercial decision [89]. The other project is Mikawa power plant in Omuta in Japan with a capacity of 49 MW, that has been recently retrofitted to combust biomass with CCS. This project is sponsored by Japan’s Ministry of the Environment [77].

Given this background, it was decided to assess the global technical potential and challenges of co-firing with CO₂ capture to achieve zero or negative emission in the context of BECCS.
3.2 Methodology

The rationale for selecting pulverised coal (PC) combustion and power generation is because it is the principal technology in the coal-based power sector worldwide. Therefore, biomass co-firing of PC power generation with CCS is likely to have the most substantial impact on reducing CO₂ emissions. The basis of the modelling was direct combustion of biomass and coal with ratios ranging from 0 to 20% biomass on a mass basis, with the PC plant assumed to be equipped with MEA post-combustion CO₂ capture technology. In this study the capacity of the PC plant is 600 MW with a net plant efficiency (LHV) of 42.1% which after integration with CCS drops to 30.6% [72]. Boiler efficiency and CO₂ capture rate are both 90%.

3.2.1 MEA CO₂ capture technology

Before entering the CO₂ capture unit, the flue gas is passed through a pre-treatment unit consisting of a selective catalyst reactor (SCR) to remove NOx (NO and NO₂) and a flue gas desulphurisation unit (FGD) to remove SOx (SO₂ and SO₃) from the flue gas. SOx and NOx damage the MEA system by forming corrosive stable salts. The SO₂ content of flue gas after FGD is around 30 ppmv [227], so a secondary FGD is required to clean the SO₂ to a level of 10 ppmv, at which the MEA system is not adversely affected. Fly ash is removed using cyclones and suspended particulates are collected by the electrostatic precipitators (ESPs) [228].

The CO₂ capture process in a chemical absorption process is based on reversible reactions of CO₂ with an amine absorbent. Among several amines studied, MEA has the highest CO₂ carrying capacity [71, 229]. In an MEA unit, sorbent solution absorbs CO₂ from the flue gas in an absorber at 30–40 °C and 1 bar, and is regenerated releasing CO₂ in a stripper at 100-120 °C and around 2 bar [229-231]. At temperatures around 50 °C a 30 wt% MEA solution can absorb 0.45 mole CO₂ per mole of MEA [229]. The absorption capacity of a MEA solution is very sensitive to the operating temperature and efficiency decreases at higher temperatures. Typically, up to 90% of the incoming CO₂ is captured in the MEA process. The purity of the separated CO₂ stream is > 99% [232]. The presence of SO₂ and O₂ in the flue gas degrades the MEA and lowers its capture performance. Steam heating required for the stripper, and pumping the liquid is the source of energy penalty in a MEA system. The overall efficiency penalty due to an MEA installation in a PC plant, is estimated to be in the range of 10–15% [72, 73].
3.2.2 Fuels

Three different categories of biomass resources have been considered in this investigation of the technical implications of co-firing with black coal in pulverised coal combustion. These biomass types are woody (wood, wood residue, forest residues), herbaceous agricultural (switchgrass, straw), and aquatic (marine macro-algae). These biomass resources are known as second and third generation biomass. Table 3-1 shows the proximate analysis and compositions of the biomass and coal fuels used in this study.

Table 3-1- Proximate and ultimate analysis of the fuels [233-239]

<table>
<thead>
<tr>
<th>Fuel</th>
<th>Proximate Analysis (As received)</th>
<th>Ultimate Analysis (Dried ash-free)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>VM</td>
<td>FC</td>
</tr>
<tr>
<td>Wood and woody Biomass</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wood</td>
<td>77.5</td>
<td>14.5</td>
</tr>
<tr>
<td>Wood residue</td>
<td>57.4</td>
<td>12.2</td>
</tr>
<tr>
<td>Forest residue</td>
<td>34.5</td>
<td>7.3</td>
</tr>
<tr>
<td>Herbaceous and agricultural biomass</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Switchgrass</td>
<td>70.8</td>
<td>12.8</td>
</tr>
<tr>
<td>Straw</td>
<td>64.3</td>
<td>13.8</td>
</tr>
<tr>
<td>Aquatic biomass</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marine macro-algae</td>
<td>45.1</td>
<td>23.1</td>
</tr>
<tr>
<td>Solid fossil fuel</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black Coal</td>
<td>30.8</td>
<td>43.9</td>
</tr>
</tbody>
</table>

The detailed ash composition of the fuels is provided in Table 3-2. The ash content of biomass in general is much less than coal. Woody biomass has the least ash content (0.2–4.5%), herbaceous biomass is in the mid-range (4.5–10 wt%) and aquatic biomass has the highest percentage of ash (>20%). In addition to the ash content, the ash composition varies in for the different sources. For instance, woody biomass has higher CaO. The herbaceous biomass on
the other hand, has the highest percentage of SiO\(_2\) and alkali metal oxides (K\(_2\)O and Na\(_2\)O). The ash composition of aquatic biomass is very different to the first two categories, with marine macro-algae very rich in SO\(_3\) and very low in SiO\(_2\).

**Table 3-2-Ash composition of the fuels [233-239]**

<table>
<thead>
<tr>
<th>Fuel</th>
<th>Ash Composition %</th>
<th>SiO(_2)</th>
<th>CaO</th>
<th>K(_2)O</th>
<th>P(_2)O(_5)</th>
<th>Al(_2)O(_3)</th>
<th>MgO</th>
<th>FeO(_3)</th>
<th>SO(_3)</th>
<th>Na(_2)O</th>
<th>TiO(_2)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Wood and woody Biomass</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wood</td>
<td>23.15</td>
<td>37.35</td>
<td>11.59</td>
<td>2.9</td>
<td>5.75</td>
<td>7.26</td>
<td>3.27</td>
<td>4.95</td>
<td>2.57</td>
<td>1.2</td>
<td></td>
</tr>
<tr>
<td>Wood residue</td>
<td>53.15</td>
<td>11.66</td>
<td>4.85</td>
<td>1.37</td>
<td>12.64</td>
<td>3.06</td>
<td>6.24</td>
<td>1.99</td>
<td>4.47</td>
<td>0.57</td>
<td></td>
</tr>
<tr>
<td>Forest residue</td>
<td>20.65</td>
<td>47.55</td>
<td>10.23</td>
<td>5.05</td>
<td>2.99</td>
<td>7.2</td>
<td>1.42</td>
<td>2.91</td>
<td>1.6</td>
<td>0.4</td>
<td></td>
</tr>
<tr>
<td><strong>Herbaceous and agricultural biomass</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Switchgrass</td>
<td>66.25</td>
<td>10.21</td>
<td>9.64</td>
<td>3.92</td>
<td>2.22</td>
<td>4.71</td>
<td>1.36</td>
<td>0.83</td>
<td>0.58</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td>Straw</td>
<td>57.14</td>
<td>6.7</td>
<td>25.82</td>
<td>2.74</td>
<td>0.76</td>
<td>1.67</td>
<td>0.53</td>
<td>3.89</td>
<td>0.7</td>
<td>0.05</td>
<td></td>
</tr>
<tr>
<td><strong>Aquatic biomass</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marine macro-algae</td>
<td>1.65</td>
<td>12.39</td>
<td>15.35</td>
<td>9.76</td>
<td>0.85</td>
<td>12.5</td>
<td>1.87</td>
<td>25.74</td>
<td>19.88</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Solid fossil fuels</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black Coal</td>
<td>54.06</td>
<td>6.57</td>
<td>1.6</td>
<td>0.5</td>
<td>23.18</td>
<td>1.83</td>
<td>6.85</td>
<td>3.54</td>
<td>0.82</td>
<td>1.05</td>
<td></td>
</tr>
<tr>
<td>Cp(kJ/kg·K) at 298° k</td>
<td>0.731</td>
<td>0.756</td>
<td>0.897</td>
<td>0.278</td>
<td>0.775</td>
<td>0.924</td>
<td>0.651</td>
<td>0.628</td>
<td>1.106</td>
<td>0.690</td>
<td></td>
</tr>
<tr>
<td>ΔH(_f) (kJ/kg)</td>
<td>-15163</td>
<td>-11324</td>
<td>-3855</td>
<td>-5178</td>
<td>-16435</td>
<td>-14931</td>
<td>-5161</td>
<td>-4942</td>
<td>-6712</td>
<td>-11832</td>
<td></td>
</tr>
</tbody>
</table>

### 3.2.3 Impact on plant efficiency

Different levels of co-firing have different impacts on the efficiency of a PC power plant. The main impact is likely to be on the boiler efficiency. The major cause of the efficiency penalty in a PC boiler is heat loss from several sources. The major heat losses are due to dry flue gas, hydrogen in the fuel, water in the fuel and air, incomplete combustion, surface radiation and convection, unburned solid fuel (ash) and heat loss of fly and bottom ash.

The method used to calculate the boiler efficiency using a blend of coal and biomass fuels is:

\[
\eta_{boiler} = \frac{Q_{in} - Q_{loss}}{Q_{in}}
\]  

(Equation 3-1)

Where Q\(_{in}\) is the input energy of the fuel and Q\(_{loss}\) is the sum of all heat losses in the boiler, which are calculated through equations below;
\[ Q_{\text{in}} = m_{\text{coal}} \times LHV_{\text{coal}} + m_{\text{biomass}} \times LHV_{\text{biomass}} \]  

(Equation 3-2)

Where \( m \) and LHV represent mass and lower heating value of the input fuels.

\[ Q_{\text{loss}} = Q_{L1} + Q_{L2} + Q_{L3} + Q_{L4} + Q_{L5} + Q_{L6} + Q_{L7} + Q_{L8} \]  

(Equation 3-3)

In Equation 3-3, \( Q_{L1} \) is loss due to dry flue gas, which is the greatest boiler loss. \( Q_{L2} \) and \( Q_{L3} \) represent the heat loss due to \( H_2 \) and \( H_2O \) in the fuel. Combustion of \( H_2 \) produces \( H_2O \) and some heat is lost to evaporate this \( H_2O \). \( Q_{L4} \) is the heat loss due to \( H_2O \) in air. \( Q_{L5} \) is the heat loss due to incomplete combustion of carbon in the fuel. \( Q_{L6} \) represents the heat loss due to surface radiation and convection. In this study, it is assumed that impact of biomass co-firing at this level (\( \leq 20\% \)) on heat loss due to dry fuel gas (\( Q_{L1} \)), incomplete carbon combustion (\( Q_{L5} \)), radiation and convection (\( Q_{L6} \)) and moisture in air (\( Q_{L4} \)), are insignificant [203, 206]. Heat losses due to moisture and \( H_2 \) in fuel (\( Q_{L2} \) and \( Q_{L3} \)) are already considered in the LHV calculation. Thus, the major impacts to be estimated are heat loss due to unburnt solid fuel i.e. bottom and fly ash (\( Q_{L7} \)) and heat lost by fly and bottom ash (\( Q_{L8} \) and \( Q_{L9} \)).

\( Q_{L7} \) represents the heat loss due to unburned solid fuel (ashes) and is calculated through Equation 3-4.

\[ Q_{L7} = m_{\text{ash}} \times LHV_{\text{ash}} \]  

(Equation 3-4)

In Equation 3-4, \( m_{\text{ash}} \) is the total mass of fly and bottom ash collected (kg). LHV\(_{\text{ash}}\) is the total LHV of bottom and fly ash that is calculated using Equation 3-5. Main ash components of the fuels (as seen in Table 3-2) are SiO\(_2\), CaO, K\(_2\)O, P\(_2\)O\(_5\), Al\(_2\)O\(_3\), MgO, Fe\(_2\)O\(_3\), SO\(_3\), Na\(_2\)O and TiO\(_2\).

\[ LHV_{\text{ash}} = LHV_{\text{SiO}_2} + LHV_{\text{CaO}} + LHV_{\text{K}_2\text{O}} + LHV_{\text{P}_2\text{O}_5} + LHV_{\text{Al}_2\text{O}_3} + LHV_{\text{MgO}} + LHV_{\text{Fe}_2\text{O}_3} + LHV_{\text{SO}_3} + LHV_{\text{Na}_2\text{O}} + LHV_{\text{TiO}_2} \]  

(Equation 3-5)

\( Q_{L8} \) and \( Q_{L9} \) are the heat loss of fly and bottom ash respectively (see equations 3-6 and 3-7 below).

\[ Q_{L8} = (T_f - T_a) \sum^n_i m_{f,i} \times c_{p-f,i} \]  

(Equation 3-6)

\[ Q_{L9} = (T_f - T_a) \sum^n_i m_{b,i} \times c_{p-b,i} \]  

(Equation 3-7)
In equations 3-6 and 3-7, \( m_{f,i} \) and \( m_{b,i} \) represent the mass of fly and bottom ash components collected (kg) and \( C_{P,f,i} \) and \( C_{P,b,i} \) are the specific heat of fly ash and bottom ash components. \( n \) is the number of different compounds in the ash.

### 3.2.4 CO₂ Emissions

Co-firing, even at low levels, could have an impact on emissions from the combustion process. Biomass materials, in general have higher volatile matter, which causes sudden explosive combustion and affects the ignition and combustion temperature [203, 209]. That could influence the kinetics of combustion and change the amount of incomplete combustion and produce more subsequent emissions of CO [209]. The nitrogen (N) and sulphur (S) content of the fuel determines the NOx and SOx emission. Woody and herbaceous biomass materials contain much less N and S than coal, which makes them favourable options. Ash content of the fuel is another factor affecting the emission profile of the co-firing plant. Biomass and coal fuels have very different ash composition. Biomass is higher in alkali metals and alkaline earth metals. In this study, SOx and NOx emissions were not calculated as the focus is on CO₂ emission and ash formation. It is often assumed that biomass, if produced sustainably, is a renewable resource and a CO₂-neutral fuel. Therefore, capturing the CO₂ from biomass combustion is counted as a negative emission. CO₂ emissions from co-firing in a PC plant equipped with carbon capture is calculated according to Equation 3-8.

\[
CO₂ – \text{emission} = CO₂ – \text{coal}(1 - \eta_{cap}) - CO₂ – \text{biomass} \cdot \eta_{cap} \\
\text{(Equation 3-8)}
\]

\( CO₂ – \text{coal} \) and \( CO₂ – \text{biomass} \) are the CO₂ emission from coal and biomass combustion (Kg/MWh), and \( \eta_{Cap} \) represents the CO₂ capture rate.

### 3.2.5 Slagging

Slag formation in a furnace occurs at temperatures above 1000 °C, when ash particles start to fuse and sinter on the surface. The general mechanism of slagging is mass transfer of ash particles from the flue gas to the surfaces [211]. Adhesion of ash particles on the surface determines the rate of slag formation. A receptive surface and molten and sticky ash particles are the factors that drive the adhesion process [203]. In addition to furnace design and combustion conditions, slagging is mainly a fuel-related process and depends on fusion behaviour of fuel compositions. This is a problem for all solid fuel-fired boilers especially biomass combustors. It has been observed that basic compounds in ash lower the melting point
and acidic ones increase it. This correlation is used to formulate an index for slag formation (equations 3-9 and 3-10) [223, 224].

\[
\frac{B}{A} = \frac{Fe_2O_3 + CaO + MgO + Na_2O + K_2O + P_2O_5}{SiO_2 + Al_2O_3 + TiO_2} \quad \text{(Equation 3-9)}
\]

\[
SI = \left(\frac{B}{A}\right)S^d \quad \text{(Equation 3-10)}
\]

In Equation 3-9, B/A represents the proportion of basic to acidic compounds in ash. Equation 3-10 is used to estimate the slagging index, SI, and \(S^d\) is the sulphur content of dry fuel. According to Pronobis [224], SI <0.6 shows low slagging tendency, SI=0.6-2.0 means medium slagging, SI=2.0-2.6 is for high and SI >2.6 shows extremely high slagging tendency.

### 3.2.6 Fouling

Deposition of sodium and potassium compounds on the cooler surfaces is the main cause of fouling. The main mechanism of fouling is through volatilization/condensation of these compounds [203]. At temperatures greater than 1000 °C volatile compounds present in ash condense on the cooled surfaces, forming a deposit. At lower temperatures (< 700°C) the deposits tend to detach from the surface [211]. Reducing the flue gas temperature is a way to control fouling. The fouling index (Equation 11) for biomass materials is based on their alkali metal content [224].

\[
FI = \frac{B}{A} (Na_2O + K_2O) \quad \text{(Equation 3-11)}
\]

In Equation 3-11, FI ≤ 0.6 shows low fouling inclination, 0.6 < FI ≤ 40 means high fouling risk and FI > 40 represents extremely high tendency to fouling.
3.3 Results

Co-firing with different biomass materials affects the operational conditions and emissions in a PC plant. Some of the major technical impacts are explained below;

3.3.1 CO₂ emissions profile

The impact of co-firing on the final CO₂ emissions of the PC plant with CCS was calculated. Equation 3-8 was used to calculate the final CO₂ emission at different co-firing ratios. The CO₂ captured from biomass combustion is considered to be a negative emission and is subtracted from the 10% emission from coal combustion that is not captured. CO₂ emission from a PC plant using only black coal with no CCS is around 805 Kg/MWh. After combining with CCS this reduces to around 93 Kg/MWh- see Figure 3-1. Therefore, the impact of biomass co-firing on the CO₂ emission of the plant with CCS is marginal. Nevertheless, the option of reaching zero or even negative emission would only be possible with co-firing. The carbon content of the all biomass types is around 50% in proximate analysis, which means the impact of their co-firing with coal on CO₂ emission is almost the same. As illustrated in Figure 3-1, in all cases, 17% co-firing on a mass basis reduces the CO₂ emissions to zero level. Negative emission would then be possible at higher co-firing ratio. When 20% of woody, herbaceous and agricultural biomass is co-fired, negative emission up to 26 Kg/MWh could be delivered. It should be noted that direct co-firing is applied in this model, wherein more than 20% co-firing ratio is typically not practical. Even 15% co-firing is only feasible in modern power plants. In more conventional PC power plants which allow co-firing up to maximum 10%, CO₂ emission would reduce to around 39 Kg/MWh (–57%) and negative emissions will not be obtained. Due to its lower carbon content, co-firing macro- algae results in a higher CO₂ emission than the rest of biomass types used in this study.
3.3.2 Impact of co-firing on efficiency

The main parameters of biomass co-firing that diminish the plant efficiency are the lower heating value, higher moisture content, lower energy density and inhomogeneity. Different levels of co-firing have different impacts on the efficiency of a PC power plant. The main impact is likely to be on efficiency of the boiler. Figure 3-2 shows the impact of the co-firing ratio on the total efficiency of the PC plant with MEA. As seen, co-firing wood leads to the lowest efficiency penalty; 20% co-firing wood leads to about 0.8% efficiency penalty. On the other hand, macro-algae with lower LHV and higher ash content causes the highest loss in efficiency, approximately 1.2%. Either efficiency penalty results in less output of power, which means economic loss, or more input fuel to sustain the stable power generation. To put this in perspective, to maintain the power generation at a constant level, 0.8% energy loss due to co-firing wood would require increasing the fuel input by 11% (Kg/KWh). In the other words, a higher cost of electricity generation. This is even worse for the other biomass sources used. For instance, in case of macro-algae the additional fuel input required at 20% co-firing would be 13.5% (Kg/KWh). As mentioned before, direct co-firing even in modern power PC plants cannot accommodate more than 20% biomass. In addition, compensating the efficiency penalty with more biomass will exacerbate the technical complications of co-firing. Hence, the more likely implications will be loss of output power.
3.3.3 Slagging & Fouling

Slagging and fouling are two of the main concerns in co-firing biomass plants. The presence of alkali metal compounds with low melting temperatures helps slag deposition. Slagging degrades the heat transfer efficiency in the furnace. As seen in Figure 3-3, co-firing biomass up to 20% for most of biomass materials does not affect the slagging index (SI). The SI for co-firing with all woody and herbaceous is less than 0.6, which lies in the low slagging inclination range. The only biomass that increases the risk of slag formation is macro-algae that contains the highest S and basic to acidic compounds ratio (B/A) among the biomass materials.
As seen in Figure 3-4, the only visible change in fouling index occurs in the case of macro-algae co-firing. Macro-algae has the highest S (2.6 wt%) and Na (20 wt% of ash) content. These two compounds have a direct impact on fouling propensity. Therefore, a 20% co-firing macro-alga, on a mass basis, raises the fouling index from 0.45 to 3.5, which means there is a high risk of fouling. Similarly, compared to other biomass resources, straw is richer in K and has higher B/A which results in higher risk of fouling, but the increase in fouling tendency even at its maximum, is still much less than that of macro-algae.
3.3.4 Impact on CO₂ capture process

One important influence on the efficiency of CO₂ capture technology is the concentration of CO₂ in the flue gas. Usually the vol% CO₂ of the flue gas exiting from a pulverised coal plant is around 15% [72]. Biomass materials contain higher volatile and moisture content. This might affect the concentration of CO₂ and the efficiency of capture consequently. The CO₂ concentration in the flue gas from co-firing the biomass materials was calculated relative to the sum of N₂, O₂ and CO₂. It is assumed that all the SO₂ in the flue gas has been separated by an FGD unit before entering the MEA unit. It was observed that CO₂ vol% in the flue gas entering the CO₂ capture unit after co-firing biomass was between 14–15 % and did not significantly changed compared to no co-firing case.

Oxidised compounds of Si, Al, Fe and Ca make up about 90% of the fly ash content [223]. Impurities in flue gas can affect the performance of the capture plant. Solvent degradation and stable salt formation are the two main risks for the amine-based capture process. For instance, the presence of acid compounds such as SO₂, NO₂, HCl and HF, and oxidation with O₂ causes MEA degradation [240]. Biomass fuels have less sulphur than coal. Concentrations of exiting O₂ and SO₂ from co-firing remain the same, except for macro-algae, which decreases with
increasing co-firing ratio. The chlorine content of woody biomass is much less than that of coal, but herbaceous biomass is higher in Cl. Although most of the fly ash (>99%) is removed from the flue gas by the EPS, it has been observed that very small concentrations (about 10 ppm) of transition metals like copper or iron is enough to catalyse amine oxidative degradation [240]. It is believed that Fe ions accelerate oxidative degradation by generating more free radicals [241]. As presented in Table 3-2, the Fe$_2$O$_3$ content of herbaceous and algae ash is less than 2.5%, which is lower than coal (8.6%). On the other hand, woody biomass has a more diverse Fe content. Wood and wood residue contain respectively 3.27% and 6.24% iron oxide in their ash and forest residue has only 1.24% Fe$_2$O$_3$. In general, since the total ash content of biomass is much less than coal, co-firing will reduce the risk of Fe deposition in the MEA process and lower the risk of amine degradation.

### 3.3.5 Impact on ESP collector

Fly ash produced from biomass combustion is different to that produced from coal. Biomass fly ash tends to be finer, forms a sub-micron fume and differs in composition [203, 242]. This change affects the efficiency of the electrostatic precipitator (ESP), in that increasing co-firing lowers the efficiency of the ESP [203]. The amount of fly ash escaping the EPS may not be significant but over a prolonged operation, but a small concentration of fly ash matter will build up in the amine-based capture equipment and cause serious damage due to degradation, corrosion and plugging [242]. However, for a biomass-to-coal thermal ratio of less than 20%, this impact tends to be insignificant [243].
3.4 Discussion

According to the results of this study, direct co-firing of up to 20% biomass in a modern PC plants equipped with CO$_2$ capture (at a 90% capture rate) and storage could deliver negative emissions of up to around -26 Kg CO$_2$/MWh. To put this in a global perspective, if all the coal-based electricity generating 9,690 TWh a year in the world is refurbished to accommodate this level of co-firing, a total negative 252 Mt CO$_2$ could be achieved per annum. Compared with the current emission of coal-based electricity (10 Gt CO$_2$ per year), it translates to mitigation of 10.25 Gt CO$_2$ per year or 102% from coal-fired power industry or eliminating more than 31% of the anthropogenic GHG emission. This could greatly contribute to achieving the COP 21 target to achieve zero net GHG emission by mid-century [3]. Direct-cofiring could be applied in the existing coal-fired power plants with minimum modifications. In other words, without phasing out the functional plants and compromising energy security, co-firing with CCS could turn coal-fired power plants from being a big contributor of CO$_2$ emission into a potential technology to remove CO$_2$ from the atmosphere.

To provide this level of negative emission around 9.6 EJ or 600 Mt biomass would be required. Hence, the availability of biomass is a key factor to achieve this amount of CO$_2$ mitigation. There is a very wide range of estimations of the global potential of bioenergy. Leatherdale et al. estimated that bioenergy supply could contribute 2–20% to global primary energy demand by 2050 [244]. The European energy agency estimated the potential for sustainable bioenergy production of approximately 340 EJ/ year in 2050 [7]. In a study by Koljonen et al. under scenarios with sustainability constraints, the global potential of bioenergy is between 200 and 500 EJ/year by 2050, that would cover up to 50% of projected world primary energy demand [48]. In a study by Haberl et al. bioenergy potential varies from 30–1000 EJ/ year by 2050 and with sustainability constraints it reduces to 160–270 EJ/year [49]. Based on the biomass potential estimated in the literature, there would be sufficient biomass supply to use in co-firing plant to achieve this level of negative emission. However, availability of biomass resources is limited by factors such as climate change, land use change, soil production and technical and economic developments. Besides, biomass is a highly dispersed feedstock and it could considerably affect the supply-chain management and transportation cost. The other issue is the sustainability of biomass production. In recent years, environmental concerns have questioned the sustainability of energy crop production. This has become an incentive to consider organic wastes and residues for co-firing. These sources could provide a new market
for residues and improve economic stability and job creation especially in developing countries with adequate and mature biomass market such as Brazil and Southeast Asia [210].

Though co-firing biomass with coal could result in an efficiency penalty, it offers higher net efficiency than dedicated biomass combustion plants. In other words, co-firing biomass offers more energy generation for less biomass input (30–90%) [40]. Most modern coal-fired power plants are of high capacity (more than 500 MW) and can accommodate up to 15% biomass without needing any modifications in the steam boiler [40, 185]. Therefore, under an energy and environment trade-off paradigm wherein supplying secure and low-cost energy ranks as the first priority and mitigating the emission comes the second, co-firing is a more favourable option than dedicated bioelectricity. According to Al-Mansour et al. [187] the most influential co-firing parameters for developing co-firing power plants around the world has been factors such as secure biomass supply, cost of biomass feedstock and transport. In addition, their study showed that environmental considerations and regulations were the second-most important driver.

Despite its promising potential, co-firing with CCS faces several technical, economic and social challenges. Co-firing is likely to increase the levelised cost of electricity production and lack of adequate economic incentives such as effective carbon price and CO₂ emission reduction credits hinders its large-scale deployment. The other challenge is the absence of an organised global market for biomass, which could affect the availability and security of biomass supply during seasonal and climate variations. Regarding synergy with CCS, the mismatch between the CO₂ source and sink is another complication which is common for all types of power plants with CCS [201]. In general, there is a negative perception regarding coal-fired power with CCS among public and environmental campaigns. The main concern is that it might favour fossil-fuel based power industry and delay fundamental mitigation measures. However, capturing CO₂ from co-firing coal with biomass and the potential to deliver zero or negative emission could reduce or eliminate this stigma and also the “not-in-my-backyard” (NIMBY) effect that has traditionally been an important public acceptance barrier to CCS [193].

Co-firing in the existing power plants equipped with CCS could be an effective near-term mitigation options, which has promising features for delivering secure energy for the growing population with zero or negative CO₂ emission. However, it will only be achievable in the
presence of supportive economic and policy schemes and under the condition that biomass production meets strict sustainability criteria.
3.5 Conclusions

Capturing CO$_2$ resulting from co-firing coal with biomass for power generation, provides a method to extend utilization of carbon capture and storage (CCS) to non-fossil fuel resources. Co-firing with CCS, through BECCS, is one of the few options to mitigate CO$_2$ emissions from the power sector and provide overall “negative” emissions.

This study developed and used a quantitative assessment to predict how different types of biomass with different compositions affect the CO$_2$ emission, boiler efficiency, and extent of slagging, fouling and the performance of the CO$_2$ capture process.

- Among the different co-firing combinations with a 20% biomass-to-coal ratio, woody biomass resulted in the lowest reduction in plant efficiency (0.8%), and macro-algae showed the highest efficiency penalty of > 1.2%. All the biomass materials are perceived as “carbon neutral” fuels so that the CO$_2$ resulting from their combustion would be regarded as a “negative emission”.
- Co-firing ratios up to 20% resulted in negative CO$_2$ emission, from 93 kg CO$_2$/MWh at zero co-firing to -26 kg CO$_2$/MWh.
- Although the relative concentrations of alkali and earth alkaline metals in biomass are higher than coal, slagging tends to decrease slightly due to a considerably lower ash and sulphur content of woody and herbaceous biomass. Similarly, co-firing with biomass (except for macro-algae) does not affect the risk of fouling. Even with co-firing biomass, the CO$_2$ concentration in the flue gas remains in the range of 14-15% and therefore does not significantly change the capturing efficiency of the MEA plant.
- Direct co-firing with CCS has the potential to mitigate the emission of power sector to zero or even negative level, without compromising energy security, thereby significantly contributing to achieving the COP 21 target of zero net GHG emissions in the second half of the 21st century.
Chapter 4

Municipal Solid Waste as a Resource for BECCS
4.1 Introduction

One of the resources used to generate bioenergy is organic waste collected from municipal solid waste (MSW). MSW is one of the main by-products of urbanization. Currently, the global urban population is estimated to be around 3 billion, which generates 1.3 Gt of solid waste annually. By 2100 MSW generation may increase to 4 Gt [245, 246]. This amount of solid waste, if not managed, is potentially a large source of methane emissions, air pollution, human and ecosystem health risks and underground water contamination. According to IPCC, about 3% (1.4 Gt CO$_2$-eq/year) of global GHG emission in 2010 was from solid waste “management” [247] and methane emitted from landfills accounted for 12% of the global anthropogenic methane emission [248].

One way of managing MSW is through waste-to-energy (WTE) technologies. MSW incineration and landfill gas (LFG) utilization are two of the most commonly utilised WTE technologies. Worldwide, around 2,000 conventional WTE facilities use more than 130 Mt MSW each year to produce energy [249]. It is estimated that around 765 incineration WTE plants using 83 Mt MSW annually are operating worldwide [250]. More than 1150 plants around the world collect and utilize LFG methane as a source of renewable energy. The equivalent energy value of the LFG utilized is more than 0.2 EJ per annum. The degree of advancement and technical and environmental standards of these WTE plants largely depends on the level of development of the countries in which they are implemented. Compared to underdeveloped or developing countries, developed countries (where more than half of the MSW in 2010 was generated [247]) usually have stricter environmental standards and deploy more efficient WTE technologies. WTE incinerators have been widely used in many European countries. In the U.S, 71 WTE power plants use about 29 Mt MSW to generate 14,000 GWh annually [251].

In the U.S and Australia, recovering landfill methane has been a more dominant WTE pathway. Air-quality regulations, renewable energy incentives and tax are some of the incentives for LFG utilization. Due to low landfill tipping fee and negative public perceptions, MSW incineration has not been widely adopted in many of these countries [252]. China generates around 300 Mt MSW each year. From early 2000, China has prioritised deploying large scale WTE plants to reduce emissions. To deal with the low calorific value of its MSW, China has developed circulating fluidised bed (CFB) technology. There are currently around 28 CFB plants, with the biggest handling 800 tonnes MSW per day [246]. Worldwide, the incentives
to deploy WTE have been mostly environmental to reduce GHG emission from the waste sector and energy security to use the non-recyclable, non-reusable portion of the waste to produce energy and reduce the volume of the landfilled waste by around 90%. The WTE market was around 25 billion USD in 2015 globally and is likely to increase by 44% by 2020, with Europe having the largest share of about 48% of the whole market [246].

Population growth, the rate of urbanization and consumption behaviours, economic development, and waste management programs are factors that influence the nature of global waste generation. The waste management hierarchy, with some variation, normally follows a step-wise process of prevention, minimisation, reuse, recycle, WTE and landfill. There are always materials which could not be recovered but can be used for energy production. The level of recycling and WTE differs from region to region. Since MSW is a continuously renewed fuel, the WTE could run continuously and be a predictable and secure source of energy technology.

4.1.1 MSW incineration

Incineration of waste produces energy by burning the carbon content of the waste. In this technology the mass of the final quantity of material requiring disposal is reduced by about 90%. However, incineration is costly and requires emission controls on any produced hazardous gases.

Incineration is conventionally carried out through two processes; grate, or fluidised bed combustion. In grate combustion, the fuel is dried and ignited as it moves through the first grate. In the second grate, complete combustion takes place. The incinerator temperature is 850 °C to 950 °C. In fluidised bed combustion, the fuel is mixed with sand for better combustion and higher combustion efficiency. It is more suitable for homogenous waste and some pre-treatment in terms of size reduction is needed. After combustion, the flue gas is cleansed of heavy metals, acid gases and dioxins [249, 253-255]. The lifetime of these WTE technologies is more than 20 years with 85-90% annual capacity factor and net electricity efficiency of 18–27% [254, 255].

In Europe the Renewable Energy Directive recognized biodegradable waste as a renewable resource and consequently the energy produced from this waste is considered as renewable energy. In the U.S only 14% of the MSW is incinerated. There are 87 WTE plants in the U.S which generate 0.4% of the country’s power [249].
Thermal WTE constituted 90% of the total waste market in 2015 and in China and India WTE is growing rapidly and replacing landfilling [256].

One of the best examples of MSW incineration technologies available is the Afval Energie Bedrijf CHP plant in Amsterdam, in operation since 2007. It is the largest incineration plant in the world (114.2 MW) and is able to process 1.5 Mt of MSW per year with an electricity generation efficiency of 30% [256, 257].

**4.1.2 LFG combustion**

Landfill gas (LFG) is responsible for most of the GHG emissions from the waste sector. Development of LFG utilization plants, using vertical wells or horizontal collectors, and mandates to reduce landfill gas emission, has led to significant reduction in methane emission. Currently, more than 1150 LFG plants are operating around the world [249]. The collected LFG is used as fuel for industrial boilers and power generation in internal combustion engines, gas turbine and steam turbines.

Most of the LFG power plants are small scale, in the range of 1–15 MW. This has resulted in challenges regarding economy of scale which hinders its diffusion in the energy sector.

Internal combustion engines, the most commonly used engines in LFG electricity generation projects, tend to be used for projects in the 800 kW to 3 MW capacity range, while gas turbines are typically used for projects that have capacities of 3 MW or more [258].

Gas compression and treatment systems to condition gas for end-user equipment, pipelines to transport LFG to the end user, and condensate management systems for removing condensate along the pipeline, are the main component of a LFG power plant [258].

Gas turbines usually have an efficiency of 20 to 28 percent at full load with landfill gas. Efficiencies drop when the turbine is operating at partial load [259, 260]. Gas turbines have relatively low maintenance costs and low nitrogen oxide emissions when compared to internal combustion engines. Gas turbines require high gas compression, which uses more power, therefore reducing the efficiency. Gas turbines are also more resistant to corrosive damage than internal combustion engines. LFG energy project costs may include costs for gas collection and flaring, electricity generation, direct use, or other project options. Due to the high capital cost of electricity generating equipment, it is often advantageous to size the project at (or near) the minimum gas flow expected during the 15-25 year project life [261].
The ecosystem, geology, land use, land ownership, community acceptance and infrastructure (i.e. pipelines, road, and waste management systems) are some of the most important environmental, social and technical considerations before establishing a sanitary landfill.

In the U.S, the Clean Air Act (CAA) Amendments/New Source Performance Standards (NSPS) [262] set out regulations to control landfill methane emissions. A large amount of LFG is recovered, and national tax credits and local initiatives support its development. In the EU, landfill directive (1999/31/EC) mandates collection and flaring of LFG at landfill sites and reducing methane emission by 35% in 2016 compared with 1995. LFG recovery is an important part of Clean Development Mechanisms (CDM) projects. Certified emission reduction (CER) credits gained from LFG plant in developing countries can be traded and sold by developed countries to meet their emission reduction targets under the Kyoto protocol [249].

The aim of this study is to provide an insight into the global potential for using municipal solid waste as a resource for bioenergy with carbon capture and storage (BECCS). Based on economic viability and environmental impacts, the sustainability of these technologies is investigated.
4.2 Methodology

Techno-economic and environmental impact assessments were conducted for the two WTE systems, with and without CCS; MSW incinerated in a circulating fluidised bed (MSW-CFB), LFG combusted in a gas turbine (LFG-GT), MSW-CFB with CCS (MSW-CCS) and LFG-GT with CCS (LFG-CCS). The two systems without CCS (MSW-CFB and LFG-GT) were used as reference to evaluate the environmental and economic implications of coupling CCS with the WTE technologies.

Best data available from the literature, reports from national and international organizations and online databases were used for the analysis. In all cases MSW generation and transportation is outside of the system boundary.

4.2.1 MSW combustion plant

In the MSW-CCS model, MSW is combusted in an incineration plant to produce electricity. The electricity generated is exported to the grid. The residues from the MSW incineration plant are mostly ash which is landfilled. Circulating fluidised bed (CFB) is used to combust MSW; the reason for using CFB is the ability to burn fuels with high moisture and low heating value with highest efficiency and the fact that CFB does not require fuel pre-processing. The gas velocity in CFB reactors is between 3–9 m/s which results in lower SOx, NOx and HCl formation. Compared with moving grate reactors which are more common for MSW combustion, CFB reactors offer higher efficiency and less bottom ash [263]. One of the challenges of MSW combustion is the presence of trace elements such as Si, Al, K, Mg, Ca and Na which result in formation of heavy metals and salts in the fly ash and bottom ash. Thermal stabilisation by melting and sintering is applied to treat these residues before landfill [264-268]. Electrostatic precipitators are used to filter the dust and most of the heavy metals.

4.2.2 LFG combustion plant

In a LFG-CCS system, wet MSW is collected and stored in a sanitary landfill facility, where LFG is produced through the activities of methanogenic microorganisms. Methane comprises only 35–65 vol% of LFG. The remainder is CO₂ (15–50 vol%), H₂ (0–3 vol%), O₂ (0–1 vol%), H₂S (0–3 vol%) and N₂ (10-15%) [269]. In this model, it is assumed that the sanitary LFG facility is equipped with a gas cleaning unit. Therefore, the final product sent through the pipeline to the gas turbine contains 50%:50% CO₂/CH₄. In some plants, flaring is used to oxidise a fraction of the methane collected [270], but in this study it is assumed that all of the
CH₄ collected is used for power production. The methanogenic phase of organic waste decomposition is a steady process and typically continues for 20 years.

The LFG generation flow rate is not linear and depends on several parameters such as composition of waste, the mass of solid, water content, pH and ambient temperature. The percentage of LFG collected in a landfill site is the collection efficiency. The LFG collection starts within 1 year of the waste being deposited. The collection efficiency depends on factors such as the operational conditions and cover type. In an LFG unit with intermediate soil cover, LFG collection efficiency will range from about 55%–95% [271]. In more advanced facilities with bioreactor LGF cells, collection efficiency could reach close to 100% [272, 273]. The LGF collection efficiency in this study is assumed to be a mid-range of 75% [274, 275]. This rate has been recommended as default collection efficiency for new landfill sites by the U.S Environmental Protection Agency [276]. Waste cover is an important factor that should be designed to minimise risks of fire, LFG leakage, odour and health. At the end of each waste disposal cycle, the site should be carefully covered. The leakage, especially when the site is still being supplied with waste, makes the collection efficiency less than 100%. Every Mt of MSW is estimated to produce approximately 140 litre/second LFG [277]. In some landfills such as bioreactors, more moisture is injected into the well to accelerate the LFG production. In this study it is assumed that no interference in the decomposition process is applied. The landfill is capped to prevent any contact of the surrounding environment with the waste. The whole landfill is protected from the surroundings by a landfill liner. The leachate collection system prevents damage from leachate overflow and leakage. All the capping and covering is done after the site has reached its full capacity, by using geo-membrane capping and liner or a compacted clay liner and then soil and vegetation on top [278].

The LFG collection system can be configured as vertical wells or horizontal collectors. Vertical wells are more commonly used as they are more reliable and less likely to be damaged by flood or failure. However, they need more complex equipment that might increase costs compared to horizontal wells. In this study, a vertical well was assumed. The vertical well is schedule 80 PVC pipe with diameter of 100 cm extracting the LFG at a point ¾ of the depth of the site depth. Vacuum blowers send the LFG into pipelines for transportation to a gas cleaning facility [279].

After extraction, LFG requires dehumidification and removal of impurities and particulates. In this study LFG is combusted in a gas turbine, which means siloxane and hydrogen sulphide
must also be removed. Siloxane in the waste converts to SiO$_2$ during LFG combustion and deposits on the internal surfaces of gas turbines causing damage and increasing maintenance, while sulphur compounds in the waste lead to formation of corrosive sulphides.

The typical methods for impurity removal are adsorption beds and biological scrubbers. Primary treatment includes drying and filtering of the particulate matter. Secondary treatment removes siloxanes and sulphur compounds.

In this study, a 10 MW gas turbine was assumed for LFG combustion. One of the benefits of LFG combustion is elimination of non-methane volatile organic compounds (NMOC) which reduces their related health risk. Other benefits include avoiding emission of SOx and NOx, particulate matter and trace elements. A combined cycle gas turbine which utilises the exhaust heat from the turbine to produce electricity has an efficiency up to 40% [277]. Gas turbines have relatively low O&M cost compared to internal combustion engines and are more corrosion resistant.

### 4.2.3 Carbon capture and storage system

The CO$_2$ capture process in our model is post-combustion chemical absorption using MEA. After separation, the CO$_2$ is compressed to 150 bars and sent to the CO$_2$ sink and stored underground in suitable geological formations. Pipelines and compression equipment for transporting the CO$_2$ and injection wells are components of the CO$_2$ storage system. The CO$_2$ capture and storage processes are identical in the MSW-CCS and LFG-CCS models.

#### 4.2.3.1 CO$_2$ capture unit

A conventional MEA-based system comprising a pre-treatment unit, CO$_2$ capture unit and a CO$_2$ compression unit is used in this study. MEA is a benchmark solvent in post combustion CO$_2$ capture from coal-fired power plants. It is commercially available and has been extensively used in industrial applications [153, 227, 280, 281].

The pre-treatment unit consists of a selective catalyst reactor (SCR) equipment to remove NOx (NO and NO$_2$) and a flue gas desulphurisation unit (FGD) to remove SOx (SO$_2$ and SO$_3$) content of the flue gas. SOx and NOx damage the MEA system by forming corrosive stable salts. The SO$_2$ content of flue gas after FGD is around 30 ppm$_v$ [227], so a secondary FGD is required to clean the SO$_2$ to a level (10 ppm$_v$) at which the MEA system is not adversely affected. Fly ash is removed using cyclones and suspended particulates are collected by the
electrostatic precipitators (ESPs) [228]. After the pre-treatment unit the flue gas goes through an absorber column. In the pre-treatment unit the temperature of the flue gas which after leaving the FGD units is around 58 °C is cooled down to the level suitable for the MEA absorber column. In the absorber, CO₂ exothermically reacts with 30 wt% aqueous MEA solution, using a lean solvent loading of 0.25 mol CO₂/mol MEA and forms a water-soluble salt. The CO₂ rich solvent goes into the stripper and is heated to 100- 140 °C at close to atmospheric pressure. In the stripper the CO₂ desorbs from the MEA solution and leaves the stripper saturated with water. After the capturing process a stream of CO₂ with > 99% purity is compressed to 150 bars and sent for transportation and storage. It is assumed that 90% of the CO₂ is sequestered from the flue gas. This rate of CO₂ capture for a MEA system with the abovementioned characteristics has been frequently reported in the literature [72, 75, 227, 282]. The recovered MEA is cooled down to 35 °C and returns to the absorber. Water and MEA vapour are washed from the CO₂-lean gas leaving the absorber and then released to the atmosphere. Steam heating required for the stripper, and pumping the liquid is the source of energy penalty in a MEA system. The total energy required to run such a system is around 3.9 GJ/t CO₂, the water used for cooling is 106 m³/t CO₂ and MEA solvent is 20 m³/t CO₂ [153]. Such MEA capture systems could decrease the thermal efficiency of the plant by 25-40% and increase the costs of electricity generation by 70–100% [227]. The thermal energy requirement decreases with increasing lean solvent loading. Abu-Zahra et al [153] studied the effect of lean solvent loading and showed that higher loading of 0.32–0.33 mol CO₂/mol MEA results in a lower thermal energy requirement of 3.45 GJ/t CO₂.

4.2.3.2 CO₂ transport and storage system

For the purposes of this study, it is assumed that CO₂ is transported via a carbon-manganese steel pipeline with an external diameter of 600 mm and a thickness of 20 mm [283], to an onshore storage site. CO₂ at supercritical conditions (above 32 °C and 73 atm with a density of CO₂ approaching 1000 kg/m³) is transported via pipelines [68, 284]. Pipeline transportation is more viable and mature for onshore transportation and is especially suitable when the CO₂ source is a power plant with a lifetime of more than 20 years [68]. The pipeline is buried in a trench of 1 m depth and equipped with over-pressure protection and leakage detection systems and block valves every 16–32 km [68, 70]. In this study, the cost of transporting CO₂ along a 500 km pipeline is around $8/t for a mass flow of 25 Mt CO₂/year [68, 80, 283].
Regions with sedimentary basins are potentially suitable for CO\textsubscript{2} storage [70], but tectonic setting, geology (reservoirs, seals, porosity, permeability), and hydrology of the basin are all factors that need to be considered when assessing the suitability of an area for geological storage of CO\textsubscript{2}. For the purpose of this modelling, it is assumed that the CO\textsubscript{2} is stored in a sedimentation storage well at a depth of 1500 m, that CO\textsubscript{2} injection pressure is monitored and that there is ongoing monitoring to ensure that there is no leakage of CO\textsubscript{2} from the well or from the storage formation with the expectation that over 99% of injected CO\textsubscript{2} will be retained in the storage reservoir over 1000 years [56]. Any contribution to global warming from release of stored CO\textsubscript{2}, acidification of underground water or damage to ecosystems is judged to be very low provided the storage site is well characterised and best practice is followed for all on-site operations and monitoring. In this study it is assumed the total cost of CO\textsubscript{2} injection, storage and monitoring is $15/t CO\textsubscript{2} [68, 80, 285]. However, parameters such as depth, permeability of the formation, and transport distance all impact on this base cost of storage.

4.2.4 Cost of Electricity Production

To investigate the economic viability of MSW-CCS and LFG-CCS electricity production systems, techno-economic assessments were conducted. The levelised cost of electricity production (LCOE) was calculated for each system. The LCOE of black coal combusted in a pulverised coal combustion plant (Coal-PC) with and without CCS technology were used as the baseline.

LCOE\textsubscript{default} [$/MWh] was calculated using Equation 4-1. All the costs in this study are in USD. The levelised cost of electricity production is the capital and operating costs spent in a period of time (a year) per the net amount of electricity generated in that period. LCOE\textsubscript{default} is the cost of electricity production in the absence of any emission policy or subsidy;

\[
LCOE\textsubscript{default} = \frac{\text{capital cost} \times \frac{(1-i \rho)}{1-[1+(1+i \rho)^{-N}]} + O&M_{\text{fixed}} + CF \times O&M_{\text{variable}}}{CF \times MWh} \quad \text{(Equation 4-1)}
\]

In Equation 4-1, O&M\textsubscript{fixed} is the sum of all fixed annual operating costs; O&M\textsubscript{variable} represents the sum of all variable annual operating costs, including fuel, using a 100 percent capacity factor. CF is the plant capacity factor (assumed to be constant over the operational period), MWh represents the annual net megawatt-hours of power generated at 100 percent capacity factor. i is the interest rate, \( \rho \) the inflation rate and N the investment lifetime. Techno-economic data of the different processes used in these models are provided in Table 4-1.
To be able to evaluate the impact of different emission policies on LCOE, Equation 4-2 is applied. As can be seen, carbon price \( P_{\text{carbon}} \), revenues from negative \( \text{CO}_2 \) emission \( P_{\text{negative} \ \text{CO}_2} \) and renewable energy certificates (REC) are included as possible emission policies in this study.

\[
LCOE = LCOE_{\text{default}} \left[ \frac{\$}{\text{MW}h} \right] + P_{\text{carbon}} \left[ \frac{\$}{\text{tCO}_2} \right] \times \text{Emission} \left[ \frac{\text{tCO}_2}{\text{MW}h} \right] - \text{REC} \left[ \frac{\$}{\text{MW}h} \right] - P_{\text{negative} \ \text{CO}_2} \left[ \frac{\$}{\text{tCO}_2} \right] \times \text{negative Emission} \left[ \frac{\text{tCO}_2}{\text{MW}h} \right]
\]

(Equation 4-2)

Because the electricity generated from MSW incineration and LFG combustion is accepted as renewable energy, these technologies receive renewable energy credits. Additionally, the \( \text{CO}_2 \) emission from organic waste is from biogenic sources and no \( \text{CO}_2 \) tax is imposed on it. Moreover, the \( \text{CO}_2 \) captured and stored will be counted as negative carbon and it is assumed that it will receive credit for negative emission. Table 4-1 lists the techno-economic data for MSW, LFG and Coal-based power plants- with and without CCS-, based on the best information available in the literature.

Cost of \( \text{CO}_2 \) emission avoided is the cost of the CCS processes added per tonne of \( \text{CO}_2 \) captured; see Equation 4-3;

\[
\text{Cost of } \text{CO}_2 \text{ avoided} = \frac{LCOE_{\text{CCS}} - LCOE_{\text{(no CCS)}}}{\text{CO}_2 \text{ (no CCS)} - \text{CO}_2 \text{ (CCS)}}
\]

(Equation 4-3)

The net \( \text{CO}_2 \) emission per megawatt-hour of electricity (tonne \( \text{CO}_2/\text{MWh} \)) generation is calculated according to Equation 4-4; where \( \text{CO}_2 \), is the total \( \text{CO}_2 \) emission of the power plant, \( \eta_{\text{CO}_2, \text{capture}} \) is the efficiency of carbon capture process and \( X_b \) is the fraction of biogenic \( \text{CO}_2 \). It is assumed all the biogenic \( \text{CO}_2 \) is GHG neutral, hence if it is emitted to the atmosphere it is equal to zero and if captured and stored underground is regarded as a negative emission.

\[
\text{Net } \text{CO}_2 \text{ emission} = \frac{\text{CO}_2 \times (1 - \eta_{\text{CO}_2, \text{capture}}) \times (1 - X_b) - \text{CO}_2 \times \eta_{\text{CO}_2, \text{capture}} \times X_b}{\text{MWh}}
\]

(Equation 4-4)

In this study, \( \eta_{\text{CO}_2, \text{capture}} \) of the MEA capture unit is 90%. \( X_b \) depends on the organic content of the feedstock to the power plant. In this study \( X_b \) for LFG is assumed to be 1. Plastic and other inert components such as glass and metal make 37 % of MSW. It assumed that the carbon content of plastic, which is non-biogenic carbon, is 0.63 wt%. Therefore, the total non-biogenic \( \text{CO}_2 \) content of the outlet \( \text{CO}_2 \) is 17 %.
### Table 4-1- Techno-economic input data

<table>
<thead>
<tr>
<th>Process</th>
<th>Capacity (MW)</th>
<th>Thermal efficiency (% (LHV))</th>
<th>Capacity factor (%)</th>
<th>Lifetime (years)</th>
<th>Capital ($/kW)</th>
<th>Variable O&amp;M ($/kW/yr)</th>
<th>Fixed O&amp;M ($/kW/yr)</th>
<th>Fuel cost (delivered) ($/t)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LFG collection-cleaning plant [19][286]</td>
<td>10</td>
<td>Mean: 35</td>
<td>70</td>
<td>25</td>
<td>Mean: 1500</td>
<td>Mean 3.6 ($/MWh)</td>
<td>Mean: 57</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 28-42</td>
<td></td>
<td></td>
<td>Range: 1350-1650</td>
<td>Range: 2.9-4.3 ($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>LFG power plant gas turbine [286, 287]</td>
<td>10</td>
<td>Mean: 30</td>
<td>90</td>
<td>25</td>
<td>Mean: 6000</td>
<td>Mean: 29 ($/tonne MSW)</td>
<td>Mean: 192</td>
<td>0</td>
</tr>
<tr>
<td>MSW plant [19][254, 288]</td>
<td>10</td>
<td>Mean: 30</td>
<td>90</td>
<td>25</td>
<td>Mean: 2400</td>
<td>Mean: 4.5 ($/MWh)</td>
<td>Mean: 36</td>
<td>Mean: 50</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 40-44</td>
<td></td>
<td></td>
<td>Range: 2100-2700</td>
<td>Range: 3.3-5.7 ($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coal plant [14][80, 289]</td>
<td>150</td>
<td>Mean: 42</td>
<td>90</td>
<td>30</td>
<td>Mean: 2100</td>
<td>Mean: 9 ($/MWh)</td>
<td>Mean: 103</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 40-44</td>
<td></td>
<td></td>
<td>Range: 1760-2440</td>
<td>Range: 5.6-12.4 ($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Capture process [69, 290-292]</td>
<td>150</td>
<td>Mean: 35</td>
<td>100</td>
<td>30</td>
<td>Mean: 1600</td>
<td>Mean: 9 ($/MWh)</td>
<td>Mean: 103</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 40-44</td>
<td></td>
<td></td>
<td>Range: 1760-2440</td>
<td>Range: 5.6-12.4 ($/MWh)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

LFG pipeline and condensation = 205 ($/km) [286]
CO₂ Transportation (500-2000 km) = 8 ($/tonne CO₂) [293]
CO₂ Storage and monitoring = 15 ($/tonne CO₂) [70]
CO₂ capture efficiency of the MEA unit is 90%
Interest rate (i) =10%
Inflation rate (ρ)=0%

Compositions of MWS and black coal are listed in Table 4-2 and 4-3.

### Table 4-2- Elemental compositions of MSW and black coal [80, 254, 288, 289, 294]

<table>
<thead>
<tr>
<th>Composition</th>
<th>MSW</th>
<th>Black coal</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td>C (wt%)</td>
<td>49.50</td>
<td>45.50-53.50</td>
</tr>
<tr>
<td>H (wt%)</td>
<td>5.60</td>
<td>3.80-7.40</td>
</tr>
<tr>
<td>O (wt%)</td>
<td>32.40</td>
<td>26.50-38.30</td>
</tr>
<tr>
<td>N (wt%)</td>
<td>1.33</td>
<td>0.90-1.76</td>
</tr>
<tr>
<td>S (wt%)</td>
<td>0.51</td>
<td>0.40-0.62</td>
</tr>
<tr>
<td>Moisture</td>
<td>34.20</td>
<td>29.50-38.90</td>
</tr>
<tr>
<td>LHV (MJ/Kg)</td>
<td>16.80</td>
<td>30.41</td>
</tr>
</tbody>
</table>

74
Table 4-3: Composition of MSW [245, 250, 256, 269, 270, 272, 295]

<table>
<thead>
<tr>
<th>Composition of MSW</th>
<th>weight %</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
</tr>
<tr>
<td>Food</td>
<td>29.4</td>
</tr>
<tr>
<td>Mixed paper</td>
<td>17.0</td>
</tr>
<tr>
<td>Garden and green</td>
<td>14.3</td>
</tr>
<tr>
<td>Textiles</td>
<td>1.4</td>
</tr>
<tr>
<td>Rubber and leather</td>
<td>0.9</td>
</tr>
<tr>
<td>Plastic</td>
<td>26.5</td>
</tr>
<tr>
<td>Inert waste (metal, glass, etc.)</td>
<td>10.5</td>
</tr>
</tbody>
</table>

4.2.5 Environmental impact Assessment

Life cycle assessment (LCA) was carried out using SimaPro software version 8.2.0.0 developed by PRé Sustainability Group. The LCA study includes four main steps; defining the goal and scope, life cycle inventory of all the inputs and outputs, lifecycle impact assessment, and interpretation of the results. Allocation modelling was applied to assess the environmental impacts of the two wastes – based BECCS systems with the same functional unit of 1 kg of the wet waste delivered to the landfill. In these assessments, the ALCAS Best Practice LCIA method [296] was used. ALCAS uses the GWP (100) according to IPCC’s fifth assessment report [247]. In order to see the effect of the method, the ReCiPe Hierarchist (ReCiPe Midpoint (H) V1.12 / World Recipe H) [297] method was applied. World ReCiPe midpoint Hierarchist method represents normalisation values of the world. The impact categories considered in this study were Global warming (GWP100a), Abiotic depletion (Fossil fuels), Ozone layer depletion (ODP), Eutrophication, Acidification, Particulate matter, Human toxicity (cancer), Human toxicity (non-cancer), Freshwater ecotoxicity and water scarcity. Most of the data regarding MSW incineration and LFG collection and combustion facilities are taken from Ecoinvent-3 [61] and the AusLCI database (developed by Australian LCA Society (ALCAS)) [298]. Data regarding the MEA CO₂ capture unit was derived from the project conducted by Hooper et al. [299, 300]. CO₂ and CH₄ emissions based on the degradable organic content (DOC) of the waste were calculated based on methodology used in the National Inventory Report [269]. LCA of reference systems without CCS (MSW-CFB and LFG-GT) and BECCS systems (MSW-CCS and LFG-CCS) were modelled. Table 4-3 shows the average composition of MSW globally. The corresponding data for the electricity mix, construction materials and
background processes, changes for each context. All the other data regarding operational conditions of the plants are the same.

Figures 4-1 and 4-2 illustrate the two systems with the major components and the system boundaries. In both cases MSW generation and transportation is outside of the system boundary. In Figure 4-1, MSW is combusted in an incineration plant to produce electricity, which is exported to the grid. Figure 4-2 shows the model used for landfill gas generation and combustion in a gas turbine with CO₂ capture and storage (LFG-CCS).

![System boundary for MSW-CCS inventories](image)

**Figure 4-1- System boundary for MSW-CCS inventories**
All the waste after separation of the reusable and recyclables are transported to the waste storage site. The waste is assumed to be composed of some inorganic material (17%), mainly plastic. The CO₂ emission from plastic is non-biogenic and is deducted from the total incineration plant CO₂ emission.

In this study, waste decomposition and methane generation in a sanitary landfill site was based on first order decay (FOD) model, as proposed by IPCC Guidelines for National Greenhouse Gas Inventories 2006 [60]. Organic decomposition in a LFG site produces mainly CH₄ and CO₂ and small percentages of H₂S, non-methane volatile organic compounds (NMVOC-around 2%), N₂O, NOx and CO.

In FOD the reaction rate is proportional to the amount of degradable organic carbon decomposable under anaerobic conditions and it is assumed that degradable organic content of the waste decays over a few decades. Therefore, during the first decade of landfill, most decomposition occurs. The timeframe for complete decomposition in FOD is recommended to be at least 50 years. In this study approximately 99% of the decomposition was completed during the first two decades after landfilling. The lifetime of the sanitary landfill collection site and the LFG combustion plant is 25 years. Hence, the time for total decomposition in this
study is 25 years. The amount of CH\textsubscript{4} and CO\textsubscript{2} generated during the lifetime of the sanitary landfill site are calculated via equations 4-5 to 4-9 [60].

\begin{align*}
\text{CH}_4\text{--captured} &= \text{CH}_4\text{--generated} \times \eta_{LFG\text{--capture}} \quad \text{(Equation 4-5)} \\
\text{CH}_4\text{--generated} &= \sum_{t=0}^{n} W_t \times \text{DOC} \times \text{DOCF} \times \text{MCF} \times F \times \frac{16}{12} \times (1-e^{-k}) \quad \text{(Equation 4-6)} \\
\text{CH}_4\text{--emitted to the ambient} &= \text{CH}_4\text{--generated} \times (1-\eta_{LFG\text{--capture}}) \times (1-\text{Ox}) \quad \text{(Equation 4-7)} \\
\text{CO}_2\text{--generated} &= \text{DOC} \times \text{DOCF} \times (1-F) \times (1-\text{NMVOC}) \frac{44}{12} \quad \text{(Equation 4-8)} \\
\text{CO}_2\text{--emitted to the ambient} &= \text{CO}_2\text{--generated} \times (1-\eta_{LFG\text{--capture}}) \quad \text{(Equation 4-9)}
\end{align*}

In the equations above;

\begin{itemize}
    \item \text{CH}_4\text{--generated}: total methane generated during the lifetime of the landfill site [tonne]
    \item \text{CH}_4\text{--captured}: total methane collected from the landfill site [tonne]
    \item \eta_{LFG\text{--captured}}: efficiency of the LFG capture, which is 75\% in this study
    \item t: time [year]
    \item F: fraction of CH\textsubscript{4}, by volume, in generated landfill gas (fraction) , assumed 50\% in this study
    \item 16/12: molecular weight ratio CH\textsubscript{4}/C
    \item 44/12: molecular weight ratio CO\textsubscript{2}/C
    \item W\(_t\): mass of waste disposed in year \(t\) [tonne]
    \item DOC: degradable organic carbon in the year of deposition [tonne C/ tonne waste]
    \item DOC\(_f\): fraction of DOC that can decompose
    \item MCF : CH\textsubscript{4} correction factor for aerobic decomposition in the year of deposition, for managed anaerobic site MCF equals 1 [60]
    \item k: reaction constant
    \item Ox: Landfill oxidation factor, assumed zero in this study
    \item NMVOC= Non-methane volatile organic compounds emission is assumed to be 2\% of LFG [301]
\end{itemize}
The final waste is assumed to be inert, after complete decomposition of the degradable carbon content of the waste. Equation 4-10 is used to calculate the amount final waste left after the decomposition is completed [270].

\[ \text{Final waste} = W_t \times (1 - \text{DOC} \times \text{DOC}_f) \]  
(Equation 4-10)

DOC, DOCf, k and moisture content of the waste materials is provided in Table 4-4.

Table 4-4: Degradable organic content (DOC), fraction of the degradable organic carbon that decomposes under anaerobic conditions (DOCf), degradation rate (k) and moisture of the waste contents [270, 301]

<table>
<thead>
<tr>
<th>Waste Type</th>
<th>DOCf</th>
<th>DOC</th>
<th>K</th>
<th>Moisture content</th>
</tr>
</thead>
<tbody>
<tr>
<td>Food</td>
<td>0.84</td>
<td>0.15</td>
<td>0.197</td>
<td>0.7</td>
</tr>
<tr>
<td>Mixed paper</td>
<td>0.49</td>
<td>0.4</td>
<td>0.06</td>
<td>0.12</td>
</tr>
<tr>
<td>Garden and green</td>
<td>0.47</td>
<td>0.2</td>
<td>0.104</td>
<td>0.12</td>
</tr>
<tr>
<td>Textiles</td>
<td>0.5</td>
<td>0.24</td>
<td>0.06</td>
<td>0.12</td>
</tr>
<tr>
<td>Rubber and leather</td>
<td>0.5</td>
<td>0.39</td>
<td>0.06</td>
<td>0.12</td>
</tr>
</tbody>
</table>
4.3 Results

End-of-pipe CO₂ emission from LFG-GT, MSW-CFB and Coal-PC, with and without CCS, is illustrated in Figure 4-3. The CO₂ emissions in Figure 4-3 are calculated using Equation 4. It is assumed that all the LFG combusted is from an organic source therefore it is CO₂ neutral. In the LFG-CCS case, for each MWh produced, 1.35 tonnes CO₂ is captured, which all is counted as negative CO₂. 17% of the carbon content in MSW comes from non-organic material and the CO₂ emission from this source is considered as non-biogenic. Hence, the emission without CCS is positive (0.12 t CO₂/ MWh). This reduces to -78 t CO₂/ MWh when CCS is added to a MSW incineration plant, which means the total CO₂ avoided by MSW-CCS is 0.9 tonnes CO₂/ MWh. Adding CCS to a coal plant results in 0.7 t CO₂/ MWh avoided. Though Coal-CCS system benefits from better combustion parameters (see Table 4-1), both BECCS options have higher avoided CO₂ per MWh generated.

![Figure 4-3: CO₂ emission from LFG-GT, MSW-CFB and Coal-PC with and without CCS](image)

4.3.1 Cost of electricity production

The levelised cost of electricity (LCOE) was calculated for MSW-CFB and LFG-GT and Coal-PC, with and without CCS, in the business-as-usual (BAU) scenarios, with no emission policy that would serve to drive uptake of these technologies. Figure 4-4 shows the LCOE (see
Equation 4-1) and cost of avoided CO\textsubscript{2} (see Equation 4-3) for these systems under the baseline scenario.

![Figure 4-4 LCOE with and without CCS and cost of CO\textsubscript{2} avoided under business-as-usual scenario](image)

As seen, under the business as usual scenario, with no emission reduction policies, in both cases with and without CCS, the LCOE in pulverised coal combustion (Coal-PC) power plants is lower than LFG in gas turbine (LFG-GT) and MSW incineration plants in circulating fluidised bed (MSW-CFB). The baseline used to assess the economic feasibility of these options is coal-based power generation. Coal power is used as the reference because of its high share of the global electricity market. In the year 2013-2014, power generation from coal was 40.8% globally [302]. PC plants benefit from economy of scale and higher thermal efficiency. Adding CCS increases the LCOE by $72–$95/MWh. The LCOE of LFG is lower than MSW, mainly due to lower O&M cost for a GT compared to CFB. The increase in cost by adding CCS is almost the same for MSW-CFB and Coal-PC. For LFG-GT, adding CCS produces the highest cost of $95 to each MWh of electricity produced. However, when the cost of avoided CO\textsubscript{2} emission is taken into account LFG-CCS has the lowest cost ($70/t CO\textsubscript{2}) compared to MWS-CCS ($81/t CO\textsubscript{2}) and Coal-CCS ($103/t CO\textsubscript{2}). That is due to higher emissions avoided through BECCS compared to Coal-CCS (see Figure 4-3). With no emission policy, the BECCS
options studied here, are significantly more costly and are unlikely to be competitive in a coal-dominant power market.

In the second scenario, the impact of introducing a carbon pricing scheme was studied. It was assumed that $P_{\text{negative CO}_2}$ and REC are equal to 0. Carbon pricing has been one of the policies used to mitigate the CO$_2$ emissions from the energy sector. Figure 4-5 illustrates how the LCOE of MSW-CCS, LFG-CCS and Coal-CCS changes with carbon price. Since electricity from LFG and MSW is considered as renewable, it is exempted from carbon pricing as used in number of countries to encourage mitigation of CO$_2$ emission from the power and industry sectors in particular, whether through a carbon tax or a cap-and-trade system. The European emission trading system (EU ETS) [108] includes 31 countries and covers a wide range of emitting sectors. The price of the carbon emission permit currently fluctuates around 5.6 ± 0.6 USD per tonne CO$_2$ [109]. Under a carbon tax scheme, the carbon price required to make MSW-CCS and LFG-CCS price-competitive with coal (with or without CCS) is between $130–$200 per tonne of CO$_2$. Considering the status of carbon pricing schemes around the world, it seems highly unlikely that such a carbon price will be implemented in the near future because of strong political and commercial opposition.

![Figure 4-5- Impact of carbon price on LCOE of MSW-CCS, LFG-CCS, Coal-CCS and coal w/o CCS](image)

82
The other emission policy investigated in this study was a renewable energy certificate scheme (REC). Under such a scheme, certificates (RECs) are given per megawatt-hour of electricity produced from recognised renewable power suppliers. Renewable Energy Certificate schemes have been successfully implemented in a number of countries including, the U.S, Australia and especially in Europe, where they have strongly supported deployment of renewable electricity. To apply the RET, REC for each 1 MWh of compliant renewable energy are issued. In other words, energy retailers and large energy users are obliged to purchase a share of their electricity from renewable energy producers. Under the RET, energy from wind, solar, geothermal, hydroelectricity, and bioenergy are recognised as renewable sources. It is assumed under RET both LFG-CCS and MSW-CCS are eligible for RECs. Looking at the results in Figure 4-6, the value of a REC has a significant impact on the cost of electricity production from BECCS. At a REC higher than $80/MWh, the cost of electricity production from both MSW-CCS and LFG-CCS becomes cheaper than Coal-CCS. At a REC higher than $150/MWh BECCS technologies become cheaper than coal-based power (without CCS). The REC is set through a deregulated power market and its value depends on factors such as policies regarding renewable energy incentives, supply and demand, renewable energy standards and technological changes [303]. It potentially makes the cost of MSW-CCS and LFG-CCS cheaper than some of the other renewable technologies such as onshore wind with a global average LCOE of $90-$130/MWh, or solar PV with a LCOE of $120–$180/MWh [80, 304].
In the fourth scenario, the only policy implemented is a negative CO₂ refunding scheme. In this scenario, BECCS technologies receive a refund for each tonne of CO₂ removed from the atmosphere. In such a scheme, CO₂ produced from bioelectricity generation is biogenic and biogenic CO₂ is taken as neutral GWP. Therefore, when it is captured and stored, it is counted as a negative emission. Since negative emission achieved through the application of BECCS is a relatively new concept, there is currently no scheme that takes the value of negative emission into account. Again, in this case the Coal-CCS plant does not receive credit for the CO₂ captured, but LFG-CCS and MSW-CCS benefit from this scheme. As shown in Figure 4-7, at a Pₚₙₑₐ₅ᵋₐ₅ᵋₑ₅ₑtant CO₂ about $80/t CO₂-negative, with biogenic CO₂ captured, the LCOE of both LFG-CCS and MSW-CCS become lower than Coal-CCS. A negative emission scheme has a higher impact on LFG-CCS due to the larger amount of CO₂ captured by this technology. Around 1.35 t CO₂/MWh is captured from end of pipe in LFG-CCS. When Pₚₙₑₐ₅ᵋₐ₅ᵋₑ₅ₑtant CO₂ is higher than $120/t CO₂-negative the revenues from LFG-CCS exceed the costs and its LCOE becomes negative. CO₂ captured form MSW-CCS is 0.9 t CO₂/MWh so this system reacts to Pₚₙₑₐ₅ᵋₐ₅ᵋₑ₅ₑtant CO₂ emission at a lower rate.
Each of these policies has its advantages and drawbacks. Among the policies that introduce cost or revenues based on CO$_2$ emissions, a negative emission refunding scheme and RET have a similar effect on the generation costs of the BECCS options and impose lower costs than a carbon pricing scheme. However, the feasibility of implementing these various schemes is very different. Although carbon pricing policies already exist and are still in effect in some countries, increasing it to the level required for BECCS to become economically viable far exceeds the current values and would demands mitigation ambitions from the policy makers that are far in excess of anything at the present time. On the other hand, currently there is no emission policy that encourages negative emission. Both schemes rely on measuring the CO$_2$ emitted or captured accurate monitoring measures and agreed standards.

The cost of waste-based BECCS technologies could be reduced by introducing a tipping fee into the cost assessment. Waste is considered as a negative externality that must be priced for all the collection and management processes. Under such a system, waste producers have to pay a tipping or gate fee per tonne of waste delivered to the WTE producers for recovering the waste energy. Tipping fees, usually set by a contract, are a source of income for WTE plants and constitutes a considerable proportion of their income. However, the tipping fee must be
lower than the landfill fee to encourage producers and therefore sometimes it is subsidised by the authorities. Currently, WTE tipping fees are highest in the UK ($148 /t) compared to a landfill fee of $153 /t. This is much lower in U.S ($68 /t) [246]. Another source of income for incineration WTE is from selling the ash from waste incineration. The ash is mostly composed of non-combustibles such as metals and glass that could be used in construction (road) and in the scrap metal market.

4.3.2 Environmental impact assessment

The LCA of the reference systems (LFG-GT and MSW-CFB) and BECCS systems (LFG-CCS and MSW-CCS) systems was modelled. Biogenic carbon flows are inventoried and reported separately from fossil-based carbon flows. CO₂ emissions from biogenic sources are assumed to be “carbon-neutral” in LCA studies. However, biogenic GHG flows are included. Figure 4-8 illustrates the comparative impact assessment of the LFG-GT, MSW-CFB, LFG-CCS and MSW-CCS modelled in this study.

As can be seen, in both BECCS systems the net global warming effect is significantly reduced compared to the baseline systems. MSW-CCS has the lowest global warming impact and LFG-GT has the highest GHG emission. The net GHG reduces considerably in LFG-CCS compared to LFG-GT; however, it is still positive net emission. This is mainly due to the fugitive methane from the landfill site. As discussed in the methodology section the efficiency of LFG collection is 75%. In other words, 25% LFG escapes to the atmosphere, of which around 50% is CH₄ and some sulphur compounds. This amount of LFG emitted results in higher acidification, photochemical oxidation and human toxicity impacts. Water scarcity is higher for both BECCS options compared to no CCS systems, which is mainly due to high water used (106 m³/t CO₂) for cooling in the MEA process. However, Table 4-5 shows that these variations are minimal, though because of scaling in Figure 4-8 the differences are scaled up.
The MSW-CCS systems create net negative emissions at around -0.7 kg CO$_2$ eq per kg of wet MSW incinerated. The net emission of LFG-CCS systems is approximately 0.56 CO$_2$ eq per kg of wet MSW delivered to the sanitary landfill site. It is important to note that biogenic emissions are considered in these assessments. The positive CO$_2$ emission in the LFG-CCS systems is due to the non-captured LFG. Methane is a more potent GHG than CO$_2$; with a global warming potential (GWP) factor of 25.25 CO$_2$ equivalent [247].

In these systems the electricity produced from MSW-CCS and LFG-CCS is sent to the grid and substitutes for the power generation mix from dominantly fossil fuel-based suppliers. Therefore, both MSW-CCS and LFG-CCS systems have a positive impact on depletion of abiotic resources by replacing fossil fuel used for power generation. The global average share of fossil fuel-based power generation is approximately 66.7% [302] in the electricity mix.
Figure 4-9 illustrates the apportioned global warming impact of different sub-processes in the LFG-CCS and MSW-CCS system per functional unit (1 kg waste delivered). The cut-off rule for CO$_2$-eq emission was 5%, i.e. any process contributing less than 5% of the net GHG emission is not included in Figure 4-9. The sum of the total positive and negative CO$_2$-eq emissions is the same as in Table 4-5. As seen the main source of negative emissions is from electricity sent to the grid; it is assumed the renewable electricity produced by these BECCS systems replaces electricity mix in the grid which is mainly supplied by fossil fuel–based power plants. In both cases CO$_2$ capture and transport contributes to positive CO$_2$ emission. The major difference is the GHG emission due to the fugitive LFG from the sanitary landfill site which leads to a net positive CO$_2$ emission for LFG-CCS. Additional processes required for sanitary landfill construction and pipelines to transport the LFG to the power station result in 0.05 kg CO$_2$-eq emission.
The environmental impact assessment shows that MSW-CCS technology has the potential to provide net negative emissions. However, the model is based on assumptions regarding technological data and waste composition and changing these parameters could have a significant impact on the negative emission potential of MSW-CCS. One crucial factor is the impact assessment method used. When ReCiPe Midpoint (H) method was applied, net negative emission for MSW-CCS was -0.2 kg CO$_2$-eq for each kg of MSW, considerably higher net negative GHG compared to the ALCAS method; In case of LFG-CCS, net CO$_2$-eq under ReCiPe Midpoint (H) method was 0.64 per kg of waste.

The LFG produced in a LFG well is collected and sent to the power station. However, not all of the LFG is captured. In a sanitary LFG site with an intermediate cover, the collection efficiency is in the range of 55%–95% [271-273]. The remainder is released to the atmosphere with 50% methane content, which has GWP of 25.25 CO$_2$-eq. In these assessments the mid-range default collection efficiency of 75% was chosen. To see the impact of LFG collection efficiency on the net CO$_2$-eq emission, a sensitivity analysis using a high (95%) and low range (55%) collection rate in the LCA models was conducted. It was seen that using 55% LFG...
capture rate increased the CO\textsubscript{2}-eq emission per each kg of MSW used in LFG-CCS model from 0.56 to 1.42 kg CO\textsubscript{2}-eq. On the other hand, a high capture rate of 95\% reduced the net CO\textsubscript{2}-eq emission to -0.29 kg. Based on these results, it can be said that LFG collection efficiency is one of the most crucial factors influencing the net emissions in an LFG-CCS system.

In this study potential of the WTE systems coupled with CCS to deliver negative emission was assessed assuming that biogenic CO\textsubscript{2} has no GWP. This assumption was based on the common perception that biomass is generally a carbon neutral energy resource because the CO\textsubscript{2} emissions from biomass combustion will be absorbed by plants through photosynthesis [46, 50, 60, 61]. This has recently been challenged especially for some long rotation biomass resources (especially from forestry) [62], which the rate of harvest might exceed the time required for the plant to be replenished [63]. It is less likely to have a significant impact on net CO\textsubscript{2} emission from the WTE systems. The organic portion of MSW in this study is composed of waste from high value products. Hence, the potential environmental impacts associated with production of dedicated energy crops, such as disturbing the natural carbon cycle, are avoided in the MSW-based BECCS systems.
4.4 Discussion

Based on the estimate of generating 1.3 Gt MSW globally and a heating value of 16.8 MJ/kg used in this study, MSW incineration has the energy value of 21.8 EJ. This translates to around negative 0.91 Gt CO₂ if all the waste is utilised in the MSW-CCS (-0.7 kg CO₂/ kg waste) and around negative 0.38 Gt CO₂ through advanced LFG-CCS with more than 95% LFG collection efficiency (-0.29 kg CO₂/ kg waste).

The COP 21 target is to hold atmospheric increase temperatures to well below 2°C. According to IPCC models, to be able to stabilise at this level by the end of century, around 5–20 Gt CO₂-eq should be removed annually, starting from mid-century [1]. The total MSW produced annually in the world is around 1.3 Gt. This means that even if only a small proportion of the MSW generated annually is used for power generation with CCS, MSW-based BECCS technologies could contribute significantly to the COP 21 targets.

Despite the potential for considerable emissions reduction, there are challenges in the deployment of these waste-based BECCS options and these are summarised below:

- There is great diversity in the amount and the composition of MSW generation and access to modern and efficient waste-to-energy technologies around the globe. Therefore, the economic and environmental models should be built based on accurate regional and national data.
- Technical challenges regarding the dispersed nature of MSW resources, access to a reliable CO₂ sink and the necessary logistics means that waste-based BECCS systems need extensive adaptive planning.
- Though MSW compared to other biomass is a more secure feedstock, its availability depends on parameters such as human population, economic development, the degree of industrialisation, public habits, and waste management regimes. Development of more sustainable waste management paradigms such as a circular economy to minimize waste generation and maximum re-use and recycling would influence availability of waste for energy production. However, the current status of waste management is far from zero-waste ideal and if BECCS is considered as a mid-term solution, the lack of organic waste is less likely to be problematic.
- Under current energy policy paradigms, BECCS is considerably more costly than base-load fossil fuel power. Reinforcing some of the existing policies such as a renewable
energy certificate system or introducing new policies such as a negative emission refunding system would support BECCS.

- There is a very limited understanding regarding BECCS and even some antipathy regarding bioenergy and CCS among stakeholders and decision makers. It is important to address any misunderstanding as soon as possible. Specifically, there is a need to conduct a comprehensive study of the global status and potential of BECSS and its potential at regional and national levels, its risks, opportunities, and how it might contribute to energy demand and emission reduction targets.

In addition, the socio-economic impacts of these technologies should be carefully investigated and addressed. For instance, there are concerns about the proximity of the WTE plants to residential areas and unless adequate measures are taken, these plants could be a source of emissions and a risk to human health. Another concern is that policies to encourage waste to energy may lead to increasing waste generation and discourage recycling. But in practice, countries with higher standards and regulations to reduce waste and increase recycling have a higher share of WTE. There is no evidence that WTE increases unsustainable waste management. Nonetheless, it is important to consider the regional social and political waste management circumstances before implementing any WTE technology.

In terms of employment, a typical WTE plant of 50,000 tonnes per annum capacity requires 6–18 workers daily. For instance in U.S, in 2014 the WTE industry employed 5,350 workers at the 85 plants, with an additional associated 8600 jobs outside the sector [246].

The concept of capturing CO$_2$ from WTE has moved on from solely a theoretical solution to manage emissions from waste sector. Recently a waste to energy agency in the Oslo municipality (EGE) conducted a feasibility study to assess the opportunities for CO$_2$ capture from a waste incineration plant at Klemetsrud [87]. Assessments show that the project could technically capture up to 315,000 tonnes of CO$_2$ annually with 90% CO$_2$ capture rate by 2020 [87]. Around 50-60% of this CO$_2$ is biogenic. Another WTE power plant with CCS is ARV-Duiven in Duiven, the Netherlands. ARV-Duiven power plant with 70 MW capacity incinerates MSW to produce around 126 GWh electricity. From 2018, ARV-Duiven is planning to capture up to 50 Ktonnes CO$_2$ per annum using MEA capture process [88]. These two projects will be valuable contributions to the development of waste-based BECCS technologies and to understanding how to adapt existing models for other regions with different conditions.
4.5 Conclusions

In this study, the global potential of using municipal solid waste as resource for bioenergy with carbon capture and storage (BECCS) was investigated. Two BECCS systems, municipal solid waste incineration with carbon capture and storage (MSW-CCS) and landfill gas combusted in a gas turbine with carbon capture and storage (LFG-CCS) were modelled.

- In the case of business-as-usual with no emission policy in effect, both BECCS options are more costly than coal-based power generation. Since BECCS options in this modelling had higher net CO$_2$ avoided, the total cost of avoided emission ($/tonne CO$_2$) was lower than coal-CCS technology. However, introducing renewable energy certificates or a negative emission refunding system has a significant impact on the economic viability of these technologies in coal-dominant power markets.

- Environmental impact assessment of the proposed BECCS models showed that both systems are environmentally benign with considerable potential for net CO$_2$-eq emission reduction. In the case of the global MSW-CCS model, for each kg of wet MSW incinerated around -0.7 kg CO$_2$-eq was created.

- The LFG-CCS model showed significant sensitivity to the collection efficiency of the LFG. When the collection efficiency increased from a mid-range of 75% to a high range of 95% the net emission reduced from 0.56 to -0.29 kg CO$_2$-eq. This translates to around negative 0.91 Gt CO$_2$ if all the 1.3 Gt MSW generated globally per year were to be utilised in the MSW-CCS or around -0.38 Gt globally through LFG-CCS.

- Degree of advancement of waste management systems, technical and environmental standards and the policy and regulatory frameworks are the main factors influencing the choice between these waste-based BECCS options.

- Despite the considerable potential for MSW-based BECCS systems in creating negative CO$_2$ emission, the dispersed nature of MSW resources, access to cost effective and efficient waste-to-energy and CO$_2$ capture technologies and lack of sufficient economic support schemes and policy instruments are barriers to their large-scale deployment. However, under the right policy settings generating electricity through MSW-based BECCS could be used to remove a significant amount of the CO$_2$ from the atmosphere and contribute to meeting COP 21 targets.
Chapter 5

Agricultural and Forestry Residues as Resources for BECCS
5.1 Introduction

Availability of bioenergy is one of the key parameters determining the potential of BECCS in the future. The potential of bioenergy is limited by factors such as production efficiency, land, water and productive soil availability and climatic conditions. The future technical potential of bioenergy depends on many factors such as climate change, land use change, and technical and economic developments.

There is a very wide range of estimations of the global potential of bioenergy. Leatherdale et al. [244] estimated that bioenergy supply could contribute 2–20% to global primary energy demand in 2050. The European Energy Agency estimated the potential for sustainable bioenergy production of approximately 340 EJ/year in 2050 [7]. In a study by Koljonen et al. [48] under scenarios with sustainability constraints, the global potential of bioenergy is between 200 and 500 EJ/year by 2050. This would meet up to 50% of projected world primary energy demand. In a study by Haberl et al. [49], bioenergy potential varies from 30 to 1000 EJ/year by 2050 and with sustainability constraints it reduces to 160–270 EJ/year.

Expanding production of dedicated energy crops for bioenergy might lead to some negative externalities such as depletion of water resources [128, 130, 135], water contamination [129, 130, 135], soil degradation [128-131], land-use change [46, 50, 127], loss of biodiversity [131, 132] and competition with food production [171].

Globally, humanity has taken about 26% (3.3 Gha) of the planet’s land area for crop land and pasture [142], which consumes around 70% of the world’s fresh water [305]. In addition, of the total 13 Gha of land on the planet, land suitable for crop production is 4.5 Gha, of which only 2.7 Gha is available for agricultural [146, 148]. Increasing the bioenergy supply from energy crops will require expansion of agricultural lands. For instance, Souza et al. [141] estimates that to provide an average 200 EJ bioenergy, around 200–500 Mha land will be required. However, of the land available for expanding bioenergy crops, 800 Mha of forest, 200 Mha of protected land and 60 Mha occupied with human settlements must be excluded [136]. It means unsustainable energy crop production could lead to major direct or indirect land-use change. According to Le Quéré et al. [137], in the last decade about 9% of global net GHG emission were related to land use change—principally associated with deforestation and expanding agricultural land use.
Another challenge of expanding bioenergy from dedicated energy crops is to maintain food security. With current production, there are 32 countries facing food crises with approximately 870 million people estimated to be undernourished and 1 billion malnourished [152]. Despite the current inefficiencies to cover the food demand of the growing population- around 9 billion by 2050 [168]- the food production has to grow by 60% by 2050 [154]. FAO estimated that to produce this amount, an additional 72 Mha of land is required [169]. In this estimation, population growth, diet change and improvement in agricultural productivity have been considered [125, 169]. A study by Tilman et al. shows much higher land expansion of 890 Mha to cover food demand in 2050 [148]. This puts even more constraints on expanding land for energy crop cultivation. Profitability of biomass for energy will be an incentive for the producers to dedicate land to energy crops rather than food crops.

Agriculture is currently one of the main contributors to global warming by emitting GHG and it is a victim of climate change as well. Agricultural activities account for approximately 58% of the total N₂O and 47% of the total CH₄ emission [152]. Expanding bioenergy crop production might exacerbate the GHG emission from this sector.

5.1.1 Organic residues: a resource for BECCS

Using organic waste from municipal, agricultural and forestry sectors for BECCS is a way to avoid the ecological and social challenges of dedicated energy crops. Organic residues are perceived to have less environmental impact as they are inevitable by-products of high value food and fodder production. Forest residue includes those available from chopping, branches, bark, foliage and stumps. In 2014 around 14 Gt forest residue was produced [146]. Russia with 5.7 Gt and Indonesia with 2.2 Gt were the top producers [146]. Every year, approximately 40 dry t/ha of lignocellulosic residues are produced, most of which are underutilized [306]. These lignocelluloses are mainly corn stover, wheat straw, rice husk, and sugarcane bagasse [46, 50, 306]. Two types of agricultural residues are associated with crop production: field (primary such as straw) and processing (secondary such as sugarcane bagasse) residues. One of the abundant sources of agricultural residue is sugarcane bagasse. Sugarcane is one the principal agricultural crops with an annual global production of 1.6 Gt [307]. Brazil with an annual production of 625 Mt, is the top producer of sugarcane [307]. Bagasse is the residue left after the juice is extracted from the sugarcane. Each tonne of sugarcane generates around 280–600 kg wet bagasse [159, 307, 308]. Converting bagasse to a value-added product could provide sustainable economic benefits to the producers.
Availability of both agricultural and forestry residue directly depends on climate situations, population, supply and demand, technical development, economic growth and land use changes. Globally there are very limited waste management mechanisms available to exploit organic waste and residues at present. The residues and waste, if not managed sustainably, are a source of GHG emissions. CO$_2$ released from microbial decomposition of organic matter, CH$_4$ emission from anaerobic decay especially in case of paddy rice residues, and N$_2$O produced by microbial transformation of N in the soil and plants are the main sources of GHG emissions from this sector [154]. In 2011, total N$_2$O emissions from crop residues was 197 Mt CO$_2$-eq, nearly 4% of total emissions from agriculture [138]. Asia and America with 47% and 27% are the biggest N$_2$O emitters from this sector. The global GHG emissions from crop residues is predicted to reach 235 Mt CO$_2$-eq in 2050 [138].

Utilising these residues in a BECCS system essentially turns a negative externality into a valuable good and provides a source of income for local farmers, industry and the state, plus generates energy and could deliver negative emissions [1, 30, 50].

### 5.1.2 Potential of organic residues

Current production of bioenergy is around 55 EJ/year – equivalent to 12% of current energy production from fossil fuels [54, 309]. Estimates in the literature of the amount of bioenergy from organic residues vary from 5 EJ/year to 270 EJ/year. Some of the most recent studies are presented below.

Chum et al. [50] estimated the potential of residues originating from forestry, agriculture and organic wastes to be around 40 to 170 EJ/year, with a mean estimate of around 100 EJ/year. They found China, the U.S and India have the highest potentials. According to Berndes et al. [310], the total amount of residues that will be generated in the food and forest sectors by the year 2100 is estimated to be about 270 EJ/year. In a study by Koljonen et al. [48], between 20 and 150 EJ/year of biomass could be supplied by organic waste from agricultural, forestry and municipal solid wastes. According to the IPCC AR5 [1] around 100 EJ/year can be supplied from agricultural and forestry residues, dung and organic waste. Gregg et al. [309] showed that the sustainable global bioenergy from residues is expected to increase to 100 EJ/year by 2100. Azar et al. [16] found 80–100 EJ/year could be supplied by residues from agriculture and forestry. In a study by Luckow et al. [105], bioenergy mainly from forestry and agricultural residues could supply 120–160 EJ/year by 2050. This potential will increase to 200–250
EJ/year mainly from dedicated energy crops and waste-to-energy utilisation in 2100 [105]. These estimations are in the range of the sustainable bioenergy potential projected in most models [30, 46, 47, 50].

Other studies suggest a more conservative range for residue-based bioenergy; Fischer and Schrattenholzer [311] estimated 35 EJ/year potential for bioenergy from agricultural residues and 100 EJ/year from forestry residues by 2050. According to Long et al. [145], the global bioenergy production from agricultural and forestry residues and wastes will be about 76–96 EJ by 2050. Hakala et al. [312] showed that global potential of residues by 2050 will be 54–57 EJ/year. Deng et al. [144] found the global bioenergy potential from agricultural residues and forestry residues is between 7 EJ and 30 EJ [144]. Leatherdale et al. [244] estimated that the bioenergy supply from forest and agricultural residues could around 5.5–12 EJ by 2050.

Haberl et al. found that with regard to the maximum biospheric capacity of net primary production (NPP), the maximum potential for bioenergy from organic residues across municipality, agricultural and forestry sectors is limited to 60 EJ/year [49, 53-55], meaning any projection of bioenergy potential higher than 60 EJ/year exceeds the biophysical limits [53].

This study investigates the potential for using residues from major agricultural crops and forestry as feedstock for BECCS. Based on economic viability and environmental impacts, the sustainability of this technology is also investigated.
5.2 Methodology

Techno-economic and environmental impact assessments were conducted for forestry residue (FR-CCS) and five BECCS options using agricultural residues: a) bagasse (in BG-CCS), b) barley straw (in BS-CCS), c) corn stover (in CS-CCS), d) rice straw (in RS-CCS) and e) wheat straw (in WS-CCS) combusted in circulating fluidised beds equipped with CCS. Table 5-1 shows the elemental composition of the fuel used in this study.

Table 5-1- Elemental compositions of the organic residues and coal [159, 217, 235, 313-315]

<table>
<thead>
<tr>
<th>Composition</th>
<th>Bagasse (BG)</th>
<th>Barley Straw (BS)</th>
<th>Corn Stover (CS)</th>
<th>Rice Straw (RS)</th>
<th>Wheat Straw (WS)</th>
<th>Forestry Residue (FR)</th>
<th>Black Coal</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
</tr>
<tr>
<td>C (wt%)</td>
<td>48.6</td>
<td>42.2-52.0</td>
<td>45.7</td>
<td>43.0-48.4</td>
<td>49.3</td>
<td>72.60-83.80</td>
<td>78.2</td>
</tr>
<tr>
<td>H (wt%)</td>
<td>5.9</td>
<td>5.4-6.4</td>
<td>6.1</td>
<td>5.3-6.9</td>
<td>6.4</td>
<td>4.14</td>
<td>6.5</td>
</tr>
<tr>
<td>O (wt%)</td>
<td>42.8</td>
<td>37.5-48.1</td>
<td>38.3</td>
<td>35.2-41.4</td>
<td>43.3</td>
<td>10.75-16.45</td>
<td>13.6</td>
</tr>
<tr>
<td>N (wt%)</td>
<td>0.16</td>
<td>0.14-0.18</td>
<td>0.4</td>
<td>0.36-0.44</td>
<td>0.9</td>
<td>0.7</td>
<td>0.6</td>
</tr>
<tr>
<td>S (wt%)</td>
<td>0.04</td>
<td>0.03-0.045</td>
<td>0.045</td>
<td>0.045-0.138</td>
<td>0.04</td>
<td>0.03</td>
<td>0.04</td>
</tr>
<tr>
<td>Moisture</td>
<td>45</td>
<td>39.6-50.4</td>
<td>26.8-33.2</td>
<td>16.29</td>
<td>12.12</td>
<td>4.55</td>
<td>15.5</td>
</tr>
<tr>
<td>LHV (MJ/Kg)</td>
<td>13.6-28.3</td>
<td>10.56-15.18</td>
<td>13.6-18.4</td>
<td>15.16</td>
<td>15.51</td>
<td>13.6</td>
<td>15.5</td>
</tr>
</tbody>
</table>
5.2.1 Modelling BECSS power plants

In these BECCS models, biomass is combusted in a circulating fluidised bed (CFB) plant to produce electricity. The electricity generated is exported to the grid. The residue from the combustion plant is mostly ash which is put into landfill. The reason for using CFB is its ability to burn fuels with high moisture and low heating value with high efficiency. In addition, CFB does not require fuel pre-processing. CFB reactors offer higher efficiency and less bottom ash than fixed bed boilers [263].

5.2.1.1 Carbon Capture and storage system

The CO$_2$ capture process in our model is post-combustion chemical absorption using monoethanolamine (MEA) with 90% CO$_2$ capture efficiency. After separation, the CO$_2$ is compressed to 150 bars and sent to the CO$_2$ sink and stored underground in suitable geological formations. Pipelines and compression equipment for transporting the CO$_2$ and injection wells are components of the CO$_2$ storage system. The CCS system used in this chapter is the same as the CCS system used in Chapter 4, Section 4.2.3.

5.2.2 Cost of electricity production

Techno-economic assessments in this chapter are the same as those described in Chapter 4, Section 4.2.4.

Table 5-2 lists the techno-economic data for BECCS systems and Coal-CCS, based on information available in the literature.
### Table 5-2: Techno-economic data of the BECCS plants (Costs are in 2017 USD)

<table>
<thead>
<tr>
<th>Process</th>
<th>Capacity (MW)</th>
<th>Thermal efficiency (% LHV)</th>
<th>capacity factor%</th>
<th>Lifetime (years)</th>
<th>Capital ($/kW)</th>
<th>Variable O&amp;M ($/MWh)</th>
<th>Fixed O&amp;M ($/kw/yr)</th>
<th>Fuel cost ($/t)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>CFB Power Plant [218, 309, 316-320]</td>
<td>100</td>
<td>Mean: 40</td>
<td>90</td>
<td>30</td>
<td>Mean: 3000</td>
<td>Mean: 4.5 ($/MWh)</td>
<td>Mean: 96</td>
<td>Bagasse Mean: 6.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Barley Mean: 32.1</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Straw Range: 20.2-44</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Corn Mean: 41.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Stover Range: 32.5-51.3</td>
</tr>
<tr>
<td>Coal plant [14][80, 289]</td>
<td>150</td>
<td>Mean: 42</td>
<td>90</td>
<td>30</td>
<td>Mean: 2400</td>
<td>Mean: 4.5 ($/MWh)</td>
<td>Mean: 36</td>
<td>Mean: 50</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 40-44</td>
<td></td>
<td></td>
<td>Range: 2100-2700</td>
<td>Range: 3.3-5.7 ($/MWh)</td>
<td>Range: 28-44</td>
<td>Range: 39-61</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>水稻 straw Mean:25.1</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>麦秆 straw Range: 18.6-31.6</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>稻秆 straw Mean: 21.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>稻秆 straw Range: 16.5-25.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>稻秆 residue Mean:48</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>稻秆 residue Range: 32-64</td>
</tr>
<tr>
<td>Capture process [69, 290-292]</td>
<td>150</td>
<td>-</td>
<td>100</td>
<td>30</td>
<td>Mean: 2100</td>
<td>Mean: 9 ($/MWh)</td>
<td>Mean: 103</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 1760-2440</td>
<td></td>
<td></td>
<td>Range: 1760-2440</td>
<td>Range: 5.6-12.4 ($/MWh)</td>
<td>Range: 87-119</td>
<td></td>
</tr>
</tbody>
</table>

*Cost of dry biomass at the power plant gate

Residue transportation = 18 ($/ton/100 km) [313]
Residue handling and storage = 12 ($/ton) [319]
CO₂ Transportation (500-2000 km) = 8 ($/ton CO₂) [320]
CO₂ Storage and monitoring = 15 ($/ton CO₂) [70]
CO₂ capture efficiency of the MEA unit is 90%
Interest rate (i) =10%
Inflation rate (ρ)=0%
5.2.3 Environmental impact assessment

Life cycle assessment (LCA) was carried out using SimaPro software version 8.2.0.0 developed by PRé Sustainability Group. The LCA study includes four main steps; defining the goal and scope, life cycle inventory of all the inputs and outputs, lifecycle impact assessment, and interpretation of the results. Allocation modelling was applied to assess the environmental impacts of the BECCS systems with the functional unit of 1 KWh electricity generated. In these assessments, the ALCAS Best Practice LCIA method [296] was used. ALCAS uses the Global Warming Potential (GWP 100) according to IPCC’s fifth assessment report. The impact categories considered in this study were global warming (GWP100a), abiotic depletion (fossil fuels), Ozone Layer Depletion (ODP), eutrophication, acidification, particulate matter, human toxicity (cancer), human toxicity (non-cancer), and water scarcity. Most of the inventory data are taken from Ecoinvent-3 [61] and the AusLCI database (developed by Australian LCA Society (ALCAS)) [298]. Data regarding the MEA CO₂ capture unit was derived from the project conducted by Hooper et.al [299, 300].

Figure 5-1 illustrates the system boundary for the inventories of the BECCS models using agricultural and forestry residues. As can be seen, in this study a cradle-to-grave LCA has been conducted which includes all the processes from production of residues to storage of CO₂. Forest residues are assumed to be collected as part of non-commercial thinning and whole-tree logging operations on private and state-owned forest lands; national forests, national parks, and other federal lands are explicitly excluded from this module.
Figure 5-1- System boundary for residue-based BECCS models inventories
5.3 Results

5.3.1 Techno-economic assessments

CO₂ emissions and LCOE were calculated for power plants with and without CCS. Equation 4-4 was used to calculate the net CO₂ emissions with CCS. The CO₂ capture rate of the MEA process in this study is 90%. After CCS, CO₂ emission from the coal-fired plant is reduced from 0.81 t/MWh to 0.11 t/MWh. All the BECCS systems deliver net negative CO₂ emission of -1.20 to -1.35 t/MWh and therefore more avoided CO₂ emission than Coal-CCS. The LCOE in Table 5-3 is calculated using Equation 4-1 (see the equation in Chapter 4), under a business as usual (BAU) scenario- where there is no emission policy in effect. Under a BAU scenario, not surprisingly, the LCOE of the BECCS systems modelled in this study are higher than that of coal-fired power plants -with or without CCS. The cost of capital and variable costs of CFB power plants used for BECCS systems are the same. The main factor of having different LCOE among BECCS is the different price of feedstock. As seen in Table 5-2, the cost of feedstock varies from $6.3/t for bagasse up to $48/t for forestry residue.

Adding CCS to the power plants increased their LCOE by up to 140%. All the BECCS systems offer up to 30% lower cost of avoided CO₂ emission than Coal-CCS. The lower avoided cost of CO₂ emissions could significantly determine economic potential of BECCS systems compared to other mitigation technologies on the market.

Table 5-3- CO₂ emission, LCOE with and without CCS and cost of avoided emission

<table>
<thead>
<tr>
<th>Technology</th>
<th>Biogenic CO₂%</th>
<th>Net CO₂ Emission (t/MWh)</th>
<th>Avoided Emission (t/MWh)</th>
<th>LCOE ($/MWh)</th>
<th>Cost of CCS ($/MWh)</th>
<th>Cost of Avoided Emission ($/t)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>No CCS</td>
<td>CCS</td>
<td>No CCS</td>
<td>CCS</td>
<td>No CCS</td>
</tr>
<tr>
<td>Coal-PC</td>
<td>0</td>
<td>0.81</td>
<td>0.11</td>
<td>0.70</td>
<td>132.13</td>
<td>72.28</td>
</tr>
<tr>
<td>BG-CFB</td>
<td>1</td>
<td>0.00</td>
<td>-1.32</td>
<td>1.32</td>
<td>170.55</td>
<td>100.06</td>
</tr>
<tr>
<td>BS-CFB</td>
<td>1</td>
<td>0.00</td>
<td>-1.20</td>
<td>1.20</td>
<td>212.84</td>
<td>117.86</td>
</tr>
<tr>
<td>CS-CFB</td>
<td>1</td>
<td>0.00</td>
<td>-1.20</td>
<td>1.20</td>
<td>187.90</td>
<td>91.13</td>
</tr>
<tr>
<td>RS-CFB</td>
<td>1</td>
<td>0.00</td>
<td>-1.35</td>
<td>1.35</td>
<td>193.04</td>
<td>102.75</td>
</tr>
<tr>
<td>WS-CFB</td>
<td>1</td>
<td>0.00</td>
<td>-1.24</td>
<td>1.24</td>
<td>175.48</td>
<td>88.71</td>
</tr>
<tr>
<td>FR-CCS</td>
<td>1</td>
<td>0.00</td>
<td>-1.29</td>
<td>1.29</td>
<td>174.53</td>
<td>89.35</td>
</tr>
</tbody>
</table>
The first emission policy investigated in this study was a renewable energy certificate scheme (REC). Under such a scheme, certificates (RECs) are given on the basis of per megawatt-hour of electricity produced from recognised renewable power suppliers. Power plants using agricultural residues as feedstock are eligible for RECs. In this study it is assumed that BECCS systems using these biomass resources will receive REC as well. As seen in Figure 5-2, introducing a REC into the energy market influences the final LCOE of renewables significantly. At a REC of $40–$80 /MWh, the LCOE of BECCS systems will be lower than Coal-CCS. At a REC higher than $120–$160 /MWh BECCS can become a cheaper power production option than a coal-fired power plant without CCS.

![Figure 5-2- Impact of REC on LCOE of BECCS, Coal-CCS and coal w/o CCS](image)

The second emission mitigation policy considered was carbon pricing. Under such a scenario, power suppliers using fossil-fuel are obliged to pay tax per tonne of CO₂ emitted to the atmosphere. The effect of reintroducing a carbon tax into the energy market on the LCOE of coal-fired power plants modelled in this study is illustrated in Figure 5-3. The BECCS systems are assumed to be exempted from a carbon tax as the CO₂ they emit is biogenic. Raising the carbon price from 0 to $200 /t CO₂ increases the LCOE of the coal-CCS system by 16% ($132/MWh to $153/MWh). A black coal-fired plant without CCS emits around 0.81 t
CO₂/MWh to the atmosphere; a carbon price higher than $140 /t CO₂ is needed to raise the LCOE of coal-fired to a level that BECCS options could become cost-comparable in the electricity market. However, given the current status of carbon pricing around the world, it seems highly unlikely that such a carbon price will be implemented in the near future given the current level of political and commercial opposition.

The third emission mitigation policy considered in this study was that of a negative CO₂ credit. In this scenario, BECCS technologies receive a refund for each tonne of CO₂ removed from the atmosphere. In this case the Coal-CCS plant does not receive credit for the CO₂ captured, but BECCS options do benefit. As seen in Figure 5-4, a negative CO₂ credit of $30–$70 /t CO₂, Negative makes the BECCS systems lower in cost than the LCOE of Coal-CCS. A much greater value, of $90–$130 /t CO₂, Negative is needed to make BECCS systems competitive with base-load coal-fired power generation. Based on the economic evaluation undertaken in this study, a negative CO₂ credit scheme results in lower cost of BECCS deployment compared with conventional carbon pricing. However, negative emission achieved through the application of BECCS is a relatively new concept and there is currently no scheme that takes the value of negative emissions into account.
Environmental impact assessment

The LCA of the six BECCS options in this study was modelled using SimaPro 8.2.0.0 software and the ALCAS Best Practice LCIA [296] method was applied for the impact assessment. The impact assessment of these systems presented in Table 5-4 shows that the differences are minimal. As can be seen, in BECCS systems the net global warming effect is significantly reduced compared to the baseline systems. In some categories such as ozone depletion, photochemical oxidation, human toxicity and water scarcity BECCS options have a lower environmental performance compared to those with no CCS.

Figure 5-5 illustrates the apportioned global warming impact (CO₂ equivalent) of different subprocesses in the BECCS systems system per functional unit of 1 kWh electricity generated). The cut-off rule for CO₂equiv emission was 5%, i.e. any process contributing less than 5% of the net GHG emission is not included in Figure 5-5. The sum of the total positive and negative CO₂equiv emissions is the same as in Table 5-4. All BECCS options result in net negative CO₂ emission. The main sources of negative emissions are end-of-pipe CO₂ stored underground (-1.4 to -1.8 Kg CO₂equiv per kWh electricity generated) and electricity sent to the grid (-0.8 to -
0.9 Kg CO₂ eq per KWh electricity generated); it is assumed the renewable electricity produced by these BECCS systems replaces electricity mix in the grid which is mainly supplied by fossil fuel-based power plants.

Figure 5-5 illustrates the apportioned CO₂ eq emission for the main processes included in BECCS systems during their lifecycle. As is evident, in all cases CO₂ production of the residues and CO₂ capture process contribute to positive CO₂ emission. The amount of CO₂ due to the capture process is very similar for all BECCS options and is around 0.2 t CO₂ eq per MWh electricity generated). The main difference is the emission from residue production. The emission of residue production is directly dependent on the CO₂ intensity of the agricultural practices used to produce the main crop that residue has been derived from.

For instance, currently the conventional method (which is applied to produce approximately 90% of rice globally) to cultivate rice is by flooded rice fields (paddy rice) which lead to CH₄ from anaerobic decomposition of organic matter in paddy fields. Asia with 89% has been the largest GHG emitter from rice cultivation [138]. With 522 Mt CO₂ eq annual GHG emissions, rice cultivation is responsible for 10% of total agriculture emission, which makes it one the most GHG emission intensive sub-sectors [321]. Again, due to the flooded rice production method, the water consumption is highest among the BECCS options. According to the LCA model used in this study for 1 KWh electricity generated from rice straw combustion 0.16 m³ water is used. Adding CCS to a power plant using rice straw (RS-CCS) increases the water consumption to 0.26 m³/ KWh. The additional water is mainly used for cooling in the MEA system. The net water consumption for other BECCS options is almost negligible.
Table 5-4: Lifecycle impact assessment of the BECCS systems per each KWh of electricity generated

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Bagasse</th>
<th>Barley Straw</th>
<th>Corn Stover</th>
<th>Rice straw</th>
<th>Wheat Straw</th>
<th>Forest Residue</th>
</tr>
</thead>
<tbody>
<tr>
<td>Global warming (GWP100a)</td>
<td>kg CO2eq</td>
<td>-2.39E+00</td>
<td>-2.21E+00</td>
<td>-2.18E+00</td>
<td>-2.11E+00</td>
<td>-2.07E+00</td>
<td>-2.21E+00</td>
</tr>
<tr>
<td>Abiotic depletion (Fossil fuels)</td>
<td>MJ NCV</td>
<td>1.60E+00</td>
<td>1.57E+00</td>
<td>1.55E+00</td>
<td>1.53E+00</td>
<td>1.50E+00</td>
<td>1.56E+00</td>
</tr>
<tr>
<td>Ozone layer depletion (ODP)</td>
<td>kg CFC11+12eq</td>
<td>-1.20E+00</td>
<td>-1.16E+00</td>
<td>-1.12E+00</td>
<td>-1.09E+00</td>
<td>-1.06E+00</td>
<td>-1.12E+00</td>
</tr>
<tr>
<td>Photochemical oxidation</td>
<td>kg CH4eq</td>
<td>3.08E-04</td>
<td>3.04E-04</td>
<td>2.99E-04</td>
<td>2.95E-04</td>
<td>2.92E-04</td>
<td>2.99E-04</td>
</tr>
<tr>
<td>Acidification</td>
<td>kg SO2eq</td>
<td>-1.22E+03</td>
<td>-1.19E+03</td>
<td>-1.17E+03</td>
<td>-1.15E+03</td>
<td>-1.13E+03</td>
<td>-1.10E+03</td>
</tr>
<tr>
<td>Eutrophication</td>
<td>kg PM2.5</td>
<td>2.98E+05</td>
<td>2.94E+05</td>
<td>2.90E+05</td>
<td>2.86E+05</td>
<td>2.82E+05</td>
<td>2.88E+05</td>
</tr>
<tr>
<td>Particulate matter</td>
<td>CTUh</td>
<td>1.99E+00</td>
<td>1.95E+00</td>
<td>1.91E+00</td>
<td>1.87E+00</td>
<td>1.83E+00</td>
<td>1.89E+00</td>
</tr>
<tr>
<td>Human toxicity, cancer</td>
<td>CTUh</td>
<td>1.08E+07</td>
<td>1.04E+07</td>
<td>1.00E+07</td>
<td>9.60E+07</td>
<td>9.20E+07</td>
<td>9.80E+07</td>
</tr>
<tr>
<td>Human toxicity, non-cancer</td>
<td>CTUh</td>
<td>2.0E+07</td>
<td>1.9E+07</td>
<td>1.8E+07</td>
<td>1.7E+07</td>
<td>1.6E+07</td>
<td>1.5E+07</td>
</tr>
<tr>
<td>Water Scarcity</td>
<td>m³/eq</td>
<td>2.33E+04</td>
<td>2.33E+04</td>
<td>2.33E+04</td>
<td>2.33E+04</td>
<td>2.33E+04</td>
<td>2.33E+04</td>
</tr>
</tbody>
</table>
The CO₂ emission from the infrastructure used for the power plant is negligible when it is apportioned over the lifetime of the project and total EL generated.

As seen in Table 5-4, the BECCS option using bagasse as feedstock (BG-CCS) results in a higher potential to remove CO₂ per MWh electricity generated compared to other residues. One of the main reasons for this is the lower GHG emission during sugarcane production. Unlike other crops in this study which are annual, sugarcane is a perennial crop. Perennial plants are crops that remain productive for more than two years. Hence, these crops are favoured on sites susceptible to soil erosion since no tillage is required and root and soil organic carbon formation is higher for perennial crops compared to annual crops. This results in less erosion and increased soil quality. In addition, nitrate leaching is lower on land cropped with perennial crops than annual crops [45]. According to Immerzeel et al. [160] perennial crops have less impact on biodiversity than annual crops. Perennial crops have lower GHG emission as well. Because of higher yield less tillage and having more soil organic matter due to deeper roots and less N₂O due to using less fertiliser. There are R&D programs to develop perennial version of typically annual crops such as wheat, corn and rice [45]. In this study, crop production in the case of a bagasse base BECCS (BG-CCS), contributes to less than 6% of total LCA GHG emissions with 0.013 t CO₂ per MWh EL generated (Figure 5-5).
Figure 5.5- Apportioned CO\textsubscript{2}-eq emission of the BECCS cradle to grave LCA processes
5.4 Discussion

5.4.1 Sustainable residue removal

Agricultural residues are either collected on the field or after the processing of primary product. The amount of field-based residues varies for different crops and depends on the residue to yield ratio. Not all the residues are economically feasible to collect. The selling price of straw and the cost to collect and store the straw determines if it is feasible to collect [315, 319]. Factors such as soil fertility, crop rotation, crop yield, climate situation, slope of the land, and farming practices are important in estimating the sustainable residue removal [319]. Estimates of the amount of recoverable agricultural residues in the literature vary between 10–32 EJ/year and total recoverable forestry residue of 10–16 EJ/year [322].

One of the limitations of residue removal is its impact on soil carbon. The soil carbon stock is a consequence of input organic matter and residues and output C oxidation due to decomposition and erosion and leaching [323]. The residue on the soil improves the soil quality, reduces soil bulk density, water retention and transmission properties. Unsustainable expansion of using organic waste for energy may lead to loss in soil carbon stock [128-131]. Cherubini et al. [324] estimated that 50% removal of straw could contribute to changes in soil carbon by 0.2 t C/ha/year and 100% removal could lead to 0.35 t C/ha/year soil carbon loss. Globally the soil carbon pool is around 2500 Gt C, compared to about 560 Gt C stored in biomass and 760 Gt C in atmosphere [323, 325]. Loss of soil carbon stock causes a large amount of relic C in the soil organic matter that has been stored in the land for thousands of years, to be released to the atmosphere.

According to Scarlat et al. [326], the sustainable removal rate for cereal crops is 40%, and for maize and rice is 50%. Other assessment use a 30% rate for all crops considering sustainable removal for maintaining soil carbon and competitive usage [327]. Recent analyses published by the World Bank [328] assumes that 40% of total produced residues can be used for bioenergy production for most crops – the exceptions would be rice straw and sugarcane bagasse where 100% of the residues can be removed. The U.S Department of Agriculture [313] recommends at least 30% of soil surface be covered with residues to avoid wind and water erosion. In Africa, the fraction of collectable residues for most crops is less than 30% because of low yields. In other regions, the collectable fraction of most crop residues is over 20% [308, 313].
With regard to forest residues, it should be noted that it is not carbon neutral if the rate of CO₂ emission from its combustion is faster than its natural decomposition. Time perspective is an important factor determining the GHG emission counting. It is important to identify the forest residue types with a highest decomposition rate to harvest [329]. Global forestry residue potential cannot be aggregated easily, because it is very site specific and depends on several factors such as tree species, management type, silvicultural practices and final product [330]. Residues usually constitute 25–45% of the harvested volume of wood [331]. Although removal of the residues lowers the risk of fire and damage from insects and diseases [331], unsustainable removal could have severe ecological implications [332]. Of the residue produced from logging, wood-processing and tending/thinning a minimum of 1000 t/km² of forestry residues from final harvest should remain on site to maintain soil properties [330, 333]. In a long term soil productivity study, Curzon et al. [334] demonstrated that increased soil disturbance resulting from the removal of forestry residues may have a negative effect on structural development and forest productivity. In addition to that, forest residues are a source of animal feed and fuel in poor communities. Another concern is biodiversity conservation. Protecting unique ecosystems and critical habitat and balancing the vegetation structure should be included in forest residue management [331].

In this study, a more conservative removal rate of 25% for all residues is chosen; it is assumed that this rate allows covering residue required for competitive utilisation such as animal feed and bedding and residue left is sufficient to preserve soil carbon. This is lower than most of a removal rates recommended in the literature but still considerably higher than the current share of agricultural residues used for energy production. Table 5-5 shows the residue availability of the crops used in this study. As it can be seen, in all cases if 25% of the residue is collected for energy production there would be still enough left for recommended share for soil cover. For instance, in the case of sugarcane with highest yield among others (74.3 t/ha) around 45 t/ha bagasse (residue to crop of 0.6) is produced. Whereas the bagasse required to be retained on soil is 1.24 t/ha, which is less than 3% of the total bagasse yield. If 25% of the bagasse is used in a BECCS system, more than 72% is still left for other purposes. This ratio is less for wheat straw where from every 3.2 t/ha wheat about 4.2 t/ha wheat straw is produced which more than 67% (2.81 t/ha) needs to be left to protect the soil carbon.
Table 5-5: Availability of Sustainable crop residues

<table>
<thead>
<tr>
<th>Crop /Residue</th>
<th>Residue to Crop ratio</th>
<th>Dry matter (%)</th>
<th>Residue Retention (t/ha)</th>
<th>Global Crop Yield (Mt/year)</th>
<th>Average Global Yield (t/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Barley Residue: Barley Straw</td>
<td>1.5</td>
<td>88.7</td>
<td>1.09</td>
<td>144.5</td>
<td>2.6</td>
</tr>
<tr>
<td>Corn Residue: Corn Stover</td>
<td>1</td>
<td>86.2</td>
<td>2.20</td>
<td>1037.8</td>
<td>5.9</td>
</tr>
<tr>
<td>Rice Residue: Rice Straw</td>
<td>1.4</td>
<td>88.6</td>
<td>0.94</td>
<td>741.5</td>
<td>6.1</td>
</tr>
<tr>
<td>Wheat Residue: Wheat Straw</td>
<td>1.3</td>
<td>89.1</td>
<td>2.81</td>
<td>729.0</td>
<td>3.2</td>
</tr>
<tr>
<td>Sugarcane Residue: Bagasse</td>
<td>0.6</td>
<td>26.0</td>
<td>1.24</td>
<td>1884</td>
<td>74.3</td>
</tr>
</tbody>
</table>

5.4.2 Availability of residues for BECCS

To have an estimation of how much these major crop residues could sustainably contribute to both CO₂ removal and energy generation through BECCS, Equations 5-1 to 5-3 were used.

\[ R_S = \text{Yield}_{\text{Crop}} \times \frac{R}{C} \times X_r \]  
\[ E_S = LHV_{\text{residue}} \times R_S \]  
\[ CO_2, \text{removal} = X_{CO_2} \times R_S \]

In the equations above, \( R_S \) represents the annual sustainable residue harvest [t/year], \( E_S \) is the annual sustainable energy content of the crop residue [MJ/year], \( CO_2,\text{removal} \) represents the sustainable annual CO₂ removed by the residue through BECCS [t CO₂/year], the R/C is the residue to crop ratio, \( X_r \) is the sustainable fraction of residue removal for BECCS (25% in this study), \text{Yield}_{\text{crop}} \) is the annual yield of the crop [t/year], \( X_{CO_2} \) is the CO₂ removed and stored through BECCS [t CO₂/t residue] . Forest residue includes those available from chopping and branches, bark, foliage and stumps. It is estimated that the sustainable forest residue for energy production is about 650 Mt/year [244, 322, 330]. Data from tables 5-1 to 5-5 were used to calculate \( R_S \), \( E_S \) and \( CO_2,\text{removal} \). Results are listed in Table 5-6.
As can be seen in Table 5-6, a sustainable harvest of the residues used in this study to produce energy and remove CO₂ from atmosphere through a BECCS system could remove annually 3.32 Gt CO₂ (all LCA GHG emissions) and 2.76 Gt CO₂ (if the CO₂ captured and stored at the end of pipe (EOP) is counted). This is higher than the lower range of CO₂ required to be removed in the < 2 °C scenario which is 2 Gt CO₂ per year by 2050 [1, 21, 30]. At the same time the total energy content of the residues harvested for BECCS is around 26 EJ/year. This is within the range estimated by Hoogwijk et al. [322] of the agricultural residues (10–31 EJ/year) and higher than estimation by Baruya et al. [146] of 4 EJ/year from cereal crops. The global crop yield production (Mt/year) presented in the Table 5-6 were obtained from latest data provided by FAOSAT regarding global production in 2014 [335]. It is expected that growing demand for agricultural products especially in developing countries will lead to an increase in crop production of 60% by 2050 [154]. Increase in crop production means increase in the crop residue at an almost same rate. Therefore, estimation of the total sustainable amount of the five main residues harvested for energy production in this study will be approximately 1,748.3 Mt (compared to the current value of 1,092.7 Mt/year). This much residue will have the potential to annually remove 2.7 Gt CO₂ (EOP) or 3.4 Gt CO₂ (LCA) by 2050 if used through BECCS. This could potentially have a significant contribution to achieving COP 21 targets to put the world on track to avoid +2 °C. The energy content of these crop residues by 2050 based on this estimation will then be about 25.6 EJ, which is within the range forecast in some models; 5.5–12 EJ [244], 20 to 150 EJ/year [48] and 10–32 EJ/year [322].

### Table 5-6- Potential of agricultural residues to deliver negative emission and energy through BECCS

<table>
<thead>
<tr>
<th>BECCS system</th>
<th>Residue</th>
<th>Rs (Mt/year)</th>
<th>LHV (MJ/kg)</th>
<th>CO₂ Emission (tCO₂/t residue)</th>
<th>CO₂ removal (GtCO₂/year)</th>
<th>Es (EJ/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>LCA</td>
<td>EOP</td>
<td>LCA</td>
</tr>
<tr>
<td>BS-CCS</td>
<td>Barley Straw</td>
<td>54.18</td>
<td>15.08</td>
<td>-2.28</td>
<td>-1.5</td>
<td>-0.12</td>
</tr>
<tr>
<td>CS-CCS</td>
<td>Corn Stover</td>
<td>259.45</td>
<td>16.29</td>
<td>-2.12</td>
<td>-1.6</td>
<td>-0.55</td>
</tr>
<tr>
<td>RS-CCS</td>
<td>Rice Straw</td>
<td>259.53</td>
<td>12.12</td>
<td>-0.61</td>
<td>-1.4</td>
<td>-0.16</td>
</tr>
<tr>
<td>BG-CCS</td>
<td>Bagasse</td>
<td>282.63</td>
<td>14.6</td>
<td>-2.28</td>
<td>-1.6</td>
<td>-0.64</td>
</tr>
<tr>
<td>WS-CCS</td>
<td>Wheat Straw</td>
<td>236.93</td>
<td>15.51</td>
<td>-2.34</td>
<td>-1.6</td>
<td>-0.55</td>
</tr>
<tr>
<td>FR-CCS</td>
<td>Forest Residue</td>
<td>650</td>
<td>15.36</td>
<td>-2.00</td>
<td>-1.65</td>
<td>-1.3</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>-3.32</strong></td>
<td><strong>-2.76</strong></td>
<td><strong>26</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
5.4.3 Feasibility of implementing BECCS using residues

According to data published by FAOSTAT, in 2014 a total land of about 645 Mha was cultivated to produce barley, corn, rice (paddy), sugarcane and wheat around the world [335]. Wheat production with a total of 220 Mha and barley production with 49.4 Mha occupy the largest and smallest amounts of land, respectively.

Table 5-7: Crop production, Land: Land Harvested (Mha), Yield: crop yield (t/ha), Production: total production (Mt), FAOSTAT database [335]

<table>
<thead>
<tr>
<th>Region</th>
<th>Barley</th>
<th>Corn</th>
<th>Rice</th>
<th>Sugarcane</th>
<th>Wheat</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Land</td>
<td>Yield</td>
<td>Production</td>
<td>Yield</td>
<td>Production</td>
</tr>
<tr>
<td>Africa</td>
<td>4.4</td>
<td>4.4</td>
<td>3.5</td>
<td>271</td>
<td>21.1</td>
</tr>
<tr>
<td>Northern America</td>
<td>3.3</td>
<td>3.3</td>
<td>3.3</td>
<td>372</td>
<td>97.8</td>
</tr>
<tr>
<td>Central America</td>
<td>1.7</td>
<td>1.7</td>
<td>2.4</td>
<td>303.6</td>
<td>25.4</td>
</tr>
<tr>
<td>South America</td>
<td>3.9</td>
<td>3.9</td>
<td>2.9</td>
<td>372</td>
<td>97.8</td>
</tr>
<tr>
<td>Asia</td>
<td>11.0</td>
<td>11.0</td>
<td>11.0</td>
<td>303.6</td>
<td>25.4</td>
</tr>
<tr>
<td>Europe</td>
<td>25.4</td>
<td>25.4</td>
<td>25.4</td>
<td>303.6</td>
<td>25.4</td>
</tr>
<tr>
<td>Oceania</td>
<td>3.9</td>
<td>3.9</td>
<td>3.9</td>
<td>372</td>
<td>97.8</td>
</tr>
<tr>
<td>World</td>
<td>49.4</td>
<td>49.4</td>
<td>49.4</td>
<td>303.6</td>
<td>25.4</td>
</tr>
</tbody>
</table>

As seen in Table 5-7, Europe at 65% produces the largest share of barley in the world. North America with 36% and Asia with 30% are the biggest corn producers in the world. Almost 90% of the world’s rice is produced in Asia. South America with 45% and Asia with 40% are main sugarcane producers in the world. Asia at 44%, Europe 35% and North America 12% are the largest wheat producers in the world. Among all the crops studied in this study, sugarcane with an average yield of approximately 74 t/ha has the highest yield. The majority of these
crops are produced in the Americas, Europe and Asia mostly in developed or developing countries. Access to bioenergy and CCS technologies is one of the key parameters to develop BECCS in these regions. There are already CCS and BECCS projects operating or under planning in some of these regions. Globally there are 38 large-scale CCS projects with a total CO$_2$ capture capacity of approximately 70 Mt per annum. There are large-scale CCS projects in Canada, Europe, South America, Australia and Asia and the Middle East [75]. In addition, there are around twenty BECCS projects identified around the world, mostly located in North America and Europe [30, 83]. Access to significant agricultural residues, bioenergy and CCS and BECCS technologies and infrastructure facilitates development of BECCS using agricultural residues in these countries. Besides, utilising these residues to produce energy for the growing population while mitigating the emissions from their energy sector in developing countries, especially in Asia, could be a double advantage arising from implementing BECCS in these regions.

5.4.4 BECCS and food security

One very important feature of using residue from main food crops is avoiding food and energy competition. No land-use change for bioenergy development will be required. The availability of these residues is entangled with demand for food growth and as the population is expected to grow to about 9 billion by mid-century, so too will the food demand and residues. To feed the growing population, production of the selected crops is expected to grow 15 Mt/year to 28 Mt/year [336]. Therefore, even with more efficient agricultural production practices and increasing the yield, it is very unlikely that residue production by 2050 will see a declining trend. Current practice to develop perennial types of the crops such as wheat, barley and corn which are currently annual is a way to mitigate the net CO$_2$ of production and improving soil carbon and water retention [161, 162] and could mean even a greater potential to remove CO$_2$ if the residues are used in a BECCS system.

5.4.4 Social benefits of BECCS

More than 80% of bioenergy is used for traditional heating and cooking by 2.7 billion people living in developing and under-developed regions [46, 47]. Around 1.2 billion people in these areas do not have access to electricity [141, 337]. One example is India, where nearly 70% of cooking energy and 32% of its primary energy requirement is met with biomass [58]. Sustainable and affordable modern bioenergy could improve the energy security, social equity and economic growth in these poor areas [141]. Most bioenergy plants serving the local
community are small-scale and located in close proximity to the feedstock production site. Although that limits its domain, at the same time, provided there is a constant and predictable supply of biomass, it reduces the risk of outage due to any deficiency in supply from a more distant centralized source [338]. One of the main resources for biomass feedstock is waste from agricultural, forest and industrial sectors. In most of the counties there is no policy to manage these wastes. BECCS utilizes these waste as feedstock, which otherwise would be landfilled and a source of emission. That essentially turns a negative externality to a valuable good and makes it a source of income for local farmers, industry and state [50]. Sustainable BECCS can constructively contribute to achieving the United Nations Sustainable Development Goals (SDGs) [339]; By offering clean energy to decarbonise energy sector contributes to SDG-13 which is to take urgent action to combat climate change and its impacts; By providing sustainable energy for rural and remote areas contributes to SDG-7 which is to ensure access to affordable, reliable, sustainable and modern energy for all. If deployed sustainably, BECCS could offer significant carbon removal, enhancing the flexibility of the mitigation portfolio, helping sustainable land management and providing social benefits to rural communities [30].
5.5 Conclusions

Annually around 14 Gt residue from forestry and 4.4 Gt of residues is generated from main crops production, namely barley, wheat, corn, sugarcane and rice, around the world. This is a significant source of underutilised biomass. Utilising by producing energy together with CCS (BECCS) could offer significant energy generation and contribute to GHG mitigation considerably. However, not all this amount could be recovered. Residue collection technology, field specification factor and climate situation are factors affecting the amount of residue recoverable. Not all the residues are economically feasible to collect. The selling price of residue and costs to collect and store it, determines if it is feasible to collect. Another limitation is the amount of residues required for the competitive usages such as animal food and bedding and soil coverage. In this study, it was assumed that using 25% of the total residue produced complies with strict sustainability constrains. The results of this study showed that the sustainable potential of BECCS using agriculture and forestry residues to produce electricity, is 26 EJ/year, which translates to removing 2.76 Gt CO$_2$ per annum. In addition, using residues for BECCS avoids the common predicaments of expanding energy crops production such as land-use change, water scarcity and competition with food. However, there are challenges of implementing it at scale which are briefly listed below;

- Dispersed nature of agriculture residues which might lead to high transportation costs and GHG emission; It means sustainable supply chain management is essential.
- Technical challenges regarding the access to a reliable CO$_2$ sink and the necessary logistic; means that residue-based BECCS systems need extensive adaptive planning.
- Feedstock security for some residues might be a challenge especially seasonal variations and availability fluctuations due to local and global climate changes.
- Lack of policy schemes to support BECCS deployment and acknowledging its potentially significant role in delivering negative emission, hinders its development.
- High cost of BECCS in absence of economic incentive makes it presently unfavourable for investors and power suppliers.
Chapter 6

Opportunities for Application of BECCS in the Australian Power Sector
6.1 Introduction

Australia has one of the highest GHG emissions per capita among developed countries [340]. Its Intended Nationally Determined Contribution (INDC) to the Paris Agreement is to reduce emissions by 26 to 28 per cent on 2005 levels by 2030 [4]. The Paris Agreement places the onus on countries to meet the mid-term mitigation targets and increase their reduction target every five years in order to achieve global zero net emission in the second half of the 21st century. Australia’s current emissions are around 600 Mt CO$_2$-eq per year [341] or about 1% of global emissions. It has some mitigation policies in place to help achieving its emission reduction targets. These include two important policy tools already implemented in Australia, namely the Emission Reduction Fund (ERF) - and its related safeguard mechanisms, and the Renewable Energy Target (RET). The RET is expected to deliver a 200 Mt CO$_2$ reduction between 2015–2030 [4], with the Australian electricity sector emissions expected to decline to less than 0.25 t CO$_2$-eq /MWh by 2030 and to 0.1 t CO$_2$-eq /MWh by 2050 [342]. This is despite an expected growth in gross electricity generation of 49% (to 377 TWh) by 2050. It is predicted that renewable energy will need to contribute at least 70% of total electricity generation between 2030 to 2050 to make this target achievable [343].

Despite its potential contribution to decreasing emissions and supplying renewable energy, there has to date been no comprehensive assessment of BECCS in Australia. Critical to any such assessment is the availability and compatibility of the sub-systems, namely the bioenergy sector and the opportunities for establishing associated CCS systems. Below, the status of Australian bioenergy and CCS and their potential/challenges are discussed.

6.1.1 The Bioenergy sector in Australia

In Australia, bioenergy makes up around 4% of the Total Primary Energy Supplies (TPES) and 78% of renewable energy is dominated by an annual usage of 4–60 Mt of firewood for heating [344]. Bioenergy in Australia has an estimated value of more than of AUD 400 million per annum [345]. Other than its use as a cheap fuel for heating, the main drivers for development of bioenergy in Australia have been mitigating GHG and particulate emissions, the oil price and the growing demand for green transport fuels, sustainable development in rural areas, and the transition toward a less carbon intensive energy future [42]. There are a range of estimates for the potential of bioelectricity in Australia. For instance, the Clean Energy Council (CEC) Bioenergy Roadmap [338] estimated that bioenergy could provide 10.6 TWh/year by 2020 and this contribution could increase to 73 TWh/year by 2050. The Australian Business Roundtable
on Climate Change foresees that bioelectricity could supply 19.8 - 30.7% of Australia’s demand by 2050 [134]. According to the CEC [338], waste and by-products from high value primary crops would become the only long-term sustainable source of biomass for bioenergy in Australia. Bioenergy in Australia has mainly relied on organic wastes from the municipal, agricultural and forestry sectors. The main resources are sugarcane bagasse, forestry wastes, wood processing wastes, urban green waste, urban wood waste, landfill gas, agricultural wastes and wet organics. Currently, the bioenergy sector is mainly reliant on wood and bagasse with 42 per cent and 44 per cent, respectively [42].

The 114 bioenergy plants in Australia have a total installed capacity of 812 MW, constituting less than 1% (around 2 TWh) of total national power generation [343]. Around 60% of this bioenergy capacity is from agricultural residues - mostly bagasse. The remainder is made up of 124 MW from forest residues and 209 MW from landfill gas and sewage [346]. The largest bioenergy plant in Australia is the Pioneer Sugar Mill in Queensland with 68 MW capacity, using bagasse [347]. The total annual sugarcane production is about 35.5 Mt [42]. Bagasse is the by-product of sugarcane food manufacturing with a yield of 0.6 kg bagasse per 1 kg of dry sugarcane [307]. Bagasse is used as a feedstock to supply the energy required for running the sugarcane mill. In this way it helps to reduce costs of production and facilitates energy supply in remote areas. Between 10 to 20 tonnes of sugarcane residue per hectare is used to cover the soil for protection and to improve the soil erosion, water content, bulk density, and carbon stock [159].

Municipal Solid Waste (MSW) is another potentially important source of bioenergy. Australia ranks seventh among OECD (Organisation for Economic Co-operation and Development) countries in MSW generation per capita [348]. In 2010-2011 around 14 Mt of MSW was generated in Australia [349]. Around 17% of this was used to produce landfill gas (LFG) [350]. In recent decades, the percentage of the LFG captured for energy recovery has increased from zero in 1990 to 3 Mt in 2010 [295]. New South Wales and Victoria are the states with the highest level of LFG recovery [295]. After bagasse (66%), LFG (22%) is the largest source of bioenergy production in Australia [349]. Australia has one of the lowest rates of waste incineration for energy recovery (< 1%) amongst OECD countries [295]. One reason for this could be the low landfill tipping fee in Australia, though this varies from State to State. For instance in Queensland there is no fee for landfilled MSW but in Victoria the fee is about AUD 58.5 per tonne [295]. Different waste and energy policies have caused inhomogeneity in the
use of energy from waste across the country. However, some national initiatives such as the national waste policy, the Carbon Farming Initiative, the energy white paper and the renewable energy target, encourage mitigation of emissions from the waste sector [351]. Landfill gas can be flared or used to generate renewable electricity or heat. This electricity can be sold back into the electricity grid. Landfill operators can also generate tradable renewable energy certificates under the Renewable Energy Target scheme [352].

Bioenergy facilities have economic benefits but can also offer social benefits such as stable long-term employment for local and regional communities compared to purely agricultural-based communities, which may only offer seasonal employment or fewer job opportunities. Bioenergy enhances revenue to local producers by selling residues and in some cases fertilizers produced from organic waste digesters. Bioenergy facilities are typically installed in close proximity to the input resources thereby reducing the need to build new electricity networks in surrounding rural areas. Additionally, this reduces transmission losses arising from long distance electricity delivery. However, it is also important to assess the social impact of bioenergy facilities located close to the production sites in rural areas, including land ownership, labour conditions and equitable access to food, land and energy [353-356].

6.1.2 CCS in Australia

According to the Global CCS institute, there are 38 large-scale CCS projects under development around the world, seven of which are operational [75]. Australia has been one of the pioneers in CCS research and there are several CCS projects but none commercially operating as yet. Because of its geology and particularly its many sedimentary basins, Australia has a very large storage resource – enough to meet its potential storage needs for many decades, if not centuries. Currently there are no operational large scale CCS projects in Australia [81, 357], but there are a number that are progressing

- The Gorgon project operated by Chevron Corporation, will be the world largest natural gas CCS project and will separate, capture and geologically store 3 to 4 Mt of CO$_2$ per annum from high CO$_2$ content natural gas. It expected to commence injection of CO$_2$ in 2018, continuing for up to 40 years.
- The Callide oxyfuel pilot project successfully captured CO$_2$ from an existing 30 MW coal-fired power plant; the project has now been concluded.
- South-West Hub: the concept is to capture and store CO$_2$ emissions from industrial sources. Several wells have been drilled, but the future of the project is uncertain.
- The CarbonNet Project, located in the Gippsland Basin is currently evaluating offshore storage potential and economic opportunities for a potential project of 1 to 5 Mt CO$_2$ per annum.
- The CTSCo Project is evaluating storage opportunities in the Surat Basin in Queensland.
- The CO2CRC Otway Project in Victoria, the world’s most comprehensive storage research project is investigating storage in a depleted gas field and in saline aquifers. It has successfully injected, stored and monitored approximately 80 Kt of CO$_2$ rich gas.

Despite its potential as a large-scale mitigation option CCS has not yet contributed to significantly reducing emissions globally at scale. Excluding CO$_2$ injected for enhanced oil recovery, the current capacity of the CCS project around the world is around 40 Mt CO$_2$ per annum. This is far less than the capacity required to make a significant contribution to the 2°C scenario emission trajectory (around 4000 Mt per annum by 2040) [29, 75]. Between 2010 and 2016 more than 20 large scale CCS projects were cancelled across Europe, the U.S and Australia [358]. The CCS flagship program in Australia was reduced from AUD 1.9 billion in 2009 to AUD 500 million in 2015 [359]. This is mainly due to uncertain policy support and economic feasibility [29].

CCS is a heterogeneous technology involving capture, transportation and storage options. The cost of the CCS chain depends on several factors. In Australia depending on the location, the cost of transport and storage is between AUD 8 to AUD 55 /t CO$_2$ avoided [290, 357, 358]. Typically, the capture process for coal-fired power generation can constitute more than 50% to the total cost of a CCS project. According to CO2CRC [80] the capital cost of CCS is expected to decrease by 30–50% by 2030 [81]. CO$_2$ transport and storage hubs can further reduce the cost of CCS by cutting the cost of infrastructure and sharing facilities [82].

Whilst long term reliable storage of CO$_2$ has been one of the concerns regarding CCS, the Otway research facility has demonstrated safe long-term storage of CO$_2$ and new monitoring methods such as permanently deployed arrays, could reduce the cost of offshore and onshore monitoring significantly (by ten to hundred millions of dollars) [360]. Having large storage potential in its onshore and offshore sedimentary basins and know-how suggest that BECCS may be a particular opportunity for Australia, but of course this is very dependent on the ready availability of biomass and the overall feasibility of BECCS systems.
The present study offers some insights into the potential contribution of bioenergy with carbon capture and storage (BECCS) in achieving long-term decarbonising of the Australian energy sector and considers the availability of sustainable bioenergy resources and the economic viability and environmental impact of BECCS.
6.2 Modelling BECCS systems

Techno-economic and life cycle assessments were conducted for four organic-based bioenergy technologies with and without CCS: a) landfill gas combusted in a gas turbine (LFG-GT), b) bagasse (BG-CFB), c) forest residue (FR-CFB) and d) municipal solid waste (MSW-CFB) combusted in circulating fluidised beds (all without CCS) and their corresponding options with CCS: LFG-CCS, BG-CCS, FR-CCS and MSW-CCS.

The four systems without CCS (LFG-GT, BG-CCS, FR-CCS and MSW-CCS) were used as a reference to evaluate the environmental and economic implications of coupling CCS with these organic-based bioenergy technologies. Data available from published literature, reports from national and international organizations and online databases was used in the analysis.

6.2.1 Source and sink matching

One important factor determining the technical feasibility and economic viability of BECCS is the transportation of biomass to a bioenergy plant and of captured CO₂ to a geological storage site. Comparing the location of the storage sites projects in the CO2CRC map [361] with the map of bioenergy facilities developed by Geoscience Australia [42], there appears to be some natural source–sink matching between the biomass production sites and CO₂ storage locations. This is potentially very important for large-scale BECCS deployment in Australia.

Figure 6-1 shows the distribution of bioenergy facilities around the Australia. As seen, the majority of bioelectricity plants are located on the east coast in close proximity to croplands and forests. All bagasse facilities (blue squares on the map), with a total capacity of around 400 MW, spread within 1500 km of the Queensland east coast. A few wood waste facilities (purple circles on the map) with a total capacity of ~ 30 MW are on the boarder of New South Wales and Queensland. Figure 6-2, illustrates the Australian CCS projects and basins with potential for CO₂ storage. Comparing figures 6-1 and 6-2, it can be seen that there is some potential for geological storage in close proximity to bioelectricity plants, especially in Queensland.
Figure 6-1- Distribution of bioenergy electricity and heat generation facilities in Australia [42]

Figure 6-2- Australia’s basins for CO₂ storage potential and CCS projects [361]

The Surat Basin is an asymmetric intracratonic basin with an area of 300,000 km² in central southern Queensland and central northern New South Wales, within 400 to 600 km of the emissions hubs [285]. It includes sedimentary formations with up to 2.9 Gt of CO₂ storage
potential [362] and an annual potential to store 50 Mt CO\textsubscript{2} [285]. The Surat Basin storage potential is currently under evaluation [358].

In this study it is assumed that bioelectricity plants are located in the south east of Queensland within a 500 km distance from a storage well in the Surat Basin (Figure 6-2).

6.2.3 BECCS power plants

In the BG-CCS, FR-CCS and MSW-CCS models, biomass is combusted in a circulating fluidised bed (CFB) plant to produce electricity. The electricity generated is exported to the grid. The residue from the combustion plant is mostly ash which is landfilled. The reason for using CFB is its ability to burn fuels with high moisture and low heating value with highest efficiency. In addition, CFB does not require fuel pre-processing. The gas velocity in CFB reactors is between 3–9 m/s which results in lower SOx, NOx and HCl formation. CFB reactors offer higher efficiency and less bottom ash than fixed bed boilers [263]. One of the challenges of waste combustion is the presence of trace elements such as Si, Al, K, Mg, Ca and Na which result in formation of heavy metals and salts in the fly ash and bottom ash. Thermal stabilisation by melting and sintering is applied to treat these residues before landfill [264-268]. Electrostatic precipitators are used to filter the dust and most of the heavy metals.

In the LFG-CCS system, wet MSW is collected and stored in a sanitary landfill facility, where LFG is produced through the activities of methanogenic microorganisms. Methane comprises only 35–65 vol% of LFG. The remainder is CO\textsubscript{2} (15–50 vol\%), H\textsubscript{2} (0–3 vol\%), O\textsubscript{2} (0–1 vol\%), H\textsubscript{2}S (0–3 vol\%) and N\textsubscript{2} (10–15\%) [269]. In this model, it is assumed that the sanitary LFG facility is equipped with a gas cleaning unit. Therefore, the final product sent through the pipeline to the gas turbine contains 50%:50% CO\textsubscript{2}/CH\textsubscript{4}. In some plants, flaring is used to oxidise a fraction of the methane collected [270], but in this study it is assumed that all of the CH\textsubscript{4} collected is used for power production. The methanogenic phase of organic waste decomposition is a steady process and typically continues for 20 years. The LFG generation flow rate is not linear and depends on several parameters such as composition of waste, the mass of solid, water content, pH and ambient temperature. The collection efficiency is expressed as the percentage of LFG collected in a landfill site. In an LFG unit with intermediate soil cover, the LFG collection efficiency will range between 55%–95% [271]. In more advanced facilities with bioreactor LGF cells, collection efficiency could reach close to 100% [272, 273]. In this study, it is assumed that the LFG is collected from a vertical well with an
LGF collection rate of 95%. The vertical well for production of the gas is schedule 80 PVC pipe with a diameter of 100 cm. Vacuum blowers send the LFG into pipelines for transportation to a gas cleaning facility [279]. After extraction, LFG requires dehumidification and removal of impurities and particulates. In this study, LFG is combusted in a gas turbine, which means siloxane and hydrogen sulphide must also be removed. Siloxane in the waste converts to SiO₂ during LFG combustion and is deposited on the internal surfaces of gas turbines, causing damage and increasing maintenance, while sulphur compounds in the waste lead to formation of corrosive sulphides. In this study, a 10 MW combined cycle gas turbine which utilises the exhaust heat from the turbine to produce electricity with an efficiency up to 40% [277] was assumed for LFG combustion. Gas turbines have relatively low O&M cost compared to internal combustion engines and are more corrosion resistant.

6.2.3.1 Carbon Capture and storage system

The CO₂ capture process in our model is post-combustion chemical absorption using MEA. After separation, the CO₂ is compressed to 150 bars and sent to the CO₂ sink and stored underground in suitable geological formations. Pipelines and compression equipment for transporting the CO₂ and injection wells are components of the CO₂ storage system. The CO₂ capture and storage processes are identical in all the BECCS models.

6.2.3.1.1 CO₂ capture unit

The CO₂ capture unit used in this chapter is the same as the CCS system used in Chapter 4, Section 4.2.3.1.

6.2.3.1.2 CO₂ Transport and storage system

The CO₂ capture unit used in this chapter is the same as the CCS system used in Chapter 4, Section 4.2.3.2.

6.2.4 Cost of electricity production

To investigate the economic viability of BG-CCS, FR-CCS, MSW-CCS and LFG-CCS electricity production systems, techno-economic assessments were conducted. The levelised cost of electricity production (LCOE) was calculated for each system. The LCOE of black coal combusted in a pulverised coal combustion plant (Coal-PC) with and without CCS technology was used as the baseline. In the year 2013–2014, the percentage of power generation from coal was 61% in Australia [363].
Techno-economic assessments in this chapter are the same as those described in chapter 4, section 4.2.4.

Compositions of the fuels are listed in Table 6-1. It is assumed that LFG sent to the gas turbine is 50:50% CO₂/CH₄ and its LHV is 18.3 MJ/kg. Table 6-2 lists the techno-economic data for BG-CCS, FR-CCS, MSW-CCS, LFG-CCS and Coal-CCS, based on information available in the literature.

Table 6-1- Elemental compositions of the fuels [159, 217, 235, 313-315]

<table>
<thead>
<tr>
<th>Composition</th>
<th>Bagasse</th>
<th>Forest Residue</th>
<th>MSW</th>
<th>Black coal</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td>C (wt%)</td>
<td>48.6</td>
<td>45.2-52.0</td>
<td>50</td>
<td>45.8-54.2</td>
</tr>
<tr>
<td>H (wt%)</td>
<td>5.9</td>
<td>5.4-6.4</td>
<td>6</td>
<td>5.6-6.4</td>
</tr>
<tr>
<td>O (wt%)</td>
<td>42.8</td>
<td>37.5-48.1</td>
<td>42</td>
<td>39.0-45.0</td>
</tr>
<tr>
<td>N (wt%)</td>
<td>0.16</td>
<td>0.14-0.18</td>
<td>0.6</td>
<td>0.55-1.25</td>
</tr>
<tr>
<td>S (wt%)</td>
<td>0.04</td>
<td>0.03-0.045</td>
<td>0.1</td>
<td>0.06-0.14</td>
</tr>
<tr>
<td>Moisture</td>
<td>45</td>
<td>39.6-50.4</td>
<td>45</td>
<td>36.2-53.8</td>
</tr>
<tr>
<td>LHV (MJ/Kg)</td>
<td>14.60</td>
<td>15.36</td>
<td>16.80</td>
<td>4.55-6.45</td>
</tr>
</tbody>
</table>
### Table 6-2: Techno-economic input data

<table>
<thead>
<tr>
<th>Process</th>
<th>Capacity (MW)</th>
<th>Thermal efficiency (% (LHV))</th>
<th>capacity factor%</th>
<th>Lifetime (years)</th>
<th>Capital ($/kW)</th>
<th>Variable O&amp;M</th>
<th>Fixed O&amp;M ($/kw/year)</th>
<th>Fuel cost (delivered) ($/t)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>LFG collection-cleaning plant [286]</td>
<td>10</td>
<td>-</td>
<td>100</td>
<td>25</td>
<td>Mean: 1500</td>
<td>-</td>
<td>Mean: 57</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 1350-1650</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LFG power plant gas turbine [286, 287]</td>
<td>10</td>
<td>Mean: 35</td>
<td>70</td>
<td>25</td>
<td>Mean: 400</td>
<td>Mean 3.6</td>
<td>Mean: 57</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 28-42</td>
<td></td>
<td></td>
<td>Range: 360-440</td>
<td>($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 2.9-4.3</td>
<td>($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 39-75</td>
<td>($/MWh)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>MSW plant [254, 288]</td>
<td>10</td>
<td>Mean: 30</td>
<td>90</td>
<td>25</td>
<td>Mean: 6000</td>
<td>Mean 29</td>
<td>Mean: 192</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 23-37</td>
<td></td>
<td></td>
<td>Range: 4950-7050</td>
<td>($/tMSW)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 21-37</td>
<td>($/tMSW)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 175-209</td>
<td>($/tMSW)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bagasse and Forest residue plant [218, 316, 317]</td>
<td>10</td>
<td>Mean: 40</td>
<td>90</td>
<td>30</td>
<td>Mean: 3000</td>
<td>Mean 4.5</td>
<td>Mean: 96</td>
<td>Mean (Bagasse):4000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 37-43</td>
<td></td>
<td></td>
<td>Range: 2750-3250</td>
<td>($/MWh)</td>
<td></td>
<td>Mean (Bagasse):28-52</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 4.2-4.7</td>
<td>($/MWh)</td>
<td></td>
<td>Mean (Forest residue):48</td>
</tr>
<tr>
<td>Coal plant [80, 289]</td>
<td>150</td>
<td>Mean: 42</td>
<td>90</td>
<td>30</td>
<td>Mean: 2400</td>
<td>Mean 4.5</td>
<td>Mean: 36</td>
<td>Mean: 50</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 40-44</td>
<td></td>
<td></td>
<td>Range: 2100-2700</td>
<td>($/MWh)</td>
<td></td>
<td>Range: 28-44</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Range: 3.3-5.7</td>
<td>($/MWh)</td>
<td></td>
<td>Range: 39-61</td>
</tr>
<tr>
<td>Capture process [69, 290-292]</td>
<td>150</td>
<td>-</td>
<td>100</td>
<td>30</td>
<td>Mean: 2100</td>
<td>Mean 9</td>
<td>Mean: 103</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Range: 1760-2440</td>
<td></td>
<td></td>
<td>Range: 5.6-12.4</td>
<td>($/MWh)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Price of dry biomass at the power plant gate

LFG pipeline and condensation = 205 ($/km) [286]
CO₂ Transportation (200-500 km) = 8 ($/tCO₂) [293]
CO₂ Storage and monitoring = 15 ($/tCO₂) [70, 285]
CO₂ capture efficiency of the MEA unit is 90%
Interest rate (i) =10%
Inflation rate (ρ)=0%
6.2.5 Environmental impact assessment

Life cycle assessment (LCA) was carried out using SimaPro software version 8.2.0.0 developed by PRé Sustainability Group. The LCA study includes four main steps; defining the goal and scope, life cycle inventory of all the inputs and outputs, lifecycle impact assessment, and interpretation of the results. Allocation modelling was applied to assess the environmental impacts of the BECCS systems with the functional unit of 1 KWh electricity generated. In these assessments, the ALCAS Best Practice LCIA method [296] was used. ALCAS uses the Global Warming Potential (GWP 100) according to IPCC’s fifth assessment report. The impact categories considered in this study were global warming (GWP100a), abiotic depletion (fossil fuels), Ozone Layer Depletion (ODP), eutrophication, acidification, particulate matter, human toxicity (cancer), human toxicity (non-cancer), freshwater ecotoxicity and water scarcity. Most of the inventory data are taken from Ecoinvent-3 [61] and the AusLCI database (developed by Australian LCA Society (ALCAS)) [298]. Data regarding the MEA CO₂ capture unit was derived from Hooper et al [299, 300]. CO₂ and CH₄ emissions are based on the degradable organic content (DOC) of the waste calculated using the methodology in the National Inventory Report [269]. Table 6-3 shows the composition of MSW in Australia.

Table 6-3- Composition of Australian MSW [245, 250, 256, 269, 270, 272, 295]

<table>
<thead>
<tr>
<th>Composition of MSW</th>
<th>weight %</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
</tr>
<tr>
<td>Food</td>
<td>38.7</td>
</tr>
<tr>
<td>Mixed paper</td>
<td>6.7</td>
</tr>
<tr>
<td>Garden and green</td>
<td>18.9</td>
</tr>
<tr>
<td>Textiles</td>
<td>1.8</td>
</tr>
<tr>
<td>Rubber and leather</td>
<td>1.2</td>
</tr>
<tr>
<td>Plastic</td>
<td>11.5</td>
</tr>
<tr>
<td>Inert waste (metal, glass,  etc.)</td>
<td>21.2</td>
</tr>
</tbody>
</table>

Figures 6-3 to 6-5 illustrate the BECCS systems with the major components and the system boundaries.

It is assumed that bagasse is produced as a by-product of the sugar industry. The AusLCI database used, includes the transport of sugarcane to the sugar refinery, the processing of sugarcane to sugar and bagasse collection.
In this study it is assumed that 25% of final harvest could be utilised for energy production. In addition, forest residues are assumed to be collected as part of non-commercial thinning and whole-tree logging operations on private and state-owned forest lands; national forests, national parks, and other federal or state lands are explicitly excluded from this module. Both bagasse and forestry residues are assumed to be supplied to the power plants through 100 km road transportation.

Figure 6-3- System boundary for BG-CCS and FR-CCS inventories

In the case of MSW-CCS (see Figure 6-4) and LFG-CCS (see Figure 6-5), MSW generation and transportation is outside of the system boundary. All the waste after separation of the reusable and recyclables are transported to the waste storage site. The waste is assumed to be composed of some inorganic material (7.2%), mainly from plastic. The CO₂ emission from plastic is non-biogenic and is deducted from the total incineration plant CO₂ emission.
Figure 6-4- System boundary for MSW-CCS inventories

Figure 6-5- System boundary for LFG-CCS inventories
The waste decomposition and methane generation in a sanitary landfill site was based on first order decay (FOD), as proposed by IPCC Guidelines for National Greenhouse Gas Inventories 2006 [60]. The LFG generation model in this chapter is the same as described in Chapter 4, Section 4.2.5.
6.3 Results

6.3.1 Cost of electricity production

CO$_2$ emissions and levelised cost of electricity production (LCOE) were calculated for LFG combusted in a gas turbine (LFG-GT), MSW, bagasse and forest residue combusted in a CFB (MSW-CFB, BG-CFB and FR-CFB) and pulverised coal combustion (Coal-PC) with and without CCS (Table 6-4).

100% of the CO$_2$ from combustion of LFG, bagasse and forest residue is biogenic, hence when this CO$_2$ is sequestered and stored it is regarded as negative CO$_2$. In the case of MSW, it is assumed that 7.2% of the CO$_2$ emission originates from inorganic combustibles such as plastic (see Table 6-3). The CO$_2$ capture rate of the MEA process in this study is 90%. After CCS, CO$_2$ emission of the coal-fired plant is reduced from 0.81 t/MWh to 0.11 t/MWh. All the BECCS systems deliver net negative CO$_2$ emission of -0.89 to -1.35 t/MWh and therefore more avoided CO$_2$ emission than Coal-CCS.

Under a business as usual (BAU) scenario where there is no emission policy in effect. Under a BAU scenario, not surprisingly, the LCOE of the BECCS systems modelled in this study are higher than that of coal-fired power plants -with or without CCS. The high production cost of BECCS, due to its high investment and O&M costs, would not be competitive in the power market. The LCOE of BECCS systems in this study is $160–$212/MWh which lies in the upper range of LCOE through BECCS of $70–$230 /MWh, reported in Kemper’s study [30].

The cost of LFG-CCS ($160 /MWh) is lower than other BECCS systems due to the lower capital and operating cost of a gas turbine compared to a solid combustion furnace. Circulating fluidised bed combustion technology used for MSW, bagasse and forest residue has a higher capital and operating cost than the pulverised coal-fired plant. MSW-CCS is the most expensive system, as it requires costly modifications in order to manage its high moisture content and flue gas cleaning requirements. However, all the BECCS systems offer up to 30% lower cost of avoided CO$_2$ emission than Coal-CCS. The lower avoided cost of CO$_2$ emission could significantly determine economic potential of BECCS systems compared to other technologies on the market.

The cost of negative carbon in this study is $70–$80/t CO$_2$. This is in line with the estimate by McGlashan et al. [18], which found costs for a large BECCS power plant between $59/t CO$_2$
and $111/t CO_2$. In a study by McLaren [19] a range of $70–$250/t CO_2 is suggested for BECCS. This study indicated that in absence of rapid development, the cost of BECCS is more likely to be around $150/t CO_2 [19].

Table 6-4: CO_2 emission, LCOE with and without CCS and cost of avoided emission

<table>
<thead>
<tr>
<th>Energy Technology</th>
<th>Biogenic CO_2 Emission (%)</th>
<th>CO_2 Emission (t/MWh)</th>
<th>Avoided CO_2 Emission (t/MWh)</th>
<th>LCOE($/MWh) No CCS</th>
<th>Cost of CCS ($/MWh)</th>
<th>Cost of Avoided CO_2 Emission ($/t)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LFG-GT</td>
<td>100</td>
<td>0</td>
<td>-1.35</td>
<td>67</td>
<td>160</td>
<td>94</td>
</tr>
<tr>
<td>MSW-CFB</td>
<td>93</td>
<td>0.05</td>
<td>-0.89</td>
<td>140</td>
<td>213</td>
<td>74</td>
</tr>
<tr>
<td>BG-CFB</td>
<td>100</td>
<td>0</td>
<td>-1.32</td>
<td>70</td>
<td>171</td>
<td>100</td>
</tr>
<tr>
<td>FR-CFB</td>
<td>100</td>
<td>0</td>
<td>-1.29</td>
<td>85</td>
<td>175</td>
<td>89</td>
</tr>
<tr>
<td>Coal-PC</td>
<td>0</td>
<td>0.81</td>
<td>0.11</td>
<td>60</td>
<td>132</td>
<td>72</td>
</tr>
</tbody>
</table>

Socio-institutional circumstances are key factors determining the future of BECCS. Currently, BECCS like other NETs is not recognised in the INDCs submitted in support of the Paris Agreement [3]. Emission policies can have a major impact on the cost of electricity production. Unlike other renewable energies such as wind and solar, bioenergy lacks integrated governmental and public support. It has made investment in BECCS in Australia and elsewhere high risk with controversies regarding its social and environmental impacts among public and experts [7]. According to Bhave et al. [225] the most significant barrier of BECCS deployment is lack of economic and regulatory support rather than technical capacity.

In this study the effect of three different policies was assessed in an Australian context. The first emission policy investigated in this study was a renewable energy certificate scheme (REC). Under such a scheme, certificates (RECs) are given on the basis of per MWh of electricity produced from recognised renewable power suppliers. Renewable Energy Certificate schemes have been successfully implemented in a number of countries including, the U.S, Australia and especially in Europe, where they have strongly supported deployment of renewable electricity. Under a RET scheme energy retailers and large energy users are obliged to purchase a share of their electricity from renewable energy producers. Power plants using LFG, MSW, agricultural and forestry residues as feedstock are eligible for RECs. In this study it is assumed that BECCS systems using these biomass resources will receive REC as well. As seen in Figure 6-6, introducing a REC into the energy market influences the final LCOE of renewables significantly. At a REC higher than $45/MWh, the LCOE of LFG-CCS,
BG-CCS and FR-CCS will be lower than Coal-CCS. Currently the price of a REC for large scale generation is around $66/MWh [364], which means that under such circumstances deploying BECCS systems can be more economically attractive than Coal-CCS. At a REC higher than $115/MWh BECCS can become a cheaper power production option than a coal-fired power plant without CCS. MSW-CCS is a more expensive BECCS option than the others and would require a higher REC to be able to compete with coal-fired power generation.

Figure 6-6- Impact of REC on LCOE of BG-CCS, FR-CCS, MSW-CCS, LFG-CCS, Coal-CCS and coal w/o CCS

The second emission mitigation policy considered was carbon pricing. Under such a scenario, power suppliers using fossil-fuel are obliged to pay tax per tonne of CO$_2$ emitted to the atmosphere. As the main incentive for BECCS is its capacity to produce negative emissions, the avoided cost of CO$_2$ emission determines its economic potential compared with other technologies on the market. Therefore, one of the decisive factors to estimate the cost of BECCS is the carbon price. Carbon price has been identified as the most effective policy mechanism to support BECCS deployment [102]. Carbon pricing was introduced in Australia in 2011 but abolished in 2014 [365]. The effect of a reintroducing a carbon tax into the Australian energy market on the LCOE of coal-fired power plants modelled in this study is illustrated in Figure 6-7. The BECCS systems are assumed to be exempted from a carbon tax as the CO$_2$ they emit is biogenic. Raising the carbon price from 0 to $200/t CO$_2$ increases the
LCOE of the coal-CCS system by 16% ($132/MWh to $153/MWh). A black coal-fired plant without CCS emits around 0.81 t CO$_2$/MWh to the atmosphere; a carbon price higher than $125/t CO$_2$ is needed to raise the LCOE of a coal-fired plant to a level that BECCS options could become cost-comparable in the electricity market. A study by CO2CRC in 2015 [80] showed that wind and Solar PV will be competitive with coal-based power with a $30/t CO$_2$ and $70/t CO$_2$ price on CO$_2$ emissions, respectively. So, in the absence of supportive policy schemes to target negative emissions, BECCS will be considerably more costly than other renewables.

Given the current status of carbon pricing schemes in Australia, it seems highly unlikely that such a carbon price will be implemented in the near future given the current level of political and commercial opposition. The situation is no brighter in other countries. Some developed countries have levied a tax on CO$_2$ emission but there is no integrated price for CO$_2$. The current carbon price under EU ETS fluctuates around 8.5 Euro per tonne CO$_2$ [109], which is far less that the level required for sharp abatements. Lack of effective and wide-spread carbon price hinders BECCS large-scale deployment and limits it to niche applications in Australia and elsewhere.

Figure 6-7- Impact of Carbon Pricing on LCOE of BG-CCS, FR-CCS, MSW-CCS, LFG-CCS, Coal-CCS and coal w/o CCS
The third emission mitigation policy considered in this study was that of a negative CO₂ credit. In this scenario, BECCS technologies receive a refund for each tonne of CO₂ removed from the atmosphere.

In this case the Coal-CCS plant does not receive credit for the CO₂ captured, but BG-CCS, FR-CCS, LFG-CCS and MSW-CCS do benefit. As seen in Figure 6-8, the effect of a negative emission policy on LCOE of BG-CCS, FR-CCS and LFG-CCS is greater than for MSW-CCS. Due to the 7.2% of non-biogenic CO₂ produced in MSW-CCS, its net negative CO₂ delivered is less than other BECCS options with 100% biogenic CO₂. A negative CO₂ credit of $35/t CO₂ makes the LCOE of BG-CCS, FR-CCS and LFG-CCS lower than the LCOE of Coal-CCS. A much greater value, of $95/t CO₂, is needed to make BECCS systems competitive with base-load coal-fired power generation. Mac Dowell and Fajardy [366] found that to incentivise large-scale BECCS deployment a minimum 12% internal rate of return with £75/t CO₂ (equal to $105/t CO₂, 2018 rate) price of negative emission is required. Though they applied a different techno-economic approach, their findings are not far from the negative emission price recommended in this study, $95/t CO₂. Based on the economic evaluation undertaken in this study, negative CO₂ credit scheme supports deployment of BECCS systems at a lower cost than conventional carbon pricing. However, negative emission achieved through the application of BECCS is a relatively new concept and there is currently no scheme in Australia or elsewhere that takes the value of negative emission into account. One of the few studies on the role of negative emission pricing is a study by Mathews [367] which investigated the possible mechanisms to certify negative carbon. According to this study, designating carbon credit per 1 tonne of CO₂ removed or avoided and integration of technical specifications into clean development mechanism (CDM) are the most effective policy instruments for BECCS support.
6.3.2 Environmental impact assessment

The LCA of the four BECCS options in this study was modelled using SimaPro 8.2.0.0 software and the ALCAS Best Practice LCIA [296] method was applied for the impact assessment. Historically, unsustainable biomass harvest and forest clearing has led to a major loss of natural forests and degradation of productive lands, large GHG emissions, and loss of biodiversity and carbon stock [133]. However, the results of LCA in this study (see Table 6-5) show that using organic wastes to produce energy through BECCS systems can have positive environmental impacts in almost all impact categories. In BECCS systems the net global warming effect is significantly reduced compared to the baseline systems. In some categories such as ozone depletion, photochemical oxidation, human toxicity and water scarcity, BECCS options have a lower environmental performance compared to those with no CCS. All of the BECCS systems studied lead to net negative GHG emission. In Table 6-5, the LCA of energy technologies with and without CCS are listed per KWh electricity generated. As shown in Table 6-5, net GHG emissions per KWh of electricity generated for LFG-CCS, MSW-CCS, BG-CCS and FR-CCS are -0.66, -1.48, -1.81 and -1.78 kg CO$_2$-eq respectively. These results show that the organic waste-based BECCS systems studied can deliver net negative emission taking into account all the processes involved in their lifecycle. The LCA with a different functional unit of 1 kg biomass utilised was conducted separately for each BECCS option. Under this LCA,
net GHG emissions for LFG-CCS (per kg of waste delivered to the landfill site- see Figure 6-5), MSW-CCS (per kg of MSW transposed to incineration plant- see Figure 6-4), BG-CCS and FR-CCS (per kg organic residue produced—see Figure 6-3) were -0.44, -0.8, -2.74 and -2.7 kg CO₂-eq respectively.

It is worth noting that there are other forms of carbon savings in these systems, compared to BECCS systems using dedicated crops which are not considered in these models. For instance, invasive agricultural techniques, land clearing, deforestation and interrupting soil carbon often involved in producing energy crops intensifies the net carbon emission which are avoided by using waste/by-products instead.

Table 6-5- Lifecycle impact assessment of the BECCS systems per 1 KWh of electricity generated

<table>
<thead>
<tr>
<th>Impact Category</th>
<th>Unit</th>
<th>Unit</th>
<th>LFG-GT</th>
<th>MSW-CFB</th>
<th>BG-CFB</th>
<th>FR-CFB</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>CCS</td>
<td>No CCS</td>
<td>CCS</td>
<td>No CCS</td>
</tr>
<tr>
<td>Global warming (GWP100a)</td>
<td>kg CO₂-eq</td>
<td>-6.56E-01</td>
<td>1.7E-01</td>
<td>-1.48E+00</td>
<td>5.0E-01</td>
<td>-1.81E+00</td>
</tr>
<tr>
<td>Abiotic depletion (Fossil fuels)</td>
<td>MJ NCV</td>
<td>-6.41E+00</td>
<td>-8.0E+00</td>
<td>-5.27E+00</td>
<td>-9.7E+00</td>
<td>-6.98E+00</td>
</tr>
<tr>
<td>Ozone layer depletion (ODP)</td>
<td>kg CFC₁₁₋₁₂ eq</td>
<td>2.90E-08</td>
<td>1.3E-08</td>
<td>1.08E-08</td>
<td>-3.7E-10</td>
<td>1.56E-08</td>
</tr>
<tr>
<td>Photochemical oxidation</td>
<td>kg C₄H₆ eq</td>
<td>2.56E-04</td>
<td>2.7E-04</td>
<td>9.70E-06</td>
<td>-2.6E-05</td>
<td>2.53E-04</td>
</tr>
<tr>
<td>Acidification</td>
<td>kg SO₂-eq</td>
<td>2.93E-04</td>
<td>-4.6E-04</td>
<td>-3.29E-04</td>
<td>-9.5E-04</td>
<td>-2.67E-04</td>
</tr>
<tr>
<td>Eutrophication</td>
<td>kg PO₄₋₄ eq</td>
<td>2.75E-04</td>
<td>-9.2E-05</td>
<td>-8.79E-05</td>
<td>-2.8E-04</td>
<td>-4.52E-05</td>
</tr>
<tr>
<td>Particulate Matter</td>
<td>kg PM2.3</td>
<td>2.51E-04</td>
<td>8.4E-05</td>
<td>-2.70E-05</td>
<td>-6.3E-05</td>
<td>5.16E-05</td>
</tr>
<tr>
<td>Human Toxicity (cancer)</td>
<td>CTUh</td>
<td>1.51E-08</td>
<td>6.8E-09</td>
<td>5.87E-10</td>
<td>-2.2E-10</td>
<td>5.77E-09</td>
</tr>
<tr>
<td>Human toxicity (non-cancer)</td>
<td>CTUh</td>
<td>4.36E-08</td>
<td>2.8E-08</td>
<td>-1.75E-08</td>
<td>-2.5E-08</td>
<td>1.09E-07</td>
</tr>
<tr>
<td>Freshwater Ecotoxicity</td>
<td>CTUe</td>
<td>-3.49E+04</td>
<td>-3.4E+04</td>
<td>-2.68E+04</td>
<td>-3.4E+04</td>
<td>-3.54E+04</td>
</tr>
<tr>
<td>Water Scarcity</td>
<td>m³-eq</td>
<td>4.85E-03</td>
<td>1.7E-03</td>
<td>2.73E-03</td>
<td>4.7E-05</td>
<td>3.12E-03</td>
</tr>
</tbody>
</table>

Bioenergy requires more water than coal-fired power generation and using fertilizer and pesticides can increase water pollution. However, in waste-based BECCS systems, the crucial impacts such water eutrophication and water scarcity are all minimal. On the other hand, water consumption in the BECCS systems is more than the cases without CCS. This is mainly due to the water used in the CO₂ capture process. In this study it is assumed that in the CCS process the water used for cooling in the MEA–based CO₂ capture unit is around 106 m³/t CO₂ [153]. Compared to dedicated energy crops, using residues and waste for BECCS has a much lower
impact on water scarcity. Water intensity of negative emissions through BECCS was estimated around 0.06 m³/kg CO₂ by Smith et al. [11] and 0.3–1.3 m³/kg CO₂ by Fajardy et al. [124]. Both of these studies focus on dedicated crops for BECCS. In this study the water consumption of residue-based BECCS resulted in lower water usage. BG-CCS and FR-CCS used 0.0017 m³/kg CO₂ and 0.0014 m³/kg CO₂, respectively. Water consumption by LFG-CCS systems was higher, with 0.0074 m³/kg CO₂. It is particularly important for Australia with an arid environment which is already facing challenges of water scarcity and lack of fertile land.

It is assumed that the electricity generated in these systems is sent to the grid and potentially replaces electricity fossil fuel-based power [363]. All the BECCS systems seem to improve depletion of fossil fuel resources and lead to negative fossil fuel depletion. This is source of significant negative emission in BECCS systems.

Biodiversity loss is an important environmental impact which was not part of this LCA. However, if natural land transformation and water contamination are taken as the main factors of biodiversity loss, then these assessments show that there is no significant risk of biodiversity loss regarding these BECCS systems. The LCA results show that the BECCS systems modelled in this study are environmentally benign with overall positive impacts.

6.3.3 Trade-off analysis

When choosing the most sustainable BECCS alternative, it is important to consider trade-offs. The objectives of the decision makers determine the outcome of these trade-offs. Figure 6-9 shows the trade-off between global warming potential, water use and LCOE (under no emission policy) of the BECCS systems. These three parameters are among the most crucial economic and environmental impacts of a BECCS system, and have been the cause of controversy both for bioenergy systems and more recently for BECCS systems [6, 35, 54, 124]. The low value of water use in the BECCS systems is due to the assumption in the LCA model that no additional water use is required for production of the residues. Association with the CCS process results in a moderate increase in water consumption. All the BECCS systems have net negative CO₂eq in their LCA, which is due to the fundamental assumption that organic waste and residues are carbon-neutral [63], so that when used for energy production and with associated CCS, the net emissions are negative. Under a regime when economic viability has the higher priority LFG-CCS with lowest LCOE would be the optimal choice. MSW-CCS with highest LCOE is on the other end of this spectrum. On the other hand, when environmental concerns are prioritised, BG-CCS with lowest GWP and water consumption ranks first.
Similarly, under a trade-off regime between environmental and economic performance, BG-CCS with relatively low LCOE and highest negative emission potential and lowest water usage seems to be the optimal choice. FR-CCS with slightly higher LCOE and GWP ranks second.

![Figure 6-9: GWP, water consumption and LCOE trade-off](image)

**6.3.4 Effect of different BECCS configurations on its overall performance**

Biomass is an inherently dispersed resource with relatively low energy content. Hence, long transportation distance could become an economic and environmental challenge. That is why most bioenergy plants are small-to-medium size and adjacent to the biomass production fields [36]. This could lead to another challenge in many areas namely that of connecting each of these bioenergy plants to the CO₂ storage site, for this would significantly add to the costs of small-scale BECCS plants. Some solutions such as establishing CO₂ transport and storage hubs [82] or biomass combustion hubs or transportable combustion systems that could be moved to sites of biomass accumulation, could cut down CCS costs considerably. BG-CCS, which proved to be the optimal choice in an economic-environmental trade-off, was selected to examine these configurations. In the reference case showed in Figure 6-3, the overall source to
sink distance is 600 km, comprised of 100 km from biomass source to the power plant and 500 km from power plant to the CO₂ storage site.

To evaluate the effect of different sink and source configurations on lifecycle CO₂ and LCOE of BECCS, two systems were studied. As seen in Figure 6-10-a, in the first arrangement (biomass hub), a BG-CCS plant with 50 MW capacity is located next to a CO₂ storage site with a minimum distance and the bagasse is transported by road from five different sources each within an average of 600 km distance. d₁ – d₅ are the length of CO₂ pipeline. In the second configuration (CO₂ storage hub- Figure 6-10-b) five BG-CCS plants with 10 MW capacity are located next to bagasse resource, so there is no feedstock transportation cost applied. Each power plant is connected to a central CO₂ storage well. It is assumed that the average distance of these bioenergy plants to the storage site is 600 km. In both cases, the efficiency of all power plants is 40% and capacity factor is 90%.

![Figure 6-10- Centralised biomass hub system (a) and centralised CO₂ storage hub system (b)](image)

The techno-economic results show that CO₂ storage hub arrangement results in 25% lower levelised cost of electricity ($176/MWh) compared to the biomass hub configuration ($235/MWh). This is mainly because of high biomass transportation cost ($18/t/100km [313]) and relatively low cost of CO₂ transportation ($12/t CO₂ for 500–2000 km [293]). On the other hand, the LCA of the two systems showed the biomass hub system as the favourable option. Results are illustrated in Figure 6-11. The total CO₂-eq emission is -1.3 t CO₂/MWh from the CO₂ storage hub system and is -1.8 t CO₂/MWh for the centralised biomass hub configuration.
The main difference is the CO₂ emission from biomass transportation and CO₂ transportation. In this study it is assumed that CO₂ transportation pipelines are made of carbon-manganese steel, which has a CO₂-intensive manufacturing process. Constructing around 30,000 km of these pipelines for the storage hub arrangement adds a considerable 0.35 t CO₂/MWh to its GWP. As seen in Figure 6-11, CO₂ emission due to biomass transportation is higher in the biomass hub system, at 0.16 t CO₂/MWh. CO₂ removal from electricity sent to the grid and CO₂ stored is 0.33 t CO₂/MWh higher in the biomass hub system. Comparing these configurations shows that economic and environmental trade-off determines the optimal BECCS source and sink configuration.

![Figure 6-11: Apportioned CO₂ emission of sub-processes of BG-CCS (biomass hub) and BG-CCS (CO₂ storage hub) configurations, per functional unit (1 MWh electricity generated in these systems)](image)
6.4 Discussion

6.4.1 Technical potential for CO\textsubscript{2} mitigation using BECCS in an Australian context

The CEC Bioenergy Roadmap provides an appraisal of the Australian biomass resources and their potential electricity generation projected to 2020 and 2050 [338]. According to this report the quantity of bagasse, landfill gas, MSW and forest residue available is 5, 9.46, 3.49 and 8.8 Mt/year, respectively [338]. It should be noted that it is not clear what sustainability criteria were used in the CEC Bioenergy Roadmap report to estimate the availability of biomass. Organic residues are perceived to have less environmental impacts as they are inevitable by-products of high value food and fodder production. However, the non-sustainable removal of residues may lead to soil carbon and soil nutrient degradation [48]. In this study it is assumed that collecting 25% of agricultural and forestry residues meets the sustainability criteria to protect soil and alternative usages such as for animal food and bedding.

Using these amounts as inputs into the BG-CCS, LFG-CCS, MSW-CCS and FR-CCS models gives the total electricity generation and negative CO\textsubscript{2} emission potential for each of these BECCS systems in the future. Table 6-6 shows total CO\textsubscript{2} emissions (LCA net emissions and end of the pipe (EOP) emissions), along with the electricity production per year to 2050. EOP emission is the CO\textsubscript{2} directly captured from the power plant, which unlike LCA CO\textsubscript{2}, can be accurately monitored. In calculating the EOP the CO\textsubscript{2} captured from biomass is considered as negative emissions and the rest (10%) sent to ambient is regarded as neutral. The CO\textsubscript{2} emission from the non-organic composition of MSW is taken as positive.

**Table 6-6- Annual net CO\textsubscript{2} emission and electricity generation**

<table>
<thead>
<tr>
<th>BECCS Option</th>
<th>Biomass</th>
<th>Quantity (Mt)</th>
<th>Sustainable to collect (Mt)</th>
<th>CO\textsubscript{2} emission (Mt/year)</th>
<th>EL generation (GWh/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BG-CCS</td>
<td>Bagasse</td>
<td>5</td>
<td>1.25</td>
<td>-2.00</td>
<td>1369</td>
</tr>
<tr>
<td>FR-CCS</td>
<td>Forest residue</td>
<td>8.8</td>
<td>2.2</td>
<td>-3.63</td>
<td>2534</td>
</tr>
<tr>
<td>LFG-CCS</td>
<td>LFG</td>
<td>9.46</td>
<td>9.46</td>
<td>-13.43</td>
<td>6438</td>
</tr>
<tr>
<td>MSW-CCS</td>
<td>MSW</td>
<td>3.94</td>
<td>3.94</td>
<td>-5.91</td>
<td>3310</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td><strong>-24.97</strong></td>
<td><strong>-16.67</strong></td>
<td></td>
<td><strong>13,651</strong></td>
</tr>
</tbody>
</table>
As seen in Table 6-6, the total CO₂ captured (EOP) from the waste-based BECCS systems is 24.97 Mt CO₂/year. This is about 4% of Australia’s current emission (600 Mt CO₂-eq/year). The total Australian carbon budget to 2050, taking into account its contribution to a 2°C target is 10 Gt CO₂-eq. Therefore the projected quantity of negative emissions delivered by BECCS systems is significant. In addition to delivering negative emissions, BECCS systems modelled in this study could supply up to around 13.65 TWh of power by mid-century. This is about seven times Australia’s current total bioenergy production (< 2 TWh) and around 3.6% of the expected gross electricity generation in 2050 (377 TWh).

It is worth noting that selection of the four biomass resources for this study was based on their anticipated long-term availability and the maturity in the Australian energy sector. However, total biomass resource includes a vast range of biomass sources, such as wet organics from dairy and meat industries, energy crops (such as mallee tree), algae, waste from pulp and paper industry and other crop residues from grains and cotton cultivation. Utilising the full capacity of the available biomass -which is mostly waste or by-product of other industries- through BECCS technologies provides a plausible opportunity for the Australian energy sector to mitigate its emissions.

Compared to BECCS systems using agricultural and forestry residue, MSW-based BECCS systems (such as MSW-CCS and LFG-CCS in this study) provide more secure feedstock and are less sensitive to seasonal and climate variation. However, availability of MSW depends on parameters such as human population, economic development, and degree of industrialization, public habits, and waste management regimes. Development of more sustainable waste management paradigms such as a circular economy to minimize waste generation and maximum re-use and recycling would influence availability of waste for energy production. However, the current status of waste management is far from a zero-waste ideal and if BECCS is considered as a mid-term solution, the lack of organic waste is unlikely to be a problem. Unlike wind and solar power, MSW-base bioenergy is not intermittent and can be a base load supplier of electricity while abating the emissions.

The long term negative emission and power generation estimates in Table 6-6 illustrate the technical potential of BECCS systems. The techno-economic assessments showed that in the absence of a carbon pricing scheme, BECCS will need a REC higher than $115/MWh to become competitive with unmitigated coal-fired power generation. It is suggested that the cost of capture process could reduce by around 50% by 2050 [29, 360], which would have a
significant impact on BECCS costs. However, BECCS would still need strong policy support to remain economically feasible, such as a renewable energy certificate system, a carbon pricing scheme or a negative emission credit or pricing system, will be essential to support the deployment of BECCS.

6.4.2 Policy mechanisms for BECCS

The first and most important step to support BECCS deployment in Australia and elsewhere is for it to be recognised as a clean energy system, which could contribute to achieving long term mitigation targets. Currently, there is very limited understanding of BECCS and even some antipathy regarding bioenergy and CCS among some stakeholders and decision makers. Unlike other renewables, the bioenergy industry in Australia has not had the integrated and well-established collaborations among its stakeholders. Forums and advocacy, to build confidence in BECCS among stakeholder, policy makers, investors and the public is essential. BECCS needs to be considered an eligible clean technology under existing schemes such as the Emissions Reduction Fund (ERF), the Renewable Energy Target (RET), and the National Carbon Offset Standard, and any future mitigation schemes. Below, some of the support policies for BECCS are discussed.

Amendment of RET

One emission policy option considered in this study is a renewable energy certificate scheme (REC). Under such a scheme, certificates (RECs) are given per megawatt hour of electricity produced from recognised renewable power suppliers. Renewable Energy Certificate schemes have been successfully implemented in a number of countries including, U.S, Australia and especially in Europe, where they have been strongly supported for deployment of renewable electricity. In 2000 the Australian government introduced the Renewable Energy Target (RET) scheme, under which 20% of Australia’s electricity (41,000 GWh) must be produced from renewable sources by 2020. The scheme was amended in 2015 to set the ceiling for RECs at 33,000 GWh by 2020. To apply the RET, the Federal Government’s Office of the Clean Energy Regulator issues Renewable Energy Certificates (REC) for each MWh of compliant renewable energy [134]. In other words, energy retailers and large energy users are obliged to purchase a share of their electricity from renewable energy producers. Under the RET, energy from wind, solar, geothermal, hydroelectricity, and bioenergy are recognised as renewable sources [133]. Currently the price of a REC for large scale generation is around $66 /MWh [364]. Although RET recognises bioenergy as a renewable source there has been little
consideration of whether BECCS would be eligible for receiving RECs. It is essential that BECCS is acknowledged as a renewable technology under RET.

**Carbon pricing**

To recognise and reward negative emission seems to be the key mechanism to underpin BECCS deployment. In 2011 a carbon pricing scheme came into effect in Australia; the scheme was to cover around 60% of Australia’s total emissions. The initial price of carbon was set at AUD23 per tonne of carbon dioxide equivalent [368]. The scheme was repealed in 2014. Under a scenario with no other emission policy in effect, a carbon price of $95/t CO$_2$ is required to make BECCS-LCOE price-competitive with unmitigated coal-fired power. However, the carbon tax required to support clean technologies such as BECCS will be even lower if it plays in a more diverse portfolio of emission mitigation mechanisms. In other words, in a mix with an existing policy- e.g. RET-, a lower carbon price would have the same effect. For instance, at the current REC of approximately $66/MWh, a carbon price of $60 make the LCOE of the BECCS options price-competitive with coal-fired power without CCS. Considering the current status of carbon pricing schemes around the world, including Australia, it seems highly unlikely that such a high carbon price will be implemented in the near future Nonetheless, carbon pricing can be one of most effective emission policies for meaningful abatement in CO$_2$ emission and for providing strong incentives for development and deployment of clean technologies.

**Negative emission credit**

A support policy which exclusively targets negative emission technologies is a negative emission credit schemes. Under such a scheme, BECCS receives a refund or credit for each tonne of CO$_2$ removed from the atmosphere. Since negative emission achieved through the application of BECCS is a relatively new concept, there is currently no scheme that takes the value of negative emission into account. A more indirect support mechanism is through a cap and trade CO$_2$ mechanism, under which the CO$_2$ emitter could buy negative emission credit from BECCS suppliers to fulfil their allocated cap emission. Mathews [367] studied the possible mechanisms to certify negative carbon. This study found that designating carbon credit per 1 tonne of CO$_2$ removed or avoided as the most effective policy instruments. According to Mac Dowell and Fajardy [366] a minimum $105/t CO_2$ negative emission credit is required to incentivise large-scale BECCS deployment.
To assure the carbon-negativity of BECCS, robust counting and monitoring of CO₂ emission due to land-use change must be conducted [154, 369]. Existing policy mechanisms such as “reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries” (REDD+) under the UNFCCC [370] and setting strict standardisation criteria for sustainable bioenergy are ways to ensure to minimise the environmental impact of BECCS supply-chain [46, 154, 369].

**Tax and refund scheme**

Introducing an integrated carbon price, which is equal for the emitted and stored CO₂ is a way to trade the negative emission and improve the role of BECCS in the energy market and cut down its production costs. Although a negative emission scheme could be an effective support policy for BECCS deployment, its potentially high cost could be a challenge. One way of addressing this would be to combine it with a carbon tax scheme, so that CO₂ emitters pay the CO₂ removers. The price would be determined through a CO₂ market. In this fashion, both CO₂ abatement among polluters and incentives for deployment for BECCS investors would be encouraged. According to Ricci [78] compared to emission tax and negative emission subside, a synergy of tax and refund is less costly in terms of welfare and revenue from the tax allows a much larger deployment of BECCS.

**Increasing MSW tipping fees**

Australia is a signatory to COP 21. With 2.2 t of waste generation per capita, Australia is the seventh highest in MSW generation among OECD countries [350]. Different waste and energy policies have caused inhomogeneity in utilizing energy from waste across the country. One reason for underutilization of waste to energy (WET) technologies in Australia could be low landfill tipping fees in some states. For instance in Queensland there is no fee for landfilled MSW but in Victoria it is about $45 per tonne [295]. Waste is considered a negative externality that must be priced to cover all the collection and management processes. Under such a system, waste producers must pay a tipping fee or gate fee per tonne of waste delivered to the WTE producers for recovering the waste energy. Tipping fees, usually set by a contract, are a source of income for WTE plants and constitutes a considerable proportion of their income. However, the tipping fee has to be lower than the landfill fee to encourage WTE producers and therefore sometimes it is subsidised by the authorities. Currently, WTE tipping fees are highest in the UK ($148 /t) compared to a landfill fee of $153/t. This is much lower in the U.S ($68/t) [246].
In the electricity cost estimation in this study tipping fees are neglected. However, the cost of waste-based BECCS technologies could be significantly subsidised by introducing a tipping fee.

### 6.4.3 Feasibility of BECCS deployment in Australia

Any estimation of the long-term effect of BECCS on GHG emissions and the energy mix, relies in no small measure on the quantity of biomass available. Availability of waste/by-product biomass resources is a key factor for the sectors, including the sugarcane industry, forestry and waste management. The quantities of the biomass feedstock in Table 6-6 are subject to many variables such as population growth, climate change, market dynamics and energy and environmental regulations.

In order to build a national database of biomass resources and bioenergy facilities across Australia, the Rural Industries Research and Development Corporation (RIRDC) has launched an Australian biomass for bioenergy assessment (ABBA) project. The ABBA project will develop an interactive tool which provides a detailed analysis of the types, volumes and locations of potential bioenergy feedstock in each state. The $6 million project started in December 2015 [371].

Technical availability of suitable long-term CO₂ storage sites is another challenge for BECCS. There has been almost 20 years of research and development on CCS in Australia, including locating suitable storage sites and developing state of the art monitoring techniques [361].

Australia’s Intended Nationally Determined Contributions (INDCs) to the Paris agreement in COP 21 is to reduce emissions by 26-28% below 2005 by 2030. This is to be achieved by a large contribution from renewables, retirement of coal-fired plants, low demand growth and energy efficiency enhancement. But more stringent measures will be required in the second half of the century to achieve a net zero emission target. The power sector-electricity sector is responsible for one third of total emission in Australia and decarbonising this sector will be an essential precursor for a net zero carbon economy, to which BECCS could have the potential to make a significant contribution. To do this, BECCS will need the same level of support as other renewable technologies.

BECCS is a combination of two core technologies, bioenergy and CCS, which are sometimes viewed negatively by the public and by decision makers, despite the fact that together these
two technologies are seen as essential if the rise in temperature is to be well below 2 degrees C. Therefore, public outreach is important to address any misunderstandings regarding BECCS and the vital role that it may need to play to mitigate climate change in the future. This must be underpinned by technological innovation and greater knowledge of Australia’s bioenergy and storage potential.
6.5 Conclusions

Australia will benefit from global mitigation of greenhouse gas emissions given that it is a relatively hot and arid country that in the future could be subject to adverse climate changes, including more bushfires and greater water scarcity. In recognition of this, Australia is among the countries which signed the Paris Agreement in 2015. Having significant resources of organic waste for bioenergy production and the accumulated practical knowledge through ongoing carbon capture and storage projects, makes Australia a good candidate for deployment of bioenergy with carbon capture and storage (BECCS).

In this study techno-economic and environmental impact assessments were conducted for four BECCS options; landfill gas combusted in a gas turbine, bagasse and forest residue and municipal solid waste combusted in circulating fluidised beds, all equipped with an amine-based capture facility and CO$_2$ transport and storage.

- BECCS options for Australia have the potential to deliver a total of 25 Mt CO$_2$/year in negative emissions by 2050. This is about 4% of Australia’s current emissions.
- In addition, these BECCS systems could supply up to around 13.7 terawatt-hours of power by mid-century or around 3.5% of the expected gross electricity generation in 2050.
- Deployment of BECCS as a reliable supplier of electricity has the potential to enhance the flexibility and diversity of Australia’s energy portfolio whilst at the same time offering the possibility to mitigate emission by removing CO$_2$ emissions from the atmosphere.
- Extending some of the existing policies such as the renewable energy certificate system, introducing a carbon pricing scheme or establishing new policies such a negative emission refunding system, will be essential if BECCS is to be economically feasible.
- Public outreach is also important to address any misunderstandings regarding BECCS and the vital role that it may need to play to mitigate climate change.
Chapter 7

Adaptive Management System for Sustainable Deployment of BECCS
7.1 Introduction

BECCS as a synergy of two complex systems, namely bioenergy and CCS, encompasses all their complexities plus considerations regarding matching optimal bioenergy with CCS. Therefore, implementing BECCS in a sustainable way requires a supply-chain network design, which includes all its steps from production and storage of biomass, to the point of its consumption for energy, capturing the CO$_2$, its transportation and storage. In other words, there is a need for an integrated adaptive approach to model social, economic and environmental interactions of all these different processes throughout the lifecycle of a BECCS system. An adaptive management system (AMS) is a solution to deal with BECCS complexity and its multifaceted nature. An AMS allows simplifying the complex problems by hierarchical categorisation of the impacts and criteria under each category without losing the holistic perspective of the problem [372-375]. Historically, planning energy systems in a non-adaptive approach has led to challenges such as carbon lock-in phenomenon which is one of the main socio-economic impediments to transition towards a carbon-free future [117, 376].

Sustainability is often described as a three-pillar concept, which encompasses economic–environmental–social dimensions. It is however, a dynamic concept and the relative importance of the sustainability criteria evolve depending on the climate change, technical development and socio-economic situations, development agenda and political plans. It means that flexibility, resilience, security and reliability are essential characteristics of a sustainable energy system. These characteristics are all embedded in an AMS. AMS is an iterative decision-making process that reduces the uncertainty over time via system monitoring [372, 374, 375]. Multi-criteria decision analysis (MCDA) is an essential component of adaptive management systems, which assist the stakeholders with a timely, participatory, and comprehensive evaluation of the sustainability of the energy systems and to assess the impact of each choice on the outcomes. This is of especially high value for BECCS systems with evolving biomass production and collection and conversion systems on the bioenergy side and CO$_2$ capture, transport and storage technologies on the CCS side. Decision-making for complex energy systems such as BECCS takes place at strategic and operational levels. Strategic level issues include long-term planning such as selection of the energy technology pathways, supply and demand management, building a network connecting the elements of a BECCS system and taking all measures to assure the sustainability of the whole system [377]. Strategic planning for BECCS requires inclusion of stakeholders in every step of the process from biomass suppliers to power plant investors and CCS contractors. The network design has to comply
with the sustainability criteria defined with optimal balance of minimum cost and maximum environmental and social benefits. The operational planning comes after strategic planning and relates to short to medium-term planning and more tactical details of the whole system such as selection of the collection system, transportation and storage methods, etc. [377]. In this study, the focus is at the strategic level, with some operational planning for BECCS.

7.1.1 Multi-criteria Decision Analysis (MCDA)

Linear programming techniques are most commonly used for problems with infinite options such as BECCS. In this technique, the alternatives are compared against the criteria defined with different weights in an objective function [378]. Several MCDA using different methods such as weighted sum, priority setting, outranking, fuzzy set methodology and their combinations are employed for energy systems decision-making. These methods can be divided in three main groups; multi-attribute utility theory (MAUT), outranking (including PROMETHEE (preference ranking organization method for enrichment evaluation) and ELECTRE (elimination and choice expressing reality), and the analytic hierarchy process (AHP). All these methods follow the same concept of ranking alternatives against criteria defined and objectives to achieve the goal of decision making but are different in the mathematical methods and interpreting the scores [378-386].

AHP and the closely related analytic network process (ANP) are two of the most widely used and best structured MCA approaches [372], which are becoming prevalent because of their comprehensiveness in theory and the simplicity in application [387, 388]. More extensive discussion on the advantages and shortcomings of the AHP method were discussed by Ramanathan and Ganesh [389]. They claimed that the main reasons for the AHP method’s popularity are its simplicity, flexibility, intuitive appeal and its ability to handle both quantitative and qualitative criteria in the same framework. However, the method also has some drawbacks. According to Loken [379], the main disadvantage is that AHP is that it is very time-consuming when the number of alternatives and/or criteria is large [379].

7.1.2 Applications of MCDA in planning sustainable energy systems

To the best knowledge of the author, there is no AMS, or MCDA specifically designed for BECCS systems to date. However, there are several examples of applying these concepts to bioenergy and CCS technologies separately. Following are some of the recent examples; Elghali et al. used MCDA to develop a methodology for sustainable bioenergy deployment
Buchholz et al. [390] evaluated the potential of MCDA to facilitate the design and implementation of sustainable bioenergy projects in Uganda [391]. Wang et al. reviewed the corresponding methods in different stages of multi-criteria decision-making for sustainable energy system [387]. Samsatli et al. [392] presented a novel mixed-integer linear programming for Biomass Value Chain Model (BVCM) which accounts for the economic and environmental impacts of a multiple bioenergy pathways in UK. In the BVCM model, some bioenergy with CCS pathways are included but not in the context of BECCS. Shackley and McLachlan applied a MCDA method to examine the stakeholders’ reactions through extensive interviews of the mitigation options of capturing CO₂ from power stations and storing it in suitable off-shore geological reservoirs [393]. Torjai et al. [394] used the sustainability hierarchy model to optimise the supply-chain of herbaceous biomass-based combined heat and power technology. Karagiannidis and Perkoulidis [395] applied ELECTRE method to develop a conceptual model for the evaluation of different anaerobic digestion technologies suitable for organic municipal solid waste. Fontana et al. [396] used PROMETHEE method to evaluate the consequences of land-use change to ecosystem service provision. Scott et al. [397] used an Analytic Hierarchy Process–Quality Function Deployment (AHP–QFD) to select and optimally order appropriate bioenergy suppliers. Troldborg et al. [398] applied the PROMETHEE method for assessing, comparing, and ranking different renewable energy technologies in UK. Iakovou et al. [377] introduced a logistics and supply chain management system for utilization of waste biomass for energy production. Myllyviita et al. [381] proposed a theoretical multi-dimensional framework based on a modified version of AHP for assessing sustainability criteria and applied it in a case of wood-based bioenergy production in eastern Finland. Volkart et al. [399, 400] conducted an interdisciplinary evaluation of the potential renewable and non-renewable power supply options for Switzerland in 2035, taking into account economic, environmental, social, and security of supply-related aspects. In their MCDA method, they evaluated different energy scenarios including CCS. Branda and Missaoui [401] presented a methodology to assess different electricity system transformation for Tunisia, using criteria such as costs, energy security, environmental impact and social welfare. Shmelev et al. [402] used an energy system model MARKAL to provide investment costs and CO₂ emissions for UK energy system decarbonisation scenarios. Sheinbaum et al. [403] applied the MCDA method to create a methodological framework for sustainable energy development analysis of the Mexican energy policy. Castelazo and Azapagic [404] developed a decision-support framework for facilitating sustainable development of energy systems. Rahman et al. [405] presented a methodological technique to assist with formulating, evaluating, and promoting energy policy.
in Bangladesh. Browne et al. [386] used MCDA to assess policy measures for residential heating energy and domestic electricity consumption, using an Irish city-region as a case study. Torvanger and Meadowcroft [406] studied the governmental challenges regarding decision making about mitigation technologies and access to resources and the energy policy background of the region. Hopkins et al. [407] proposed an integrated assessment on resource recovery from waste in UK which included values across environmental, social, economic and technical dimensions.

7.1.3 Main criteria for MCDA

Choosing the sustainability criteria in a MCDA has a decisive effect on the outcomes of the decision-making model. Energy systems are typically ranked according to aggregated score of the economic, environmental and social impacts [387, 390, 404, 408, 409]. Selection of criteria for each of these impact clusters depends on the goal and objectives of the decision-making model. Buchholz et al. [409] conducted an expert appraisal to find and rank the critical criteria for sustainability assessments of bioenergy systems and identified 35 criteria. Elghali et al. [408] studied the most important criteria for sustainable bioenergy deployment. Most of these criteria are common among different MCDA but their ranking varies. For CCS decision making the main criteria studied have been reservoir capacity, monitoring, storage security, public perceptions, costs, CO2 capture efficiency, local injectivity, total pore volume and permanence of storage [410, 411]. In most CCS decision-making, these criteria are a source of uncertainties as well. Other than technical criteria, socio-political factors play an important role in planning CCS. According to Wang et al. [387] efficiency, investment cost, CO2 emission and job creation are the most common criteria in the technical, economic, environmental and social attributes, respectively. Another study identified GHG emissions and the price of electricity generation as the two most commonly applied criteria in MCDAs of renewable energy developments [398]. Strantzali et al. [412] describes land use, CO2 emissions, job creation and social acceptability as the fundamental criteria which determine the decision of the stakeholders. These criteria could be quantitative or qualitative, objective or subjective by nature. Subjective and quantitate criteria are usually more difficult to score. Social and cultural sustainability are often ignored because of difficulties related to identifying and quantifying their indicators which are often qualitative and subjective [381, 405].

This study proposes a comprehensive adaptive management system for BECCS, which comprises creating the BECCS models, evaluating their economic- environmental- social
impacts, ranking them against a set of sustainability criteria in a multi-criteria decision analysis and testing the robustness of the results through a sensitivity analysis.
7.2 Methodology

7.2.1 Analytic hierarchy process (AHP)

In this study, the analytic hierarchy process (AHP) is used to assess the sustainability of the BECCS options. AHP was developed by Saaty [388] and is an effective tool to prioritise the alternatives to assist decision makers in a transparent, interactive and efficient manner. AHP comprises a hierarchy model, which includes a goal, criteria, and alternatives. Alternatives are ranked in a pairwise comparison against the criteria categories [381, 385, 388]. This pairwise comparison has proven extremely intuitive and practical for stakeholders [413]. In an AHP, it is possible to include subjective and objective criteria of a decision. It is also possible to check the consistency of the evaluation process [388]. AHP is considered one of the most promising options to be used in sustainability assessments, because it is comprehensible to apply and it incorporates the preferences of decision makers in an advanced manner [381].

The general steps in an AHP are:

1- Establishing the context of the decision making, the goal of the planning and identifying the main stakeholders

2- Identify the alternatives which are to be considered for the appraisal

3- Identify the main cluster of criteria and their corresponding impact categories that the alternatives should be examined against

4- Identify the main sub criteria in each cluster criteria

5- Scoring the criteria in each cluster based on the importance of contribution to the goal

6- Weighting the cluster criteria and sub-criteria to reflect their relative importance

7- Check the consistency of the weighting system

8- Combine the scores and weights to calculate the overall weight of each alternative

Three more steps are added to the AHP to integrate it into an adaptive system:

9- Analyse the results

10- Run sensitivity analysis
11- Review, monitor and adjust the whole system according to the outcome

In AHP it is assumed that $m$ criteria are considered to evaluate $n$ alternatives; the following explains the methodology of AHP used in this study;

First a $m \times m$ real matrix ($C$) is created to facilitate pairwise comparison of the criteria. Each entry $C_{ij}$ of the matrix $C$ represents the importance of the $i$th criterion relative to the $j$th criterion. If $C_{ij} > 1$, then the $i$th criterion is more important than the $j$th criterion, while if $C_{ij} < 1$, then the $i$th criterion is less important than the $j$th criterion. If two criteria have the same importance, then the entry $C_{ij}$ is 1 and $C_{ii} = 1$ for all $i$. $C_{ij} \times C_{ji}$ must be always equal to 1. The relevant important of the criteria is set based on a 1-9 scale.

- $C_{ij} = 1$ if $i$ and $j$ are equally important
- $C_{ij} = 3$ if $i$ is slightly more important than $j$
- $C_{ij} = 5$ if $i$ is more important than $j$
- $C_{ij} = 7$ if $i$ is strongly more important than $j$
- $C_{ij} = 9$ if $i$ is absolutely more important than $j$

When all the entries of the matrix $C$ are calculated, a normalised pairwise comparison matrix $C_n$ with $\bar{c}_{ij}$ entries are built (see Equation 7-1). In the $C_n$ matrix the sum of all entries in each column is equal to 1.

$$\bar{c}_{ij} = \frac{c_{ij}}{\sum_{k=1}^{m} c_{kj}}$$  \hspace{1cm} (Equation 7-1)

Matrix $C_n$ is used as the base to create the criteria weight vector ($w$), a $m$-dimensional column vector, using Equation 7-2.

$$w_i = \frac{\sum_{k=1}^{m} \bar{c}_{ik}}{m}$$  \hspace{1cm} (Equation 7-2)

The same method is used to build the weighing factor for sub-criteria within each criterion by pairwise comparison matrix $A$ with $a_{ij}$ entry of sub-criterion $j$ under criterion $i$. 
The impact assessments are normally composed of several indices with different units, which could be quantitative or qualitative. Equation 7-3 is used to standardise the impacts of each alternative under each sub-criterion. \( \hat{x}_{ijk} \) is between 0 and 1.

\[
\hat{x}_{ijk} = \frac{x_{ijk} - \min_j(x_{ijk})}{\max_j(x_{ijk}) - \min_j(x_{ijk})}
\]  
(Equation 7-3)

The sustainability index (overall score, \( S_k \)) of each alternative is calculated through Equation 7-4; The last step is to rank the alternative based on their overall sustainability score in decreasing order. The alternative with highest score ranks the most sustainable choice.

\[
S_k = \sum_{ij} (w_i \times a_{ij} \times \hat{x}_{ijk})
\]  
(Equation 7-4)

\( W_i \) is the weight factor for criterion i, \( a_{ij} \) is the weighting factor of the sub-criterion j under criterion i and \( \hat{x}_{ijk} \) is the standardised impact score of alternative k with respect to criterion j under criterion i.

In AHP a consistency index (CI) is calculated for each pairwise comparison matrix. CI is calculated through Equation 7-5;

\[
CI = \frac{\lambda - m}{m-1}
\]  
(Equation 7-5)

Where \( \lambda \) is the average of the elements of the vector whose \( ith \) element is the ratio of the \( ith \) element of the vector \( C \times w \) to the corresponding element of the vector \( w \) and \( m \) is the number of compared criteria. Ideally, CI should be equal to 0 but a Random Index (RI) is used to define consistency ratio, CR, as the limit for acceptable inconsistency. So, Equation 7-6 is used to check the consistency:

\[
CR = \frac{CI}{RI} < 0.1
\]  
(Equation 7-6)

Table 7-1 lists the RI for problems with \( m \leq 10 \)

<table>
<thead>
<tr>
<th>m</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>RI</td>
<td>0</td>
<td>0.58</td>
<td>0.90</td>
<td>1.12</td>
<td>1.24</td>
<td>1.32</td>
<td>1.41</td>
<td>1.45</td>
<td>1.51</td>
</tr>
</tbody>
</table>

163
The schematic of the AMS for BECCS systems modelled in this study is presented in Figure 7-1.

Figure 7-1- Schematic of the AMS for BECCS systems, adapted from [388, 408, 412, 414]

7.2.2 Sustainability criteria and sub-criteria

In this study, the main impact criteria are technical, economic, environmental, social and policy. Policy criteria have an important role on credibility and feasibility of energy systems but are often missed out in MCDA [408, 412, 414]. The main criteria for evaluating the sustainability of a BECCS system were identified and are listed in Table 7-2. The functional unit is 1 KWh electricity produced from the BECCS option.
Table 7-2: Criteria and sub-criteria system for BECCS sustainability evaluation

<table>
<thead>
<tr>
<th>Criteria</th>
<th>Sub-Criteria</th>
<th>Unit</th>
<th>Optimise</th>
<th>Methodology</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Technical</td>
<td>Thermal Efficiency</td>
<td>%</td>
<td>Maximise</td>
<td>System Modelling</td>
<td>The thermal efficiency of the BECCS alternative is indicative of the reliability, possibility to manage feedstock input efficiently, and consequently lower cost of electricity generation. It is the most used technical criterion to evaluate energy systems [387]</td>
</tr>
<tr>
<td></td>
<td>Maturity</td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment / Technical review</td>
<td>Technical maturity of an alternative indicates the level of research and development and operational/demonstration on a national and international level. How widespread it is and if it has been deployed to its theoretical limit of efficiency and full technical potential. Number and scale of deployment of the BECCS alternative is used to score this criterion</td>
</tr>
<tr>
<td></td>
<td>Resource security</td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Technical review</td>
<td>This criterion regards security of the resource used for BECCS under fluctuations in technical, climate, economic and social circumstances. A more secure BECCS alternative is the one using locally produced and storable feedstock</td>
</tr>
<tr>
<td></td>
<td>Credibility</td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>Credibility is a measure of how realistic and feasible implementation of the BECCS alternative is in the real world [411]</td>
</tr>
<tr>
<td></td>
<td>Resource availability</td>
<td>tonne/year</td>
<td>Maximise</td>
<td>Technical review</td>
<td>Annual amount of the resource available for electricity generation in the BECCS system</td>
</tr>
<tr>
<td></td>
<td>Annual negative CO₂</td>
<td>Mt CO₂/Year</td>
<td>Maximise</td>
<td>System modelling</td>
<td>The net negative CO₂, which the BECCS alternative could deliver in a year. This criterion only accounts for the end-of-pipe CO₂ captured and stored</td>
</tr>
<tr>
<td></td>
<td>Avoided CO₂</td>
<td>t CO₂/MWh</td>
<td>Maximise</td>
<td>System modelling</td>
<td>Amount of CO₂ which is avoided through CCS at the end of pipe per MWh electricity generated</td>
</tr>
<tr>
<td>Economic</td>
<td>LCOE</td>
<td>$/MWh</td>
<td>Minimise</td>
<td>Techno-economic assessment</td>
<td>Levelised cost of electricity production during the plants lifetime (all the investment, operating and maintenance and fuel cost are included in calculating the LCOE)</td>
</tr>
<tr>
<td></td>
<td>Cost of avoided CO₂</td>
<td>$/t CO₂</td>
<td>Minimise</td>
<td>Techno-economic assessment</td>
<td>Cost of avoided each tonne of CO₂ through BECCS system</td>
</tr>
<tr>
<td>Environmental</td>
<td>GWP</td>
<td>kg CO₂-equiv/ KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Global warming potential of the system measured in total greenhouse gas equivalent emitted throughout the LCA of the BECCS alternative per KWh electricity generated</td>
</tr>
<tr>
<td></td>
<td>Acidification</td>
<td>kg SO₂-equiv/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Total SO₂ equivalent throughout the LCA of the BECCS option per KWh electricity generated to assess the acidification impact of the system</td>
</tr>
<tr>
<td></td>
<td>Particulate matter</td>
<td>kg PM2.3/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Total solid particles and liquid droplets with diameters ≤ 2.3 microns emitted to the air per each KWh of electricity generated</td>
</tr>
<tr>
<td></td>
<td>Abiotic depletion</td>
<td>MJ NCV/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Total amount of abiotic natural resources i.e. fossil fuels in forms of coal, oil and natural gas consumed throughout the lifecycle of the BECCS system</td>
</tr>
<tr>
<td></td>
<td>Human toxicity, cancer</td>
<td>CTUh/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Impact of the emissions in a BECCS system in Comparative Toxic Unit for human (CTUh) on human health</td>
</tr>
<tr>
<td></td>
<td>Ozone layer depletion</td>
<td>kg CFC-11-equiv/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Impact on ozone layer depletion through measuring the equivalent CFC-11 emission.</td>
</tr>
<tr>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
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<tr>
<td>----------------</td>
<td>------------------</td>
<td>------------------</td>
<td>------------------</td>
<td>-----------------------------------------------------------------</td>
<td></td>
</tr>
<tr>
<td><strong>Water Scarcity</strong></td>
<td>m³ eq/ KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Total water usage for generation of 1 KWh electricity in a BECCS system</td>
<td></td>
</tr>
<tr>
<td><strong>Eutrophication</strong></td>
<td>kg PO₄₃⁻ eq/ KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>Equivalent phosphate amount of substances emitted in the LCA of a BECCS system</td>
<td></td>
</tr>
<tr>
<td><strong>Land Use Change</strong></td>
<td>m²/KWh</td>
<td>Minimise</td>
<td>LCA model</td>
<td>The area of natural land transformed due to implementation of the BECCS system; this area is the sum of all area used for production of the biomass, land occupied for the infrastructure and CO₂ storage facilities</td>
<td></td>
</tr>
<tr>
<td><strong>Access to Energy</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>This criterion is an indication of how implementing BECCS could contribute to social benefits such as equal opportunity of the communities to access energy, especially in less developed or remote regions</td>
<td></td>
</tr>
<tr>
<td><strong>Food security</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>Impact of utilising the biomass resource in a BECCS system on access to food and food price</td>
<td></td>
</tr>
<tr>
<td><strong>Social acceptability</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>The overall opinion and level of public perception regarding the BECCS system, it is not a measurable index and usually is assessed through previous experiences, surveys and questionnaires asked from general public, NGOs and key stakeholders. Social acceptability has an extremely important impact on the credibility of BECCS and the pace of its deployment</td>
<td></td>
</tr>
<tr>
<td><strong>Job creation</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>Capacity of a BECCS system to create jobs through its whole lifecycle. Creating more jobs contribute to higher social benefit and influences the social acceptability and policy support of the BECCS system. It could be scored both qualitatively and quantitatively (e.g. number of jobs per MWh generated) [386, 401]</td>
<td></td>
</tr>
<tr>
<td><strong>Policy support</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>This criterion is a measure of the extent of existing or planned policy/regulatory mechanisms to support deployment of the BECCS system or those in effect that could be amended for BECCS inclusion in the portfolio of the clean energy pathways</td>
<td></td>
</tr>
<tr>
<td><strong>Recognition among main stakeholders</strong></td>
<td>Qualitative (0-1)</td>
<td>Maximise</td>
<td>Expert judgment</td>
<td>The reputation of the BECCS systems as a potential mid-term solution for CO₂ removal and key component of the portfolio towards COP 21 goals among stakeholders and decision makers in the governments, international organisations, industries and NGOs</td>
<td></td>
</tr>
</tbody>
</table>
6.2.3 Modelling BECSS alternatives

The BECCS alternatives assessed in this study all utilise organic waste/residues from municipal, agricultural and forestry sectors. Two BECCS systems, MSW-CCS and LFG-CCS use municipal solid waste (MSW); five systems, BG-CCS (feedstock: bagasse), BS-CCS (feedstock: barley straw), CS-CCS (feedstock: corn straw), RS-CCS (feedstock: rice straw) and WS-CCS (wheat straw) use agricultural residues or by-products and FR-CCS uses forestry residue to produce electricity. Results from Chapters 4 and 5 are the inputs to the AMS in this Chapter. A general schematic of the whole BECCS lifecycle is presented in Figure 7-2.
7.2.4 Impact evaluation

The impact values of the BECCS alternative for the sub-criteria, as defined in Table 7-2, are listed in Table 7-3 below;

Table 7-3- Impact evaluation of the BECCS alternatives for each sub-criterion

<table>
<thead>
<tr>
<th>Criteria</th>
<th>Sub-Criteria</th>
<th>Unit</th>
<th>BECCS Alternatives</th>
</tr>
</thead>
<tbody>
<tr>
<td>Thermal Efficiency</td>
<td>%</td>
<td>MSW-CCS</td>
<td>LFG-CCS</td>
</tr>
<tr>
<td>Maturity (N2)</td>
<td>Qualitative (0-1)</td>
<td>1</td>
<td>0.6</td>
</tr>
<tr>
<td>resource security (N3)</td>
<td>Qualitative (0-1)</td>
<td>0.9</td>
<td>0.9</td>
</tr>
<tr>
<td>Credibility (N4)</td>
<td>Qualitative (0-1)</td>
<td>1</td>
<td>0.7</td>
</tr>
<tr>
<td>Global resource availability (N5)</td>
<td>EJ/year</td>
<td>21.8</td>
<td>21.8</td>
</tr>
<tr>
<td>Global annual negative CO(_2) emission (N6)</td>
<td>Gt CO(_2)/Year</td>
<td>1.96</td>
<td>0.49</td>
</tr>
<tr>
<td>Avoided CO(_2) (N7)</td>
<td>t CO(_2)/MWh</td>
<td>0.94</td>
<td>1.35</td>
</tr>
<tr>
<td>LCOE (default- no CO(_2) policy) (N8)</td>
<td>S/MWh</td>
<td>212.84</td>
<td>160.15</td>
</tr>
<tr>
<td>Cost of avoided CO(_2) (N9)</td>
<td>S/t CO(_2)</td>
<td>78.4</td>
<td>70.2</td>
</tr>
<tr>
<td>GWP</td>
<td>kg CO(_2)-eq/KWh</td>
<td>-1.53</td>
<td>1.29</td>
</tr>
<tr>
<td>Acidification</td>
<td>kg SO(_2)-eq/KWh</td>
<td>-5.41</td>
<td>-5.09</td>
</tr>
<tr>
<td>Particulate matter</td>
<td>kg PM2.3/KWh</td>
<td>-6.9E-04</td>
<td>-5.4E-04</td>
</tr>
<tr>
<td>Abiotic depletion</td>
<td>MJ NCV/ KWh</td>
<td>-1.8E-03</td>
<td>-1.51E-03</td>
</tr>
<tr>
<td>Human toxicity, cancer</td>
<td>CTU/h/ KWh</td>
<td>-3.9E-09</td>
<td>2.30E-09</td>
</tr>
<tr>
<td>Ozone layer depletion</td>
<td>kg CFC-11-eq/ KWh</td>
<td>-4.2E-08</td>
<td>-3.19E-08</td>
</tr>
<tr>
<td>Water Scarcity</td>
<td>m(^3) eq/ KWh</td>
<td>2.7E-03</td>
<td>1.99E-03</td>
</tr>
<tr>
<td>Eutrophication</td>
<td>kg PO(_4)-eq/ KWh</td>
<td>-2.1E-03</td>
<td>-1.92E-03</td>
</tr>
<tr>
<td>Land Use change</td>
<td>m(^2)/KWh</td>
<td>3.4E-06</td>
<td>4.73E-05</td>
</tr>
<tr>
<td>Access to Energy (N11)</td>
<td>Qualitative (0-1)</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Food security (N12)</td>
<td>Qualitative (0-1)</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Social acceptability (N13)</td>
<td>Qualitative (0-1)</td>
<td>0.7</td>
<td>0.7</td>
</tr>
<tr>
<td>Job Creation (N14)</td>
<td>Qualitative (0-1)</td>
<td>0.5</td>
<td>0.5</td>
</tr>
<tr>
<td>policy support (N15)</td>
<td>Qualitative (0-1)</td>
<td>0.4</td>
<td>0.2</td>
</tr>
<tr>
<td>Recognition among main stakeholders (N16)</td>
<td>Qualitative (0-1)</td>
<td>0.3</td>
<td>0.3</td>
</tr>
</tbody>
</table>

168
The explanatory notes regarding each impact assessment are provided below;

**N1:** Thermal efficiency of the power plants is based on data found in the literature [218, 254, 286-288, 316, 317, 415].

**N2:** There are examples of MSW-CCS systems already in operation or planned in the near future [87, 88]. Although there are no LFG-CCS plants planned or operated, more than 1150 LFG combustion plants are operating around the world [249]. The knowledge and maturity of LFG power plants could be a basis for LFG-CCS. No BECCS system using agricultural residues is under operation or planned. However, using these feedstocks to produce energy is a rather mature technology.

**N3:** The global urban population is estimated to be around 3 billion, which generates 1.3 Gt of solid waste annually and by 2100, MSW generation may increase to 4 Gt [245, 246]. Parameters such as human population, economic development, public habits, and waste management regimes determine the security of MSW as an energy resource. However, considering the current state of waste management and generation around the world it can be said that MSW is a rather secure feedstock at least for the mid-term future. Therefore, the LFG and MSW score highest security among other resources in this study. Security of supply of agricultural residues directly depends on the main products, and agricultural production is often very susceptible to natural changes such as floods, droughts, and regional conflicts. Therefore, BECCS using agricultural and forestry residues are given a lower score in security.

**N4:** Credibility of a BECCS option is defined as the feasibility to implement in the real world. It is a rather ambiguous criterion, considering that the circumstances to implement a new technology vary widely from region to region. There are already MSW-CCS plants in operation, so it can be said this BECCS option is highly credible. For other BECCS pathways in this study with no operating or even small-scale demonstration projects, the credibility will be an indication of maturity, policy support and social acceptance.

**N5:** Resource availability of the alternatives has been calculated based on the data obtained from [245, 246, 249, 308, 309, 313, 318, 323, 416].

**N6:** Annual negative CO\(_2\) emission delivered by a BECCS system is calculated based on multiplying the total available biomass (m) by the negative emissions delivered by the BECCS system per unit of the input biomass (Equation 7-7).
Annual negative CO$_2$ emission \( \frac{Gt_{CO_2}}{\text{year}} \) = \( m \left( \frac{Gt_{biomass}}{\text{year}} \right) \times \text{Negative CO}_2 \left( \frac{Gt_{CO_2}}{Gt_{biomass}} \right) \) 

(Equation 7-7)

**N7:** Net CO$_2$ avoided per MWh electricity generated (tCO$_2$/MWh) is calculated through the mass balance of the BECCS models, according to Equation 4-3 (Chapter 4, Section 4.2.4);

**N8:** The levelised cost of electricity production, LCOE [$/MWh], was calculated using Equation 4-1 (Chapter 4, Section 4.2.4).

**N9:** Cost of CO$_2$ emission avoided is the cost of the CCS processes added per tonne of CO$_2$ captured; see Equation 4-2 (Chapter 4, Section 4.2.4).

**N10:** Environmental impacts were derived from LCA results in Chapter 4, Section 4.3.2 and Chapter 5, Section 5.3.2.

**N11:** Energy is one of main prerequisites for social and economic development and it is crucial that everyone has equal access to energy. In order to minimise the transportation cost and time, MSW-based BECCS systems with a medium to large capacity (like waste to energy plants without CCS), such as MSW-CCS and LFG-CCS in this study, are typically in the vicinity of high population municipalities, where the waste generation is high. These towns are most likely provided with electricity from other suppliers as well. It means access to a more diverse and flexible electricity mix, which will enhance access to energy during peak hours and also enhance access to more remote areas. BECCS using agricultural and forestry residues (like the same bioenergy plants without CCS) are typically installed in close proximity to where the resources are produced, thereby supplying rural areas and farmlands with locally generated electricity and reducing the need for costly infrastructures and networks to import electricity from distant suppliers [338]. Additionally, this reduces transmission losses arising from long distance electricity delivery. It is also important to assess the social impact of BECCS facilities located close to the production sites in rural areas, including land ownership, labour conditions and equitable access to food, land, and energy [353-356].

**N12:** Food security has been one of most frequently used criteria in MCDA of bioenergy planning [404, 417]. One of controversial challenges of expanding bioenergy from dedicated energy crop is maintaining food security. With current production, there are 32 countries facing food crises with around 870 million people estimated to be undernourished and 1 billion malnourished [152]. Despite the current inefficiencies to cover food demand the food
production has to grow by 60% to feed around 9 billion people in 2050 [125, 154, 168, 169]. In absence of required advancements in food production and in light of limited land and water resources, this put even more constrains on expanding lands for energy crop cultivation; On the other hand, organic residues are perceived to have less environmental impact as they are inevitable by-products of the high value food and fodder production [48]. Using residues is a way to mitigate the negative ramifications of energy crop production [308, 323]. Besides, the sustainable amount of residue removal to maintain soil productivity used in this study was calculated based on strict sustainability criteria [326-328]. Therefore, BECCS system presented in this study are deemed to have no negative impact on food security.

**N13:** BECCS is not known to the public and even to key stakeholders. To date there has been limited research into the social acceptability of BECCS. Some research shows that BECCS often receives lower priority than other clean energy technologies, such as solar and wind power, but a higher priority than fossil energy with CCS or nuclear power [14, 30, 418]. It can be said that acceptability for BECCS is similar to that for bioenergy without CCS. It is assumed that using organic waste and residues for BECCS will receive less criticism and opposition compared to BECCS systems using dedicated energy crops or food crops. It is important to clarify how using the organic waste/residue in BECCS offers a multiple advantage of producing energy, removing CO₂ from the atmosphere and avoiding negative environmental impacts that might rise due to lack of management or unsustainable management of these wastes/residues. In this study BECCS systems using municipal waste are given slightly higher than score in the social acceptability sub-criterion. The reason is concerns regarding removal of residues on soil conditions and also competition with alternative usages such as animal feed and bedding.

**N14:** BECCS facilities can offer social benefits such as stable long-term employment for local and regional communities compared to purely agricultural-based communities, which only offers seasonal employment and fewer job opportunities. It will create revenue for local producers by selling residues. In terms of employment, it could be measured quantitatively by number of jobs per unit of electricity generation or number of jobs per unit of biomass utilised. A BECCS system encompasses a wide range of inputs and processes throughout its lifecycle and is specific depending on the components of each BECCS system (i.e. feedstock, conversion technology, CCS process). To date, there is no reliable data on the number of jobs created in a BECCS system. It is assumed that the CCS part of all the BECCS systems in this study is identical, so it is reasonable to assume that the corresponding jobs specific to the CCS
operations is the same. Under this presumption, job creation of these BECCS systems will be ranked according to job creation potential of their bioenergy side. A typical waste-to-energy plant of 50,000 tonnes per annum capacity requires 6-18 workers daily [246]. For biomass combustion the total number of jobs including O&M and fuel production from agricultural residues is about 1.6 jobs/MW$_{el}$ [386]. It is difficult to draw reliable conclusions, but it is safe to assume that BECCS using agricultural residues due to harvest, collection, storage, transportation could create more jobs that MSW-based BECCS systems.

**N15:** The high cost of BECCS systems means that strong policy and government support is crucial for their global proliferation. This fact has been seen in successful cases such as the Illinois Basin Decatur Project /Illinois Industrial CCS (IL-ICCS) project which in 2011 with support from the U.S Department of Energy was established and is the first large-scale BECCS project in the world [86]. A recent MSW-based BECCS project at Klemetsrud with support of waste to energy agency in the Oslo municipality (EGE) has been assessed [87]. Assessments show that the project could technically capture up to 315,000 tonnes of CO$_2$ annually with 90% CO$_2$ capture rate by 2020 [87]. Another waste-to-energy power plant with CCS is ARV-Duiven in Duiven, the Netherlands that has been established by the support of national government and the provinces. The ARV- Duiven power plant with 70 MW capacity incinerates MSW to produce around 126 GWh electricity and capture up to 50 Ktonnes CO$_2$ per annum using an MEA capture process, from 2018 [88]. Globally there have been 21 BECCS projects, mostly located in North America, Europe and Scandinavia [30, 83]. Five projects have been cancelled mostly due to lack of economic feasibility and the remainder are either completed or under evaluations/planning [30]. There are already policy mechanisms, especially in developed countries, which recognise biodegradable waste as a renewable resource [249, 256, 262, 352] but to date, no policy tool, which rewards negative CO$_2$ emission has been devised.

**N16:** In a historical agreement at the 2015 United Nations Conference of Parties (COP 21) in Paris most countries have set out a global action plan to put the world on track to avoid dangerous climate change by limiting global warming to well below 2 °C [3]. Close to 90% of the IPCC scenarios show that to achieve this target around 5-20 Gt CO$_2$-eq needs to be removed annually, starting from mid-century [1, 12]. The global potential of BECCS as a CO$_2$ removal technology in the recent IEA global models was estimated 14 Gt CO$_2$ between 2015 - 2050 [29, 41]. Despite the crucial role that BECCS could play in achieving ambitious mitigation
pathways it is not yet included in energy policy agendas anywhere in the world. So it is necessary to accelerate investments in BECCS R&D projects and initiate political and societal debates to build discourse around BECCS at global and regional levels [419].

7.2.5 Scoring and weighting the criteria

Prioritising the criteria depends on the characteristics of the system under assessment, defined goals and objectives of the MCDA, and also the targeted stakeholders. For instance, in a study by Myllyviita et al. [381] on the sustainability of bioenergy systems, economic and environmental criteria are equally important and are assigned higher weight than social criteria. Akadiri et al. [414], considered economic aspects more important than environmental and social impacts, and other studies consider environmental and social criteria equally important [409]. Assigning weights to the criteria in this study was based on a strict sustainability discipline, which perceives all aspects of sustainability to be equally important. So, technical, economic, environmental, social and political criteria have the same weight of 0.2. Weighting the sub-criteria reflects their relative importance in a BECCS system under a baseline scenario which has no immediate indispensable action plan for mitigation. In the technical criteria those parameters that indicate the potential for delivering negative emission are assigned more weight. Resource availability and security are of secondary priority. Among economic sub-criteria LCOE is given a higher priority as it is a parameter commonly used to compare different power generation technologies. Cost of avoided CO₂ is especially important when comparing BECCS with other CO₂ removal pathways, fossil fuel-based CCS or energy conservation measures. Among environmental criteria, global warming potential (GWP) is given the highest priority, considering that delivering negative CO₂ emission is the most prominent feature of BECCS systems. Two other parameters causing controversial debates regarding BECCS environmental impacts, namely natural use change and water scarcity were given high weightings. Sustainability assessments often fail to include policy/regulatory aspects of energy systems. Nevertheless, for BECCS, as a rather novel mitigation pathway, policy support and recognition is essential and will be a decisive influence on its future development. In this study, policy support is given a higher priority than recognition among stakeholders. The consistency of weight allocation to the criteria and sub-criteria was checked and as shown in Table 7-4 in all cases consistency ratio (CR) is within the acceptable range of < 0.1.
<table>
<thead>
<tr>
<th>Criteria</th>
<th>Sub-Criteria</th>
<th>Weight</th>
<th>CR</th>
</tr>
</thead>
<tbody>
<tr>
<td>Technical</td>
<td>Thermal Efficiency</td>
<td>0.08</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td>Maturity</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td></td>
<td>resource security</td>
<td>0.14</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Credibility</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Resource availability</td>
<td>0.12</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Annual negative CO₂ emission</td>
<td>0.20</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Avoided CO₂</td>
<td>0.24</td>
<td></td>
</tr>
<tr>
<td>Economic</td>
<td>LCOE</td>
<td>0.65</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Cost of avoided CO₂</td>
<td>0.35</td>
<td></td>
</tr>
<tr>
<td>Environmental</td>
<td>GWP</td>
<td>0.26</td>
<td>0.03</td>
</tr>
<tr>
<td></td>
<td>Acidification</td>
<td>0.05</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Particulate matter</td>
<td>0.06</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Abiotic depletion</td>
<td>0.05</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Human toxicity, cancer</td>
<td>0.15</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ozone layer depletion</td>
<td>0.06</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Water Scarcity</td>
<td>0.11</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Eutrophication</td>
<td>0.05</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Land-Use Change</td>
<td>0.18</td>
<td></td>
</tr>
<tr>
<td>Social</td>
<td>Access to Energy</td>
<td>0.24</td>
<td>0.03</td>
</tr>
<tr>
<td></td>
<td>Food security</td>
<td>0.53</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Social acceptability</td>
<td>0.11</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Job creation</td>
<td>0.12</td>
<td></td>
</tr>
<tr>
<td>Policy</td>
<td>Policy support</td>
<td>0.75</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Recognition among main stakeholders</td>
<td>0.25</td>
<td></td>
</tr>
</tbody>
</table>
7.3 Results

The result of the AHP decision-making model under a baseline scenario was evaluated. The baseline scenario considers all the criteria to be equally important. The optimal BECCS system has to obtain a highest overall sustainability index but it does not need to have highest score in each criterion. As seen in Figure 7-3, LFG-CCS with SI= 0.68 and RS-CCS and BS-CCS with SI=0.49 have respectively the highest and lowest sustainability index among BECCS alternatives. On average, municipal waste-based BECCS alternatives have a higher rank than agricultural residue-based BECCS systems. LFG-CCS is the most technically feasible BECCS alternative, whereas technical score of BS-CCS is only 0.04. BG-CCS and FR-CCS with SI of 0.62 and 0.60, respectively rank after LFG-CCS and are the most sustainable alternative among residue-based BECCS alternatives. If strict sustainability discipline were applied in this MCDA, FR-CCS would be the most sustainable alternative as it acquires an overall high SI and higher than average score in each criterion.

![Figure 7-3: Overall Sustainability Index (SI) of BECCS alternatives under baseline scenario](image-url)
7.3.1 Sensitivity analysis

In order to evaluate the impact of weighting of sustainability criteria on the outcome of the MCDA, a sensitivity analysis was carried out. In three different scenarios, the sustainability criteria were assigned different weightings, which are listed in Table 7-5.

Table 7-5- weightings of sustainability criteria and their consistency ratio under four scenarios

<table>
<thead>
<tr>
<th>Criteria</th>
<th>Baseline</th>
<th>Scenario No.1</th>
<th>Scenario No.2</th>
<th>Scenario No.3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Technical</td>
<td>0.2</td>
<td>0.37</td>
<td>0.18</td>
<td>0.10</td>
</tr>
<tr>
<td>Economic</td>
<td>0.2</td>
<td>0.09</td>
<td>0.33</td>
<td>0.13</td>
</tr>
<tr>
<td>Environmental</td>
<td>0.2</td>
<td>0.35</td>
<td>0.33</td>
<td>0.35</td>
</tr>
<tr>
<td>Social</td>
<td>0.2</td>
<td>0.10</td>
<td>0.08</td>
<td>0.35</td>
</tr>
<tr>
<td>Policy</td>
<td>0.2</td>
<td>0.10</td>
<td>0.08</td>
<td>0.07</td>
</tr>
<tr>
<td>CR</td>
<td>0</td>
<td>0.01</td>
<td>0.00</td>
<td>0.02</td>
</tr>
</tbody>
</table>

Description of the scenarios and the result of their corresponding MCDA are in the following:

**Scenario No.1:** In this scenario, it is assumed that the most important criteria for a BECCS alternative are its technical and environmental impacts. Hence, the BECCS alternative with highest negative CO\(_2\) emissions and energy production with minimum environmental impacts are most important. In this scenario policy, social and economic criteria are of lower priority. The implicit assumption in scenario No.1 is that the socio-political circumstances have to support the BECCS alternative, which promises the highest net negative CO\(_2\) whatever the cost. Under this scenario, the choice of BECCS systems is technology driven, having minimum environmental impact as the limit. As illustrated in Figure 7-4, assigning a higher priority to technical and environmental criteria resulted in the change in criteria score and overall SI of all alternatives. Under scenario No.1 LFG-CCS ranks as the most sustainable BECCS alternative with SI = 0.65, followed by MSW (SI=0.62) and BG-CCS (SI=0.58). Compared to the baseline scenario, scores of technical and environmental criteria has increased and scores of economic, social and policy has declined for all the alternatives. This scenario has benefitted MSW-CCS, which delivers high negative emissions with low environmental impacts but at the highest cost.
Scenario No.2: In this scenario, economic and environmental criteria have the highest priority and policy and social criteria are of lower importance. In other words, the BECCS alternatives with minimum economic cost and environmental impact and maximum technical potential is deemed as the most sustainable choice. The rationale of this scenario is to impose least cost on the economy. In scenario No.2 environmental impacts are to be minimised but the main feature of BECCS to deliver negative emission is of secondary priority. Results of the MCDA under scenario No.2 are shown in Figure 7-5. As seen, like the previous scenarios, LFG-CCS with $\text{SI}=0.72$ ranks the most sustainable alternative. LFG-CCS promises a high annual negative emission at the lowest cost. BG-CCS with $\text{SI}=0.66$ and FR-CCS with $\text{SI}=0.64$ were the second and third best alternatives.

Figure 7-4: Overall Sustainability Index (SI) of BECCS alternatives under scenario No.1
Scenario No.2

In the third scenario, social criteria along with environmental criteria have the highest priority. Under this scenario, the most sustainable BECCS alternative is the one which provides highest social benefit with minimum environmental impacts. The technical capacity and cost of BECCS systems to deliver negative CO\textsubscript{2} emission is not emphasised in this scenario. Giving social criteria the highest priority is not common in MCDA of energy systems. Unlike economic, environmental, and most technical sub-criteria, which were qualitatively calculated and assessed, the social sub-criteria in this study are qualitative which were scored based on expert judgment. This makes an MCDA driven by highest social benefit more susceptible to uncertainty. The results of MCDA under scenario No.3 are presented in Figure 7-6. Compared to the baseline scenario, the BECCS alternatives utilising agricultural residues gained higher overall sustainability index. This is mostly due to the assumption that these BECCS alternatives are likely to offer a higher rate of job creation than MSW-based BECCS systems. In this scenario, FR-CCS and BG-CCS with SI= 0.75, ranks the most sustainable choice. Their overall SI is dominant by the high score in environmental and social criteria, but they acquire low scores on all other criteria. LFG-CCS and WS-CCS with SI= 0.74 were identified as second priority. Considering the significant gaps between the scores of different criteria, none of the alternatives could fulfil the strict sustainability discipline.
7.3.2 Trade-off analysis

The performance of the BECCS alternatives under criteria and sub-criteria is different. It means trade-offs are inevitable when choosing the most sustainable alternative. The objectives of the decision-making model determine the outcome of these trade-offs.

Figure 7-7 shows the trade-off between levelised cost of electricity production ($/MWh) and technical capacity to deliver negative emission (Gt CO\textsubscript{2}/year) of the BECCS systems. These two parameters are among the most crucial aspects of a BECCS system. Under a business-as-usual scenario with no carbon policy in effect, the LCOE of the BECCS systems is higher than that of coal-fired power plants -with or without CCS. The high production cost of BECCS, due to its high investment and O&M costs, would not be competitive in the power market. The LOCE of BECCS systems in this study is of $160–$212/MWh which lies in the upper range of LCOE through BECCS of $70–$230 /MWh, reported in a review study by kemper [30]. The cost of LFG-CCS ($160/MWh) is lower than other BECCS systems due to the lower capital and operating cost of a gas turbine compared to a solid combustion furnace. Circulating fluidised bed combustion technology used for MSW, agricultural and forest residue has a higher capital and operating cost than the pulverised coal-fired plant. Although MSW-CCS
offers the highest technical capacity for negative emission, but it is the most expensive system, as it requires costly modifications in order to manage its high moisture content and flue gas cleaning requirements.

The total negative emission potential of the residue and waste based BECCS systems modelled in this study is 4.8 Gt CO$_2$/year (see Figure 7-8). This 4.8 Gt CO$_2$/year is made up of agricultural residues (1.7 Gt CO$_2$/year), forestry residues (1.1 Gt CO$_2$/year) and organic municipal solid waste (MSW) (2 Gt CO$_2$/year). The range of the negative emission potential of BECCS envisaged in majorities of IAMs, including those in IPCC AR5, to achieve 2 °C is up to 20 Gt CO$_2$/year. According to this study, using organic MSW, agricultural and forest residue under strict sustainability criteria could contribute up to 24% of the maximum negative emission required for ambitions emission targets. The total energy available via sustainable biomass resources is estimated to be 48 EJ (Figure 7-8). Whilst this may be enhanced by for example major break-throughs in biotechnology, it would be dangerous to assume that this will happen.
Reaching this capacity requires large-scale BECCS deployment and for that recognising and crediting negative emissions are the key mechanisms needed.

![Negative CO₂ emission (Gt CO₂/year)](chart1)

![Resource Availability (EJ/year)](chart2)

Figure 7-8- Sustainable BECCS potential to contribute to the 2 °C target

Water-use and land-use change as the main environmental sub-criteria were used to demonstrate the trade-off between environmental and technical performance of the BECCS alternatives (see Figure 7-9). Historically, unsustainable biomass harvest and forest clearing in some areas, has led to loss of a considerable proportion of natural forests and degradation of productive lands [46, 50, 127] and depletion of water resources [128-130, 135]. Emissions from agriculture, forestry and other land-use change (AFLOU) account for approximately 10 to 12 Gt CO₂-eq/year of anthropogenic GHG emissions; with 5–5.8 Gt CO₂-eq/year from agricultural production and 4.3–5.5 Gt CO₂-eq/year from land use and land-use change activities [46]. The area of land needed for bioenergy production depends on the productivity of the land, efficiency of production practices and the type of biomass. The area available for bioenergy production from energy crops in the literature varies from 80 to 2400 Mha [11, 16, 49, 124, 140-144]. However of the land available for expanding bioenergy is only 40 Mha [136]. In Figure 7-9, the negative value of land-use change of the BECCS systems using agricultural residues is due to the assumption in the LCA model that no land transformation is required for production of the residues. Besides, these BECCS systems export electricity to the grid. This electricity replaces power from suppliers, which use land for construction and infrastructure.
Around 70% of the total withdrawal of all fresh water is used in agriculture. The water needed to grow dedicated crops to deliver a mid-range of 12.1 Gt CO₂/year negative emission through BECCS, in 2100 would be approximately 720 km³ [11]. That would make 18% of current human withdrawal. Compared to dedicated energy crops, using residues and waste for BECCS has a much lower impact on water scarcity. Water use of the most BECCS alternatives is close to zero. The only BECCS system with considerable lifecycle water consumption is RS-CCS with 0.26 m³/KWh. Currently the conventional method to cultivate rice, which is applied to produce approximately 90% of rice globally, is by flooded rice fields (paddy rice) [138]. Due to water intensity of the flooded method, the water consumption of RS-CCS is highest among the BECCS options. According to the LCA model used in this study for each KWh electricity generated from rice straw combustion 0.16 m³ water is used. Adding CCS to a power plant using rice straw increases the water consumption to 0.26 m³/ KWh. The additional water is mainly used for cooling in the MEA system. According to Figure 7-9, MSW-CCS with highest negative emission capacity and relatively low water use and land-use change is the optimal choice in this trade-off.

Figure 7-9- Water use, land-use change and negative emission trade-off for BECCS alternatives
7.4 Discussion

The AMS proposed in this study is a generic model and could be applied to any energy system. Though some of the sub-criteria such as annual negative emissions are specific to BECCS technologies, the rest are relevant to any power generation system. The aim of comparing the sustainability of BECCS alternatives was to assist decision makers and stakeholders to choose among the range of BECCS alternatives in order to fulfil planning objectives, especially if they have access to a variety of organic resources and energy technologies within environmental, financial and time constrains.

In this study, average global data was used to develop the AMS. However, it is crucial to adapt the objectives and criteria of an AMS used for implementing a BECCS system according to the regional parameters. For instance, response to BECCS will be different among developed and developing countries as their development agendas and priorities vary. In developed countries, where the majority of BECCS projects have been established, the agenda has been to use BECCS to replace carbon intensive alternatives while attaining major mitigation in the energy sector [30]. Conversely, in developing countries with significant under-utilised biomass resources (especially organic waste) [335], BECCS could be a solution to cut emissions while providing energy for the growing population. Furthermore, BECCS in developing countries could contribute to achieving sustainable development goals by enhancing parameters such as energy security, job creation, and economic growth [420]. However, in these regions, access to the high cost and complex BECCS technology could be a challenge. Therefore, introducing supportive plans such as inclusion of BECCS in clean development mechanisms (CDM) authorised by UNFCC [421] is a possible way to fast-track BECCS development at several scales in developing or even under-developed countries. To assure the carbon-neutrality/negativity of BECCS, robust accounting and monitoring of CO₂ emissions due to land use change must be conducted [154, 369]. Existing policy mechanisms such as “reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries” (REDD+) under the UNFCCC [370] and setting strict standardisation criteria for sustainable bioenergy help to minimise the environmental impact of the BECCS supply-chain [46, 154, 369].

One of the main challenges in this study was to rate the social and policy criteria of BECCS alternatives. With very limited practical experience of BECCS deployment, scoring these
criteria was based on best guess or generalised knowledge on CCS and bioenergy systems. Although the BECCS discourse has progressed within the scientific and expert sphere and is present in climate models, public perception and acceptance is still one of its main challenges. Similar studies of social acceptance of other industries, CCS and bioenergy in particular, could give an indication of how BECCS would be received by the public [30, 404, 417, 418, 422-425]. This could be a step forward to accelerate BECCS deployment which will be necessary if deep cuts in emissions, including decreases in existing atmospheric CO₂ is to be achieved. Assessment of social benefits such as job creation and access to energy are parameters having a positive impact on its community acceptance. Some trade-offs among the sustainability criteria are needed, particularly with respect to the social impacts. These trade-offs can be explored within the decision-support framework to reveal how different stakeholders’ preferences affect the outcomes of sustainability assessment [404]. To date, studies of the public understanding of BECCS are limited. A study by Fridahl [14] is one the few research projects on public acceptability of BECCS [14]. This study found that BECCS was viewed more favourably than fossil fuels with CCS and nuclear. However, compared to other mitigation options such as solar and wind, BECCS is assigned a relatively lower priority. Conversely, a study by Dowd et al. [418] on the social science of BECCS, showed that BECCS has a lower acceptability than fossil fuels with CCS. There is a profound mismatch in the degree of attention given to BECCS between the expert and public spheres. Therefore, access to knowledge could have a bilateral importance; improving public knowledge to rectify misconceptions on one hand, and on the other hand, educating advocates and developers on the holistic perspective of the community’s social, economic and cultural [418]. Active engagement with media has a key impact on managing the public perspective of BECCS [418]. Upham and Roberts [426] showed that collaboration with local NGOs and trusted stakeholder could facilitate development of BECCS projects, their surveys showed that there is lack of trust in information originating from industry or government. Seigo et al. [423] found that comparative assessment of the economic costs and benefits of BECCS and other mitigation alternatives affects its acceptability. Diaz-Chavez et al. [355] emphasised that reporting of corporate sustainability to the public could improve gaining community’s trust.

The scenarios examined in this study reflect the effect of socio-economic, technical and environmental drivers on the sustainability ranking of BECCS systems. Varying the configuration of BECCS alternatives under each scenario is an indication of how sustainability is a contextual concept. Selection of a scenario determines the objective of the decision-making
analysis. Assigning the first priority to the technical potential of BECCS to deliver negative emission with minimum environmental ramifications reflects the crucial need to remove CO$_2$ from the atmosphere. This is in line with the role of BECCS to remove CO$_2$ foreseen in integrated assessment models to reach zero emission by 2050 and net negative emission by the end of century. To achieve this means removing 5-20 Gt CO$_2$-eq from the atmosphere annually starting from mid-century [1, 12, 21]. The recommendation regarding the timeline to deploy BECCS at scale to reach this quantity of negative emission is not included in these models, which has caused uncertainty regarding the pace of BECCS large-scale deployment. The timing of the implementing the mitigation pathways that include BECCS could be crucial with any delay in mitigation may lead to more uncertainties. For instance, the long residence time of CO$_2$ in the atmosphere, uncertainties in the future availability of biomass, population growth and energy demand, and irreversible climate changes are some of the likely risks of a delayed mitigation plan.

An adaptive management system as proposed in this study, facilitates the decision-making process to evaluate the sustainability of the BECCS alternatives and introduces a systematic methodology to analyse the synergies and trade-offs between different criteria and scenarios. However, BECCS is a rather novel concept and there are still several key uncertainties regarding the optimal scale and timing of its deployment, availability of biomass in the future, the interlinkage between natural carbon cycles and biomass harvest, CCS storage capacity and permanency, legal policy/regulatory background and so forth. To eliminate the uncertainties, more in-depth assessment of its key parameters from global down to local levels is essential. To facilitate this procedure, strong governmental support is required [14], as has been the case with all BECCS projects so far. Coherent policy programs which have been advocated in the context of sustainable development [427] applies equally for deployment of BECCS. Adapting policy measures that target different but interlinked areas such as land and water conservation, food security, energy security and emission mitigation, is a possible way to achieve a policy coherence, which support BECCS deployment and long-term certainty.
7.5 Conclusions

In several climates studies [1, 29, 30, 124], bioenergy with carbon capture and storage (BECCS) has been given a key role in achieving ambitious mitigation targets such as those of COP 21 [3]. However, the focus on the techno-economic dimensions in assessing BECCS potential to produce energy and deliver negative emission [16, 34] will be challenging. Knowledge gaps regarding the complexities of BECCS systems and direct and indirect interactions with natural ecosystem, social development and political dynamics are factors that could potentially hinder its large-scale development. Hence, holistic sustainability assessment of the technical, economic, environmental, social and policy aspects of the different BECCS systems is crucial. The main objective of this study was to present a model for sustainable BECCS implementation, which identifies the key sustainability criteria and methodology to adapt alteration in the whole system.

- This study proposes an adaptive management system, which employs an analytical hierarchy to assess several BECCS alternatives and rank them against a set of key sustainability criteria. Techno-economic analysis, lifecycle environmental impact assessment along with social and policy assessment were applied to evaluate the performance of the BECCS alternatives under each criterion. The biomass resources for the BECCS alternatives in this study were organic waste from the municipal, agricultural and forestry sectors.
- A sensitivity analysis using scenarios with different sustainability paradigms for mitigation was conducted. The results endorse BECCS alternatives using municipal solid wastes under all scenarios.
- The dearth of references on the social and policy aspects of BECCS made scoring their corresponding criteria one of the challenges encountered in this study.
- In general, limited access to reliable data regarding socio-political parameters such as extent of policy support, social benefits and social acceptance of BECCS as a generic concept for CO$_2$ removal and different specific BECCS options, was one of the limitations of the present work.
Chapter 8

Conclusions
The uptake of renewable technologies has been spectacular in many ways, yet the global economy is still very dependent on fossil fuels and there is every prospect that emissions (and attendant global temperatures) will overshoot the agreed Paris targets. Hence the importance of negative emissions and the great stress placed on BECCS in many of the global mitigation models. Despite this importance, there are few quantitative assessments of the real opportunities that sustainable BECCS might offer. This thesis makes a contribution to addressing this knowledge gap by considering such issues as BECCS and co-firing, what the real potential is for BECCS if sustainability is to be fully considered, what the technical and economic challenges are for BECCS and what policy and other measures could be taken to accelerate the uptake of BECCS.

This concluding chapter attempts to address these and related questions;

8.1 Is Co-firing coal with biomass a viable option in a BECCS context?

Despite the need for the world to move to a zero-carbon economy as quickly as possible, fossil fuels and especially coal continues to be widely used for power generation. Therefore, the question arises whether there is merit in applying co-firing in power plants equipped with CCS as a near-term mitigation option. This would obviously require retrofit of CCS and significant investment. But perhaps it could provide a bridging technology to deliver secure energy for a growing population and cost-effectively lower CO₂ emissions.

This study suggests that:

- Direct co-firing of up to 20% biomass in a modern pulverised coal combustion plants equipped with CO₂ capture and storage could deliver negative emissions of up to around -26 kg CO₂/MWh.
- To put this into a global perspective, if all the coal-based electricity generating capacity (9,690 TWh a year) in the world were to be refurbished (including retrofit CCS) to accommodate this level of co-firing global negative emissions of 252 Mt CO₂ per annum could be achieved.
- This would be a modest but valuable contribution to achieving the Paris Agreement.
8.2 What are the opportunities for application of BECCS in the Australian power sector?

Like many other countries, Australia has committed to meeting its international obligations to decrease its greenhouse gas emissions including transitioning toward decarbonising its emission-intense energy sector. However, it is facing the dual problems of increasing electricity cost and decreasing energy security.

This study suggests that:

- Based on the quantity of biomass resources available, BECCS options in Australia have the potential to remove a total of 25 Mt CO$_2$/year from the atmosphere as negative emissions.
- BECCS systems could supply Australia with up to 13.7 TWh of renewable power by mid-century which is around 4% of expected gross electricity generation in 2050.
- Deployment of BECCS as a reliable supplier of clean electricity would potentially enhance the flexibility and diversity of Australia’s energy portfolio and remove carbon dioxide from the atmosphere. Presently, there is no other technology that offers this promise.

8.3 How important is organic residue/waste as a resource for BECCS?

Increased unsustainable energy crop production will lead to environmental impacts such as loss of a considerable proportion of remaining natural forests, degradation of productive lands, increased GHG emissions, loss of biodiversity and carbon stock and depletion of water resources. Maintaining food security and affordability whilst increasing the deployment of BECCS is one of serious social challenges facing any expansion of dedicated bioenergy crops.

Using organic waste from municipal, agricultural and forestry sectors for BECCS is a way to avoid the ecological and social challenges of dedicated energy crops. Organic residues are perceived to have less environmental impacts as they are inevitable by-products of high value food and fodder production. Globally there are very limited waste management mechanisms in place to exploit organic waste and residues at present. The residues and waste if not managed sustainably are sources of GHG emission.
This study suggests that:

- Given the social uncertainties and environmental impacts resulting from dedicated energy crop production, expansion of future bioenergy crops will be restricted by the availability of land and water.
- To address these challenges, priority should be given to the use of organic waste and residues for BECCS.
- Utilising these wastes in a BECCS system essentially turns a negative externality into a valuable good and provides a source of income for local farmers, industry and the state, whilst at the same time generating energy and delivering negative emission.

8.4 Could municipal solid waste be a valuable resource for BECCS?

Municipal solid waste (MSW) is a major by-product of urbanisation. Currently, the global urban population generates 1.3 Gt of solid waste annually, but very little of this is used for energy production. Challenges include the dispersed nature of MSW resources and the lack of economic support schemes, such as those that commonly apply to solar and wind. Nonetheless, MSW-based BECCS technologies have significant potential for abating and removing considerable amounts of the greenhouse gases from the atmosphere, thereby contributing significantly to the COP 21 emission reduction targets.

This study suggests that:

- Municipal solid waste could be a major resource for BECCS with an annual global potential of 2 Gt of negative emissions and 22 EJ of energy.
- To realise this potential will require legislative, financial and regulatory support.

8.5 What is the potential for agricultural and forestry residues to be a resource for BECCS?

Globally around 14 Gt of residue from forestry and 4.4 Gt residues from crop production (mainly barley, wheat, corn, sugarcane and rice), is generated annually. This is a significant underutilised biomass. However, not all of this biomass could be recovered. Residue collection technology, field specification parameters and climate constraints are factors affecting the amount of residue recoverable. One of the limitations on residue removal is its impact on soil carbon. Unconstrained expansion of using organic waste for energy may lead to the unsustainable loss of soil carbon stock.
This study suggests that:

- Assuming a conservative removal rate of 25% for all residues would allow adequate residues for utilisation such as animal feed and bedding and preserve soil carbon.
- Globally, agricultural and forestry residues could sustainably provide up to 2.8 Gt CO$_2$ negative emissions and 26 EJ energy per annum.
- Utilising residues for sustainable BECCS would enhance the flexibility of the mitigation portfolio and provide social and economic benefits to rural communities.

### 8.6 How sustainable is BECCS?

Whilst it could be argued that there are currently opportunities for sustainable cropping of corn for ethanol, this is unlikely to be sustainable in the longer term because of inevitable conflict with meeting the food needs of an ever-expanding global population. Therefore, sustainable BECCS will be primarily on the basis of negative emission delivered through organic waste and residues. A global potential negative emission (Gt CO$_2$/year) delivered by sustainable BECCS of 4.8 Gt CO$_2$/year compared to the CO$_2$ removal range envisaged in the majority of IAMs, including those in IPCC AR5 of up to 20 Gt CO$_2$/year. This 4.8 Gt CO$_2$/year is made up of agricultural residues (1.7 Gt CO$_2$/year), forestry residues (1.1 Gt CO$_2$/year) and organic municipal solid waste (2 Gt CO$_2$/year). In other words, of the total negative emission required in IAMs only 24% is achievable through what is regarded here as sustainable BECCS. Similarly, whilst globally the average amount of bioenergy that can be generated is assumed in a range of IAMs, to be of the order of 5-270 EJ/ year with an average of 100 EJ/ year by 2050 the total energy available via sustainable biomass resources is estimated to be only about half of this, at 48 EJ. Whilst this may be enhanced by for example major break-throughs in biotechnology, it would be dangerous to assume that this will happen.

This study suggests that:

- On the basis of existing biotechnologies and the sort of sustainable approach taken to bioenergy in this thesis, the values assumed for BECCS - based negative emissions in most of the IAMs are overly optimistic.
- Globally, sustainable BECCS could deliver approximately 48 EJ of energy and 4.8 Gt CO$_2$/year of negative emissions, made up of agricultural residues (1.7 Gt CO$_2$/year), forestry residues (1.1 Gt CO$_2$/year) and organic municipal solid waste (2 Gt CO$_2$/year).
8.7 Is an Adaptive Management System useful when assessing the opportunities for sustainable deployment of BECCS?

To deal with the complexities of BECCS systems, facilitate decision-making and enable evaluation of the sustainability of BECCS systems, an adaptive management system (AMS) has been used in this thesis. AMS employs an analytical hierarchy to assess BECCS systems and rank them against a set of key sustainability criteria. Techno-economic analysis and lifecycle environmental impact assessment, along with social and policy assessments were applied to evaluate the performance of BECCS alternatives under each criterion. An adaptive management system as proposed here, provides a systematic methodology to analyse the synergies and trade-offs between different criteria and scenarios.

This study suggests that:

- An Adaptive Management Systems approach provides a rigour to evaluation of BECCS that is lacking in evaluations based solely on a quantitative approach to biomass, which ignores or undervalues holistic sustainability as an essential component of BECCS.
- A sensitivity analysis using scenarios with different sustainability paradigms for mitigation endorses BECCS using municipal solid wastes under all scenarios.

8.8 How feasible is large-scale BECCS deployment globally?

Whilst BECCS offers many advantages, the reality is that under current energy policies, BECCS is considerably more costly than base-load fossil fuel-based power or wind or solar power. In the absence of any economic incentives, the levelised cost of electricity production from sustainable BECCS systems, lies between $160–$210/MWh, compared to a conventional (unmitigated) coal-fired power plant, which is less than $60/MWh. Based on the techno-economic assessments undertaken in this study, a carbon price of $130/t CO₂ – $200/t CO₂ would be needed for BECCS to be cost-competitiveness with conventional unmitigated coal-fired power generation. BECCS will not be competitive under any existing scheme that is based solely on the cost of produced electricity. That will be only achievable under a globally integrated policy with an effective price on CO₂, including a realistic price on negative emissions. The current carbon price under the European Union emission trading system (EU ETS) fluctuates around 8.5 Euro per tonne CO₂, which is far less that the level required for deep cuts in emissions.
However, in many ways this cost comparison is not valid in a carbon-constrained situation. Nor it is valid to compare BECCS-based power with intermittent wind power or solar. Perhaps a more valid comparison is with deep (hot rock) geothermal or solar thermal, which have the potential to produce renewable base-load power, but for the present there are few facilities that are doing so. Unlike BECCS, these systems are receiving attention from governments, endorsement by NGO’s and research funding. Despite this, many of the global mitigation models assume a level of knowledge regarding the potential of BECCS that does not presently exist. Nor is there a level of government research support consistent with its potential importance, as evident in many models. Unlike other sources of clean energy, the level of government support for BECCS RD&D is at best modest and in many instances non-existent. This is at variance with the fact that many internationally accepted climate models have BECCS as an essential part of their mitigation strategy.

This study suggests that:

- It is essential to establish a consistent negative emission narrative.
- Unless there is international and national support, including regulatory and financial support, BECCS will not progress.
- An effective and widely-applied carbon price or credit, together with supporting legislation and for some years to come, direct support for BECCS RD&D to bring down costs, are essential to encourage large-scale deployment of BECCS.
- Whether based on a carbon price, a carbon credit or renewable energy certificates, geologically stored CO\textsubscript{2} and emitted (or mitigated CO\textsubscript{2}) must be placed on the same basis, if BECCS-based negative emissions are to reach their full potential to help control global warming.
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210


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Title:
Bioenergy with carbon capture and storage; sustainability, challenges, and potential

Date:
2018

Persistent Link:
http://hdl.handle.net/11343/214528

File Description:
Bioenergy with Carbon Capture and Storage (BECCS); Sustainability, Challenges, and Potential

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