Forest Soil Carbon Stocks and Life Cycle Assessment of Short Rotation Forestry

A thesis submitted to the University of Limerick in fulfilment of the academic requirements for the Degree of Doctor of Philosophy by

Michael A. Clancy

Under the supervision of:

Dr. Kenneth A. Byrne (University of Limerick)

Submitted to Department of Biological Sciences, Faculty of Science and Engineering
University of Limerick, Ireland, November 2018
Abstract

Inventories of forest soil carbon (C) stocks are necessary to determine spatial and temporal C stock changes and support climate change mitigation policy development. Afforested podzols and peaty podzols were sampled to measure their soil organic carbon (SOC) concentration and bulk density (BD), with the aim of improving baseline soil organic carbon density (SOCD) estimates for Irish forests. Podzols are not always distinguished from peaty podzols and both qualify as mineral soil types. The estimated SOCD for the podzol sites ranged from 129–139 Mg C ha\(^{-1}\), while the peaty podzols had 229–385 Mg C ha\(^{-1}\). The major disparity in their SOCD implies the need to disaggregate podzols and peaty podzols in conducting soil C inventories, with the need for development of C emission factors for peaty podzols to reduce uncertainty in SOCD estimates. Soil bulk density (BD) is a principal component in estimating the stock of any soil nutrient or other substances, such as organic carbon. There are several well-known methods of estimating soil BD such as coring, clod, and pit excavation, with variants of each method covered in the literature. In many older soil surveys, for reasons related to cost or the time and resources available to complete the work, the soil BD was not measured, often due to a high volume of rock fragments (RFs) in the soil. This study used the core and pit excavation methods to sample short rotation forestry (SRF) soils and determine their SOC concentration, BD, and RF content, and used those to estimate the SOCD to a depth of 40 cm. Novel methods of sampling and calculating BD for soils with high RF content were also devised, with results that compared more favourably to the pit excavation results than provided by the core method. Using SRF biomass for bioenergy has the potential to contribute to the Ireland’s increasing commitments under the EU Renewable Energy Directive to use more renewable energy and reduce greenhouse gas (GHG) emissions by 2020 and beyond. When sustainable forest
management practices are employed SRF can also help offset Ireland’s GHG emissions from the combustion of decreasing reserves of peat and fossil fuels through their displacement in industrial power plants. This study investigated the GHG balance of a eucalyptus (*E. nitens*) SRF plantation over three 10 year rotations through the use of Life Cycle Assessment (LCA) methodology, combined with specialised LCA software tools and databases. The mean GHG balance of the SRF scenarios (i.e. 2441 t CO$_2$-eq ha$^{-1}$ and 1135 t CO$_2$-eq ha$^{-1}$, at the 30 % and 50 % co-firing rates, respectively) are both approximately seven times greater than the GHG balance of the Sitka spruce scenarios (i.e. 339 t CO$_2$-eq ha$^{-1}$ and 168 t CO$_2$-eq ha$^{-1}$, at the same respective co-firing rates). Hence, the mean GHG emission reductions from the SRF biomass scenarios outperform those from the Sitka spruce scenarios at both the per-MWh$_e$ and per-hectare levels.

**Keywords:** Afforestation, podzol, bulk density, rock fragments, bioenergy.
Declaration

I, the undersigned, declare that the work in this thesis is entirely my own and to the best of my knowledge contains no material previously written, published or submitted for merit or award by this University or by any other academic establishment, except where acknowledgement has been made in the text.

Signed: ____________________________ Date: ___________
Acknowledgements

Firstly, I would like to thank my supervisor Dr. Ken Byrne. Ken gave me the unexpected but very much appreciated opportunity to undertake my postgraduate studies, which to-date has culminated in the body of work herein. During the last several years working under Ken’s guidance, I’ve been the beneficiary of his great generosity of spirit, patience, and unfailing support to help me succeed in achieving my goals. I have marvelled at the range and depth of his knowledge on many subjects and learned a great deal from him. I will always value his thoughtful insights on our shared academic and non-academic interests.

My co-supervisor on the CForRep project Dr. Thomas Cummins also gave me a great start in soil science, field and laboratory work. From those early lessons in soil profile analysis and description to his guidance in academic writing, I hope to continue developing those critically important skills.

I am very grateful to my postgraduate funding provider, the Department of Agriculture, Forestry and the Marine (DAFM), Ireland, without which I could not have undertaken the path to this PhD. I was lucky enough to work on two great projects funded by DAFM, namely CForRep and ShortFor with a number of exceptional project partners: UCD: Dr. Brian Tobin, Dr. Conor O’Reilly, Dr. Kevin Black, WIT: Mr. Tom Kent, Dr. Enda Coates, Ana de Miguel, Brian Nolan, Coillte: Padraig O’Tuama, Mary Leahy, TCD: Dr. Liwen Xiao, Teagasc: Dr. Niall Farrelly, Dr. Nuala Ni Fhlatharta. I am very grateful for all their support, and in particular the assistance given to me by Prof. Rachel Creamer and Dr. Brian Reidy, both formerly with Teagasc and the Irish Soil Information System project. One other person I had the great fortune to meet and learn from was the late Dr. Otto Spaargaren, and even though the time we spent together was
very short, he made a profound impression on me with his expertise in soil science, and his humility.

To my UL colleagues: Dr. Richard Lane, Dr. Jonay Jovani, Dr. Caitlin Rigney, and Dr. Jean O’Dwyer, I will always be grateful for your great friendship, support, encouragement, and the fun times we have shared to-date. I wish you all the best in your current and future roles. I would also like to thank Laura Walshe, Kevin Kilcoyne, and Dr. Arit Efretui for their assistance and positivity during what was often testing field and laboratory work.

I have always been blessed with great support from my brothers and sisters, Noel, John (deceased), James, Patricia, Carol and Dan, for which I am eternally grateful. I wish to acknowledge the role my deceased father, John Clancy, played in shaping my attitude to education and instilling a sense of curiosity in the world around us. My greatest regret is that I didn’t finish my PhD before my mother, Mary Clancy, passed away in March this year. She was a constant well of support and encouragement, prayers and lighted candles, all for a successful outcome. I will never forget her kind words, smiles, and frequent uncontrollable laughter that lightened our lives during her last years with us.

Finally, I want to dedicate this work to my wonderful wife Edel, and my children Rachel, James, Robert, and Isabelle, for all the sacrifices they have endured while I was a student. Without Edel’s unyielding love and support I could never have started, never mind finished this long undertaking. I will never stop trying to repay you in kind for never losing faith and carrying me all the way. A very sincere “Thank you” to all, and go n-éirí an bóthar leat.
# Table of Contents

Abstract .............................................................................................................................. i  
Declaration ...................................................................................................................... iii  
Acknowledgements ......................................................................................................... v  
Table of Contents ........................................................................................................... vii  
List of Tables ................................................................................................................... xiii  
List of Figures ................................................................................................................ xvi  
List of Abbreviations ..................................................................................................... xxv  
List of publications ....................................................................................................... xxxvii

## Chapter 1 - Introduction ............................................................................................. 1

1.1 Background ............................................................................................................... 3

1.2 Aims and objectives of the thesis ........................................................................... 6

1.3 Thesis structure ........................................................................................................ 8

## Chapter 2 - Literature Review .................................................................................. 9

2.1 The global carbon cycle and our changing climate ............................................. 11

2.1.1 Greenhouse gases and associated global warming ........................................ 13

2.1.2 International climate change mitigation policy development and the role of afforestation ........................................................................................................... 15

2.2 Irish forestry development and use of forest biomass in offsetting greenhouse gas emissions .................................................................................................................. 18

2.2.1 Irish conventional Sitka spruce forestry ............................................................ 20

2.2.2 Short Rotation Forestry and other bioenergy crops .................................... 21

2.2.3 Irish forestry yield modelling ........................................................................... 24

2.3 The role of forest soils in climate change mitigation ........................................... 25

2.3.1 Soil organic matter and soil carbon pools ..................................................... 25

2.3.2 Land-use change, site preparation and disturbances ...................................... 29

2.3.3 Forest soil organic carbon change ................................................................... 31
3.4.2 Podzol soil organic carbon concentration 72
3.4.3 Podzol soil organic carbon concentration analysis 74
3.4.4 Podzol soil organic carbon density 74
3.4.5 Soil organic carbon correlation with bulk density for the podzol sites 75
3.4.6 Peaty podzol bulk density 76
3.4.7 Peaty podzol soil organic carbon concentration 76
3.4.8 Peaty podzol soil organic carbon concentration analysis 78
3.4.9 Peaty podzol soil organic carbon density 78
3.4.10 Soil organic carbon concentration correlation with bulk density for the peaty podzol sites 79

3.5 Discussion 81

3.6 Conclusion 83

Chapter 4 - The influence of soil bulk density sampling and calculation methods on estimates of soil organic carbon density ................................................................. 85

4.1 Abstract 87

4.2 Introduction 88

4.3 Materials and Methods 92

4.3.1 Site selection and description 92

4.3.2 Field methods 95

4.3.3 Laboratory methods 99

4.3.4 Soil bulk density calculation methods 101

4.3.5 Method for adjustment of soil bulk density measured with cores to take account of large rock fragments 102

4.3.6 Estimation of the soil organic carbon density using the core, pit, and core-scaling BD methods 104

4.4 Results 104

4.4.1 Soil bulk density of core samples calculated using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods 105

4.4.2 Soil bulk density of pit samples calculated using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods 109
4.4.3 Comparison of the mean soil bulk density using two field sampling and three calculation methods

4.4.4 Rock fragment contents of core and pit samples

4.4.5 “Core-scaling” rock fragment mass and volume, and comparison of derived BD results with core and pit $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$.

4.4.6 Soil organic carbon concentration

4.4.7 Estimated core, pit, and core-scaling soil organic carbon density, calculating using the $\rho_{total}$, $\rho_{fe}$, $\rho_{hybrid}$ methods

4.5 Discussion

4.6 Conclusions

Chapter 5 - Life Cycle Assessment of the Greenhouse Gas Balance of Irish Short Rotation Forestry ................................................................................................... 135

5.1 Abstract 137

5.2 Introduction 138

5.2.1 Objectives of this study: 139

5.3 Materials & Methods 140

5.3.1 LCA Goal and Scope Definition 141

5.3.2 Life Cycle Inventory data sources and assumptions 145

5.3.3 LCIA methodology, and its modification 157

5.4 Results 158

5.4.1 Cradle-to-grave greenhouse gas balances 159

5.4.2 Cradle-to-gate and gate-to-grave greenhouse gas balances 163

5.4.3 Disaggregated biomass and peat cradle-to-gate greenhouse gas balances 168

5.4.4 Greenhouse gas balances of the biomass and peat processing and transport processes 172

5.4.5 Potential annual greenhouse gas emission reductions 175

5.5 Discussion 177

5.5.1 Research aims and previous Irish LCA studies 177

5.5.2 LCA scope and system boundaries 177
5.5.3 Study limitations and uncertainty analysis 178
5.5.4 Biomass and peat greenhouse gas balances per MWh_e 179
5.5.5 Peat greenhouse gas balance per MWh_e 180
5.5.6 Peat and biomass co-firing greenhouse gas balance per MWh_e 180
5.5.7 Greenhouse gas balances per hectare 181
5.5.8 Impact of biomass and peat land use change 181
5.5.9 Impact of intensive harvesting and related fertilizer use 182
5.5.10 The influence of tree species carbon and basic density on biomass carbon sequestration 184
5.5.11 Potential short rotation forestry contribution to EU and Irish greenhouse gas targets 184
5.6 Conclusions 185

Chapter 6 - Synthesis and recommendations ...................................................... 187
6.1 General discussion and synthesis 189
6.2 Recommendations for further research 192

Chapter 7 - References ........................................................................................... 195

Chapter 8 - Appendices .......................................................................................... 225
Appendix 1.1 – WRB Keys to Reference Soil Groups 226
Appendix 1.2 – FAO subordinate characteristics within master horizons 227
Appendix 1.3 – FAO Soil Description - Stone 228
Appendix 1.4 – FAO Soil Description - Roots 229
Appendix 1.5- Measuring rock fragment bulk density using the water displacement method 230
8.1 For Appendices 3.1 to 5.4 please see attached disk 231
8.1.1 A3.1 – Chapter 3 Results table (as published) 231
8.1.2 A4.1 – Chapter 4: Core and pit excavation data 231
8.1.3 A4.2 – Chapter 4: SOCD data 231
8.1.4 A5.1 – Chapter 5: LCA scenario data, per MWh_e 231
8.1.5 A5.2 – Chapter 5: LCA scenario data, per hectare 231
8.1.6 A5.3 – Chapter 5: LCI data sources (numbers coded in parentheses) 231
8.1.7 A5.4 – Chapter 5: LCA literature (coded as per A5.3) 231
List of Tables

Table 3.1 Sites selected for SOC and BD sampling. Alt = Altitude. Altn. = Alternate to NFI site, Limk. = Limerick, Tipp. = Tipperary.................................................................62

Table 3.2 Number of field soil organic carbon (SOC) and bulk density (BD) samples taken for laboratory analysis. ..................................................................................66

Table 3.3 Soil profile analysis for the podzol sites. ........................................................................68

Table 3.4 Soil profile analysis for the peaty podzol sites. ...........................................................69

Table 3.5 Soil bulk density (BD, g cm$^{-3}$) and standard deviation (S.D.) by depth, and inter-site mean BD and S.D. for the podzol sites...................................................72

Table 3.6 Soil organic carbon (SOC) concentration (%) and standard deviation (S.D.) by depth, and inter-site mean SOC and S.D for the podzol sites.............................73

Table 3.7 Mean SOC % by depth for podzol sites. N = population, SD = Standard Deviation, SEM = Standard Error of the Mean, CV = Coefficient of Variance.....74

Table 3.8 SOCD (Mg ha$^{-1}$) by depth and overall mean values for the podzol sites.......75

Table 3.9 Soil bulk density (BD, g cm$^{-3}$) and standard deviation (S.D.) by depth, and inter-site mean BD and S.D. for the peaty podzol sites..........................................76

Table 3.10 Soil organic carbon (SOC) concentration (%) with standard deviation (S.D.) by depth, and inter-site mean SOC and S.D for the peaty podzol sites. ...............77

Table 3.11 Mean SOC % by depth for peaty podzol sites. n = population, SD = Standard Deviation, SE = Standard Error of the Mean, CV = Coefficient of Variance. ......78
Table 3.12 Soil organic carbon density (SOCD) by depth and overall mean values for peaty podzol (PP) sites.

Table 4.1 Characteristics of the 10 short rotation forestry (SRF) sites selected for soil carbon stock sampling.

Table 4.2 The soil bulk density (BD), standard deviation (SD), and coefficient of variation (CV), sampled by core method and calculated using the $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$ methods (Throop et al. (2012)), for four 10 cm depths, and the mean values for 0–40 cm depth.

Table 4.3 Mean OD mass (g), volume (cm$^3$), and percentage of total sample mass and volume of the rock fragments (RFs) found in the 100 cm$^3$ core samples, for each of the 10 SRF sites, plus the mean values for all 10 sites.

Table 4.4 Total rock fragment (RF) mass (kg), volume (l), and percentage of total sample mass and volume of the RF, for each of the seven sites sampled using the pits. Also, the mean values for all seven sites.

Table 4.5 The mean OD mass and volume of the core sample RF, from the seven sites sampled using both the core and pit methods, scaled up to match the total volume of their respective quantitative pits.

Table 4.6 The estimated soil organic carbon (SOC) concentration (%) of the 10 short rotation forestry sites. The soil was sampled in 10 cm depth increments down to 40 cm depth.

Table 5.1 The assumed SRF and Sitka spruce biomass yields, by scenario, per hectare, for 3 × 10, and 1 × 30 year rotations respectively, with their associated solid-to-
woodchip/hogfuel conversion factors (CF), and embedded energy values (MWh_e/ha). .................................................................................................................................147

Table 5.2 Seedling production and site establishment process flows for 1 ha of forest plantation, for biomass assortments in both the Sitka spruce and SRF scenario models. ........................................................................................................................................148

Table 5.3 The short rotation forestry (SRF, represented by the Eucalyptus nitens species) and the Sitka spruce biomass assortment yields, for 3×10 year and 1×30 year rotations, respectively. ........................................................................................................................................149

Table 5.4 A consolidated inventory of the harvesting process input and output flows for the aboveground Sitka spruce pulpwood and brash, and SRF (E. nitens) stem-only roundwood and wholetree biomass assortments, along with the belowground Sitka spruce and SRF stumpwood recovered after clearfell. ........................................................................................................152

Table 5.5 Forwarding process flows for 1 hour of work on-site........................................154

Table 5.6 Chipping process (at the roadside) flows for 1 hour of work on-site. .................155

Table 5.7 Transport process for biomass woodchip and hogfuel to plant gate..............156
List of Figures

Figure 2.1 The global C cycle. The black numbers and arrows represent the pre-Industrialisation Era (c.1750) mass of the C stocks in each reservoir, in PgC (1 Pg = 1 Petagramme = 10^{15} g), and annual C exchange fluxes in PgC y^{-1}. The red numbers and arrows indicate the post-industrial era annual anthropogenic perturbation of the C cycle fluxes, averaged over the years 2000–2009. The red arrows represent: The average atmospheric CO_{2} increase (also called the ‘CO_{2} growth rate’) due to fossil fuel combustion, cement production, and net land-use change CO_{2} emissions, plus the net ocean and terrestrial ecosystems uptake of anthropogenic CO_{2} (Source: Ciais et al. 2013). ...................................................... 12

Figure 2.2 Atmospheric concentrations of the main GHGs from year 1750–2011 AD. 14

Figure 2.3 Biomass from forestry operations and residues from wood processing are used to meet internal process energy needs in the forestry and for renewable energy generation as electricity, heat or wood derived fuels. Figure: Sveaskog (Source: Berndes et al. 2016). .................................................................................. 20

Figure 2.4 Components of C balance and allocation to soil C after afforestation. .......33

Figure 2.5 A generic soil profile with layered organic and mineral horizons (Source: University of Miami, 2013). NB: The above diagram gives an overview of several of the more common organic (e.g. Oi), and mineral (e.g. B) master horizons and transitional horizons (e.g., BC) found in soil profiles. In field studies various combinations and/or subsets of these horizons, with different organic or mineral characteristics and other associated FAO (2006b) horizon designations than those shown above, may be found at varying depths in the soil profile. .......................... 40
Figure 2.6 A typical podzol profile

Figure 2.7 A typical peaty podzol profile

Figure 2.8 The four phases of a Life Cycle Assessment (LCA) study, as defined by the LCA standards (ISO 2006).

Figure 3.1 Geographic distribution of the sampled sites (Source: Google earth, 2014).

Figure 3.2 Forest pit excavation for soil bulk density (BD) sampling.

Figure 3.3 Schematic of forest soil pit bulk density (BD) core sampling at 10 cm depth increments to 40 cm, and adjacent soil organic carbon (SOC) auger sampling at 25 and 50 cm from the pit faces.

Figure 3.4 Soil organic carbon (SOC) concentration (%) change by depth, and standard deviation error bars, for the podzol sites.

Figure 3.5 The SOC % to BD (g cm$^{-3}$) correlation for the podzol sites, with the best-fit exponential function.

Figure 3.6 Soil organic carbon (SOC) concentration (%) change by depth, and standard deviation error bars, for the peaty podzol sites.

Figure 3.7 Soil organic carbon concentration (SOC %) to soil bulk density (BD, g cm$^{-3}$) correlation for the peaty podzol sites.

Figure 3.8 Mean SOC % for each depth for the podzol and peaty podzol sites, with standard deviation error bars.

Figure 4.1 Geographic locations of the 10 short rotation forestry (SRF) sites in counties Limerick, Cork, and Waterford which were sampled for SOC concentration and
soil BD. The red box on the inset map of Ireland shows the area of southern Ireland in which the sites were located. Figure 4.2 An outline (not to exact scale) of the layout and dimensions of the three pits. The pits were spaced approximately 2–3 m apart depending on tree and drain spacing. Pit B, the central pit, was at first excavated to the same depth and approximate volume as the pits on either side (Pits A and C). After all the soil (fine and coarse fraction) was removed for laboratory analysis, Pit B was then extended to an area approximately 1 m$^2$ to allow soil bulk density (BD) samples to be taken using the cores from the middle of each 10 cm depth increment, from the beneath the litter layer down to 40 cm. Figure 4.3 An area approximately 30 cm in diameter was cleared of loose litter and vegetation before excavating each soil pit to a minimum depth of 40 cm. Figure 4.4 Example of glass beads being used to measure the volume of an irregular shaped soil pit. The soil pits were double-lined with polyurethane bin liners and the volume of the pit was calculated using a known weight and volume of glass beads. The straight-edge ruler in the figure (between the tray and the pit) is 30 cm long. Figure 4.5 Schematic of the forest soil pit bulk density (BD) sampling at 10 cm depth increments, from a depth of 40 cm. The samples were taken diagonally (to reduce compaction) starting at the bottom and moving up the profile to avoid soil contamination from above when removing the 5 cm diameter, 100 cm$^3$ cores (Eijkelkamp, Netherlands). Figure 4.6 The 0–40 cm soil BD (g cm$^{-3}$) for the seven pit sampled sites using analysis of the mass and volume of both the soil fine and coarse fraction from one
quantitative pit per site, and calculated using the \( \rho_{\text{total}} \), \( \rho_{\text{fe}} \), and \( \rho_{\text{hybrid}} \) methods.

Also, the mean \( \rho_{\text{total}} \), \( \rho_{\text{fe}} \), and \( \rho_{\text{hybrid}} \) values for the seven sites sampled.

Figure 4.7a,b,c Comparison of \( \rho_{\text{total}} \), \( \rho_{\text{fe}} \), and \( \rho_{\text{hybrid}} \) (g cm\(^{-3}\)) for the 0–40 cm sampling depth using the core and pit methods, with standard deviation error bars for the core samples and the means for both methods. The results for the three almost stone-free sites (i.e. DKB, DKN, CLY) sampled using the core method are also presented.

Figure 4.8a,b The best-fit linear correlation of pit-\( \rho_{\text{total}} \) a) pit-\( \rho_{\text{hybrid}} \) and b) to pit-\( \rho_{\text{fe}} \) results for the seven sites where the quantitative pit sampling method was used.

Figure 4.9 The mass (kg) of the rock fragments from the pit samples in four size classes.

Figure 4.10 The volume (l) of the rock fragments from the pit samples in four size classes.

Figure 4.11 Comparison of the soil bulk density (BD) for the 0–40 cm depth for the seven sites, derived using the combination of the core, pit and core-scaling sampling methods, and calculated using the a) \( \rho_{\text{total}} \), (b) \( \rho_{\text{fe}} \), and (c) \( \rho_{\text{hybrid}} \) methods.

Figure 4.12 The correlation of the pit \( \rho_{\text{total}} \) (a) values to the \( \rho_{\text{fe}} \) (b) and \( \rho_{\text{hybrid}} \) (c) values calculated using the mean mass and volume of the soil fine and coarse fraction from the cores scaled up to match the pit volume, and adjusted for the mass and volume of the pit rock fragments larger than the core diameter, i.e., 5 cm.
Figure 4.13 The estimated core-, pit-, and scaled- $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ SOCD for the 0–40 cm depth, for the 10 core sampled sites, with the means for the three calculation methods. ................................................................. 126

Figure 4.14a,b,c The estimated SOCD for the 0–40 cm depth using the BD calculations from the core-, pit-, and scaled- $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$, using data from the a) core method, b) pit method (b), and (c) core-scaling sampling methods. ................. 127

Figure 5.1 A graphic overview of the integrated harvesting, forwarding, chipping and transport to the point-of-use of three biomass assortments: stems, wholetree, and stumps (Source: METLA Forest Energy Portal)................................................................. 141

Figure 5.2 The Life Cycle Assessment (LCA) system boundary for four Sitka spruce (SS1–4, green arrows) and four SRF (SRF1–4, blue arrows) biomass assortment scenarios. The system boundary encompasses all the cradle-to-grave biomass production and processing operations that have potential environmental impacts. The Sitka spruce undergoes end-of-rotation ($\times$1) clearfell harvesting of pulpwood (SS1–4), with optional thinning operations (e.g. SS3–4), recovery of brash and stumps (SS2, SS4), and fertilizer application after the Sitka spruce site establishment (SS4). The SRF undergoes end-of-rotation ($\times$3) clearfell harvesting of stems (SRF1) or wholetree assortments (SRF2–4), with optional recovery of stumps (SRF3–4) and fertilizer application after the SRF site establishment (SRF4). The LCA operations shown encompass all the material (e.g. fertilizers, machines) and energy (e.g. diesel oil, electricity) inputs, and subsequent outputs (e.g. biomass products, or greenhouse gases). ...................................................... 144

Figure 5.3 The operations included in the Life Cycle Assessment system boundary for the peat-only (Peat-0) scenario. ................................................................. 145
Figure 5.4 The area of existing forestry and areas with soil types suitable to afforestation in Ireland, within 100 km radius of the Edenderry Power Ltd. peat and biomass co-firing electricity generation plant (based on map from Schulte et al. 2016).

Figure 5.5 The GHG balance (kg CO$_2$-eq) per MWh$_e$ of the cradle-to-grave LCA of the peat, Sitka spruce, and SRF biomass scenarios, at 0, 30, 50, and 100 % substitution of peat at the combustion phase of the life cycle. Subst.: substitution.

Figure 5.6 The estimated energy potential from the woodchip and hogfuel from the Sitka spruce and SRF biomass produced in each scenario, converted to MWh$_e$ ha$^{-1}$, at 40 % moisture content (MC).

Figure 5.7 The GHG balance (t CO$_2$-eq) per hectare of the cradle-to-grave LCA of the peat, Sitka spruce, and SRF biomass scenarios, at 0, 30, 50, and 100 % substitution of peat at the combustion phase of the life cycle. Subst.: substitution.

Figure 5.8 The GHG balance (kg CO$_2$-eq) per MWhe of the cradle-to-gate (BAG, PAG) and gate-to-grave processes (PS, PC, BC, AD) at the EPL power-station. BAG: biomass-at-the-gate, PAG: peat-at-the-gate, PS: power station infrastructure, PC: peat combustion, BC: biomass combustion, AD: ash disposal. The data labels in the figure are for the three main contributors to the net GHG balance, i.e., BAG, PC, and BC.

Figure 5.9 The GHG balance (t CO$_2$-eq) per hectare of the cradle-to-gate (BAG, PAG) and gate-to-grave processes (PS, PC, BC, AD) at the EPL power-station. BAG: biomass-at-the-gate, PAG: peat-at-the-gate, PS: power station infrastructure, PC: peat combustion, BC: biomass combustion, AD: ash disposal. The data labels in
the figure are for the three main contributors to the net GHG balance, i.e., BAG, PC, and BC........................................................................................................... 167

Figure 5.10 The GHG balance (kg CO$_2$-eq) per MWh$_e$ of the aggregated biomass land-use change (BLUC), biomass carbon sequestration (BCS), biomass processing and transport (BP&T), peat land-use change (PLUC), and peat processing and transport (PP&T) operations of each scenario. NB: i) The data label values are rounded for display purposes, ii) The Sitka spruce BLUC values are shown just below the x-axis, iii) The Sitka spruce and SRF BCS values are shown below their bars. ...... 170

Figure 5.11 The GHG balance (t CO$_2$-eq) per hectare of the aggregated biomass land-use change (BLUC), biomass carbon sequestration (BCS), biomass processing and transport (BP&T), peat land-use change (PLUC), and peat processing and transport (PP&T) operations of each scenario. The data label values are rounded for display purposes. ............................................................................................................... 171

Figure 5.12 The GHG balance MWh$_e^{-1}$ of the SRF and Sitka spruce biomass production and transport phases of the 30 and 50 % peat substitution levels for all LCA scenarios. SE: site establishment, H: harvesting, F: forwarding, C: chipping, T: transport, PP: peat production, PT: peat transport. NB: The data label values are for the H, T, and PP components, and are rounded for display purposes. .................. 173

Figure 5.13 The GHG balance ha$^{-1}$ of the SRF and Sitka spruce biomass production and transport phases of the 30 and 50 % peat substitution levels for all LCA scenarios. SE: site establishment, H: harvesting, F: forwarding, C: chipping, T: transport, PP: peat production, PT: peat transport. NB: The data label values are for the H, T, and PP components, and are rounded for display purposes.......................... 174
Figure 5.14 Potential GHG emissions percentage reduction (per MWh$_{e}$ basis) by scenario from co-firing peat and biomass for a full year of the Edenderry power plant operation, compared to exclusive peat usage, i.e., the Peat-0 scenario. .... 176
### List of Abbreviations

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>AD</td>
<td>Ash disposal</td>
</tr>
<tr>
<td>ALCA</td>
<td>Attributional Life Cycle Assessment</td>
</tr>
<tr>
<td>BAG</td>
<td>Biomass at the gate</td>
</tr>
<tr>
<td>BC</td>
<td>Biomass combustion</td>
</tr>
<tr>
<td>BCS</td>
<td>Biomass carbon sequestration</td>
</tr>
<tr>
<td>BLUC</td>
<td>Biomass land use change</td>
</tr>
<tr>
<td>BP&amp;T</td>
<td>Biomass processing and transport</td>
</tr>
<tr>
<td>C</td>
<td>Carbon</td>
</tr>
<tr>
<td>CO₂</td>
<td>Carbon dioxide</td>
</tr>
<tr>
<td>CO₂-eq</td>
<td>Carbon dioxide equivalent</td>
</tr>
<tr>
<td>CH₄</td>
<td>Methane</td>
</tr>
<tr>
<td>CCS</td>
<td>Carbon capture and storage</td>
</tr>
<tr>
<td>CF</td>
<td>Conversion factor</td>
</tr>
<tr>
<td>CHF</td>
<td>LCIA characterisation factor</td>
</tr>
<tr>
<td>CLCA</td>
<td>Consequential Life Cycle Assessment</td>
</tr>
<tr>
<td>CML</td>
<td>Centrum Milieukunde, Leiden University, Netherlands</td>
</tr>
<tr>
<td>DAFM</td>
<td>Department of Agriculture, Forestry and the Marine</td>
</tr>
<tr>
<td>DCENR</td>
<td>Department of Communications Energy and Natural Resources</td>
</tr>
<tr>
<td>EC</td>
<td>European Commission</td>
</tr>
<tr>
<td>EPL</td>
<td>Edenderry Power Ltd.</td>
</tr>
<tr>
<td>EU</td>
<td>European Union</td>
</tr>
<tr>
<td>FAO</td>
<td>Food and Agriculture Organisation</td>
</tr>
<tr>
<td>FFF</td>
<td>Forestry for Fibre</td>
</tr>
<tr>
<td>GHG</td>
<td>Greenhouse gas</td>
</tr>
<tr>
<td>GPC</td>
<td>Grant and Premium Category</td>
</tr>
<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
</tr>
<tr>
<td>LCA</td>
<td>Life Cycle Assessment</td>
</tr>
<tr>
<td>LCIA</td>
<td>Life Cycle Impact Assessment</td>
</tr>
<tr>
<td>LCI</td>
<td>Life Cycle Inventory</td>
</tr>
<tr>
<td>LUC</td>
<td>Land-use change</td>
</tr>
<tr>
<td>N₂O</td>
<td>Dinitrogen oxide</td>
</tr>
<tr>
<td>OD/ODT</td>
<td>Oven-dry/ovendry-tonne</td>
</tr>
<tr>
<td>PAG</td>
<td>Peat at the gate</td>
</tr>
<tr>
<td>PC</td>
<td>Peat combustion</td>
</tr>
<tr>
<td>PLUC</td>
<td>Peat land use change</td>
</tr>
<tr>
<td>PP&amp;T</td>
<td>Peat processing and transport</td>
</tr>
<tr>
<td>ReCiPe</td>
<td>A life cycle impact assessment method</td>
</tr>
<tr>
<td>SEAI</td>
<td>Sustainable Energy Authority Ireland</td>
</tr>
<tr>
<td>SCS</td>
<td>Soil carbon stocks</td>
</tr>
<tr>
<td>SE</td>
<td>Standard error of the mean</td>
</tr>
<tr>
<td>SOC</td>
<td>Soil organic carbon</td>
</tr>
<tr>
<td>SOM</td>
<td>Soil organic matter</td>
</tr>
<tr>
<td>SRC</td>
<td>Short Rotation Coppice</td>
</tr>
<tr>
<td>SRF</td>
<td>Short Rotation Forestry</td>
</tr>
<tr>
<td>UNFCCC</td>
<td>United Nations Framework Convention on Climate Change</td>
</tr>
<tr>
<td>YC</td>
<td>Yield Class</td>
</tr>
</tbody>
</table>
List of publications

Clancy M.A., Jovani Sancho A.J., Cummins, T., and Byrne K. A. 2016. The need to
disaggregate podzols and peaty podzols when assessing forest soil carbon stocks, Irish
Forestry 72(1&2), pp. 95-110.
1. Chapter 1 - Introduction
1.1 Background

Limiting global warming and related adverse climate change in the 21st century and beyond, caused by increased anthropogenic greenhouse gas (GHG) emissions to our atmosphere, is becoming increasingly urgent (IPCC 2014b). The recent IPCC “Special Report” (IPCC 2018) addressed the need for immediate action by all governments in order to limit global warming to 1.5 °C above pre-industrial era levels. Failure to act on reducing GHG emissions is expected to lead to escalating the risk of long-term, abrupt, and irreversible changes, e.g., ocean acidification and the melting of polar ice leading to rising sea levels and coastal flooding. Reducing the risks of climate change will require far-reaching, cross-sectoral, upscaling and acceleration of mitigation and adaptation measures (IPCC 2018). Mitigating increasing levels of atmospheric GHGs, to not overshoot a 1.5 °C rise, will require the expedient implementation of a portfolio of measures. Those measures are expected to include lower energy and resource use intensity, transition of land from agriculture to energy crops and forestry, and technologies involving bioenergy and carbon capture and storage (Azar et al. 2010, IPCC 2018).

There is a large body of work on the role of forestry in mitigating GHG emissions and its potential contribution to combating climate change (e.g. Schlamadinger and Marland 1996, Mason et al. 2009, Byrne 2010, Lundmark et al. 2014, Grassi et al. 2017). Within the literature there is also much debate about the most appropriate treatment of forest soils and biomass production regarding soil organic carbon (SOC) stocks, carbon (C) neutrality, and their role in reducing global GHG emissions (e.g. Fargione et al, 2008, Schulze et al. 2012, Ros et al. 2013, Strengers et al. 2015, Ter-Mikaelian et al. 2015, Berndes et al. 2016). The 21st Conference of the Parties (COP21) to the United Nations Framework Convention on Climate Change
UNFCCC) devised aspirational proposals to enhance the SOC stocks of world soils by 0.4 % per year to compensate for anthropogenic GHG emissions (Lal 2016, Minasny et al. 2017). Those COP21 proposals (namely “4 per Thousand”) calls for voluntary plans to promote SOC sequestration (to a depth of 40 cm) through recommended soil management best practices (Lal 2016). Despite the debate around the approach, the pursuit of enhanced soil C sequestration is regarded as a win-win strategy for restoring degraded soils, improved biomass production, and offsetting GHG emissions (Lal 2004a).

There are several common methods of measuring and monitoring forest SOC stocks (hereafter, soil organic carbon density (SOCD)), in use globally. For example, on mineral soils field based repeat sampling of SOC concentration and soil bulk density (BD), the oven-dry mass per unit volume, e.g., g C cm$^{-3}$ soil, is conducted using steel cores or pit excavation methods (Throop et al. 2012, Mehler et al. 2014). In Ireland, SOC stock estimates or emission factors (EFs) for GHG efflux from forest soils have been compiled for organic soils (e.g. peatlands, Byrne and Farrell 2005, Jovani-Sancho et al. 2018), organo-mineral soils (e.g. peaty gleys, Zerva et al. 2005, Lane 2016), and mineral soils (e.g. brown earths and gleys, Black et al. 2009, Wellock et al. 2011, Premrov et al. 2017).

In 2014 the Irish Soil Information System (SIS) completed the country-wide mapping of Irish soils. The associated sampling work provided further SOC concentration and BD data using a soil auger and small volume “core” sample (98 cm$^{3}$) based field survey (Simo et al. 2014). Given the low level of forest coverage in Ireland (approximately 11 %) the SIS campaign understandably concentrated most of their resources on agricultural soils, with only 12 forestry sites sampled out of a total of 246 sites (Creamer pers. comm.). Of the 12 forestry sites sampled only one site provided BD
data due to excessive stoniness, i.e., a high occurrence of rock fragments (RFs) in the soil (Creamer pers. comm.). Therefore to improve the SOCD and related GHG emissions reporting at regional and national level, there are refinements necessary in the methods of sampling and calculating the BD of forest soils with high RF contents. The Irish Environmental Protection Agency (EPA) in a subsequent review of soil status, policy and research needs also noted that forest soils should receive more focus in future Irish soils research (Bampa et al. 2016). In the course of the podzol SOCD study, and through analysis of data from the Irish SIS field campaign, it was apparent that the use of steel cores for measuring soil BD was problematic due to the frequency of large RFs in Irish forest soils. To address the inherent challenges of both acquiring the necessary replicates of BD samples, and accounting for the volume of soil occupied by RFs in these soils when estimating their BD and SOCD, both existing and novel sampling and calculation methods were used in this study. That work was carried out under the direction of the Department of Agriculture, Forestry and the Marine (DAFM) funded ShortFor project.

While sampling and data gathering related to SOCD have been completed for many Irish soil and forest types, some mineral and organo-mineral (peaty) soils, and soils under short rotation forestry (SRF) were under-represented. The DAFM funded CForRep project (“Additions and refinements to the Irish forest carbon accounting and reporting tool”) performed a gap analysis on national soils databases to help target under-represented soils. One of the soil groups highlighted as requiring further research was the podzol group. Due to their leached, nutrient-poor status, podzols and peaty podzols are deemed marginal for agricultural purposes and therefore more suited to forestry (Farrelly and Gallagher 2015b). This study addressed the podzol data gap through the SOC and BD sampling of afforested podzol sites and the development of
SOCD estimates for those soils, and found a significant disparity in the range of SOCD between podzols and peaty podzols.

Renewable energy sources such as biomass for bioenergy offer an alternative to fossil fuels and can assist in mitigating atmospheric GHG emissions (Demirbas et al. 2009). In Ireland, SRF has the potential to increase biomass production for renewable energy, and contribute to the EU Renewable Energy Directive target of 16 % renewable energy and a 20 % reduction in GHG emissions by 2020 (Howley and Holland 2016). In the absence of dedicated SRF plantations the most likely alternative biomass supply for bioenergy in Ireland is from conventional forest (e.g. Sitka spruce) pulpwood and residues. This study also conducted a Life Cycle Assessment (LCA) of the greenhouse gas balance of several SRF biomass supply scenarios, and compared them to two reference systems, i.e., similar Sitka spruce biomass supply scenarios, and a milled-peat scenario.

1.2 Aims and objectives of the thesis

The study was divided into three research chapters (numbers 3, 4, and 5), each with a specific aim and objectives:

I. Chapter 3: To measure soil BD and SOC concentration of afforested podzols with the aim of improving baseline mineral SOCD estimates for Irish forests.

   i. As a result of sampling these soils the need arose for a discussion on disaggregating podzols and peaty podzols, and the respective methodologies employed when undertaking soil C inventories of them.
II. Chapter 4: To assess the influence of soil BD sampling and calculation methods on SOCD estimates. Two soil BD sampling methods, i.e., the core and pit methods, and three BD calculation methods were used in the determination of the SOCD of Irish SRF soils. Ten SRF sites were utilised with a range of soil types and properties. The tasks undertaken to achieve the objectives were as follows:

i. Use of the core method to measure soil BD and SOC concentration to 40 cm depth at all 10 SRF sites.

ii. Use of a novel BD sampling method using glass beads to measure the volume of large excavated pits on seven of the 10 sites, and estimate the soil BD to 40 cm depth. These seven sites had the highest relative volume of RF.

iii. Investigate the impact of three different calculation methods (from Throop et al. 2012) on the BD results, and subsequently derived SOCD.

iv. Compare the core and pit excavation methods for assessing forest soil bulk density.

v. Devise and use a novel method of expediting the estimation of soil BD using a combination of core and quantitative pit samples.

III. Chapter 5: To conduct a Life Cycle Assessment (LCA) of the greenhouse gas balance of Irish Short Rotation Forestry, using *E. nitens* as a representative tree species.

i. Using LCA methodology, assess the cradle-to-grave GHG balance of SRF biomass for bioenergy, using several biomass scenarios.
ii. Use LCA methodology to also assess the GHG balance of Sitka spruce biomass for bioenergy as a reference system, and compare the results to those for SRF biomass.

iii. Compare the results of both the SRF and Sitka spruce biomass LCAs to a fossil fuel (milled peat) reference system, and determine their potential impact on the EU Renewable Energy Directive targets for Ireland, i.e. 16% renewable energy and 20% reduction of GHG emissions, both by 2020.

1.3 Thesis structure

The following thesis is structured in eight chapters, including a list of references, followed by appendices. Chapter 1 presents an overall introduction to the thesis. The literature review, addressing the main concepts involved in this thesis, is presented in Chapter 2. Chapter 3 is a study that estimates the soil organic carbon density of afforested podzol and peaty podzol soils in Ireland. Chapter 4 assesses the influence of soil BD sampling and calculations methods on estimates of the soil organic carbon density of Irish SRF soils. Chapter 5 presents a comprehensive cradle-to-grave Life Cycle Assessment of Irish SRF biomass for bioenergy, and compares it to two reference systems, i.e., conventional Sitka spruce biomass, and also a milled-peat fossil fuel scenario. Chapter 6 discusses the main findings of the thesis, the final conclusions and recommendations for further research. Chapter 7 contains the bibliography of all the reference material used in the thesis. Finally, Chapter 8 presents the appendices referred to throughout the thesis.
2. Chapter 2 - Literature Review
2.1 The global carbon cycle and our changing climate

The global carbon (C) cycle consists of both natural and human-induced fluxes between five major integrated C pools, i.e. Earth’s atmospheric, geologic, oceanic, biotic, and pedologic (soil) systems (Lal 2004a). According to Kuzyakov (2011) “pools” represent the static components and stability of a system, and the “fluxes” account for its dynamics. The global C fluxes are driven by a series of on-going biogeochemical and climate processes that are responsible for the dynamic nature of the C exchanges between the five pools, whereby they act as both sources and sinks of C (Lal 2008).

The Fifth Assessment Report (AR5) of the Intergovernmental Panel on Climate Change (IPCC) states that C turnover between these pools (Figure 2.1), mainly in the form of carbon dioxide (CO$_2$), occurs through both fast and slow moving exchanges (Ciais et al. 2013). Those processes can range in duration from a few years for the atmosphere, to several millennia for soil and geologic fluxes (Ciais et al. 2013). The cycle of C between the different pools and the associated feedbacks were relatively stable for millennia until recent anthropogenic activity began altering the natural world’s C balance (Janzen 2004).

As far back as 1896 the Swedish scientist Svante Arrhenius detected a difference in CO$_2$ concentration between the atmosphere and the ocean’s (Grace 2004). His work later prompted Keeling’s recording of atmospheric CO$_2$ measurements (Keeling and Whorf 2005), which started in 1958 at the Scripps Institution of Oceanography on Mauna Loa island, Hawaii. This early work demonstrated that approximately half of all CO$_2$ from combustion of fossil fuels stays in the atmosphere, which inferred the existence of C sinks on land or in the ocean’s (Grace 2004). From a pre-Industrial Revolution level of 278 ppm the atmospheric concentration of CO$_2$, the most abundant greenhouse gas (GHG), has risen by 45 % to over 400 ppm and is now widely accepted
as the principal driver of climate change (Friedlingstein and Prentice 2010, Ciais et al. 2013).

**Figure 2.1** The global C cycle. The black numbers and arrows represent the pre-Industrialisation Era (c.1750) mass of the C stocks in each reservoir, in PgC (1 Pg = 1 Petagramme = $10^{15}$ g), and annual C exchange fluxes in PgC yr$^{-1}$. The red numbers and arrows indicate the post-industrial era annual anthropogenic perturbation of the C cycle fluxes, averaged over the years 2000–2009. The red arrows represent: The average atmospheric CO$_2$ increase (also called the ‘CO$_2$ growth rate’) due to fossil fuel combustion, cement production, and net land-use change CO$_2$ emissions, plus the net ocean and terrestrial ecosystems uptake of anthropogenic CO$_2$ (Source: Ciais et al. 2013).
2.1.1 Greenhouse gases and associated global warming

Since the start of the Industrial Revolution c.1750 the acceleration of land-use change (LUC) and burning of fossil fuels has led to increasing emissions of CO\(_2\), methane (CH\(_4\)) and nitrous oxide (N\(_2\)O), into Earth’s atmosphere (Figure 2.2) (Friedlingstein and Prentice 2010, Ciais et al. 2013). Together, the atmospheric accumulation of these biogeochemical trace gases, otherwise known as greenhouse gases, account for 80 % of the total “radiative forcing” which has caused an increase in the global mean temperature by 0.8 °C since the late 1800’s (Lal 2008, Ciais et al. 2013). Radiative forcing is the net change between upward and downward irradiance between Earth’s surface and the top of the atmosphere, caused by external drivers of climate change such as increased atmospheric concentration of CO\(_2\) (Ciais et al. 2013). The net increase in downward irradiance responsible for our warming planet has increased since 1750 by 2.3 W m\(^{-2}\) (IPCC 2014b). The IPCC use the global warming potential (GWP) metric to characterise the cumulative radiative forcing resulting from the pulse emission of a unit mass of a greenhouse gas (GHG), e.g., Gt CO\(_2\) (Levasseur et al. 2010). They also use the GWP metrics of different GHGs to compare potential climate change impacts, using CO\(_2\) equivalent (CO\(_2\)eq) emissions as a common scale, over specified periods, e.g., 20 or 100 years, namely GWP\(_{20}\) or GWP\(_{100}\), respectively (IPCC 2007).

According to the AR5, from the 1950’s onwards our climate system is undergoing unequivocal warming that is unprecedented over decades to millennia (IPCC 2014b). The observed consequences of this climate change, such as higher atmospheric and oceanic temperatures along with reduced amounts of snow and ice and rising sea levels, are attributed to the highest levels of anthropogenic GHG emissions in human history (IPCC 2014b). The most significant human sources of GHGs contributing to global warming are the burning of fossil fuels, and intensive
management of land converted from natural grassland and forests for agricultural use (Oelbermann 2010). While climate change is one of humanity’s greatest challenges for this century, there are efforts being made to mitigate the most damaging effects of the existing energy systems reliance on fossil fuels.

Figure 2.2 Atmospheric concentrations of the main GHGs from year 1750–2011 AD. Increases since the start of the Industrial Era c.1750 are attributed to human-induced activities (Source: IPCC 2014b).
2.1.2 International climate change mitigation policy development and the role of afforestation

Since the 1972 Stockholm meeting of Heads of Government organised by the United Nations Environmental Programme, global environmental issues such as LUC, anthropogenically altered C and nitrogen cycles, and associated global warming have raised scientific, public, and political concern. The inception of the IPCC in 1988 and the subsequent 1994 UNFCCC were devised to tackle the escalating issue of atmospheric greenhouse gases “dangerous interference” with Earth’s climate (Grace 2004).

Article 3.3 of the 1997 Kyoto Protocol (KP) to the UNFCCC, deals with net changes in GHG source emissions and sequestration by sinks as a result of direct anthropogenic afforestation, reforestation and deforestation since 1990 (IPCC, 2000). As agreed by the Parties to the KP, only verifiable changes in C stocks in a given commitment-period can be used in measuring C sequestration by Article 3.3 forests (IPCC, 2000). Article 3.4 covers additional anthropogenic activities, e.g. management of C pools through conservation tillage or wood product use, in the land use, land-use change and forestry (LULUCF) sectors (IPCC, 2000).

Following the KP call for countries to report on emissions and sinks due to human-induced LUC and forestry activities, Lund (1999) examined the uses and definitions of several key terms and made nine recommendations to the UNFCCC for standardisation of those terms. Some of the terms reviewed were afforestation, reforestation and deforestation, along with related terms such as land use, land cover, forest, and tree. The Lund (1999) study reported about 240 definitions for these terms in use around the world, reflecting significant differences in biogeophysical states, social
systems, and economies (IPCC 2000). The IPCC (2000) agreed that the terms “land cover” and “land use” are often confused and proposed the following clarification:

*Land cover is "the observed physical and biological cover of the earth's land, as vegetation or man-made features","*

and,

*Land use is “the total of arrangements, activities, and inputs undertaken in a certain land cover type (a set of human actions). The social and economic purposes for which land is managed (e.g., grazing, timber extraction, conservation).”*

The KP, agreed by 191 countries (including all European Union (EU) members), set a target of reducing anthropogenic GHG emissions by at least 5% below their respective 1990 levels between 2008 and 2012 (IPCC 2000). The subsequent Paris Agreement of 2015, adopted by 195 countries at the COP21 of the UNFCCC, set new targets which upon ratification become Nationally Determined Contributions (NDCs). The NDCs set ambitious targets intended to reach a balance between anthropogenic GHG emissions and removals in the second half of this century, and thereby limit global warming to “well below 2 °C” (Grassi et al. 2017). Ireland is a Party to the UNFCCC and to the Paris Agreement and thereby the legally binding agreements necessary to restrict the mean global temperature to less than 2 °C above pre-industrial levels (Ireland, DCENR 2015).

The EU, as part of their proposed climate change mitigation commitments to COP21, have now pledged a 40 % reduction in GHG emissions compared to 1990 levels by 2030 (Ireland, DCENR 2015). The displacement of GHG emissions from fossil fuel combustion with renewable energy sources presents a major opportunity for climate change mitigation through the development of sustainable energy provision (IPCC 2012). The IPCC (2012) have assessed the scientific literature and focused on six
renewable energy sources, i.e., bioenergy, solar, hydropower, geothermal, wind and ocean as part of a portfolio of climate change mitigation options. These renewable energy sources have the potential for integration with, or replacement of, current energy systems. Other options for reducing GHG emissions such as energy efficiency and conservation, carbon capture and storage (CCS), fossil fuel switching, and nuclear power were also assessed (IPCC 2012).

The EU formulated Renewable Energy Directive (RED) 2009/28/EC set a target of delivering 20 % of its energy needs from renewable sources by 2020, with the target for Ireland set at 16 % (Ireland, DAFM 2015a). According to the Irish governments Department of Communications, Energy and Natural Resources (DCENR) the provision of energy from non-renewable sources in Ireland currently stands at 90 %, with a goal of 84% if the binding targets are met by 2020. The Irish RED target is spread across the three main energy sectors with the following individual targets set for each: 40 % of electricity supply, 12 % of heating, and 10 % of transport (Ireland, DCENR 2015).

In tandem with these international agreements, afforestation, the process of converting land previously non-forested for at least 50 years, such as pasture or cropland to forest plantations has been increasing in recent decades and is promoted as a means of sequestering atmospheric CO$_2$ (Houghton et al. 1999, IPCC 2006, Lal, 2008, Berthrong et al. 2009). In 2005 forests covered an estimated 3952 Mha or 30 % of Earths terrestrial surface (Lorenz and Lal 2010, Malmsheimer et al. 2011). During the period 1990 to 2005 the rapid increase in afforestation, mainly in North and Central America, Oceania and Asia, has seen afforested plantations grow by roughly 2.8 Mha y$^{-1}$ to cover about 140 Mha, (FAO 2006a, Berthrong et al. 2009). Globally forests store approximately 45% of terrestrial carbon and therefore are inextricably tied to the C fluxes driving global climate change (Malmsheimer et al. 2011). Even if afforestation
has only a minimal effect on soil C stocks at the regional or country level, its effect on the global C pool could be significant if large scale conversion of agricultural land to plantation continues (Paul et al. 2002).

2.2 Irish forestry development and use of forest biomass in offsetting greenhouse gas emissions

At the founding of the Irish state in the early 1920’s the Irish forest estate occupied just over 1% of the total land area. Due to Irish government policy from the 1950’s onwards, the DAFM reported that by the end of 2012 the total forest area had grown to approximately 10.5%, i.e. 731,650 ha (Ireland, DAFM 2013). Between 1990 and 2009 Ireland added over 250,000 ha of new plantation forest, of which about 50 % is in stands of pure Sitka spruce (Picea sitchensis (Bong.) Carr.) (Black et al. 2009). These forests were planted to support a burgeoning wood processing industry and rural economic development, but also due to their potential to offset CO₂ emissions by tree growth and sequestering of C in forest biomass and soils (Black et al. 2009, Ireland, DAFM 2013). Ireland’s state owned forestry management company “Coillte” founded in 1988 owns over 445,000 ha of land, approximately 7% of Ireland’s land area and around 50 % of the forest estate (Coillte 2013).

When forest biomass is used to produce commercial wood products such as pulp or paper, bioenergy is also produced as a by-product simultaneously (Figure 2.3) (Cowie et al. 2017). The use of forestry in accordance with sustainable forest management principles to mitigate the effects of climate change, through carbon sequestration and replacement of fossil fuels, is one of the main reasons for the Irish government funded “Forestry Programme 2014–2020” (Ireland, DAFM 2015a). This
programme recognises the EC and United Nations Economic Commission for Europe and FAO outlook for European growth in consumption of forest products and wood energy, and concludes that “all components of supply will have to expand, especially harvest residues”.

The forestry programme contains Grant and Premium Categories (GPC) which align with the EU forest strategy to deliver additional forest biomass for both commercial and domestic heat and power generation, and contribute to meeting Ireland’s renewable energy targets. The GPC 12 a & b are the “Forestry for Fibre” (FFF) grants which were introduced in 2015 to promote the establishment of plantations with fast growing tree species, on sites which are generally more fertile than the marginal agricultural land of conventional afforestation (Ireland, DAFM 2015a).

The primary aim of these fast growing, short rotation forestry (SRF) plantations, is to provide wood fibre for use as raw material in wood panel production or for combustion in the bioenergy sector (Ireland, DAFM 2015a). The FFF measure together with grant aid for forest roads will help mobilise small dimension logs from early thinnings and harvest residues for renewable bioenergy use (Ireland, DAFM 2015b). Due to the aforementioned government policies and afforestation incentives, the national harvest of pulpwood and sawlogs is expected to rise from 877 and 2235 M m\(^3\) in 2013 to 1772 and 4195 M m\(^3\) in 2020, respectively (Ireland, DAFM 2015a). The available forestry wood fibre for energy is also forecast to rise from 1 to 1.5 M m\(^3\) over the period 2011-2020 (Murphy et al. 2016).
Figure 2.3 Biomass from forestry operations and residues from wood processing are used to meet internal process energy needs in the forestry and for renewable energy generation as electricity, heat or wood derived fuels. Figure: Sveaskog (Source: Berndes et al. 2016).

2.2.1 Irish conventional Sitka spruce forestry

The General Yield Class (GYC) is an index based on the maximum mean annual increment (MMAI) of cumulative timber volume, measured in units of m$^3$ ha$^{-1}$ yr$^{-1}$. Given its favourable temperate oceanic climate, tree productivity in Ireland can reach exceptional levels compared with the UK, Sweden, Canada and the USA. For example, the average yield class for Irish conifers and broadleaves is 17.7 and 7.8 m$^3$ ha$^{-1}$ yr$^{-1}$, versus 11 and 5 m$^3$ ha$^{-1}$ yr$^{-1}$ respectively in the UK (Farrelly 2010). Conventional Irish plantation forestry is dominated by Sitka spruce, i.e. 52% of the total forest estate in 2012 (Ireland, DAFM 2015a), with productivity ranging from 4 to 34 m$^3$ ha$^{-1}$ yr$^{-1}$.
depending on factors such as soil type, climate regime, and site elevation (Farrelly et al. 2009). This range shows that Irish Sitka spruce growth rates can exceed the maximum GYC index of 24. Following a study of 201 sampled sites the weighted mean GYC of Sitka spruce in Ireland was found to be 22 (21.2 m$^3$ ha$^{-1}$ yr$^{-1}$) (Farrelly et al. 2011).

To maximise the commercial return the mean rotation age for pre-1990 planted Sitka spruce was 40–60 years (UNFCCC 2011), with a forecasted mean rotation length for the years 2008–2020 of 41 years (COFORD 2009). Increases above 18 m$^3$ ha$^{-1}$ yr$^{-1}$ in the predicted yield class for Sitka spruce have been linked to the improvement in the quality of afforested land after 1990 (Dhubháin et al. 2006). The improved growth rate has led to general practice within Coillte and most private forest owners to clearfell their plantations after 37 years (80 % of the age of MMAI for GYC of 24), with forestry investment companies more recently advising further reductions in rotation lengths to 30 years (Dhubháin et al. 2006). According to the Irish Forestry Service the earliest scheduled clearfell age for Sitka spruce is at 27 years old (Ireland, DAFM 2015b).

### 2.2.2 Short Rotation Forestry and other bioenergy crops

Growing short rotation woody crops for fibre and energy uses has its origins in antiquity, but the current concept of close-spaced, fast growing (1–15 year rotations) silviculture evolved scientifically via tree breeding programs in the early 20th century (Dickmann 2006). As early as 1935 trials of SRF for fibre production using hybrid poplar clones in high density spacing were undertaken in the US (Schreiner 1970). By the late 1960s the replacement of conventional long-rotation wood production by intensively managed “silage sycamore” plantations (McAlpine et al. 1966) were proposed, and use of “mini-rotation forestry” on farmland to meet an expected deficit in fibre and timber was predicted (Schreiner 1970). The 1970s oil-crisis spurred
government interest and academic research in alternate energy sources such as woody biomass from SRF for fuel, as a means of mitigating energy security risks (Steinbeck 1999, Hardcastle et al. 2006).

Short rotation forestry is a form of forest management designed to maximise wood biomass production over rotation lengths much shorter than the 40+ years of conventional forestry. Christersson and Verma (2006) define SRF as a sustainable silvicultural practice utilising high density planting of fast growing tree species, with rotation lengths of less than 30 years, and established on former agricultural land or fertile but degraded forest sites. In short rotation plantations trees are grown either as single stem (hereafter referred to as SRF), or as coppice (hereafter referred to as SRC) systems (Wickham et al. 2010), with expected yearly wood production of more than 25 m³ or 10 t (DM) ha⁻¹ (Christersson and Verma 2006). The genera Salix (willow) and Populus (poplar) are two of the most common commercially grown SRC crops used for heat and power (Evans et al. 2007, Wickham et al. 2010). In SRF plantations tree species such as alder, birch, black locust, sycamore, and varieties of eucalyptus that do not coppice well, e.g., E. nitens, have been trialled in many European countries as well as the Americas, Africa, Asia, Australia and New Zealand (Neenan and Lyons 1980, Steinbeck 1999, Hardcastle et al. 2006, Sochacki et al. 2007, Jack and Hall 2009, Pleguezuelo et al. 2015).

Rotation lengths for SRF are typically between 8–20 years depending on species growth rates (Hardcastle et al. 2006). The objective of the FFF scheme is to address a forecasted shortfall in the supply of fibre for the energy and wood products sectors by incentivising the establishment of Italian alder, hybrid aspen, poplar, and eucalyptus plantations (Teagasc 2016). The scheme aims to achieve its goals using single stem SRF tree species to produce 150–300 m³ ha⁻¹ of wood biomass over multiple 10–15 year
rotations (Teagasc 2016). In contrast, SRC systems use shorter rotations of 2–5 years for willow and poplar (Evans et al. 2007, Wickham et al. 2010), or alternatively slightly longer 3 × 7, 7 × 3, or 3 × 10 year rotations with poplar and eucalyptus (Gabrielle et al. 2013, Schweier et al. 2017). Usually, after 15–30 years of careful management the SRC stools need to be replaced to maintain crop productivity (Rowe et al. 2009, Schweier et al. 2016).

According to Cannell (1980) biomass yields are independent of planting density for fully stocked vigorous hardwood plantations, with 6–8 t ha\(^{-1}\) yr\(^{-1}\) obtainable over 1–25 years with 250,000 or 2,000 trees ha\(^{-1}\). Site quality and management practice determine mean productivity, but the piece size and rotation length needed to attain the site carrying capacity is largely governed by planting density (Bernardo et al. 1998, Johnson et al. 2007). Short rotation coppice systems typically use planting densities of between 5,000–20,000 stools ha\(^{-1}\) (Dickmann 2006, Rowe et al. 2009), with some as low as 1,250 (Gabrielle et al. 2013) or others at > 25,000 (Wickham et al. 2010). In SRF systems, where hardwood genus such as *Eucalyptus*, *Alnus* (alder), and *Acer* (sycamore) are commonly grown to large stature, planting densities are generally between of 700–5,000 plants ha\(^{-1}\) (Johnson et al. 2007, Bogdan et al. 2009, González-Garcia et al. 2016). The Irish FFF scheme stipulates a minimum of 2,000 plants ha\(^{-1}\) (Teagasc 2016).

Given the increasing demands on agricultural land to satisfy food supply and security needs (Smith et al. 2013), combined with the uncertainties associated with climate change, the choice of tree species, site conditions (e.g. local climate and soil type) for SRF in Ireland are critical to its commercial success as a source of biomass for bioenergy (Thompson et al. 2012). The reported average yields (28 m\(^{3}\) ha\(^{-1}\) yr\(^{-1}\) on a rotation length of 12 to 15 years) for Irish SRF (*E. nitens*) plantations, managed using

23
SRF methods, make it a potential source of high volume biomass for bioenergy (Thompson et al. 2012). Irish grown Sitka spruce and E. nitens have proven to be high-yielding sources of biomass (Farrelly 2010, Thompson et al. 2012). Given the prevalence of Sitka spruce in Irish forestry it is also an existing source of biomass for bioenergy (COFORD 2017b, SEAI 2017). In contrast, there are only a small number of E. nitens trials established by Coillte and Teagasc ongoing in Ireland (Coates, pers. comm.), with the ultimate aim of producing commercial plantations to supply the Irish board industry and biomass for bioenergy (Purse and Leslie 2016).

2.2.3 Irish forestry yield modelling

The easiest way of modelling possible future scenarios is to assume that the future is similar to the present and work with known product data, or it may be more appropriate to design new models that elaborate potential future scenarios (Finnveden et al. 2009). For many years Irish forest managers relied on yield model tables developed by the UK Forestry Commission (Edwards and Christie 1981) for predicting forest growth and future timber production (Purser and Lynch 2012). The development of yield models based on Irish research data (e.g., Broad and Lynch 2006) has been ongoing since 2001 (COFORD 2017).

To date the most commercially important tree species for Irish producers, e.g., Sitka spruce, lodgepole pine, Norway spruce, Douglas fir, have been modelled. These yield models enable forecasting of future forest yields, and are available to the forestry sector through a computer software tool called GROWFOR. This interactive tool facilitates dynamic yield modelling and both operational and financial planning based on parameters such as, tree species, stocking levels (stems ha$^{-1}$) top height (m), DBH (cm). The tool can also factor in the effects of different thinning regimes over varied
rotations lengths, which can aid in planning of forest road construction and harvest operations (Purser and Lynch 2012). Unfortunately no such yield modelling tables or tools are available for potential Irish SRF tree species, and therefore current research relies on less comprehensive data from the literature.

2.3 The role of forest soils in climate change mitigation

Emissions or removals of GHG’s per hectare of forest can vary depending on site variables such as plantation age and type, soil type, tree species planted, and management practices (IPCC 2006). Examples of plantation types could be coniferous, broadleaf or mixed, and soil types could be organic, mineral or organo-mineral with several possible sub-divisions into soil taxonomic groups. To comply with the terms of the UNFCCC it is mandatory to report on the soil C pool, therefore stratification of forest characteristics at regional or sub-regional level within a country and measurement of associated soil C stocks and emissions is required to support national GHG inventory reporting (IPCC 2006).

2.3.1 Soil organic matter and soil carbon pools

Soil is generally known as the provider of nutrition and water to support agriculture and many diverse terrestrial ecosystems, but is less well recognised for its fundamental role in climate change, ground water recharge, and waste management, and as a provider of engineering and construction raw materials (Wilding and Lin 2006). The pedosphere is the shallow, porous, terrestrial surface layer of our planet that acts as an interface between our atmosphere, lithosphere, hydrosphere and biosphere, where soil forming
processes take place (Jackson 1997, Wilding and Lin 2006). Jenny (1941) identified parent material, i.e., weatherable bed-rock or mineral deposits (Subke et al. 2009), topography, climate, biota and time as soil forming factors. More recently the influence of human activity on geomorphic and pedologic processes through ploughing, irrigation, fertilization and deep working has become widely recognised as another major soil forming factor (Amundson and Jenny 1991, Bockheim and Gennadiyev 2000, Certini and Scalenghe 2011). The profound modification of the properties of some soils by human activity, called “anthrolization”, has also led to the establishment of a new soil group called Anthrosols in several of the world’s modern soil classification systems, e.g., the IUSS (2007) “World Reference Base for Soil Resources” (WRB).

The six soil forming factors interact through a combination of addition, loss, transformation, and translocation processes, known collectively as pedogenesis, to produce a soil system (Simonson 1959, Targulian and Krasilnikov 2007). Our understanding of the components and formation of soils, and in particular the concept of soil organic matter (SOM), has evolved significantly over the past 200 years (Manlay 2007). The FAO (2006b) define SOM as the decomposed, non-decomposed, and partially decomposed organic matter derived from non-living flora and fauna. In many agricultural studies up to the 1840’s soil humus was equated with SOM, but due to interdisciplinary mineral and ecological research since then, humus is now understood to represent only the well decomposed fraction of SOM, unrecognisable from its plant or animal origins (Amundson 2001, Manlay 2007). Now SOM is increasingly understood as a multifaceted bio-organo-mineral system (Manlay 2007) comprised of many different chemical compounds with varying degrees of physical accessibility (Rovira and Vellejo 2007).
Furthermore SOM is generally divided into light, medium, and heavy fractions which are also considered respectively to be labile, stable, and inert (Strosser 2007). The light fraction of SOM is characterized as free, the medium fraction is slightly occluded in organo-mineral aggregates, and the heavy fraction is tightly bound to minerals (Alvarez et al. 1998, Strosser 2007). Large accumulation of light fraction SOM can occur under permanent vegetation, e.g. forests or grassland, with its quantity and turnover rates (from a few months to a few years) significantly affected by harvesting and tillage management practices (Post et al. 2001). The heavy fraction, which is less susceptible to decomposition by soil organisms, makes up the bulk of SOM with turnover times ranging between decades and millennia (Post et al. 2001).

Many soil properties, e.g., soil pH, aeration, cation exchange capacity, water retention and filtration, and soil fauna activity are affected by SOM (Baldock and Skjemstad 2000, Palmer et al. 2002, Krull et al. 2004). The abundance or absence of SOM gives a strong indication of land fertility and degradation respectively (Manlay 2007). Soil structural properties and its main biogeochemical cycles, including storage or release of GHGs, are also closely controlled by SOM (Manlay 2007).

Soils are our largest terrestrial C pool (Figure 2.1.1), estimated to hold in total between 1500–2400 Pg C (1 Pg = 1 × 10^{15} g) to a soil depth of 1 m (excluding the surface organic horizon), which is approximately twice the size of the atmospheric C pool and three times the biotic C pool (Eswaran et al. 1993, Batjes 1996; Petrokofsky et al. 2012). The value for total C is comprised of approximately two thirds (1550 Pg) of soil organic carbon (SOC) and one third (750 Pg) soil inorganic carbon (SIC) (Batjes 1996, Lal 2008, Rawlins 2011).

Though humus and SOM have been studied for centuries, present day interest in soil C sequestration has intensified the need for accurate measurement of SOC as a
percentage of SOM. As SOM is composed of C, H, O, N, P and S, actually measuring SOM is difficult and most analytical methods work out the SOC content and estimate SOM by using a conversion factor (Krull et al. 2004). In a critical review of the conventional “van Bemmelen” conversion factor of 1.724, which dates from the early 19\textsuperscript{th} century and assumes that SOM contains 58\% C, Pribyl (2010) found that subsequent research suggests a median value of 1.9 and a theoretical value of 2 (i.e. 50 \% C) are more accurate. Pribyl’s study also points out that accurate measurement of SOC should be determined directly, and that any values estimated using a conversion factor that is not a universal physical constant could potentially have serious errors.

The process of plant photosynthesis takes CO\textsubscript{2} from the atmosphere and converts it to the energy required to grow biomass in roots, stems, branches and leaves. The SOC pool is added to by inputs from aboveground and belowground litterfall, understory sward senescence, belowground root senescence, and rhizodeposition of root exudates (Jandl et al. 2007, De Vliegher and Carlier 2007, Laganière et al. 2010). The main contributors to soil C loss through CO\textsubscript{2} efflux are the respiration of soil organisms feeding on SOM, and rhizodeposition derived root and microbial respiration (Sulzman et al. 2005, Jandl et al. 2007, Kuzyakov and Gavrichkova 2010). SIC is found mainly in carbonate minerals, e.g., calcium carbonate (CaCO\textsubscript{3}) and dolomite (MgCO\textsubscript{3}), with larger SIC concentrations (e.g., mg C g\textsuperscript{-1} soil, often expressed as a percentage or mass per unit area, e.g., g C m\textsuperscript{-2} soil) found in areas with arid soils or those formed over calcareous parent materials. Acidic or strongly weathered soils generally contain less SIC due to the originally present carbonates having been dissolved (Batjes 1996).

Through long-term pedogenic processes organic and inorganic C can also be transported down the soil profile into sediments where it can remain sequestered for millennia (Lal 2005). The net balance between soil C inputs and releases determines
whether the pool acts as a sink or a source (Laganière et al. 2010). Hence soils play a very significant role in the global C cycle due to their potential to accumulate C through both SOM and mineral inputs (Post and Kwon 2000). On a global scale there is considerable potential for C sequestration in forested and cultivated soils, possibly offsetting a significant amount of CO$_2$ emissions to our atmosphere (Sampson et al. 2000, Conant et al. 2003). Given that 1 Gt of the soil C pool is equivalent to 0.47 ppm of atmospheric CO$_2$, even relatively small changes to the soil C pool can translate into substantial changes in the atmospheric pool, further highlighting the importance of soil C in the global C cycle and efforts at climate change mitigation (Lal 2016).

2.3.2 Land-use change, site preparation and disturbances

The large total ecosystem organic and inorganic C pool and its environment are in dynamic equilibrium, and ecosystem disturbances such as those due to LUC influence C fluxes and SOCD (Lal 2005, Schrumpf et al. 2011). With LUC significant re-allocation of land and related C pools occurs, with forests and grasslands gaining C in the process, though this is mostly offset by losses due to expanding cropland, settlements and associated infrastructure (Schulze et al. 2010). In the 20 years prior to 2010, Schulze et al. (2010) reported that LUC across Europe was estimated to have created a small C sink of 9–10 Tg yr$^{-1}$, but that it was the effects of land-use intensity rather than LUC that had the greater impact on the atmospheric GHG composition.

During any LUC the soil C pool is altered by changes in the type of plant cover and level of growth, biomass removal through harvesting and mechanical soil preparation (Schrumpf et al. 2011). Conant et al. (2003) state that net losses of soil C due to increased soil erosion, decomposition, or loss of productive capacity brought about by LUC over 30 years before present, can have a significant and persistent effect
on soil C. In particular the conversion of native vegetation to cropland can result in SOC stock being reduced by 25–30% (Post and Kwon 2000, Guo and Gifford 2002, Houghton et al. 2012).

By changing land use from agriculture to afforestation the soil C pool equilibrium can be greatly disturbed by both biotic and abiotic factors, and due to the relatively long mean residence time of soil C the resulting disequilibrium may last for decades (Polglase et al. 2000). Establishing forest plantations on what was previously pasture or cropland significantly alters C cycling between vegetation and the soil by modifying the quantity and quality of fresh soil C inputs and outputs from decomposition (Polglase et al. 2000, Paul et al. 2002). Soils (to a depth of 1 m) in temperate forests store between 153 and 195 Pg C on an estimated area between 920 and 1600 Mha (Lorenz and Lal 2010).

During the afforestation process land often undergoes major mechanical soil preparation in the form of mounding or ploughing to various depths to promote rapid establishment of the plantation (Turner and Lambert 2000). Soil fertilisation, scarification, digging drains and subsoiling prior to planting are also common and are done to improve overall site productivity, drainage, and the penetration of the tree seedling roots in the soil (IPCC 2006, Lof et al. 2012). Forest plantations are also subject to various management practices, e.g., selection of tree species, rotation length, thinning and harvesting, which may also affect SOCD (IPCC 2006). Digging drains and the subsequent lowering of the water table prior to afforestation, particularly on organic soils such as peatlands, which, in their pristine state are C sinks (Byrne and Farrell, 2000) can lead to reduced soil C stocks (Byrne and Milne 2006, IPCC 2006). These disturbances lead to increased soil aeration and higher microbial decomposition rates of surface litter and SOM as measured by soil CO₂ effluxes from autotrophic and
heterotrophic respiration (Byrne and Farrell 2005, Sulzman et al. 2005). Paul et al. (2002) in a review of global data on changes in soil C after afforestation, found that in the first five years following plantation SOC often decreased due mainly to site preparation activities, and absence of significant plant growth resulting in low C inputs. The C cycle includes SOC stock changes due to both continuous growth and decay processes and discrete events, e.g., disturbances from harvesting, fire and insect outbreaks, and LUC (IPCC 2006). Those continuous processes can impact C stocks in all areas annually, while discrete disturbances produce emissions and redistribute ecosystem C in the area where the disturbance occurs, in the year of the disturbance, but may also have longer-lasting effects like decay of burnt or wind-blown trees (IPCC 2006).

2.3.3 Forest soil organic carbon change

The main driver of changes in the balance and allocation of C in growing forest plantations is the quantity and temporal changes of net primary productivity (NPP, the net quantity of C captured by trees and distributed to its components) and decomposition rates (Figure 2.4) (Polglase et al. 2000, von Lützow et al. 2006). Poeplau et al. (2011) state that with LUC from forests to grasslands to croplands there is an increasing average land-use intensity and increasing harvested fraction of the NPP, e.g., 12 ± 3%, 29 ± 8% and 47 ± 15%, respectively, for Europe (Schulze et al. 2010), therefore forests leave a larger C fraction than croplands to increase and maintain SOCD.

In a review of 95 previous studies, covering 322 sites, Poeplau et al. (2011) found that in most studies, the temporal dynamic was ignored and the impact of LUC on SOC is estimated by the net loss or gain of SOC in equilibrium after LUC (Post &
Kwon 2000, Guo and Gifford 2002, Berthrong et al. 2009, Laganiere et al. 2010). Poeplau et al. (2011) also state that SOC stock changes do not happen immediately after LUC but over a period of years to decades. A period of 20 years is assumed by the IPCC for a new C steady state to occur following LUC (IPCC 2006).

The inevitable changes that occur in the timing, quality, amount, and spatial distribution of SOC inputs following afforestation, along with the changes in the soil microenvironment, affect decomposition rates. Processes that regulate the soil C balance such as NPP of vegetation and decomposition of SOM, which are influenced by climatic conditions (e.g., temperature and CO₂ concentration), are likely to drive changes in the quantity of soil C in response to climatic warming (Palosuo et al. 2005). SOC decomposition rates increase due to higher soil temperature, moisture and aeration, or structural and chemical changes in SOM (Schlesinger 1977, Baritz et al. 2010). In contrast, Polglase et al. (2000) report that decomposition rates may decrease under the forest canopy and litter layer due to the cooler soil surface, and decomposition of soil C added via rhizodeposition may decrease due to lower soil temperatures and degraded microbial activity.

The substantial spatial variability of soil C stocks are also related to the spatial variability of soil C inputs and outputs under the influence of natural factors such as climate, parent material, vegetation, topography and human-induced factors, e.g. land use and management practices (Conant and Paustian 2002, Tan et al. 2004, von Lützow et al. 2006).
Figure 2.4 Components of C balance and allocation to soil C after afforestation.

1 = Quantities and temporal changes in gross and net primary production (GNPP)
2 = Aggradation of C in forest litter and rate of decomposition
3 = Allocation of NPP, in particular to fine roots
4 = Root lifespan and soil inputs from root exudates
5 = Rate of root decomposition to soil organic matter

Due to the many biogeochemical processes involved in SOC cycling and the complex physical structure of soil, C substrate sources are heterogeneously distributed throughout the soil profile and their availability at any given point goes through temporal changes (Subke et al. 2009, Gruneberg et al. 2010). As a result of these SOC dynamics the spatial pattern of SOCD is still not well understood (Gruneberg et al. 2010). According to Bradford et al. (2010) forest ecosystems are spatially “notoriously heterogeneous”, and the influence and relative importance of the contributing factors
can vary spatially across multiple scales (from plots and stands to landscapes and regions), making it very difficult to accurately estimate SOCD (Gruneberg et al. 2010).

2.4 Soil sampling and methods of measuring soil organic carbon

2.4.1 Sampling design challenges and considerations

To understand current trends and support climate change mitigation policy development, SOC inventories are required to determine estimated SOC stock changes at regional to national scales (Ogle et al. 2010, Mishra et al. 2012). Sampling of a total population involves taking measurements from a select subset of individuals in order to estimate the properties (or parameters) of the total population (Pennock et al. 2006). In any realistic soil science field research program sampling is necessary because of the impossibility of measuring the total population (Pennock et al. 2006).

In order to reduce the SOC sampling effort some theoretical deliberations may help, such as pre-stratifying of target sites (Heim et al. 2009), e.g., by soil groups (e.g., histosols, gleysols, podzols) and tree species. In a forest growth model designed to predict changes in C stocks, Peltoniemi et al. (2007) used stratification by stand age and standard forest management practices to reduce the standard error of the stratified mean relative to random sampling.

Soil sampling also requires taking account of the geomorphic characteristics of the landscape, e.g., landform type and position, parent material distribution, slope and drainage, and their associated catenary relationships, then gathering and analysing sufficient soil samples to reach acceptable accuracy of measurements at lowest cost (Burt 2004). Generally, precision of estimated regional or national SOC inventories
values (Mg C ha\(^{-1}\)) are increased (i.e. smaller confidence ranges) with increased sampling (IPCC 2006), but there is broad agreement that direct measurement of SOCD, and particularly the associated sampling of soil BD is a laborious, time-consuming, and costly job (Palmer et al. 2002, Conant et al. 2003, Peltoniemi et al. 2004, Smith, 2004a, Don et al. 2007, Weindorf and Zhu 2010). Conversely, it is difficult to scale sample results from a few statistically unrepresentative sites to regional or national levels (Peltoniemi et al. 2004).

In hypothesis-testing experiments, questions of scale are strongly tied to the issue of replication, and in a mensurative study, repeated, unbiased sampling of a population element constitutes replication (Pennock 2004). Replicated measurements are used to determine the estimate of experimental error, to improve the precision of results by reducing the standard deviation, to enlarge the scope of inference, and to control the error variance (Steel et al. 1997). Therefore designing a sampling plan that is representative of the inherent variability and true mean value of a forests SOC stock may require a large number of sampling points. Determining the appropriate sample size requires achieving a balance between too large a number that may lead to inefficient use of manpower and money, and too small a number that causes erroneous statistical test results (Belanger and Van Rees 2006).

According to Bradford et al. (2010) studies attempting to meet the challenges of assessing soil C spatial heterogeneity are commonly grouped into three complementary categories: a) plot-level biometric methods of measuring C pools and/or fluxes (Botkin et al. 1993), b) using micrometeorological towers for continuous monitoring of the entire ecosystems C balance (Baldocchi et al. 2001), and c) analysing imagery generated by remote sensing apparatus (Schimel 1995, Turner et al. 2003).
Bradford et al. (2008) conducted an insightful “lessons learned” review of the practical challenges of designing and undertaking landscape-scale measurement of C pools and fluxes across three heterogeneous forest landscapes. The cost and benefit characteristics of measuring C related processes, and the advantages and limitations of their sampling plot layout were examined.

In drawing upon their experience at each site they identified the following five points as central to potentially improving future sampling design:

1. Standardising the sampling design and plot layout across all study sites facilitates straightforward comparison of results between sites and previous research data using the same approach.

2. Where there is heterogeneity of landscape cover types or topography, prior stratification of the study area into a sampling approach that includes all important elements of the landscape is advisable. Also, no single approach to sampling C pools and fluxes may be useful for all sample plots.

3. Careful consideration of plot layouts is crucial to efficient representation of the landscape, while minimising bias. For example measuring the effect of short-term processes and changes may require less subplots/plot to achieve the required accuracy with less use of resources.

4. Evaluation of sampling design and intensity should be done with spatial and temporal variability in mind, and the impact on the total C storage and balance of each process to be measured. While balancing the need for comparison of data across sites, sampling efficiency requires avoidance of a one-size-fits-all strategy which may lead to oversampling elements of the landscape that exhibit low variance.
5. Consider the sampling design in terms of compatibility with other data gathering approaches. For example, work with models/modellers prior to starting measurements to ensure comparable processes are measured or gathering data at scales that are suited to ground truthing of remote sensing imagery.

2.4.2 Soil carbon stocks and minimum detectable difference

Detecting changes in soil C stocks brought about by changes in land use or management practices (as required by international accords such as the KP) requires taking large numbers of samples and precise measurements to find relatively small changes in large, spatially and temporally varied, soil C stocks (Palmer et al. 2002, Conen et al. 2003, Muukkonen 2009). According to Conant et al. (2001) average changes in soil C sequestration rates are small relative to the total mass of C in the soil, e.g., 0.5 Mg C ha\(^{-1}\) yr\(^{-1}\) versus 50.3 Mg C ha\(^{-1}\) in top 0.3 m. Given such a small signal to noise ratio in SOC stock changes, soil sampling with sufficient replication to detect small changes relative to the total amount of SOC is the only way of directly monitoring the changes due to land use and management practices (Ellert et al. 2006, VandenBygaart 2006).

One tool used for sampling purposes to measure SOC variability is the minimum detectable difference (MDD), studies of which found a broad range (i.e., 5 to 306 observations) used in calculation values (VandenBygaart 2006). VandenBygaart (2006) defined MDD, for temporal SOC change, as the smallest difference that can be detected between two time periods at a given level of statistical significance. Additionally, VandenBygaart (2006) warn that due to site heterogeneity of SOC, using a common MDD for designing a universal sampling plan is questionable.
Alternatively, Garten and Wullschleger (1999) defined the minimum detectable difference (MDD) for SOC stock measurements as the smallest mean value detectable between samples after the variation, significance, statistical power, and sample size are calculated. Their analysis demonstrated that a difference of about 50 mg SOC cm$^{-2}$ or 5 Mg SOC ha$^{-1}$, between 10–15% of existing SOC, is detectable with a sample sizes of n = 16, and a statistical power of 0.90. They further state that the lowest detectable difference in SOCD, 2–3%, is only achievable with sample sizes of n >100, equal to 10 mg SOC cm$^{-2}$ or 1 Mg SOC ha$^{-1}$. Therefore to work out the MDD of the most variable of soil properties, such as SOC, it is necessary to analyse a large number of samples. In a review of the large “CarboEurope” soil inventory dataset, Schrumpf et al. (2011) found that cropland sites, which generally have less spatial variability, had lower MDD with 100 sampling points (105 ± 28 g C m$^{-2}$ for the top 10 cm of the soil) than grassland and forest sites (206 ± 64 and 246 ± 64 g C m$^{-2}$ for 0–10 cm) respectively.

2.4.3 Soil carbon sampling methods

There are several sampling methodologies covered in the literature, e.g., repeat sampling, paired plots, chronosequence studies, and process and modelling studies, which aim to define the soil C dynamics under different sets of conditions, such as land use/soil group classes. These methodologies are designed to measure the net effect of temporal and spatial variables, along with climate differences and land management, have on soil C stocks within a site, region or country at a point in time, or help forecast the impact of future LUC (Turner and Lambert 2000, Scott et al. 2002).

Given that current remote sensing technology cannot penetrate soil surfaces, deriving regional or national SOC inventories requires careful and accurate field-based measurements of SOC (Throop et al. 2012). To calculate a specific sites soil SOC stock
requires the measurement of the mean SOC concentration and the mean soil BD, plus the fine earth fraction (<2 mm), soil depth, and stone contents (>2 mm), which all vary spatially and have different associated measurement errors (Schrumpf et al. 2011). Direct and spatially-intensive soil BD measurements are also critical in order to convert SOC concentration measurements to an areal basis (i.e. units of area or volume such as g C m\(^{-2}\) soil (Don et al. 2011).

### 2.4.4 Soil organic carbon sampling by horizon versus depth

Once the SOC stock variability due to natural and anthropogenic causes have been considered another source of variability in SOC measurements can arise from the choice of sampling, i.e. by soil profile horizon (Figure 2.5) or by depth increments (Gruneberg et al. 2010). Whether to measure SOCD by horizon or depth increments is an ongoing discussion in the literature, with only a few studies comparing the two sampling methods. On average 50 % of the total SOC stock is stored below 0.2 m in forest mineral subsoil horizons (Jobbagy and Jackson 2000). Overall to date, the consequences of the two different sampling methods for regional SOC stock variability in forest soils is not well known (Gruneberg et al. 2010).
Figure 2.5 A generic soil profile with layered organic and mineral horizons (Source: University of Miami, 2013). NB: The above diagram gives an overview of several of the more common organic (e.g. Oi), and mineral (e.g. B) master horizons and transitional horizons (e.g., BC) found in soil profiles. In field studies various combinations and/or subsets of these horizons, with different organic or mineral characteristics and other associated FAO (2006b) horizon designations than those shown above, may be found at varying depths in the soil profile.

Gruneberg et al. (2010) report that variability in soil BD, SOC concentrations and SOCD of soil groups are greater if horizons rather than depth increments are considered. They also state that when soil pedogenic processes are studied in the context of C storage, then horizon sampling is beneficial, as the unavoidable mixing that occurs with depth increments is likely to ruin important pedogenetical information.
Palmer et al. (2002) also highlighted the potential consequences of mixing different horizon SOC concentrations by using depth increment sampling. They found this could lead to lower reported SOC concentrations in mineral A horizons than found in the upper more organic soil, where only the top 5 cm of the mineral soil is sampled and the A horizons are thicker (e.g., 12 cm).

In a study of SOC lateral and vertical variability in several different Canadian soil groups by VandenBygaart et al. (2007), the aim was to ascertain the effects of sampling by pedogenetic horizon versus depth increment on the precision of SOC measurements. They report that sampling by horizon reduced SOC stock variability in the top 30 cm of ploughed gleysols, and thereby linked horizon SOC stock variability to soil horizon thickness variability. This led Gruneberg et al. (2010) to conclude that the variability of horizon thickness is crucial in estimating SOCD. Though alternatively, when horizons with very different SOC concentrations were mixed by sampling in 10 cm increments instead of by horizon (where average A horizon depth was 7 – 10 cm), the effort to detect SOC stock changes was considerably reduced (Gruneberg et al. 2010).

2.4.5 Determining carbon stock change through repeat sampling

In order to determine if soils are a source or sink for atmospheric CO$_2$, direct measurement of soil C stocks through repeat sampling of the same area over a period of time is commonly used, this is also called the “stock change” method (Smith 2004a, Gruneberg et al. 2010). Smith (2004b) states that in planning experiments to detect verifiable changes in SOC content, as used in KP related reporting, it is necessary to know how many samples are needed (e.g., MDD), but also how long before a change is detectable. His SOC dynamics modelling results, based on changes in NPP, land uses
and soil types, showed detection of SOC content changes may take 1–2 years where NPP has doubled, and over 40 years where NPP has only increased by 10 %.

In a study of sampling design efficacy Conant et al. (2003) found that consistent intra-microplot differences at all study sites enabled detection of modest changes in forest soil C, but only if the same microplots were intensively and precisely resampled. According to Conant et al. (2003), in the absence of new and more sophisticated in-situ methods of soil C analysis with greater sensitivity to detecting recent C additions over shorter time periods, we are reliant on intensive sampling plans that enable repeat sampling of the same area.

### 2.4.6 Paired plot sampling

The paired plot approach to soil C sampling aims to quantify the changes in soil C stocks arising from LULUCF, e.g., from non-forest to afforestation (Turner and Lambert 2000). In studying forest soil C stock changes paired plots consist of an afforested or reforested site and an adjacent non-forest site with similar prior land use, soil group, topography, and climate characteristics. Paired plot site selection criteria for forest soil C stock change measurements also take into account the forest type (broadleaf, conifer, mixed) and stand age characteristics (Wellock et al. 2011). The MDD associated with spatial or temporal variability of study sites can be determined by using paired plots (Garten and Wullschleger 1999).

To measure and compare the effects of LUC on soil C, Scott et al. (1999) in a New Zealand based study dug five large pits to a depth of at least 2 m in pine forest and five more in an adjacent pasture site with an equivalent soil group. Soil cores were extracted and bulked at three depth increments (i.e. 0 - 0.1, 0.1 - 0.2, 0.2 - 0.5 m) from three different pit faces to measure the soil C stock at each depth increment. By
comparing the values from the forest and non-forest plots they could then quantify the soil C stock losses or gains associated with afforestation.

2.4.7 Chronosequence sampling

The chronosequence sampling approach uses plots with incrementally increasing stand ages to determine the temporal effects of afforestation on forest soil C stocks. Regional and national C inventories need a greater understanding of the C dynamics of both the forest vegetation and soil, particularly as significant changes in forest C stocks are affected by distribution of age classes (Peltoniemi et al. 2004).

The data obtained from chronosequence studies provide a means of estimating the C stock development in a space-for-time substitution, and if subsequent research can reduce or eliminate sampling and measurement uncertainties then long-term experiments may provide more accurate forest C dynamics data (Peltoniemi et al. 2004). One significant drawback of the chronosequence approach is the inability to distinguish the effects of land use management practices at time of treatment, from the effects of time since the treatment (Yanai et al. 2000).

2.5 Podzols, peaty podzols, and podzolisation

Podzols are a well-defined “Reference Soil Group” in the WRB, but they are also known as podosols in Australia, spodosols in the USA and China, and espodossolos in the Brazilian soil taxonomies (IUSS 2007, Soil Survey Staff 2010). The term podzol comes from the Russian words pod meaning underneath, and zola meaning ash (IUSS 2007). Podzols are primarily conditioned by percolating rainwater in a temperate climate, and have soil horizon profiles heavily influenced by iron (Fe) and/or aluminium (Al) chemistry (Spaargaren 2008). Worldwide they cover approximately 485 Mha, and
are found mainly in the temperate and boreal zones of the Northern Hemisphere, with a much smaller proportion (10 Mha) found in the tropics (IUSS 2007).

The mineral parent materials are formed mostly from weathering of siliceous rock, including glacial till and alluvial and aeolian deposits of quartzite sands, which accounts for their weakly aggregated structure and moderate to well-drained sandy texture (IUSS 2007, Spaargaren 2003). In Ireland, podzols occupy about 8% of the land and are most often located in hilly and mountainous areas at altitudes above 150 m OD, where rainfall plays a significant part in their development. Due to their topographical location and associated issues of accessibility they are generally found under natural or semi-natural vegetation, where their land-use is confined to rough grazing or coniferous forest plantation (Finch and Ryan 1966). The low base parent material, often combined with an acidic litter-fall derived peaty O horizon, leads to low soil pH values (NFI 2007b, IUSS 2007).

Several types of podzol are known (see Figures 2.6 and 2.7) with peaty podzols having a relatively thick (7.5–40 cm) peaty O horizon, where the extent of organic matter humification increases with depth (Teagasc 2013). They are characterised by cheluviation/chilluviation processes involving downward movement in the soil profile of soluble metal-humus complexes (chelates) (Spaargaren 2003, 2008). These podzolisation processes create a sequence of horizons that distinguish them from other mineral soils. See Appendices 1.1 and 1.2 for details of the WRB keys to reference soil groups and subordinate characteristics within master horizons.

An eluvial, albic (from the Latin word *albus*, meaning white) or ash-grey E horizon is a common feature of podzols (Soil Survey Staff 2010). This horizon, which forms near the surface, has been leached to varying degrees of silicate clay particles, free iron and aluminium oxides or organic material. The terms “spodic” and the
equivalent term “Podzolic” in soil horizon nomenclature both refer to sub-surface illuvial B horizons, which are often brown or blackish in colour due to accumulation of SOM, or reddish horizons with accumulation of iron (Fe₂O₃) or aluminium (Al₂O₃) sesquioxides (IUSS 2007). These horizons are often found on top of an illuvial Bh horizon (accumulated humic material) and/or a strongly indurated Bf horizon known as an “iron pan Podzol” (Spaargaren 2003, Soil Survey Staff 2010).

Where an iron-pan forms it hinders root penetration and may cause surface water drainage issues, which unless broken by deep digging or ploughing may restrict the land usage (Finch and Ryan 1966). Below the Bf horizon a further iron-enriched Bs horizon will form (Spaargaren 2003, IUSS 2007). Where the non-selective leaching of clay particles occurs in an overlying A, E or Bs horizon, an illuvial silicate clay enriched argillic Bt or Btg horizon forms beneath (FAO 2006b).

**Figure 2.6** A typical podzol profile
(Source: NFI 2007)

**Figure 2.7** A typical peaty podzol profile
(Source: NFI 2007)
2.6 Life Cycle Assessment, origins and development

In the late 1960s, at a time of growing public concern over issues like energy efficiency, resource and waste management, the idea of evaluating a product’s environmental impact over its life cycle was first conceived (Guinée et al. 2011, McManus and Taylor 2015). From those early scientific studies (e.g. Sundstrom 1971, Hunt et al. 1974) Life Cycle Assessment (LCA) methodologies have been developed (Guinée et al. 1993a and 1993b) and are now widely used for the systematic evaluation of both goods and services, and have become a key component of environmental policy development across the globe (Guinée et al. 2011). Comprehensive LCAs generally cover all phases of a product’s life from acquisition of raw materials, to production, uses and waste management, which includes final disposal or recycling (Finnveden et al. 2009). Using an environmental LCA combined with cost and performance data can inform decision makers in selecting a product, or highlighting the most impactful processes in its life cycle (Curran, 2006). The term “cradle-to-grave” is used in LCA to cover all phases of a product’s life cycle, whereas the term “cradle-to-gate” or “partial LCA” deals with the delivery of a product to the consumer’s gate, or point-of-use (Finnveden et al. 2009).

With the growth in the practice and publication of scientific LCA studies from the 1990s onward (Finnveden et al. 2009), and with many divergent approaches employed, the need for development and standardisation of the LCA framework, terminology, and methodology was recognised. Through the efforts of the Society of Environmental Toxicology and Chemistry (SETAC) together with the International Organization for Standardization (ISO) the widely used “Environmental management - Life cycle assessment” standards ISO 14040 (Principles and framework) and ISO 14044 (Requirements and guidelines) were developed in 2006 (Guinée et al. 2011, McManus 2006).
and Taylor 2015). ISO 14040 and 14044 are hereafter referred to as the “LCA standards”.

2.6.1 Life Cycle Assessment framework and nomenclature

The methodologies described in the LCA standards are designed to systematically assess the phases and individual steps in providing a product, and use the perspective gained to identify and possibly avoid potential environmental impacts (ISO 2006a). Life Cycle Assessment is a form of systems analysis which aims to quantify all of the “cradle-to-grave” material and energy inputs and outputs needed for a product. It covers each step from its cradle (e.g., extraction and acquisition of raw materials), through energy and material production, to its use and end-of-life outcome, e.g., its grave (waste disposal) or recycling (ISO 2006a; Brandao et al. 2011).

In LCA nomenclature the “product system” represents a products life cycle and embodies all the elementary and product flows required to produce a reference product, e.g., a 10 year old *E. nitens* tree stem. A “functional unit” in LCA is a quantitative measure of the performance of the product system, such as one megawatt hour of electricity (MWh<sub>e</sub>) from a bioenergy product system, or one oven-dry tonne (ODT) of solid wood from a forest biomass product system. In the LCA standards the term “product” also includes services, and potential environmental impacts are expressed relative to the functional unit of a product system (ISO 2006b).

The product system is constructed via a set of “unit processes”, i.e. the smallest element of the life cycle inventory (LCI) analysis with quantifiable input and output data for all the elementary and products flows within the system. The analysis of each unit process and the compilation of the material and energy inputs and outputs are conducted relative to the provision of the defined functional unit (ISO 2006).
overall scope of the LCA is described within the “system boundary” which may contain one or more defined product systems, and which sets the criteria for which unit processes are included within those product systems (ISO 2006a).

The four phases (Figure 2.8) of a product LCA (ISO 2006a), are as follows:

1. **Goal and scope definition.** Covers the reasons for carrying out the LCA, the expected application of the study, and the target audience for the study outputs. This phase also describes the system boundary and defines the functional unit of the product system.

2. **Life Cycle Inventory (LCI).** The result from the LCI is a compilation of the material and energy inputs and the outputs (e.g. GHG emissions) from the product system over its life cycle in relation to the functional unit.

3. **Life Cycle Impact Assessment (LCIA).** The objective of the LCIA is to provide an understanding and evaluation of the magnitude and significance of the potential environmental impacts of the product system.

4. **Interpretation.** The outputs from the previous phases are evaluated in terms of the goal and scope, to enable the investigators draw conclusions and make recommendations to the target audience.
There is ongoing debate and development of recommended LCA practice beyond that covered within the four part framework of the ISO 14040 standard. Some of the developmental issues receiving attention are the distinction between attributional (ALCA) and consequential (CLCA) LCA (Curran et al. 2005, Finnveden et al. 2009, Cherubini and Strømman 2011, Zamagni et al. 2012, McManus and Taylor 2015), the handling of the “multi-functionality” problem, also known as allocation by substitution or system expansion (Finnveden et al. 2009, Wardenaar et al. 2012), biogenic carbon neutrality (Johnson 2009, O’Sullivan et al. 2016), direct and indirect LUC (McManus and Taylor 2015, Schmidt et al. 2015), and timing of emissions (Levasseur et al. 2010, Helin et al. 2013).
2.6.2 Life Cycle Impact Assessment methods

Traditionally, a Life Cycle Impact Assessment (LCIA) method is understood to be a set of environmental impact categories (Acero et al. 2015) quantified by a reference unit, e.g., climate change in kg CO$_2$eq; eutrophication in kg P equivalent; acidification in kg SO$_2$ equivalent; or particulate matter formation in kg PM10 equivalent. Each impact category has an associated set of compound flows, with some flows emanating from a variety of sources, e.g., the “climate change” category has the flows “carbon dioxide, in air” and “carbon dioxide, fossil”, which are further grouped into source related sub-categories such as “air/high population density” or “resource/in air”. The LCIA flows have a characterisation factor (CHF) based on their GWP, which is a multiplier representing the magnitude of its environmental impact, e.g., in the “climate change–air/high population density” category–sub-category the CHFs for the flow “carbon-dioxide, in air” = 1, for “methane, fossil” = 25, and for “dinitrogen-monoxide” = 298.

There are several commonly used LCIA methods available to LCA practitioners, e.g., Eco-Indicator 99 (Goedkoop et al. 2013), CML 2001 (Guinée et al. 2002), and ReCiPe 2008 (Goedkoop et al. 2013). The ReCiPe 2008 method was created by RIVM (the Dutch National Institute for Public Health and the Environment), CML (the Centre for Environmental Sciences, University of Leiden), PRé Consultants, and Radboud Universiteit Nijmegen. The ReCiPe authors include the developers of the CML and Eco-Indicator methodologies, and it was developed to combine the best aspects of those methods, taking and rebuilding the midpoint indicators from CML and the endpoint indicators from Eco-Indicator (Goedkoop et al. 2013). ReCiPe distinguishes two levels of indicators; a) Midpoint indicators, covering 18 impact categories, and b) Endpoint indicators, with categories such as damage to human health, damage to ecosystems and damage to resource availability (Acero et al. 2015). One important difference between
the newer ReCiPe and the older CML method is their treatment of atmospheric and terrestrial C sinks. The CML method has two “Flow;Category” combinations (i.e. “Carbon dioxide, in air; resource/in air” and “Carbon dioxide; resource/unspecified”) with GWP “Factors” of −1 that enable accounting for photosynthetic CO$_2$ sequestration in biomass and soil C sinks, respectively, whereas the ReCiPe method makes no such accommodation. In LCA terms, the principle output product for each process is called the “quantitative reference” or the “reference product” and is assigned the flow-type “product-flow” (GreenDelta 2016). All the input and output elementary or product flows for each processes are quantified based on the magnitude and unit of measurement of the reference product.

2.6.3 LCA studies of woody biomass

Among the many woody/lignocellulosic bioenergy crops covered in the LCA literature, e.g., Miscanthus (Murphy et al. 2013; Styles et al. 2007b), short rotation coppice (SRC) of willow (Goglio & Owende 2009; Djomo et al. 2011; Murphy et al. 2014b) or poplar (Schweier et al. 2016; Sabbatini et al. 2016), biomass production from single stem SRF is less commonly assessed. Klein et al. (2015) conducted a descriptive and quantitative analysis of existing LCA studies in forest production based on non-coppicing tree species, with a specific focus on their GWP. In that comprehensive study of 26 LCAs and two databases published over the previous 20 years, a total of 25 forestry processes were considered. Those processes, which covered the cradle-to-gate phases of the forest life cycle, were grouped into six process groups, starting with “Secondary processes” such as initial planning and seedling production, and ending with the transport of the chipped or shredded (hogfuel) woody biomass to the consumer. The four intermediate groups consisted of “Site preparation”, e.g., clearing and scarification, “Site tending”,
e.g., fencing, with fertilizer and herbicide application, and “Silvicultural operations” which includes planting, final felling, and forwarding. Only two, González-García et al. (2009) and González-García et al. (2013), of the 26 LCA studies were based on fast growing trees potentially suitable for SRF, which in both cases were Eucalyptus. Klein et al. (2015) found 12 different functional units in forest industry LCA studies, and they are generally expressed by dimension (e.g. 1 m$^3$, over or under bark), by area (e.g. hectares or square meters), by mass (e.g. 1 t, oven-dried), or by energy content (e.g. MWh, MJ). Their study also found that 1 m$^3$ over bark (ob) of recently harvested green wood was one of the most commonly used functional units.

Biomass for bioenergy LCAs cover a range of thermochemical or hydrothermal technologies, e.g. pyrolysis, gasification, liquefaction, carbonisation, and combustion, which are used in converting biomass products directly into energy or other useful products such as biochar, and liquid or gas fuels (Faaij 2006, Patel et al. 2016). Exclusive combustion of biomass, or co-firing with a fossil fuel like peat (Murphy et al. 2016), is also used in power generation or combined heat and power (CHP) plants (Guest et al. 2011). Biomass for bioenergy derived from sustainable forest management is widely considered a potential pathway to offsetting or reducing GHG emissions from fossil fuels through their displacement in the energy sector (Cherubini 2010, Murphy et al. 2013, Jonker et al. 2014, Thornley et al. 2015, Murphy et al. 2016). Where the above mentioned conversion technologies are used in conjunction with C capture and storage technologies the GHG emissions levels can be reduced to very low or even negative values (Kraxner 2003, Azar et al. 2010). Lastly, there are alternative uses of biomass other than for bioenergy that should be considered, e.g., forest wood products such as building materials and furniture offer the potential for long-term sequestration.
of biogenic C and also help mitigate GHG emissions by displacement of more GHG emission intensive products (Perez-Garcia et al. 2005, Donlan et al. 2012).
3. Chapter 3 - The need to disaggregate podzols and peaty podzols when assessing forest soil carbon stocks

The study presented in this chapter was published in the Journal of the Society of Irish Foresters, Irish Forestry, Vol. 72, Nos. 1&2, 2015. The final publication is available at:

https://journal.societyofirishforesters.ie/index.php/forestry/issue/view/909
3.1 Abstract

Inventories of forest soil carbon (C) stocks are necessary to determine spatial and temporal C stock changes and support climate change mitigation policy development. Afforested podzols and peaty podzols were sampled to measure bulk density (BD) and soil organic carbon (SOC) content with the aim of improving baseline soil C stock estimates for Irish forests. Podzols are not always distinguished from peaty podzols, and both qualify as mineral soil types. Distinct differences in mean BD, SOC %, and soil C stock values were found between sites with podzols and peaty podzols across the four depths sampled, i.e., 0–10, 10–20, 20–30, 30–40 cm. The estimated soil C stocks for the podzol sites ranged from 129–139 Mg C ha\(^{-1}\), while the peaty podzols had 229–385 Mg C ha\(^{-1}\). The major disparity in the soil C stocks implies the need to disaggregate podzols and peaty podzols in conducting soil C inventories, with the need for development of carbon emission factors for peaty podzols to reduce uncertainty in soil C stock changes.

**Keywords:** Afforestation, bulk density, soil organic carbon density, emission factor.
3.2 Introduction

Regional and national scale soil carbon (C) inventories are required to understand soil C dynamics and support climate change mitigation policy development (IPCC 2006, Ogle et al. 2010, Mishra et al. 2012). Sampling of a population involves taking measurements from a select subset of individuals to estimate the properties or parameters of the total population (Pennock et al. 2006). Stratified sampling (Heim et al. 2009), e.g. by soil group (e.g. peat soils, gleys, podzols) and tree species, can be used to reduce the soil organic carbon (SOC) sampling effort. Generally, precision of estimated regional or national SOC inventory values (Mg C ha\(^{-1}\)) is increased (i.e. smaller confidence ranges) with increased sampling (IPCC 2006).

3.2.1 Differences in definition and carbon assessment of organic and mineral soils

The Intergovernmental Panel on Climate Change (IPCC) and the Food and Agriculture Organisation (FAO) use similar criteria to distinguish between organic and mineral soils. Organic soils are also known as peatland, bog, or muck soils (Couwenberg 2009, IPCC 2014a). The IPCC mostly use the following FAO guidelines for defining organic soils, but allow greater autonomy based on country-specific historical definitions of organic soils (IPCC 2014a):

i) have a minimum thickness of 10 cm where overlying rock or ice;

ii) contain at least 12% organic C (~20 % soil organic matter (SOM) by weight) for 0–20 cm soil depth where the organic layer is <20 cm deep;

iii) hold >20 % SOC (~35% SOM) for normally unsaturated soils; and

iv) have between 12–18% SOC with clay content varying between 0–60 % (IPCC, 2014).
In contrast, most European definitions of organic soils stipulate >30 % (dry mass) of SOM in layers ≥40 cm deep (Joosten and Clarke 2002, Couwenberg 2009).

The Irish EPA defines organic soils as having >20 % SOC and depth >30 cm (Duffy et al. 2014). Teagasc, the Irish Agriculture and Food Development Authority, use a depth >40 cm and sub-divide organic soils with <50 % SOC into sandy, loamy and peaty organic soils based on the percentage of clay and sand found (Simo et al. 2014). Northern hemisphere organic soils cover around 3% of the global land area, hold approximately one third of global SOCD (Gorham 1991, Turunen et al. 2002) and between 53–62 % of Irish soil’s SOC (Tomlinson 2005, Eaton et al. 2008). These C rich soils occupy around 14–17 % of Ireland’s land area (Connolly et al. 2007, Hammond 1981) and have C stock values for their total estimated depth ranging from as low as 240 Mg C ha⁻¹ for lowland blanket peats to as high as 3,070 Mg C ha⁻¹ in lowland raised bogs (Tomlinson 2005). Due to their large C stock values, use of soil sampling techniques to measure relatively small changes in C stock can be adversely affected by a low signal-to-noise ratio (Baker and Griffis 2005). Therefore the C flux of organic soils is often assessed via eddy covariance or chamber based monitoring systems from which emission inventories for carbon dioxide (CO₂) or other greenhouse gas (GHG) fluxes are derived (Alm et al. 2007, Couwenberg 2009). The measured GHG emission quantities together with activity statistics, e.g., land area and afforestation or deforestation rates form the basis of emission factors (EFs) for a studied source/GHG combination (Duffy et al. 2014). An EF of 2.6 Mg CO₂ ha⁻¹ yr⁻¹ is reported for forests on drained organic soils in temperate climate/vegetation zones (IPCC 2014a), though the EPA (Duffy et al. 2014) use a much lower value based on data from Byrne and Farrell (2005) of 0.58 Mg CO₂ ha⁻¹ yr⁻¹. The EPA also use the same soil EF as above for peaty mineral soils but adjust the EF based on the depth (cm) of the peaty layer.
Alternatively, mineral soils are defined by the EPA as having <20 % SOC to a maximum depth of 30 cm and generally have much lower C stocks. For example the top 30 cm (excluding the litter and fine woody debris components) of Irish forested brown earths and gleys have estimated C stocks ranging from 42 Mg C ha\(^{-1}\) to 167 Mg C ha\(^{-1}\) respectively (Wellock \textit{et al.} 2011). Tomlinson (2005) also reported Irish mineral soil C stocks ranging from 137 Mg C ha\(^{-1}\) to 343 Mg C ha\(^{-1}\) for grey-brown podzolics and podzols, respectively. Smith \textit{et al.} (2006) used a cut-off point of 200 Mg C ha\(^{-1}\) to differentiate between organic and mineral soils in parameterising the Rothamsted Carbon Model (RothC; Coleman and Jenkinson 1996). Estimation of C stocks in mineral soils, to a specified depth, is typically done via stratified sampling of SOC content (%), bulk density (BD) (g cm\(^{-3}\)) and rock fragments mass and volume (Olsson \textit{et al.} 2009, Wellock \textit{et al.} 2011). Forest soil sampling methodologies for national SOC inventories vary. They include repeat standardised sampling of stock changes, which is rare (Ortiz \textit{et al.} 2011), paired plots, e.g. forested and non-forested sites on similar soils (Wellock \textit{et al.} 2011, Lawrence \textit{et al.} 2013), and chronosequence-based studies (Reidy and Bolger 2013). These methodologies are designed to measure the net effect that temporal and spatial variables, along with climate differences and land management, have on soil C stocks within a site, region or country at a point in time. They are also intended to help forecast the impact of future land-use change (Turner and Lambert, 2000, Scott \textit{et al.} 2002).

While soils are generally classified as organic or mineral soils, there is also an intermediate group of soils in the continuum (Duffy \textit{et al.} 2014) of the above mentioned SOCD ranges. These soils are listed variously in the literature as peaty, peat-topped, humus-mineral or organo-mineral soils (Duffy \textit{et al.} 2014, Montanarella \textit{et al.} 2006, Smith \textit{et al.} 2007). These peaty mineral soils, which the EPA classify as having an
organic surface layer <30 cm deep, account for over 21,000 ha of the Irish forest estate (~14%, excluding open areas) (Duffy et al. 2014). They are not as well accounted for when it comes to C stock values and sampling methodology best practice, partially due to the significant site-level spatial variability of surface organic layer thickness (Kiely et al. 2009). In their study of Irish SOC dynamics over the years 1851–2000, Eaton et al. (2008) highlighted the differences between Irish forested mineral and peat soils and the blurred distinction of soil types presented by peaty soils. They also noted that because of the prevalence of peaty soils, such as peaty podzols, Ireland’s forest soils run counter to the global trends found in the study done by Guo and Gifford (2002) in that they contain greater C stocks than grasslands. They therefore warranted a focus on further disaggregation of soil classification beyond just mineral and peat soils.

3.2.2 Podzols and peaty podzols

Podzols are primarily conditioned by percolating rainwater in a temperate climate, and have soil horizon profiles heavily influenced by iron (Fe) and/or aluminium (Al) chemistry (FAO 2001). In Ireland, podzols occupy an estimated 559,600 ha (8%) of the land area (Gardiner and Radford 1980, Tomlinson 2005), account for 10 % of the forest estate (NFI 2013), and are most often located in hilly and mountainous areas at altitudes 150 m above mean sea level (AMSL), where rainfall plays a significant part in their development (Finch and Ryan 1966). Due to their topographical location and associated issues of accessibility they are generally found under natural or semi-natural vegetation, and their land-use has often been confined to rough grazing or coniferous forest plantations (Finch and Ryan 1966).

The recently developed Irish Soil Information System (SIS) identifies and describes Irish soil types and uses a unique blend of current and traditional methods and soils data to produce a new Irish soil classification system (Creamer et al. 2014). In the
Irish SIS, soil types are identified primarily by 11 soil “Great Groups”, one of which is the podzol group of mineral soils. The Podzol Great Group in turn contains subgroups, e.g. “Typical Podzol” or “Humic Podzol”, which further classifies together soils that share similar characteristics, which is a necessary aid to understanding the complexity of Ireland’s heterogeneous soils (Creamer et al. 2014). Within the Irish SIS Typical- and Humic-Podzol subgroups there are several soils described which have a surface peaty horizon (< 40 cm thick), underlying less decomposed organic horizons and overlying mineral horizons (Creamer et al. 2014). In order to keep the focus on their SOC content (%) over their other characteristics, this study uses the term “peaty podzol” when discussing these soils. They are found predominantly in mountain and hill topographies in Ireland (Gardiner and Radford 1980), and often at altitudes just below upland blanket peat.

Soil C stocks in forest plantations have been a central theme of forest research in recent decades (Byrne et al. 2015). This has addressed a range of issues such as modelling the effects of land use management and changes on the SOC pool (Black et al. 2014), and estimating C stocks in Sitka spruce (Picea sitchensis (Bong.) Carr.) plantations (Reidy and Bolger 2013). Also, several studies have assessed the C stock in prominent Irish forest soil types, e.g. peat soils and gleys (Tomlinson 2005, Byrne and Milne 2006, Black et al. 2009, Wellock et al. 2011). The objectives of this study were to measure BD and SOC content in afforested podzols with the aim of improving baseline mineral soil C stock estimates for Irish forests. As a result of sampling these soils the need arose for a discussion on disaggregating podzols and peaty podzols, and the respective methodologies employed when undertaking soil C inventories of them needs to be discussed.
3.3 Materials & Methods

3.3.1 Site selection and description

The forest podzol sites were selected from the total Irish National Forest Inventory (NFI) population of 1827 sites, which were systematically surveyed between 2010 and 2012 (NFI, 2012). Of the total 188 Podzol sites in the NFI survey, 12 out of the 16 sites that fitted the following stratified criteria were selected for further assessment; afforested (first rotation), tree species = Sitka spruce (*Picea Sitchensis* spp.), stand-age > 20 years, soil depth > 40 cm, land owned by Coillte, and within a 80 km radius of the University of Limerick in the province of Munster. The exact locations of the NFI sites are tightly controlled by the Irish Forestry Service (FS) in order to maintain the integrity of the surveyed sites. Therefore, after formally agreeing to maintain the anonymity of the NFI sites and avoid disturbance at their immediate location, their GPS coordinates were obtained from the FS.

Many NFI sites are situated several hundred metres from the nearest road, and Coillte owned sites are secured from vehicular traffic by padlocked roadside barriers. In order to gain car access to these sites it was necessary to complete “Hazard Risk Assessment” and “Indemnity Insurance” paperwork to obtain a district-wide barrier key from Coillte. After locating the NFI site coordinates via a handheld “Satmap Active 10” GPS unit (Satmap Systems Ltd., UK) and before commencing sampling it was necessary to confirm the presence of podzol soil characteristics. After the 12 NFI sites were visited and assessed for suitability against the above selection criteria, seven sites (Table 3.1 and Figure 3.1) were eventually selected for BD and SOC sampling, and individual “site codes” were assigned. The sites codes consist of two or three letters with the first letter assigned according to place name and the remaining letters being the compass direction (e.g. E = east, NW = northwest) from the NFI site to the sampled
plot, with the stand age in parentheses directly after. The other five sites were rejected due to issues such as inability to locate Podzol characteristics or accessibility of the NFI site. Of the seven sampled sites, three (i.e. BNW, ANW, KE) were located directly adjacent to the NFI site. The other four sites being alternate locations up to 1.2 km away due to issues at the NFI site location such as clear-felling, fire disturbance, accessibility, and infestation with Rhododendron.

Table 3.1 Sites selected for SOC and BD sampling. Alt = Altitude. Altrn. = Alternate to NFI site, Limk. = Limerick, Tipp. = Tipperary.

<table>
<thead>
<tr>
<th>Site code</th>
<th>NFI or Altrn.</th>
<th>County</th>
<th>Location</th>
<th>Soil Type</th>
<th>Soil Depth (cm)</th>
<th>Alt. (m)</th>
<th>Slope (°)</th>
</tr>
</thead>
<tbody>
<tr>
<td>VNE(19)</td>
<td>Altrn</td>
<td>Tipp.</td>
<td>Vee Gap</td>
<td>Podzol</td>
<td>125</td>
<td>174</td>
<td>18</td>
</tr>
<tr>
<td>BNW(21)</td>
<td>NFI</td>
<td>Cork</td>
<td>Boggeragh</td>
<td>Podzol</td>
<td>65</td>
<td>296</td>
<td>15</td>
</tr>
<tr>
<td>SE(26)</td>
<td>Altrn</td>
<td>Cork</td>
<td>Skeheen</td>
<td>Podzol</td>
<td>50</td>
<td>294</td>
<td>9</td>
</tr>
<tr>
<td>DSW(23)</td>
<td>Altrn</td>
<td>Tipp.</td>
<td>Devilsbit</td>
<td>Peaty Podzol</td>
<td>60</td>
<td>339</td>
<td>20</td>
</tr>
<tr>
<td>GSW(33)</td>
<td>Altrn</td>
<td>Limk.</td>
<td>Glenanair</td>
<td>Peaty Podzol</td>
<td>75</td>
<td>248</td>
<td>11</td>
</tr>
<tr>
<td>ANW(40)</td>
<td>NFI</td>
<td>Limk.</td>
<td>Anglesborough</td>
<td>Peaty Podzol</td>
<td>100</td>
<td>451</td>
<td>26</td>
</tr>
<tr>
<td>KE(44)</td>
<td>NFI</td>
<td>Limk.</td>
<td>Keale</td>
<td>Peaty Podzol</td>
<td>100</td>
<td>263</td>
<td>11</td>
</tr>
</tbody>
</table>

At the three NFI adjacent sites VNE(19), SE(26), and GSW(33), after moving a minimum of 50 m away from, but still within NFI stand location, test augers of the soil to 45 cm depth where examined to confirm the soil profile as Podzolic before
commencing sampling for SOC and BD measurement. At the four alternate sites it was necessary to drive through the forest plantation and examine road and drain cuttings to locate afforested podzols which also matched the required age profile. The stand age of the alternate sites was estimated on-site based on tree height, with the correct stand age later confirmed by Coillte via GPS coordinates of the sites supplied for referencing.

The selected sites were located in mountain or hill topographies (between Latitude 52°2’ and 52°48’ N and Longitude 7°54’ and 8°51’ W) at altitudes between 145 and 388 m OD. The Irish climate is cool temperate oceanic with a mean annual temperature and rainfall of 9-10° Celsius and 1230 mm respectively between 1981-2010 (Walsh 2012). According to Met Éireann (2014) annual precipitation rates increase on eastern slopes and exposed western uplands by between 100 – 200 mm respectively for every 100 m rise in elevation, and annual rainfall may increase by 50 % at altitudes above 300 m compared to adjacent lowlands.

Figure 3.1 Geographic distribution of the sampled sites (Source: Google earth, 2014).
3.3.2 Field methods: Bulk density sampling and soil profile analysis

Sampling of the selected sites was completed during August and September, 2013. The plot sampling design was adapted from Wellock et al. (2011). At each sampling plot a pit of approximately 1.0 × 0.8 m in area, and a minimum of 0.4 m below the forest floor loose litter (Oi horizon) was excavated (Figure 3.2), across the prevailing slope. Pits were excavated at each site and their profiles analysed (see Tables 3.3 and 3.4) to ascertain their structure, horizon depth, and to estimate the abundance of rock fragments and roots in the soil. Road cuttings adjacent to the VNE(19) and GSW(33) sites were used to measure the soil profiles to depths of approximately 1.3 m.

![Forest pit excavation for soil bulk density (BD) sampling.](image)

The pit dimensions varied due to environmental factors such as the presence of large rock fragments and roots, or soil depth before reaching the C horizon, and the
minimum space required to hammer the BD sampling cores into the pit profile. The BD samples were taken from three of the four pit faces using stainless steel BD rings (Eijkelkamp Agrisearch Equipment BV, Netherlands) of a volume = 100 cm³. Four BD samples were taken from the centre of each 10 cm increment below the litter layer down to 40 cm, giving three samples for each depth increment from each pit (Table 3.2). The individual BD rings were capped on site, labelled and bagged for transport back to the laboratory. The average depth of the organic and mineral horizons within the soil profiles were recorded along with approximate root and stone contents of each horizon.

3.3.2.1 Soil organic carbon sampling

At two points, 25 and 50 cm from the centre of each of the 4 pit sides (Figure 3.3), soil samples were taken using a 2.5 cm wide by 15 cm long soil auger (Eijkelkamp Agrisearch Equipment BV, Netherlands). The auger handle was marked to denote 30 cm and 45 cm from the auger tip to aid in tracking sampling depth below the soil surface. The auger core length was divided into three 5 cm segments and marked to allow division of each 15 cm sample into two samples of 10 and 5 cm lengths.

The first 15 cm auger sample was divided with the top 10 cm going into a plastic bag, which was labelled according to site, location relative to the pit face, and depth increment (e.g. VNE-1a-1). The remaining 5 cm of the auger sample was placed in the next labelled bag, with the top 5 cm from the next auger sample added to this to make up the 10 – 20 cm depth increment sample. In this manner the soil samples, for depth increments of 0-10, 10-20, 20-30, and 30-40 cm below the forest floor loose litter (Oi horizon), were obtained and used for SOC % determination in the laboratory.
Figure 3.3 Schematic of forest soil pit bulk density (BD) core sampling at 10 cm depth increments to 40 cm, and adjacent soil organic carbon (SOC) auger sampling at 25 and 50 cm from the pit faces.

Table 3.2 Number of field soil organic carbon (SOC) and bulk density (BD) samples taken for laboratory analysis.

<table>
<thead>
<tr>
<th></th>
<th>No. of sites</th>
<th>No. of plots/site</th>
<th>No. of samples/plot</th>
<th>No. of depth increments</th>
<th>Sample totals</th>
</tr>
</thead>
<tbody>
<tr>
<td>SOC Samples</td>
<td>7</td>
<td>1</td>
<td>8</td>
<td>4</td>
<td>224</td>
</tr>
<tr>
<td>BD Samples</td>
<td>7</td>
<td>1</td>
<td>3</td>
<td>4</td>
<td>84</td>
</tr>
<tr>
<td>Overall total:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>308</td>
</tr>
</tbody>
</table>

3.3.3 Podzol soil profiles

The pits at the forest podzol sites had horizons of decomposed organic matter, the Oi, Oe, and Oa horizons respectively, ranging between 8–11 cm of cumulative depth (Table
3.3). These were followed by a humic mineral Ah horizon and transitional AE or EA horizons, displaying both A and E horizon characteristics to different degrees, and in the case of VNE(19) a subsequent strongly leached E horizon which extended to a depth of 74 cm. The road cutting adjacent to the pit location at the VNE(19) site facilitated measuring the soil horizons from the surface organic layers down to the C horizon. This site had the highest levels of rock and root abundance in the top 20 cm of the profile, with rock fragments of up to 200 mm in size commonly found in the mineral horizons. See Appendices 1.3 and 1.4 for details of the FAO Soil Description codes for rock fragment and root classification, and abundance estimation. At all three podzol sites the root abundance found in the O and Ah horizons significantly decreased in the eluvial E and illuvial B horizons.

3.3.4 Peaty podzol soil profiles

The pits at the peaty podzol sites each had organic horizons of varying thickness, reaching from 12 – 20 cm below the forest floor (Table 3.4). These horizons were followed by a humic mineral Ah horizon or mixed AE horizons, which extended to a depth of 16 – 30 cm. At the four peaty podzol sites the abundance of roots diminished significantly below the surface mineral (A) horizons.
Table 3.3 Soil profile analysis for the podzol sites.

<table>
<thead>
<tr>
<th>Site &amp; Layer</th>
<th>Horizon</th>
<th>Depth (cm)</th>
<th>Cuml. Depth (cm)</th>
<th>Rock Abundance (Code: %)</th>
<th>Rock Size Class (Code: mm)</th>
<th>Root Abundance (Code)</th>
<th>Root Size Class (Code: mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>VNE(19)</td>
<td>Oi</td>
<td>4</td>
<td></td>
<td>N: 0</td>
<td>N</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oe</td>
<td>3</td>
<td>7</td>
<td>V: 0-2</td>
<td>F: 2-6</td>
<td>M</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>Oa</td>
<td>4</td>
<td>11</td>
<td>F: 2-5</td>
<td>FM: 2-20</td>
<td>M</td>
<td>MC: 2-&gt;5</td>
</tr>
<tr>
<td></td>
<td>Ah</td>
<td>6</td>
<td>17</td>
<td>F: 2-5</td>
<td>FM: 2-20</td>
<td>M</td>
<td>MC: 2-&gt;5</td>
</tr>
<tr>
<td></td>
<td>E</td>
<td>57</td>
<td>74</td>
<td>C: 5-15</td>
<td>CS: 20-200</td>
<td>V</td>
<td>F: 0.5-2</td>
</tr>
<tr>
<td></td>
<td>Br</td>
<td>2.5</td>
<td>76.5</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bs</td>
<td>&gt;50</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BNW(21)</td>
<td>Oi</td>
<td>2</td>
<td></td>
<td>N: 0</td>
<td>N</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oe</td>
<td>4</td>
<td>6</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>MC: 2-&gt;5</td>
</tr>
<tr>
<td></td>
<td>Oa</td>
<td>3</td>
<td>9</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>MC: 2-&gt;5</td>
</tr>
<tr>
<td></td>
<td>Ah</td>
<td>7</td>
<td>16</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>M</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>EA</td>
<td>9</td>
<td>25</td>
<td>V: 0-2</td>
<td>C: 20-60</td>
<td>F</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>EB</td>
<td>&gt;27</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SE(26)</td>
<td>Oi</td>
<td>3</td>
<td></td>
<td>N: 0</td>
<td>N</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oe</td>
<td>3</td>
<td>6</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>M</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>Oa</td>
<td>2</td>
<td>8</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>M</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>Ah</td>
<td>4</td>
<td>12</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>FM: 0.5-5</td>
</tr>
<tr>
<td></td>
<td>AE</td>
<td>13</td>
<td>25</td>
<td>C: 5-15</td>
<td>CS: 20-200</td>
<td>F</td>
<td>F: 0.5-2</td>
</tr>
<tr>
<td></td>
<td>Bs</td>
<td>&gt;25</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

NB: Cuml. = Cumulative.

1Rock/root abundance codes: N: None, V: Very Few, F: Few, C: Common, M: Many
2Rock size class codes and combinations: F: Fine gravel, M: Medium gravel, C: Coarse gravel, S: Stones, FM: Fine and medium gravel, CS: Coarse gravel and stones

(1,2,3 Source: FAO 2006)
Table 3.4 Soil profile analysis for the peaty podzol sites.

<table>
<thead>
<tr>
<th>Site &amp; Layer</th>
<th>Horizon</th>
<th>Depth (cm)</th>
<th>Cuml. Depth (cm)</th>
<th>Rock Abundance</th>
<th>Rock Size Class (Code: %)</th>
<th>Root Abundance</th>
<th>Root Size Class (Code: mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>DSW(23)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Oi</td>
<td>2</td>
<td>N: 0</td>
<td></td>
<td></td>
<td></td>
<td>N</td>
</tr>
<tr>
<td>2</td>
<td>Oe</td>
<td>3</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>F</td>
<td>M: 2-5</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Oa</td>
<td>7</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>V</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Ah</td>
<td>4</td>
<td>F: 2-5</td>
<td>FM: 2-20</td>
<td>F</td>
<td>M: 2-5</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>E/B</td>
<td>16</td>
<td>C: 5-15</td>
<td>C: 20-60</td>
<td>F</td>
<td>M: 2-5</td>
<td></td>
</tr>
<tr>
<td>GSW(33)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Oi</td>
<td>4</td>
<td>N: 0</td>
<td></td>
<td></td>
<td></td>
<td>N</td>
</tr>
<tr>
<td>2</td>
<td>Oe</td>
<td>4</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>M</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Oa</td>
<td>12</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>M</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Ah</td>
<td>7</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>F: 0.5-2</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>E/A</td>
<td>27</td>
<td>F: 2-5</td>
<td>FM: 2-20</td>
<td>V</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>Bs</td>
<td>15</td>
<td>C: 5-15</td>
<td>C: 20-60</td>
<td>N</td>
<td></td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>C</td>
<td>&gt;30</td>
<td>M: 15-40</td>
<td>CS: 20-200</td>
<td>N</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ANW(40)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Oi</td>
<td>4</td>
<td>N: 0</td>
<td></td>
<td></td>
<td></td>
<td>N</td>
</tr>
<tr>
<td>2</td>
<td>Oe</td>
<td>4</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Oa</td>
<td>9</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>C</td>
<td>F: 0.5-2</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>A/E</td>
<td>13</td>
<td>M: 15-40</td>
<td>CS: 20-200</td>
<td>V</td>
<td>F: 0.5-2</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Bs</td>
<td>&gt;50</td>
<td>M: 15-40</td>
<td>CS: 20-200</td>
<td>V</td>
<td>F: 0.5-2</td>
<td></td>
</tr>
<tr>
<td>KE(44)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Oi</td>
<td>2</td>
<td>N: 0</td>
<td></td>
<td></td>
<td></td>
<td>N</td>
</tr>
<tr>
<td>2</td>
<td>Oe</td>
<td>8</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>F</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Oa</td>
<td>2</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>V</td>
<td>FM: 0.5-5</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>A/E</td>
<td>5</td>
<td>V: 0-2</td>
<td>FM: 2-20</td>
<td>V</td>
<td>MC: 2-&gt;5</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Bh</td>
<td>3</td>
<td>F: 2-5</td>
<td>CS: 20-200</td>
<td>V</td>
<td>F: 0.5-2</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>Bs</td>
<td>&gt;23</td>
<td>F: 2-5</td>
<td>CS: 20-200</td>
<td>N</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**NB:** Cuml. = Cumulative.

1Rock/root abundance codes: N: None, V: Very Few, F: Few, C: Common, M: Many
2Rock size class codes and combinations: F: Fine gravel, M: Medium gravel, C: Coarse gravel, S: Stones, FM: Fine and medium gravel, CS: Coarse gravel and stones

(1,2,3 Source: FAO 2006)
3.3.5 Laboratory methods: Estimation of soil bulk density

In the laboratory the fresh samples were extracted from the BD rings and weighed to determine their fresh weight. The BD samples were placed in brown paper bags and air-dried at room-temperature for one week before being oven-dried to constant weight at 105 °C. The dried soil BD samples were weighed and their mass (to 0.00 g) recorded for calculation of the % moisture content. The samples where then broken by hand and any visible coarse fragments (i.e. > 2 mm) such as gravel, stone, or roots, were removed. The samples were then crushed using a wooden rolling-pin before being sieved through a 2 mm aperture to separate the fine and coarse fractions. The mass of both the fine and coarse fraction was recorded. The volume of the coarse fraction was determined by the water displacement method using a fixed volume of water in a graduated cylinder.

The bulk density of the fine earth fraction (BD\textsubscript{Pfe}) of each sample was determined using the following formula from Throop \textit{et al.} (2012).

\[ BD_{Pfe} = \frac{Mass_{soil} - Mass_{rf}}{Volume_{soil} - Volume_{rf}} \]

\begin{align*}
\text{Mass}_{soil} & = \text{Mass of oven-dried BD soil sample} \\
\text{Mass}_{rf} & = \text{Mass of rock fragments} \\
\text{Volume}_{soil} & = \text{Volume of BD ring (i.e. 100 cm}^3\text{)} \\
\text{Volume}_{rf} & = \text{Volume occupied by the rock fragments}
\end{align*}

3.3.5.1 Estimation of soil organic carbon concentration and soil organic carbon density

In the laboratory the augured SOC samples were also placed in brown paper bags and air-dried at room-temperature for one week before being oven-dried to constant weight at 105 °C. The samples where then broken by hand and any visible rock fragments such
as gravel, stone, or roots, were removed. The samples were then crushed using a wooden rolling-pin before being sieved to separate the fine and coarse fractions. A 5.00 - 5.10 g sub-sample of the soil fine fraction of each SOC sample, which had already been thoroughly mixed by the crushing and sieving process, was then placed in a crucible. The combined soil and crucible weight for each sample was recorded. All the SOC samples were placed in a muffle furnace for three hours at 550 °C to determine % of SOM from the Loss On Ignition (LOI) method. The LOI values for the samples SOM concentration (%) were then converted to SOC concentration (%) using the “van Bemmelen” factor as follows:

- SOM = ((SMbc – SMac) / SMbc ) × 100

Where;

SMbc = Oven-dried (105 °C) sample mass before combustion*
SMac = Sample mass after combustion (1)

(1) Combustion of 5 g sample @ 550 °C for 3 hours

- SOC = SOM ÷ 1.724 (2)

(2) 1.724 = Van Bemmelen (1881) factor

Going one step further, the determination of the SOC stock to a given depth, hereafter referred to as the soil organic carbon density (SOCD) on an areal basis (Mg C ha⁻¹), is calculated as follows:

- SOCD = BD × SOC × D / 100

Where;
D = Sampling depth (cm)
3.4 Results

3.4.1 Podzol bulk density

The mean BD values (Table 3.5) for the forest podzols increased from 0.68 to 1.04 g cm\(^{-3}\) with each 10 cm depth increment through 0 – 30 cm, but fell to 0.89 g cm\(^{-3}\) in the 30 – 40 cm increment. The lowest values were found in site SE(26) where the coarse fraction volume in the BD sampling cores was highest, primarily due to the high rock fragment abundance in the pit. Site VNE(19) had the highest mean BD at 1.16 g cm\(^{-3}\) for the full 0 – 40 cm depth, while site SE(26) had the lowest mean BD of 0.54 g cm\(^{-3}\).

Table 3.5 Soil bulk density (BD, g cm\(^{-3}\)) and standard deviation (S.D.) by depth, and inter-site mean BD and S.D. for the podzol sites.

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>VNE(19)</th>
<th>S.D. ±</th>
<th>BNW(21)</th>
<th>S.D. ±</th>
<th>SE(26)</th>
<th>S.D. ±</th>
<th>Inter-site mean BD</th>
<th>Inter-site S.D. ±</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–10</td>
<td>0.57</td>
<td>0.28</td>
<td>0.79</td>
<td>0.20</td>
<td>0.68</td>
<td>0.06</td>
<td>0.68</td>
<td>0.20</td>
</tr>
<tr>
<td>10–20</td>
<td>1.07</td>
<td>0.14</td>
<td>1.00</td>
<td>0.10</td>
<td>0.59</td>
<td>0.12</td>
<td>0.89</td>
<td>0.25</td>
</tr>
<tr>
<td>20–30</td>
<td>1.48</td>
<td>0.03</td>
<td>1.08</td>
<td>0.19</td>
<td>0.58</td>
<td>0.30</td>
<td>1.04</td>
<td>0.43</td>
</tr>
<tr>
<td>30–40</td>
<td>1.51</td>
<td>0.03</td>
<td>0.81</td>
<td>0.18</td>
<td>0.34</td>
<td>0.10</td>
<td>0.89</td>
<td>0.52</td>
</tr>
<tr>
<td>Mean: 0–40</td>
<td>1.16</td>
<td></td>
<td>0.92</td>
<td></td>
<td>0.54</td>
<td></td>
<td>0.87</td>
<td></td>
</tr>
</tbody>
</table>

3.4.2 Podzol soil organic carbon concentration

The mean SOC concentration (%) for the podzol sites (Table 3.6) decreased with each subsequent 10 cm depth increment. The incremental decrease in SOC % in the top 30 cm was evident for the VNE(19) and BNW(21) sites (Figure 3.4), followed by a 0.07 and 0.40 SOC % increase respectively in those sites 30 – 40 cm depth increments. There
was also an SOC % increase in the 20 – 30 cm layer of the SE(26) site in comparison to
the over and underlying depth increments.

**Table 3.6** Soil organic carbon (SOC) concentration (%) and standard deviation (S.D.)
by depth, and inter-site mean SOC and S.D for the podzol sites.

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>SOC (%)</th>
<th>VNE(19)</th>
<th>S.D. ±</th>
<th>BNW(21)</th>
<th>S.D. ±</th>
<th>SE(26)</th>
<th>S.D. ±</th>
<th>Inter-site mean SOC</th>
<th>Inter-site S.D. ±</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–10</td>
<td>8.64</td>
<td>3.80</td>
<td></td>
<td>6.99</td>
<td>0.77</td>
<td>7.70</td>
<td>2.32</td>
<td>7.78</td>
<td>2.59</td>
</tr>
<tr>
<td>10–20</td>
<td>3.51</td>
<td>2.32</td>
<td></td>
<td>3.53</td>
<td>0.49</td>
<td>5.54</td>
<td>2.85</td>
<td>4.19</td>
<td>2.26</td>
</tr>
<tr>
<td>20–30</td>
<td>1.35</td>
<td>0.76</td>
<td></td>
<td>1.91</td>
<td>0.56</td>
<td>6.24</td>
<td>2.44</td>
<td>3.16</td>
<td>2.66</td>
</tr>
<tr>
<td>30–40</td>
<td>1.42</td>
<td>0.46</td>
<td></td>
<td>2.31</td>
<td>0.38</td>
<td>5.36</td>
<td>1.45</td>
<td>3.03</td>
<td>1.93</td>
</tr>
<tr>
<td>Mean: 0–40</td>
<td>3.73</td>
<td>3.68</td>
<td></td>
<td>6.21</td>
<td>4.54</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Figure 3.4** Soil organic carbon (SOC) concentration (%) change by depth, and standard
deviation error bars, for the podzol sites.
3.4.3 Podzol soil organic carbon concentration analysis

The mean SOC % decreased at each depth increment from 0 – 40 cm at the podzol sites (Table 3.7), from a high value of 7.78% to 3.03% at the lowest depth increment. The largest SOC % Standard Deviation (SD), Standard Error of the Mean (SEM), and Coefficient of Variance (CV) occurred in the 20 – 30 cm increment, with values of 2.66, 0.54, and 84.0 respectively. The lowest values for SD, SE and CV values were found in the 30 – 40 cm depth increment.

Table 3.7 Mean SOC % by depth for podzol sites. N = population, SD = Standard Deviation, SEM = Standard Error of the Mean, CV = Coefficient of Variance

<table>
<thead>
<tr>
<th>Depth below Oi horizon (cm)</th>
<th>n</th>
<th>Mean SOC (%)</th>
<th>SD</th>
<th>SE</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 10</td>
<td>24</td>
<td>7.78</td>
<td>2.59</td>
<td>0.53</td>
<td>33.3</td>
</tr>
<tr>
<td>10 – 20</td>
<td>24</td>
<td>4.19</td>
<td>2.26</td>
<td>0.46</td>
<td>53.9</td>
</tr>
<tr>
<td>20 – 30</td>
<td>24</td>
<td>3.16</td>
<td>2.66</td>
<td>0.54</td>
<td>84.0</td>
</tr>
<tr>
<td>30 – 40</td>
<td>24</td>
<td>3.03</td>
<td>1.93</td>
<td>0.39</td>
<td>63.7</td>
</tr>
<tr>
<td>Mean: 0-40</td>
<td>24</td>
<td>4.54</td>
<td>2.36</td>
<td>0.48</td>
<td>58.7</td>
</tr>
</tbody>
</table>

3.4.4 Podzol soil organic carbon density

The mean SOCD (Mg h⁻¹) in the Podzol sites decreased with each depth increment down to 40 cm (Table 3.8). The total SOCD also increased from 128.52 to 138.60 Mg h⁻¹ with stand age across the sites.
Table 3.8 SOCD (Mg ha\(^{-1}\)) by depth and overall mean values for the podzol sites.

<table>
<thead>
<tr>
<th>Depth below Oi horizon (cm)</th>
<th>Sites</th>
<th>Mean SOCD (Mg ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>VNE(19)</td>
<td>BNW(21)</td>
</tr>
<tr>
<td>0 – 10</td>
<td>49.58</td>
<td>55.14</td>
</tr>
<tr>
<td>10 – 20</td>
<td>37.55</td>
<td>35.33</td>
</tr>
<tr>
<td>20 – 30</td>
<td>19.93</td>
<td>20.58</td>
</tr>
<tr>
<td>30 – 40</td>
<td>21.46</td>
<td>18.80</td>
</tr>
<tr>
<td>SOCD: 0–40</td>
<td>128.5</td>
<td>129.8</td>
</tr>
</tbody>
</table>

3.4.5 Soil organic carbon correlation with bulk density for the podzol sites

There was a strong negative correlation (R\(^2\) = 0.72) between the SOC % and BD values at the podzol sites, using a best-fit exponential line function as shown in Figure 3.5.

The highest SOC % values were associated with the lowest BD values, which, as was seen previously were found in the depth increments nearest the forest floor.

Figure 3.5 The SOC % to BD (g cm\(^{-3}\)) correlation for the podzol sites, with the best-fit exponential function.
3.4.6 Peaty podzol bulk density

The mean BD values (Table 3.9) for the forest peaty podzols increased from 0.41 to 0.63 g cm\(^{-3}\) with each 10 cm depth increment through 0 – 30 cm, but fell from 30 – 40 cm to 0.58 g cm\(^{-3}\). The lowest mean BD value of 0.51 g cm\(^{-3}\) was found at site ANW(40) due to the high rock fragment content in the 20 – 40 cm depth increments. Site GSW(33) had the highest mean BD at 0.60 g cm\(^{-3}\) for the full 0 – 40 cm depth.

Table 3.9 Soil bulk density (BD, g cm\(^{-3}\)) and standard deviation (S.D.) by depth, and inter-site mean BD and S.D. for the peaty podzol sites.

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>DSW(23) S.D. ±</th>
<th>GSW(33) S.D. ±</th>
<th>ANW(40) S.D. ±</th>
<th>KE(44) S.D. ±</th>
<th>Inter-site mean BD</th>
<th>Inter-site S.D. ±</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–10</td>
<td>0.34</td>
<td>0.03</td>
<td>0.31</td>
<td>0.08</td>
<td>0.42</td>
<td>0.08</td>
</tr>
<tr>
<td>10–20</td>
<td>0.65</td>
<td>0.24</td>
<td>0.33</td>
<td>0.03</td>
<td>0.64</td>
<td>0.17</td>
</tr>
<tr>
<td>20–30</td>
<td>0.66</td>
<td>0.10</td>
<td>0.81</td>
<td>0.27</td>
<td>0.46</td>
<td>0.07</td>
</tr>
<tr>
<td>30–40</td>
<td>0.53</td>
<td>0.11</td>
<td>0.96</td>
<td>0.07</td>
<td>0.51</td>
<td>0.03</td>
</tr>
</tbody>
</table>

Mean: 0–40 0.54 0.60 0.51 0.56 0.55

3.4.7 Peaty podzol soil organic carbon concentration

The mean SOC concentration (%) for the peaty podzol sites (Table 3.10) also decreased with each subsequent 10 cm depth increment down to 40 cm. There was an incremental decrease in SOC % in the top 30 cm for all four sites (Figure 3.6), but two sites, DSW(23) and KE(44), showed a 0.04 and 1.33% increase respectively in the 30–40 cm depth increments.
Table 3.10 Soil organic carbon (SOC) concentration (%) with standard deviation (S.D.) by depth, and inter-site mean SOC and S.D for the peaty podzol sites.

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>DSW(23)</th>
<th>S.D. ±</th>
<th>GSW(33)</th>
<th>S.D. ±</th>
<th>ANW(40)</th>
<th>S.D. ±</th>
<th>KE(44)</th>
<th>S.D. ±</th>
<th>Inter-site mean SOC</th>
<th>Inter-site S.D. ±</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–10</td>
<td>31.8</td>
<td>7.89</td>
<td>39.5</td>
<td>6.48</td>
<td>35.5</td>
<td>5.85</td>
<td>22.3</td>
<td>4.93</td>
<td>32.3</td>
<td>8.88</td>
</tr>
<tr>
<td>10–20</td>
<td>8.33</td>
<td>3.21</td>
<td>39.2</td>
<td>14.1</td>
<td>19.6</td>
<td>6.34</td>
<td>7.24</td>
<td>1.10</td>
<td>18.6</td>
<td>15.1</td>
</tr>
<tr>
<td>20–30</td>
<td>5.46</td>
<td>1.38</td>
<td>11.3</td>
<td>11.1</td>
<td>9.13</td>
<td>2.74</td>
<td>6.13</td>
<td>0.64</td>
<td>8.01</td>
<td>5.99</td>
</tr>
<tr>
<td>30–40</td>
<td>5.50</td>
<td>0.82</td>
<td>4.24</td>
<td>5.30</td>
<td>7.84</td>
<td>2.91</td>
<td>7.46</td>
<td>1.41</td>
<td>6.26</td>
<td>3.33</td>
</tr>
<tr>
<td>Mean: 0–40</td>
<td>12.8</td>
<td></td>
<td>23.6</td>
<td></td>
<td>18.0</td>
<td></td>
<td>10.8</td>
<td></td>
<td>16.3</td>
<td></td>
</tr>
</tbody>
</table>

Figure 3.6 Soil organic carbon (SOC) concentration (%) change by depth, and standard deviation error bars, for the peaty podzol sites.
3.4.8 Peaty podzol soil organic carbon concentration analysis

The mean SOC % decreased at each depth increment from 0–40 cm at the peaty podzol sites (see Table 3.11), from a high value of 32.2% to 6.26% at the lowest depth increment. The largest SOC % SD, SE, and CV values occurred in the 10–20 cm increment, with values of 15.1, 2.67, and 81.1 respectively. The lowest values for SD, SE and CV were found in the top 10 cm.

<table>
<thead>
<tr>
<th>Depth below Oi horizon (cm)</th>
<th>N</th>
<th>Mean (%)</th>
<th>SD</th>
<th>SE</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 10</td>
<td>32</td>
<td>32.3</td>
<td>8.88</td>
<td>1.57</td>
<td>27.5</td>
</tr>
<tr>
<td>10 – 20</td>
<td>32</td>
<td>18.6</td>
<td>15.1</td>
<td>2.67</td>
<td>81.1</td>
</tr>
<tr>
<td>20 – 30</td>
<td>32</td>
<td>8.01</td>
<td>5.99</td>
<td>1.06</td>
<td>74.7</td>
</tr>
<tr>
<td>30 – 40</td>
<td>32</td>
<td>6.26</td>
<td>3.33</td>
<td>0.59</td>
<td>53.1</td>
</tr>
</tbody>
</table>

Table 3.11 Mean SOC % by depth for peaty podzol sites. n = population, SD = Standard Deviation, SE = Standard Error of the Mean, CV = Coefficient of Variance.

3.4.9 Peaty podzol soil organic carbon density

The mean SOCD (Mg h\(^{-1}\)) in the peaty podzol sites also decreased with each 10 cm depth increment down to 40 cm (see Table 3.12). The 10 – 20 cm depth increment of the GSW(33) site was the only one to show a reversal of the trend in decreasing SOC by subsequent depth increments. The total SOCD to a depth of 40 cm also varied significantly across the sites from a low of 228.51 to a high of 385.42 Mg h\(^{-1}\).
Table 3.12 Soil organic carbon density (SOCD) by depth and overall mean values for peaty podzol (PP) sites.

<table>
<thead>
<tr>
<th>Depth below Oi horizon (cm)</th>
<th>Sites</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DSW(23)</td>
</tr>
<tr>
<td>0 – 10</td>
<td>109.6</td>
</tr>
<tr>
<td>10 – 20</td>
<td>53.90</td>
</tr>
<tr>
<td>20 – 30</td>
<td>35.91</td>
</tr>
<tr>
<td>30 – 40</td>
<td>29.13</td>
</tr>
<tr>
<td>SOCD: 0–40</td>
<td>228.5</td>
</tr>
</tbody>
</table>

3.4.10 Soil organic carbon concentration correlation with bulk density for the peaty podzol sites

The logarithmic correlation ($R^2 = 0.43$) was the best-fit line function between the SOC % and BD values at the peaty podzol sites (see Figure 3.7) was lower than that for the Podzol sites. Again, the highest SOC % values were associated with the lowest BD values, but the greater overall range and variability in the SOC % values lowered the significance of the relationship.

Figure 3.8 illustrates the disparity between the podzol and peaty podzol sites SOC concentrations (%) at each of the four sampling depths.
Figure 3.7 Soil organic carbon concentration (SOC %) to soil bulk density (BD, g cm\(^{-3}\)) correlation for the peaty podzol sites.

Figure 3.8 Mean SOC % for each depth for the podzol and peaty podzol sites, with standard deviation error bars.
3.5 Discussion

The seven podzol and peaty podzol sites sampled in this study adhered to the soil classification used by Teagasc (Creamer et al. 2014) and the criteria used by the annual EPA “National Inventory Report” on GHG emissions (Duffy et al. 2014). Distinct differences were found in mean BD, SOC %, and soil C stock values between podzols and peaty podzols, across the four depths sampled. Even with their low mean BD of 0.55 g cm\(^{-3}\), the C rich surface horizons of the peaty podzols had mean soil C stocks (0–40 cm depth) that are over twice that in the podzols, i.e., an estimated 304 Mg C ha\(^{-1}\) in the former versus 132 Mg C ha\(^{-1}\) in the latter. The mean BD (0–40 cm) for the podzol sites of 0.87 g cm\(^{-3}\) was 58% higher than the respective value for the peaty podzol sites. There are very few published non-graphical sources with BD data by depth for Irish forested soils making direct comparisons with the soil types in this study impossible, therefore only comparisons with other forested mineral soils can be reported. This study’s mean BD for 0–30 and 0–40 cm for the podzol sites were the same, both were 0.87 g cm\(^{-3}\). These mean BD values are 7% lower in comparison to the mean BD (0–30 cm) of 0.94 g cm\(^{-3}\) for all 21 forested mineral soil sites assessed by Wellock et al. (2011), and 14% lower than the 1.01 g cm\(^{-3}\) for the five coniferous forest sites on surface-water gley soils studied by Black et al. (2009).

In their study of mainly “Humo-Ferric Podzol” forest soils in Canada, with their typically low density organic layers, high root abundance and stony mineral layers, Perie and Ouimet (2008) found that BD was closely correlated with SOM content (\(r^2 = 0.81\)). The peaty podzol sites in this study had a mean BD value (0–40 cm) of 0.55 g cm\(^{-3}\), 37% lower than the respective value for the podzol sites, which as seen above is already comparatively low to other forest mineral soils. Given the direct relationship in this study between SOM and SOC content via the 0.58 conversion factor, the high
estimated SOC % found in the peaty podzols at each sampled depth helps explain the low BD values found in this study. Without full soil particle and porosity analysis, and more extensive measurement of the in-situ rock fragments it is difficult to accurately assign causality for their low mean BD values, but it is thought the low BD values are attributable to a combination of the thick organic layers in the top 20 cm, and the often weakly aggregated sandy texture of podzols, as noted by FAO (2001).

The coefficient of variation (CV) values of the SOC % indicates that these soils were highly heterogeneous across all sampled depths, with the podzol sites CVs ranging from 33.3 (0–10 cm) to 84.0 (20–30 cm), with a similar range of 27.5 (0–10 cm) and 81.1 (10–20 cm) in the peaty podzol sites. There was a 65% increase in the soil C stock value between the highest value for the podzol sites and the lowest value for the peaty podzol sites, i.e., 139 Mg C ha\(^{-1}\) and 229 Mg C ha\(^{-1}\) respectively. At each sampled 10 cm depth down to 30 cm, the mean soil C stock of the peaty podzols exceeded that of the podzols by more than double. It was only in the lowest sampling depth, 30–40 cm, that this trend stopped and difference between the two soil C stock values was 58%.

When comparing the combined mean soil C stock for the three podzol sites in this study to the mean of the four coniferous forest podzol sites of Wellock et al. (2011), inclusive of F/H and mineral horizons to 30 cm for both studies, the results were within a 5% range. This study estimated the soil C stock for those same horizons and sampling depths to be 113 Mg C ha\(^{-1}\), while Wellock et al. (2011) estimated it at 117 Mg C ha\(^{-1}\). The estimated mean C stock for the peaty podzols in this study was 197 Mg C ha\(^{-1}\), a 68% increase over the Wellock et al. (2011) podzol values to the same depth.

The soil C stock (0–40 cm) for the podzol sites ranged between 129 and 139 Mg C ha\(^{-1}\), with a mean of 132 Mg C ha\(^{-1}\). This mean value is 15 Mg C ha\(^{-1}\) higher than the mean of 117 Mg C ha\(^{-1}\) from four podzol sites sampled by Wellock et al. (2011) (0–30
cm, excluding forest litter). It should be noted that the Wellock et al. SOC values were determined using a C/N elemental analyser in contrast to the LOI method used in this study, though De Vos et al. (2005) conclude both methods are comparable except in low organic C, non-calcareous soils where the former method is more reliable. The mean soil C stock for the podzol sites decreased by depth for each 10 cm increment down to 40 cm, from a high of 52 Mg C ha\(^{-1}\) in the top 10 cm to a low of 19 Mg C ha\(^{-1}\) in the bottom 10 cm, reflecting the decreasing SOC % at the same increments. The peaty podzol sites had much higher soil C stock (0–40 cm), ranging from 229 to 385 Mg C ha\(^{-1}\), with a mean of 304 Mg C ha\(^{-1}\). This mean value is 11% less than the 343 Mg C ha\(^{-1}\) for podzol soils in the Republic of Ireland as determined by Tomlinson (2005).

3.6 Conclusion

The disparity in the C stocks between afforested podzols and peaty podzols in this work underlines the need to disaggregate these soils and has implications for how they should be treated in soil C inventories to reduce uncertainty associated with soil C stock estimation. With their suitability for conifer plantations it is likely that these soils will be further utilised in any future expansion of the forest estate. In such cases of afforestation there may be potentially adverse implications for the stability of their inherent C stocks, e.g. via increased C emissions due to soil disturbance and drainage. Even if afforestation has only a minimal effect on soil C stocks at the regional or country level, its effect on the global C pool could be significant if large scale conversion of agricultural land to plantation continues (Paul et al. 2002). To establish more accurate baseline estimates against which future C stock change can be assessed, and facilitate a better understanding of the impact of afforestation on their soil C stocks, the methods employed in measuring their soil C stocks and fluxes need to be adapted.
Based on the findings of this study, the cut-off point of 200 Mg C ha\(^{-1}\) used by Smith et al. (2006) to differentiate between organic and mineral soils is deemed a useful threshold for determining the most appropriate method for monitoring C stock changes. For example, soil C stock sampling methodologies could be used for soils below the threshold, and the development of specific C EFs for soils that are above that level.
4. Chapter 4 - The influence of soil bulk density sampling and calculation methods on estimates of soil organic carbon density
4.1 Abstract

Soil bulk density (BD) is a principal component in estimating the density of any soil nutrient or other elements, such as carbon (C), therefore using the appropriate sampling and calculation methods are critical to the accuracy of soil nutrient and C inventories. There are several well-known methods of measuring soil BD such as core, clod, and quantitative pit, with variants of each method covered in the literature. In many soil surveys, for reasons related to cost, available resources, or the prevalence of rock fragments (RF), the soil BD was not measured directly. In other cases the core method was used as the sole determinant of soil BD, which potentially neglected to account for the soil volume dilution effect of RF larger than the diameter of the cores used. This study used the core and quantitative pit methods at 10 forest sites to determine the soil organic carbon (SOC) concentration (g SOC kg$^{-1}$ soil, expressed as a percentage) and BD (g cm$^{-3}$) and the RF mass and volume to a depth of 40 cm. This data was used to investigate the influence of large RF on the BD and the derived soil organic carbon density (SOCD). Three distinct methods of calculating the soil BD, which either fully include ($\rho_{\text{total}}$) or exclude the RF mass and volume ($\rho_{\text{fe}}$), or exclude their mass but include their volume ($\rho_{\text{hybrid}}$) in their equations, were also investigated to determine their influence on the SOCD estimates. A novel method, hereafter named “core-scaling”, was also devised to combine core and pit sampling methods and expedite the incorporation of large RF mass and volume in BD calculations. The literature has shown, and the results of this study concur, that in soils with RF content above 3 % of the total sample volume, substantial differences in estimated SOCD are found depending on the soil BD calculation method chosen.

**Keywords:** core, quantitative pit, rock fragments, core scaling.


4.2 Introduction

In a summary of published global estimates Scharlemann et al. (2014) found that soils are estimated to store between 504–3000 Pg C, (median = 1461 Pg, 1 Pg = $1 \times 10^{15}$ g), which is three times the amount stored in the atmosphere or terrestrial vegetation (Delgado-Baquerizo et al. 2017). The value for total C is comprised of approximately two thirds of soil organic carbon (SOC) and one third inorganic carbon (Lal 2008, Rawlins 2011). The importance of soil as the largest terrestrial C pool drives the need for better understanding of SOC pool sizes, soil C sequestration or emission potentials, and their responses to climate and land-use changes (Beem-Miller et al. 2016, Wiesmeier et al. 2012). Soil C inventories are key to constraining the global C cycle and our efforts to understand and predict future climate change and atmospheric chemistry (King et al. 2007). To understand current trends and support climate change mitigation policy development, SOCD inventories are also required to determine estimated SOCD changes at regional to national scales (Ogle et al. 2010, Mishra et al. 2012).

In order to properly reflect SOCD spatial heterogeneity and accurately identify SOCD changes at the landscape level, Jandl et al. (2014) highlighted the need for more cost-effective sampling methods. In any realistic soil science field research program sampling is necessary because of the impossibility of measuring the total population (Pennock et al. 2006). The accuracy of regional, national, and global SOCD estimates then depends on the quantity and quality of available data, and requires data on soil BD (g cm$^{-3}$), SOC concentration (g SOC kg$^{-1}$ soil, usually expressed as a percentage), and the sampled depth (Poeplau et al. 2017).

While SOC concentration and soil horizon depth are usually measured extensively as part of regional-scale or national SOC inventory studies, the greater time
and effort required to measure soil bulk density (BD) means it is often approximated using pedotransfer functions (De Vos et al. 2005, Wiesmeier et al. 2012, Reidy et al. 2016), or neglected altogether (Batjes 1996, Fay et al. 2007, Hiederer and Köchy 2011) and therefore surprisingly scarce (Taalab et al. 2015). Measuring SOCD to a specified depth, e.g., 30, 40, or 100 cm, requires two soil properties to be sampled in the field before analysis and calculations can be carried out in the laboratory. The SOC concentration and BD, which are often sampled at 5 or 10 cm increments to the specified depth (Don et al. 2007, Wellock et al. 2011) or by genetic horizons in an exposed soil profile (Premrov et al. 2018).

Soil stoniness, i.e., the soil fraction containing rock fragments (RF) > 2 mm in diameter, is another characteristic that is often ignored in soil surveys and the literature (Miller and Guthrie 1984, Torri et al. 1994), and is therefore regarded as the source of greatest uncertainty in SOCD estimates (IPCC 2003, Poeplau et al. 2017). Even when soil BD is recorded, the presence of RF in the soil, and the different field-work (Page-Dumroese et al. 1999) and calculation methods (Throop et al. 2012) used to determine BD can lead to systematically different SOCD estimates (Poeplau et al. 2017). The measurement of soil stoniness is also a key component in determining the hydraulic conductivity, soil compaction and degradation parameters often used in the development site indices for land-use and management (Tetegan et al. 2012, Farrelly et al. 2011b). Studies have also been undertaken to predict and verify the impact of land-use change, e.g., from abandoned arable land to SRF, using estimates of SOCD which included volumetric estimates of RF (Rytter 2012).

Jandl et al. (2014) discussed the need for methodological standardisation of soil C monitoring, and outlined four levels of soil sampling effort from lowest (Level 1) to highest (Level 4). Their Level 1 effort for sampling the soil BD and stoniness properties
were “no estimation” being made, while their Level 4 efforts for the same properties were site specific measurements of BD and RF volume. Indeed, Jandl et al. (2014) state that measurements of soil BD and RF content are equally important for the valid estimation of the SOC pool, and universal SOC monitoring. According to the literature there is no agreed standard for sampling soil BD (Batjes 1996, Scharlemann et al. 2014) with several methods commonly used, e.g., core, clod, and quantitative pit, hereafter “pit” (Vadeboncoeur et al. 2012). Based on a study comparing the core and pit methods, the latter is considered to be the optimal method for determining soil stone content and BD, and for estimating SOCD (Beem-Miller et al. 2016).

The main advantages of the core method over the pit method are the portability of the sampling tools, the relative speed of the sampling, and the potential to take replicate samples over a wide area. For those reasons the core method is the most widely used technique for estimating soil BD (Throop et al. 2012). While the excavation of quantitative pits can cause substantial site disturbance, they allow for deep sampling of stony soils, and direct, non-biased measurement of the soil properties, e.g., large RF mass and volume (Lyford 1964, Vadeboncoeur et al. 2014).

There are also several methods of calculating BD using:

- The total soil mass and volume
- The mass and volume of the fine earth fraction, i.e., that which will pass through a < 2 mm sieve only (Corti et al. 1998)
- The mass of the fine earth fraction divided by the volume of the total soil sample.

Those three methods, as reviewed by Throop et al. (2012) were designated $\rho_{\text{core}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ respectively. To avoid ambiguity or confusion with the core sampling method, in this study the $\rho_{\text{fe}}$ and $\rho_{\text{hybrid}}$ designations are retained, but $\rho_{\text{core}}$ is renamed to $\rho_{\text{total}}$. By
their inclusion or exclusion of the soil sample RF mass and volume, they respectively consider or ignore the dilution effect of the RF on the fine earth (<2 mm fraction) volume. In their study Throop et al. (2012) showed the large variation in SOCD estimates calculated using BD results derived from samples taken with stainless-steel cores (hereafter “cores”), and employing three different methods for calculating BD.

The objective of this chapter is to compare two soil sampling methods, i.e., the core and pit methods, and three BD calculation methods for the determination of SOCD. Ten SRF sites were utilised with a range of soil types and properties. The specific tasks undertaken to achieve that objective are as follows:

1. Use of the core method to measure soil BD and SOC concentration to 40 cm depth at all 10 SRF sites.

2. Use of a novel method using glass beads to measure the volume of large excavated pits on seven of the 10 sites, and estimate the soil BD to 40 cm depth. These seven sites had the highest relative volume of RF.

3. Investigate the impact of three different calculation methods (from Throop et al. 2012) on the BD results, and subsequently derived SOCD.

4. Compare the BD and SOCD results from the core and pit field methods, and the influence of the RF on their respective results.

5. Use of a novel method of expediting the estimation of soil BD from core and quantitative pit samples. The method involves scaling-up the mass and volume of the fine and coarse fractions (as appropriate to the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ BD calculation methods) from core samples taken in one pit, to estimate the mass and volume of the fine and coarse fractions in adjacent quantitative pits. Then adjusting the scaled up fine earth BD using the mass and volume of the large RF fraction, i.e.
the RF greater in diameter than the diameter of the cores, a new BD is calculated that accounts for the soil volume dilution effect of the large RF.

4.3 Materials and Methods

4.3.1 Site selection and description

Of fifteen SRF sites that were initially assessed, 10 were chosen for sampling with five ruled out for reasons such as site waterlogging, invasion of furze (Pteridium spp.) or bramble (Rubus fruticosus), and inaccessibility. The 10 SRF study sites were located in counties Limerick, Cork, and Waterford in southern Ireland (Figure 4.1). The sites were chosen using plantation profiles extracted from Coillte and Forest Service data (pers. comm. O’Tuama, Coates, respectively). The 10 SRF sites consisted of five eucalyptus, four alder and one sycamore plantations, and ranged in altitude above mean sea level (AMSL) from 36–247 m (Table 4.1). The sites also include all the major mineral soil types used for forestry in Ireland. The site selection criteria were as follows:

1. The sites contained a plantation of tree species, or genus where the species was not specified, suited to short rotation forestry.

2. The area within the plantation contained at least 0.5 ha of the targeted SRF genera/species. Sites of at least 0.5 ha were chosen so that the area sampled for SOC concentration and soil BD would primarily reflect the organic matter inputs from the specific tree species planted on the site.

3. The age of the SRF plantation was less than 15 years old. The age limit of 15 years was chosen to reflect the guidelines of the Grant and Premium Category 12a (GPC 12a) of the DAFM “Forestry for Fibre” scheme. The GPC 12a guidelines state that trees planted under the scheme will be felled at 10–15 years of age.
Figure 4.1 Geographic locations of the 10 short rotation forestry (SRF) sites in counties Limerick, Cork, and Waterford which were sampled for SOC concentration and soil BD. The red box on the inset map of Ireland shows the area of southern Ireland in which the sites were located.

The sampling of the sites for SOC concentration and soil BD took place between August 2015 and January 2016. The laboratory processing of the samples started in August 2015 and was completed in February 2016.
Table 4.1 Characteristics of the 10 short rotation forestry (SRF) sites selected for soil carbon stock sampling.

<table>
<thead>
<tr>
<th>Site code</th>
<th>Site name</th>
<th>Genus / species</th>
<th>Year of planting</th>
<th>Age (yrs)</th>
<th>Area (ha)</th>
<th>Initial spacing</th>
<th>Soil type</th>
<th>Former use</th>
<th>AMSL (m)</th>
<th>Slope (°)</th>
<th>Aspect</th>
</tr>
</thead>
<tbody>
<tr>
<td>CHR</td>
<td>Curraghanolomer</td>
<td>E. gunii</td>
<td>2011</td>
<td>4</td>
<td>2.9</td>
<td>2×2</td>
<td>Brown podzolic</td>
<td>SS plantation</td>
<td>236</td>
<td>3.5</td>
<td>S</td>
</tr>
<tr>
<td>GLR</td>
<td>Glannagear</td>
<td>E. gunii</td>
<td>2011</td>
<td>4</td>
<td>1.1</td>
<td>2×2</td>
<td>Brown podzolic</td>
<td>SS plantation</td>
<td>247</td>
<td>9.1</td>
<td>SW</td>
</tr>
<tr>
<td>WTB</td>
<td>Kilmacthomas 2</td>
<td>E. nitens</td>
<td>2009</td>
<td>6</td>
<td>1.9</td>
<td>2×2</td>
<td>Brown earth</td>
<td>FS nursery</td>
<td>168</td>
<td>2.5</td>
<td>SE</td>
</tr>
<tr>
<td>WTA</td>
<td>Kilmacthomas 1</td>
<td>E. nitens</td>
<td>2009</td>
<td>6</td>
<td>2.5</td>
<td>2×2</td>
<td>Brown earth</td>
<td>FS nursery</td>
<td>165</td>
<td>8</td>
<td>SE</td>
</tr>
<tr>
<td>MAC</td>
<td>Macroney</td>
<td>E. gunii</td>
<td>2010</td>
<td>5</td>
<td>18.1</td>
<td>2×2</td>
<td>Brown podzolic</td>
<td>SS plantation</td>
<td>197</td>
<td>2.5</td>
<td>SW</td>
</tr>
<tr>
<td>DKK</td>
<td>Dunkip K</td>
<td>Sycamore</td>
<td>2005</td>
<td>11</td>
<td>1</td>
<td>2×2</td>
<td>Surface-water gley</td>
<td>Rough pasture</td>
<td>41</td>
<td>0</td>
<td>Flat</td>
</tr>
<tr>
<td>CMM</td>
<td>Coom</td>
<td>Alder</td>
<td>2004</td>
<td>12</td>
<td>3.7</td>
<td>2×1.8</td>
<td>Brown podzolic</td>
<td>Rough pasture</td>
<td>221</td>
<td>0</td>
<td>Flat</td>
</tr>
<tr>
<td>DKB</td>
<td>Dunkip B</td>
<td>Alder</td>
<td>2003</td>
<td>13</td>
<td>2.8</td>
<td>1.8×1.5</td>
<td>Surface-water gley</td>
<td>Rough pasture</td>
<td>37</td>
<td>0</td>
<td>Flat</td>
</tr>
<tr>
<td>DKN</td>
<td>Dunkip N</td>
<td>Alder</td>
<td>2003</td>
<td>13</td>
<td>1.5</td>
<td>1.8×2</td>
<td>Alluvial gley</td>
<td>Rough pasture</td>
<td>39</td>
<td>0</td>
<td>Flat</td>
</tr>
<tr>
<td>CLY</td>
<td>Clonshavoy</td>
<td>Alder</td>
<td>2005</td>
<td>11</td>
<td>1</td>
<td>1.8×1.5</td>
<td>Ground-water gley</td>
<td>Rough pasture</td>
<td>36</td>
<td>0</td>
<td>Flat</td>
</tr>
</tbody>
</table>

*AMSL = Altitude above mean sea level, † SS = Sitka spruce, ‡ FS = Forest Service
4.3.2 Field methods

At each site a location at least 10 m inside the plantation boundary was chosen for SOC and soil BD sampling using the core and pit methods. Along a 4–6 m transect perpendicular to the prevailing slope, or at the non-sloping sites, to the nearest boundary, plots for three pits were selected. The plots were spaced at approximately 2–3 m intervals, depending on tree or drain spacing on the site. The pits were labelled A, B, and C. After the initial pit excavation and volume measurement for the quantitative pit method was completed, the central pit (Pit B) was enlarged to allow sampling using the core method (Figure 4.2).

Figure 4.2 An outline (not to exact scale) of the layout and dimensions of the three pits. The pits were spaced approximately 2–3 m apart depending on tree and drain spacing. Pit B, the central pit, was at first excavated to the same depth and approximate volume as the pits on either side (Pits A and C). After all the soil (fine and coarse fraction) was removed for laboratory analysis, Pit B was then extended to an area approximately 1 m² to allow soil bulk density (BD) samples to be taken using the cores from the middle of each 10 cm depth increment, from the beneath the litter layer down to 40 cm.
The cores were chamfered at one end with a 15 % cutting angle, had a diameter of 5 cm, and held a 100 cm$^3$ soil sample (Eijkelkamp, Netherlands).

After clearing the loose litter and vegetation from an area at least 30 cm in diameter (Figure 4.3), each pit was excavated until a minimum depth of 40 cm was reached. All soil fine and coarse fraction (RF, roots, and artefacts) were excavated from the pit into buckets for transport to the laboratory. Any coarse fraction that was significantly protruding from the pit walls were either cut (roots) or dug out (RF and artefacts) and retained, to make the pit walls as linear as possible. After excavation, the pit was lined with two polyurethane bin liners, which were fitted by hand to the shape of the pit. A method to measure the volume of the irregular shaped pit was devised using spherical glass beads of known weight (mean weight = 5.27 g) and volume (mean volume = 2.13 ml) to fill the pit.

**Figure 4.3** An area approximately 30 cm in diameter was cleared of loose litter and vegetation before excavating each soil pit to a minimum depth of 40 cm.
Glass beads were chosen because of a) their hardness and resistance to deformation under pressure, b) their ease of portability without spillage, and c) their reusability at all study sites. After lining the pit with two polyurethane bin-liners (of negligible volume relative to the pit volume), a 45 litre hard-plastic tote containing a known weight of glass beads was poured into the pit, stopping periodically to settle the beads by hand to ensure that they were filling the pit completely. The beads were added until they were brought above the surface level of the pit. A straight-edge ruler was used to remove any excess beads and ensure the level of the beads matched the level of the uneven pit surface. Individual beads were added and adjusted with the straight-edge ruler where necessary to fill holes in the surface layer, until the surface layer undulations matched those of the pit edge. The remaining beads in the 45 litre tote were then weighed using a digital scale (Salter 1060, accurate to 2 g) placed on a levelled rigid tray (Figure 4.4). The weight of the left-over beads was then subtracted from the total weight of all the beads, which enabled the calculation of the pit volume using the following equation:

\[
V_{pit} = \frac{(W_{gb} - W_r)}{P_{gb}} \times V_{sb}
\]

Eq. 4.1

Where:

\( V_{pit} \) = Volume of the pit (ml)

\( W_{gb} \) = Total weight of all glass beads (g)

\( W_r \) = Weight of all beads remaining after filling the pit (g)

\( W_{sb} \) = Mean weight of a single glass bead (5.27 g)

\( V_{sb} \) = Mean volume of a single glass bead (2.13 ml)

\( P_{gb} \) = Packing density of glass beads (0.55 or 55 %)
Figure 4.4 Example of glass beads being used to measure the volume of an irregular shaped soil pit. The soil pits were double-lined with polyurethane bin liners and the volume of the pit was calculated using a known weight and volume of glass beads. The straight-edge ruler in the figure (between the tray and the pit) is 30 cm long.

Subsequently, Pit B was further excavated and widened to an area approximately 1 m × 0.8 m to allow for sampling of the soil BD using the cores (per the schematic in Figure 4.5). Three samples were taken from the middle of each 10 cm depth increment from 0–40 cm depth, for a total of 12 BD core samples per pit. The BD samples were capped and kept intact within the core for transport back to the laboratory.
Figure 4.5 Schematic of the forest soil pit bulk density (BD) sampling at 10 cm depth increments, from a depth of 40 cm. The samples were taken diagonally (to reduce compaction) starting at the bottom and moving up the profile to avoid soil contamination from above when removing the 5 cm diameter, 100 cm$^3$ cores (Eijkelkamp, Netherlands).

4.3.3 Laboratory methods

4.3.3.1 Glass bead packing density

The packing density ($\rho_{gb}$) of the glass beads (1.5875 cm diameter) used in the field was first determined in the laboratory by filling a 250 ml beaker with glass beads 10 times and using the mean number of beads multiplied by their mean volume divided by the beaker volume, as in the equation below.
\[
\rho_{gb} = \frac{(X_{gb} \ast V_{gb})}{V_{bk}}
\]
Eq. 4.2

Where:
- \(\rho_{gb}\) = Packing density of glass beads
- \(X_{gb}\) = Mean number of glass beads that fit into the beaker
- \(V_{gb}\) = Mean volume of 1 glass bead
- \(V_{bk}\) = Volume of the beaker

The mean value found for \(X_{gb}\) was 65, while the mean value found for \(V_{gb}\) was 2.13 ml. Therefore the value for \(\rho_{gb}\) using the 2.13 ml glass beads was found to be 0.55, or 55 %. This value was used when measuring the volume of the pit per the method in Eq. 4.1.

4.3.3.2 Soil sample bulk density and SOC concentration measurement

On arrival at the laboratory the soil samples from the cores were placed in paper bags and oven-dried (OD) to constant weight at 105 °C. The samples were then weighed to obtain their OD mass. The samples were then broken by hand and any visible coarse fraction (i.e. > 2 mm) such as RF or roots, were removed. The size of the RF found inside the core were less than the 5 cm core diameter along their longest axis, and hereafter are referred to as “RF<5”. The RF<5 were thoroughly cleaned with a brush to remove any adhered fine earth. The soil samples were then broken using a wooden rolling-pin before being sieved through a 2 mm aperture to separate the remaining fine and coarse fractions. The mass of the fine earth (soil < 2 mm) and RF measuring 0.2–2, 2–5, 5–20 and > 20 cm (along their longest axis) were recorded. The volume of all the RF class sizes were determined by the water displacement method using a fixed volume
of water in a graduated cylinder, or using a 45 litre plastic container for the large RF (Appendix 1.5).

### 4.3.4 Soil bulk density calculation methods

The BD of each sample was determined using the \( \rho_{\text{total}}, \rho_{\text{fe}}, \) and \( \rho_{\text{hybrid}} \) methods below.

\[
\rho_{\text{total}} = \frac{M_{\text{sample}}}{V_{\text{sample}}} \quad \text{Eq. 4.3}
\]

\( \rho_{\text{total}} \) = Bulk density of the soil sample  
\( M_{\text{sample}} \) = Mass of the OD soil sample (fine earth and coarse fraction)  
\( V_{\text{sample}} \) = Volume of the soil sample (e.g. in this study, core samples = 100 cm\(^3\))

\[
\rho_{\text{fe}} = \frac{M_{\text{fine earth}}}{V_{\text{fine earth}}} \quad \text{Eq. 4.4}
\]

\( \rho_{\text{fe}} \) = Bulk density of the fine earth fraction of the soil sample  
\( M_{\text{fine earth}} \) = Mass of the OD fine earth fraction of the soil sample  
\( V_{\text{fine earth}} \) = Volume of the OD fine earth fraction of the soil sample

\[
\rho_{\text{hybrid}} = \frac{M_{\text{fine earth}}}{V_{\text{sample}}} \quad \text{Eq. 4.5}
\]

\( \rho_{\text{hybrid}} \) = Hybrid bulk density: Excludes RF mass, but includes RF volume  
\( M_{\text{fine earth}} \) = Mass of the OD fine earth fraction of the core sample  
\( V_{\text{sample}} \) = Volume of the soil sample

Once the mass and volume of the soil and RF<5 in the core samples were measured and the core-\( \rho_{\text{total}} \), core-\( \rho_{\text{fe}} \), and core-\( \rho_{\text{hybrid}} \) calculated, the SOC concentration in the sample was determined. The SOC concentration analysis was done using the same LOI method as described in chapter 3. Using a LOI temperature not exceeding 550 °C allowed for the segregation of SOC and soil inorganic C during the combustion process, per Hoffmann et al. 2014.
The soil samples from the pits were air-dried for seven days before they were initially hand-sorted into their coarse and fine fractions. The coarse and fine fractions samples were then spread out on large metal trays and oven-dried to constant weight at 105 °C. The mass of the roots found in the seven pit samples ranged from 0.06 to 1.52 % of the total sample mass (mean = 0.53 ±0.005 %), and the root volume ranged from 0.11 to 2.4 % of the total sample volume (mean = 1.02 ±0.009 %), and were not analysed further. Both the soil fine and coarse fraction mass and volume were measured and equations 4.3, 4.4, and 4.5 were then used to calculate the pit-$\rho_{\text{total}}$, pit-$\rho_{\text{fe}}$, and pit-$\rho_{\text{hybrid}}$.

4.3.5 Method for adjustment of soil bulk density measured with cores to take account of large rock fragments

A method, hereafter called “core-scaling”, was devised for adjusting the soil BD measured with samples from the 5 cm diameter cores to take account of the RF found in an excavated pit that were larger (along its longest axis) than the core diameter, hereafter “RF>5”. This method is applicable to cores of any dimensions and the RF sampled by them, simply by determining the RF dimension limits of the cores used.

In this study the core-scaling method uses:

i) the core samples with the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods (Eq. 4.3–4.5)

ii) the total volume of the associated pit

iii) the mass and volume of the RF>5, to adjust the soil BD to reflect their further dilution effect on the relative fine earth volume, and more accurately estimate the SOCD

The mass and volume of the fine earth fraction found in the 12 core samples is first scaled up by multiplying both by the result of the pit volume divided by the volume of
the core, i.e., 100 cm$^3$, as in Eq. 4.6. The results of those calculations are used to reflect
the fine earth fraction in the pit, which has already been corrected for the RF<5 mass
and volume, as if there were no RF>5 present. Then the mass and volume of the RF>5
found in the pit is subtracted from the scaled up fine earth fraction, as in the $\rho_{fe}$ method
(Eq. 4.6). The “scaled-$\rho_{fe}$” is calculated per the method example below:

1. The mean ($n = 12$) mass of the < 2 mm oven-dried soil found in the cores = 60 g.
2. The mean ($n = 12$) volume of the < 2 mm oven-dried soil found in the cores = 80
   cm$^3$.
3. After hand sorting the pit samples to separate the RF>5, then measure their oven-dry
   mass (X) and volume (Y). In this example, X = 5,000 g, and Y = 2,000 cm$^3$.
4. Scaling up for a pit, e.g., of volume = 30,000 cm$^3$, the pit represents a 300 fold
   increase over the volume sampled by the 100 cm$^3$ cores. Therefore, multiply the
   mass of the soil fine fraction (i.e. 60 g) and the volume of the fine fraction (i.e. 80
   cm$^3$) by 300 as shown in the example below.
5. Then using the equation 4.4 (for $\rho_{fe}$), recalculate the soil BD. In this example the
   resulting scaled-$\rho_{fe}$ = 0.59 g cm$^{-3}$, which is 21 % lower than the $\rho_{fe}$ as determined by
   the core method.

\[
\text{Core-}$\rho_{fe} = \frac{60 \, \text{g}}{80 \, \text{cm}^3} = 0.75 \, \text{g cm}^{-3}
\]

\[
\text{Scaled-}$\rho_{fe} = \frac{60 \, \text{g} \times 300 - \text{Mass RF}>5}{80 \, \text{cm}^3 \times 300 - \text{Volume RF}>5} = \frac{18,000 \, \text{g} - X \, \text{g}}{24,000 \, \text{cm}^3 - Y \, \text{cm}^3}
\]

E.g., $X = 5,000$ g, $Y = 2,000$ cm$^3$

\[
\text{Scaled-}$\rho_{fe} = \frac{18,000 \, \text{g} - 5,000 \, \text{g}}{24,000 \, \text{cm}^3 - 2,000 \, \text{cm}^3} = \frac{13,000}{22,000} = 0.59 \, \text{g cm}^{-3}
\]
Similar steps are followed for calculating a “scaled-$\rho_{\text{total}}$“ and a “scaled-$\rho_{\text{hybrid}}$“ BD with the appropriate inclusion or exclusion of the core sample coarse fraction mass and volume, per the soil BD calculation method chosen (e.g. Eq. 4.3 and Eq. 4.5). These steps were followed for each of the three pits excavated at the CHR, GLR, WTA, WTB, MAC, DKK and CMM sites, and the mean ($n = 3$) scaled-$\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ was used in the calculation of the soil organic carbon density (SOCD) for the site.

4.3.6 Estimation of the soil organic carbon density using the core, pit, and core-scaling BD methods

The SOCD (Mg C ha$^{-1}$) for each of the 10 SRF sites sampled using the core method were estimated using the SOC concentration and the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ soil BD for four depth increments (i.e. 0–10, 10–20, 20–30, 30–40 cm), and then summed for the total 0–40 cm SOCD. The SOCD for each of the seven sites sampled using the pit method were calculated using the mean SOC concentration for the 0–40 cm depth, as obtained from the core samples, and using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ soil BD for each pit. Lastly, the SOCD for the seven pit sites were calculated using the mean SOC concentration, as above for the pits, and the core-scaling BD method applied to the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ for each pit. Please refer to Chapter 3, section 3.2.3.2 for the detailed steps involved in calculating the SOC concentration (%) and the SOCD calculation method.

4.4 Results

The soil BD, stoniness, SOC concentration, and SOCD results for the 10 sites sampled via the core method and the seven sites sampled using the pit method are presented in Tables 4.2–4.6 and Figures 4.6–4.12, respectively. The results of the soil BD, as
measured by the core and pit methods to a depth of 40 cm are presented first. The soil stoniness, i.e. the measured OD mass and volume of the RF found in the core and pit samples are then reviewed. The percentage of the mass and volume associated with the RF relative to the total soil sample is also presented to allow comparisons between the two sampling methods, and also to show their respective influence on the three BD calculation methods, i.e., $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$. Next, the SOC concentration results for all 10 sites are shown. The effects of $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$ on the SOCD results are then traceable to the varying impact of the soil stoniness on the BD values for each of the sampled sites.

4.4.1 Soil bulk density of core samples calculated using the $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$ methods

The mean (n = 3) values for core-{$\rho_{\text{total}}$}, calculated for each 10 cm depth increment, and the mean (n = 12) for the 0–40 cm depth, are $\geq$ core-{$\rho_{\text{fc}}$}, which are $\geq$ core-{$\rho_{\text{hybrid}}$} for all 10 sites (Table 4.2). The brown podzolic CHR site had the highest core-{$\rho_{\text{total}}$} of 1.30 g cm$^{-3}$ for the 0–40 cm depth, while the lowest core-{$\rho_{\text{total}}$} of 0.80 g cm$^{-3}$ for the same depth was found at the Surface-water gley DKB site. The mean core-{$\rho_{\text{total}}$} values for the 0–10, 10–20, and 20–30 cm depths increased successively down the soil profile, from 0.93 g cm$^{-3}$ in the top 10 cm to 1.20 g cm$^{-3}$ at the 20–30 cm depth. The 20–30 and 30–40 cm depth core-{$\rho_{\text{total}}$} values were the same (1.20 g cm$^{-3}$), though the standard deviation (SD) doubled from $\pm$0.09 to $\pm$0.18 g cm$^{-3}$, respectively.

The mean core-{$\rho_{\text{fc}}$} of the 10 sampled sites for the 0–40 cm depth ranged between 1.12 g cm$^{-3}$ at the Ground-water gley CLY site and 0.79 g cm$^{-3}$ at the Surface-water gley DKB site. For each of the four sampled depth increments (0–10, 10–20, 20–30, 30–40 cm) the mean core-{$\rho_{\text{fc}}$} of the 10 sites increased successively from 0.79 g cm$^{-3}$ in
the top layer to 1.08 g cm$^{-3}$ in the bottom layer. At five of the 10 sites (GLR, DKB, DKK, DKN, and WTA) the core-$\rho_{fe}$ increased for each successive depth down the soil profile from 0–40 cm (Table 4.2). The five other sites did not follow the often observed increasing BD by depth increment, but instead had varying BD by depth. For example, the CHR site had a higher core-$\rho_{fe}$ of 0.90 ±0.21 g cm$^{-3}$ at the 0–10 cm depth than the 0.86 ±0.13 g cm$^{-3}$ found at the 30–40 cm depth.

The CLY site had the highest core-$\rho_{hybrid}$ of 1.11 g cm$^{-3}$, while the brown podzolic GLR site had the lowest 0.63 g cm$^{-3}$ for the 0–40 cm depth. As with the mean core-$\rho_{fe}$ values of each successive 10 cm depth increment down the soil profile for the 10 sites, the mean core-$\rho_{hybrid}$ increased from 0.71 ±0.14 g cm$^{-3}$ in the top layer to 0.98 ±0.15 g cm$^{-3}$ in the bottom layer. Only three sites, GLR, WTA, and DKN, had successively increasing core-$\rho_{hybrid}$ for all depth increments from the top 0–10 cm depth to the bottom 30–40 cm depth (Table 4.2). The other seven sites had decreases in core-$\rho_{hybrid}$ for at least one successive depth increment, e.g. the 0–10 cm depth at brown podzolic MAC site had a core-$\rho_{hybrid}$ value of 0.82 ±0.26 g cm$^{-3}$ while the 10–20 cm depth value was 0.59 ±0.11 g cm$^{-3}$.

The Alluvial gley DKN site had the most homogeneous core-$\rho_{total}$, core-$\rho_{fe}$, and core-$\rho_{hybrid}$ results with the mean SD ranging from 0.03 g cm$^{-3}$ (core-$\rho_{fe}$ and core-$\rho_{hybrid}$) to 0.04 g cm$^{-3}$ (core-$\rho_{total}$), and the mean coefficient of variation (CV) = 3 for all three methods. Conversely, the GLR site had the most heterogeneous core-$\rho_{total}$, core-$\rho_{fe}$, and core-$\rho_{hybrid}$ results with the mean SD ranging from 0.12 g cm$^{-3}$ (core-$\rho_{hybrid}$) to 0.28 g cm$^{-3}$ (core-$\rho_{total}$), and the CV ranging from 22 to 30, also for core-$\rho_{hybrid}$ and core-$\rho_{total}$ respectively.
Table 4.2 The soil bulk density (BD), standard deviation (SD), and coefficient of variation (CV), sampled by core method and calculated using the \(\rho\)total, \(\rho\)fe, and \(\rho\)hybrid methods (Throop et al. (2012), for four 10 cm depths, and the mean values for 0–40 cm depth.

<table>
<thead>
<tr>
<th>Method</th>
<th>Site</th>
<th>Depth 0-10</th>
<th>Depth 10-20</th>
<th>Depth 20-30</th>
<th>Depth 30-40</th>
<th>Mean 0-40</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>BD</td>
<td>SD</td>
<td>CV</td>
<td>BD</td>
<td>SD</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>CHR</td>
<td>1.28</td>
<td>0.29</td>
<td>22</td>
<td>1.35</td>
<td>0.27</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.90</td>
<td>0.21</td>
<td>23</td>
<td>1.16</td>
<td>0.16</td>
<td>14</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.61</td>
<td>0.14</td>
<td>24</td>
<td>0.92</td>
<td>0.17</td>
<td>18</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>GLR</td>
<td>0.67</td>
<td>0.40</td>
<td>59</td>
<td>1.03</td>
<td>0.39</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.51</td>
<td>0.27</td>
<td>53</td>
<td>0.69</td>
<td>0.18</td>
<td>26</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.45</td>
<td>0.22</td>
<td>49</td>
<td>0.54</td>
<td>0.09</td>
<td>17</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>WTB</td>
<td>1.18</td>
<td>0.07</td>
<td>6</td>
<td>1.23</td>
<td>0.03</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.90</td>
<td>0.09</td>
<td>10</td>
<td>1.00</td>
<td>0.12</td>
<td>12</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.74</td>
<td>0.09</td>
<td>12</td>
<td>0.84</td>
<td>0.15</td>
<td>18</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>WTA</td>
<td>1.00</td>
<td>0.17</td>
<td>17</td>
<td>1.06</td>
<td>0.08</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.72</td>
<td>0.16</td>
<td>23</td>
<td>0.81</td>
<td>0.06</td>
<td>8</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.60</td>
<td>0.12</td>
<td>20</td>
<td>0.68</td>
<td>0.04</td>
<td>6</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>MAC</td>
<td>1.20</td>
<td>0.17</td>
<td>14</td>
<td>0.79</td>
<td>0.26</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.97</td>
<td>0.13</td>
<td>13</td>
<td>0.66</td>
<td>0.17</td>
<td>26</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.82</td>
<td>0.26</td>
<td>32</td>
<td>0.59</td>
<td>0.11</td>
<td>18</td>
</tr>
<tr>
<td>(\rho)total</td>
<td>DKK</td>
<td>0.78</td>
<td>0.08</td>
<td>11</td>
<td>1.07</td>
<td>0.08</td>
</tr>
<tr>
<td>(\rho)fe</td>
<td>0.78</td>
<td>0.08</td>
<td>10</td>
<td>0.93</td>
<td>0.08</td>
<td>8</td>
</tr>
<tr>
<td>(\rho)hybrid</td>
<td>0.77</td>
<td>0.07</td>
<td>9</td>
<td>0.83</td>
<td>0.07</td>
<td>8</td>
</tr>
</tbody>
</table>
Table 4.2, continued

<table>
<thead>
<tr>
<th>Method</th>
<th>Site</th>
<th>Depth</th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>0-10</td>
<td>10-20</td>
<td>20-30</td>
<td>30-40</td>
<td>Mean</td>
<td>BD</td>
<td>SD</td>
<td>CV</td>
<td>BD</td>
<td>SD</td>
<td>CV</td>
</tr>
<tr>
<td>$\rho_{\text{total}}$</td>
<td>CMM</td>
<td>0.90</td>
<td>0.09</td>
<td>10</td>
<td>0.80</td>
<td>0.26</td>
<td>32</td>
<td>0.95</td>
<td>0.03</td>
<td>3</td>
<td>1.15</td>
<td>0.19</td>
</tr>
<tr>
<td>$\rho_{fe}$</td>
<td></td>
<td>0.87</td>
<td>0.09</td>
<td>10</td>
<td>0.78</td>
<td>0.27</td>
<td>34</td>
<td>0.94</td>
<td>0.03</td>
<td>4</td>
<td>1.13</td>
<td>0.18</td>
</tr>
<tr>
<td>$\rho_{\text{hybrid}}$</td>
<td></td>
<td>0.85</td>
<td>0.09</td>
<td>10</td>
<td>0.78</td>
<td>0.26</td>
<td>34</td>
<td>0.94</td>
<td>0.03</td>
<td>4</td>
<td>1.13</td>
<td>0.18</td>
</tr>
<tr>
<td>$\rho_{\text{total}}$</td>
<td>DKB</td>
<td>0.61</td>
<td>0.02</td>
<td>3</td>
<td>0.78</td>
<td>0.03</td>
<td>4</td>
<td>0.87</td>
<td>0.09</td>
<td>10</td>
<td>0.94</td>
<td>0.48</td>
</tr>
<tr>
<td>$\rho_{fe}$</td>
<td></td>
<td>0.61</td>
<td>0.02</td>
<td>3</td>
<td>0.77</td>
<td>0.03</td>
<td>4</td>
<td>0.87</td>
<td>0.09</td>
<td>10</td>
<td>0.90</td>
<td>0.42</td>
</tr>
<tr>
<td>$\rho_{\text{hybrid}}$</td>
<td></td>
<td>0.61</td>
<td>0.02</td>
<td>3</td>
<td>0.77</td>
<td>0.03</td>
<td>4</td>
<td>0.87</td>
<td>0.09</td>
<td>10</td>
<td>0.86</td>
<td>0.35</td>
</tr>
<tr>
<td>$\rho_{\text{total}}$</td>
<td>DKN</td>
<td>0.74</td>
<td>0.04</td>
<td>5</td>
<td>0.96</td>
<td>0.02</td>
<td>2</td>
<td>1.24</td>
<td>0.06</td>
<td>5</td>
<td>1.33</td>
<td>0.02</td>
</tr>
<tr>
<td>$\rho_{fe}$</td>
<td></td>
<td>0.74</td>
<td>0.04</td>
<td>5</td>
<td>0.96</td>
<td>0.02</td>
<td>2</td>
<td>1.20</td>
<td>0.05</td>
<td>4</td>
<td>1.32</td>
<td>0.02</td>
</tr>
<tr>
<td>$\rho_{\text{hybrid}}$</td>
<td></td>
<td>0.74</td>
<td>0.04</td>
<td>5</td>
<td>0.96</td>
<td>0.02</td>
<td>2</td>
<td>1.16</td>
<td>0.03</td>
<td>3</td>
<td>1.30</td>
<td>0.03</td>
</tr>
<tr>
<td>$\rho_{\text{total}}$</td>
<td>CLY</td>
<td>0.94</td>
<td>0.31</td>
<td>33</td>
<td>1.19</td>
<td>0.09</td>
<td>8</td>
<td>1.22</td>
<td>0.06</td>
<td>5</td>
<td>1.14</td>
<td>0.22</td>
</tr>
<tr>
<td>$\rho_{fe}$</td>
<td></td>
<td>0.94</td>
<td>0.31</td>
<td>33</td>
<td>1.18</td>
<td>0.09</td>
<td>7</td>
<td>1.21</td>
<td>0.06</td>
<td>5</td>
<td>1.14</td>
<td>0.22</td>
</tr>
<tr>
<td>$\rho_{\text{hybrid}}$</td>
<td></td>
<td>0.94</td>
<td>0.31</td>
<td>33</td>
<td>1.16</td>
<td>0.08</td>
<td>7</td>
<td>1.21</td>
<td>0.05</td>
<td>4</td>
<td>1.14</td>
<td>0.22</td>
</tr>
</tbody>
</table>

$\rho_{\text{total mean}}$  $0.93$  0.16  0  $1.03$  0.15  1  $1.20$  0.09  8  $1.20$  0.18  16  $1.09$
$\rho_{fe}$ mean  $0.79$  0.14  18  $0.89$  0.12  14  $1.06$  0.10  10  $1.08$  0.18  17  $0.96$
$\rho_{\text{hybrid mean}}$  $0.71$  0.14  20  $0.81$  0.10  13  $0.95$  0.09  10  $0.98$  0.15  16  $0.86$
4.4.2 Soil bulk density of pit samples calculated using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods

As with the soil BD results by site for the core method, the BD results for the quantitative pit method, and calculated using the $\rho_{\text{total}}$ method were $\geq$ to those using $\rho_{\text{fe}}$, which were $\geq$ to those using $\rho_{\text{hybrid}}$ (Figure 4.6). The pit-$\rho_{\text{total}}$ ranged from 0.95 g cm$^{-3}$ at the brown podzolic CMM site, to 1.48 g cm$^{-3}$ at the GLR site, with the mean pit-$\rho_{\text{total}}$ for the seven sites being 1.19 g cm$^{-3}$. The Surface-water gley DKK site had the highest pit-$\rho_{\text{fe}}$ of 0.97 g cm$^{-3}$ whereas the GLR site had the lowest pit-$\rho_{\text{fe}}$ of 0.66 g cm$^{-3}$, while the mean pit-$\rho_{\text{fe}}$ for the seven sites was 0.86 g cm$^{-3}$. The pit-$\rho_{\text{hybrid}}$ results were the reverse of the pit-$\rho_{\text{total}}$ results, with the lowest pit-$\rho_{\text{hybrid}}$, 0.35 g cm$^{-3}$ at the GLR site, to the highest, 0.93 g cm$^{-3}$ at the CMM site, with a mean pit-$\rho_{\text{hybrid}}$ of 0.69 g cm$^{-3}$.

![Figure 4.6](image-url)

**Figure 4.6** The 0–40 cm soil BD (g cm$^{-3}$) for the seven pit sampled sites using analysis of the mass and volume of both the soil fine and coarse fraction from one quantitative pit per site, and calculated using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods. Also, the mean $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ values for the seven sites sampled.
4.4.3 Comparison of the mean soil bulk density using two field sampling and three calculation methods

The soil BD for the seven sites estimated by both the core and the pit field methods, and calculated using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods were compared (Figure 4.7a, b, and c, respectively). The mean core-$\rho_{\text{total}}$ for the 0–40 cm depth, as measured by the core (1.09 g cm$^{-3}$) and pit (1.19 g cm$^{-3}$) methods have the highest values across all of the sites, followed in descending order by the $\rho_{\text{fe}}$ and $\rho_{\text{hybrid}}$ results. The core samples of the CHR site had the highest $\rho_{\text{total}}$ of 1.30 g cm$^{-3}$, while the core samples from the DKB site had the lowest $\rho_{\text{total}}$ of 0.80 g cm$^{-3}$. The GLR pit had the highest $\rho_{\text{total}}$ of 1.48 g cm$^{-3}$, while the CMM site had the lowest, i.e. 0.95 g cm$^{-3}$. Only the DKK site had a higher mean $\rho_{\text{total}}$ value (1.21 g cm$^{-3}$) from the core samples than that found in the pit sample (1.09 g cm$^{-3}$). For the seven sites sampled with both methods, all the core method $\rho_{\text{fe}}$ and $\rho_{\text{hybrid}}$ results were ≥ their counterpart pit results.

The mean core-$\rho_{\text{fe}}$ BD for the 0–40 cm depth ranged from 0.79 g cm$^{-3}$ at the DKB site to 1.12 g cm$^{-3}$ at the CLY site, whereas the pit-$\rho_{\text{fe}}$ ranged from 0.66 g cm$^{-3}$ at the GLR site to 0.97 g cm$^{-3}$ at the DKK site. The mean core-$\rho_{\text{fe}}$ for the 10 sites was 0.96 g cm$^{-3}$, while the mean pit-$\rho_{\text{fe}}$ was approximately 10 % lower at 0.86 g cm$^{-3}$. The CHR site, at 26 % had the largest % difference between the core- and pit-$\rho_{\text{fe}}$ values, 1.03 and 0.76 g cm$^{-3}$ respectively, while the smallest (0 %) occurred at the WTA site where both methods resulted in a value of 0.83 g cm$^{-3}$. The CLY site had the highest mean core-$\rho_{\text{hybrid}}$ value, i.e. 1.11 g cm$^{-3}$. The seven pit sampled sites had $\rho_{\text{hybrid}}$ values ranging from 0.35 g cm$^{-3}$ at the GLR site to 0.92 g cm$^{-3}$ at the CMM site with a mean of 0.69 g cm$^{-3}$. Of the seven sites sampled by both core and pit methods, the DKK site had the highest $\rho_{\text{hybrid}}$ values of i.e. 1.00 g cm$^{-3}$ and 0.89 g cm$^{-3}$ respectively.
Figure 4.7a,b,c Comparison of $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ (g cm$^{-3}$) for the 0–40 cm sampling depth using the core and pit methods, with standard deviation error bars for the core samples and the means for both methods. The results for the three almost stone-free sites (i.e. DKB, DKN, CLY) sampled using the core method are also presented.
The CMM site, which had the lowest RF content, also had the smallest range in values, i.e., 0.92 to 0.95 g cm\(^{-3}\) for core-\(\rho\)\(_{\text{hybrid}}\) and core-\(\rho\)\(_{\text{total}}\), respectively. The CMM site was also the only one of the seven sites to have the same \(\rho\)\(_{\text{total}}\) and \(\rho\)\(_{\text{fe}}\) results for both the core and pit methods. Due to the different treatment of the RF fraction relative to the whole soil sample in the methods, the pit-\(\rho\)\(_{\text{fe}}\) and pit-\(\rho\)\(_{\text{hybrid}}\) results are inversely correlated with the pit-\(\rho\)\(_{\text{total}}\) results, with the coefficient of determination (\(R^2\)) = 0.69 and 0.88, respectively (Figure 4.8 a and b). The best-fit correlation was the linear function.

![Figure 4.8a,b](image)

Figure 4.8a,b The best-fit linear correlation of pit-\(\rho\)\(_{\text{total}}\) a) pit-\(\rho\)\(_{\text{hybrid}}\) and b) to pit-\(\rho\)\(_{\text{fe}}\) results for the seven sites where the quantitative pit sampling method was used.
4.4.4 Rock fragment contents of core and pit samples

The mean OD mass of RF found in the cores taken at each of the 10 sites sampled using the core method was 22.8 g, and ranged from 1.1 g at the CLY site to 52.8 g at the CHR site (Table 4.3). The percentage of the total OD sample mass accounted for by the RF ranged from 1 to 43 % at the CLY and CHR sites, respectively, with a mean for the 10 sites of 20 %. The mean volume of RF in the core samples ranged from 0.4 cm$^3$ at the CLY site to 24.5 cm$^3$ at the CHR site, with an overall mean for the 10 sites of 9.8 cm$^3$ per 100 cm$^3$ core sample. The CHR site had the highest mean percentage volume of RF (25 %) in the core samples, while the CLY site had the lowest (0 %). Therefore the CHR and CLY sites, as measured by the core method, had the highest and lowest levels of stoniness by mass and volume, respectively.
Table 4.3 Mean OD mass (g), volume (cm$^3$), and percentage of total sample mass and volume of the rock fragments (RFs) found in the 100 cm$^3$ core samples, for each of the 10 SRF sites, plus the mean values for all 10 sites.

<table>
<thead>
<tr>
<th>Sites</th>
<th>CHR</th>
<th>GLR</th>
<th>WTB</th>
<th>WTA</th>
<th>MAC</th>
<th>DKK</th>
<th>CMM</th>
<th>DKB</th>
<th>DKN</th>
<th>CLY</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean RF mass (g)</td>
<td>52.8</td>
<td>47.4</td>
<td>38.0</td>
<td>34.6</td>
<td>26.8</td>
<td>21.0</td>
<td>2.3</td>
<td>2.1</td>
<td>2.0</td>
<td>1.1</td>
<td>22.8</td>
</tr>
<tr>
<td>RF mass S.D. (±)</td>
<td>25.3</td>
<td>26.6</td>
<td>8.5</td>
<td>13.1</td>
<td>21.5</td>
<td>14.7</td>
<td>1.7</td>
<td>3.2</td>
<td>2.1</td>
<td>1.6</td>
<td>11.8</td>
</tr>
<tr>
<td>Mean RF mass % of total mass</td>
<td>43</td>
<td>41</td>
<td>24</td>
<td>32</td>
<td>33</td>
<td>17</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td>1</td>
<td>20</td>
</tr>
<tr>
<td>Mean RF volume (cm$^3$)</td>
<td>24.5</td>
<td>18.9</td>
<td>15.5</td>
<td>14.4</td>
<td>12.1</td>
<td>9.2</td>
<td>1.1</td>
<td>1.1</td>
<td>0.9</td>
<td>0.4</td>
<td>9.8</td>
</tr>
<tr>
<td>RF volume S.D. (±)</td>
<td>11.8</td>
<td>10.6</td>
<td>3.5</td>
<td>5.4</td>
<td>9.7</td>
<td>6.4</td>
<td>0.8</td>
<td>1.2</td>
<td>0.9</td>
<td>0.7</td>
<td>5.1</td>
</tr>
<tr>
<td>Mean RF vol. % of total vol.</td>
<td>25</td>
<td>19</td>
<td>16</td>
<td>14</td>
<td>12</td>
<td>9</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>10</td>
</tr>
</tbody>
</table>

In contrast to the core samples, the total mass of RF in the seven pits ranged from 0.49 kg at the CMM site to 30.30 kg at the GLR site. The percentage of the total OD sample mass accounted for by the RF in the seven pits ranged from 2 to 76 %, at the CMM and GLR sites, respectively (Table 4.4). The mass of the RF>5 fraction (i.e. the RF fraction too large to be sampled within the 5 cm diameter cores), accounted for between 0 to 65 % of the total OD sample mass, with the CMM and GLR sites the lowest and highest, respectively. When the mass of the RF>5 fraction was considered as a percentage of the total RF mass in the pit samples, they accounted for 0 to 86 %, again at the CMM and GLR sites, respectively. The volume of the RF excavated from the pits ranged from 0.21 l at the CMM site to 12.35 l at the GLR sites, with a mean for the seven sites of 5.05 l.
Whereas the CHR and CLY sites had the highest and lowest percentage volume of RF in the core samples, 25 and 1 % respectively, the GLR site had the highest percentage volume (46 %) of RF in the pits sampled, while the CMM site had the lowest (1 %). The CMM and WTA sites had the lowest percentage (0 %) of the total pit volume occupied by the RF>5 fraction, while the GLR site had the highest percentage (39 %). Following a similar pattern, the pit samples from the CMM and GLR sites had the highest and lowest values for the percentage of the total RF volume accounted for by the RF>5 fraction, 0 and 84 %, respectively, with a mean of 29 % for all seven sites.

**Table 4.4** Total rock fragment (RF) mass (kg), volume (l), and percentage of total sample mass and volume of the RF, for each of the seven sites sampled using the pits. Also, the mean values for all seven sites.

<table>
<thead>
<tr>
<th>Pits:</th>
<th>CHR</th>
<th>GLR</th>
<th>WTB</th>
<th>WTA</th>
<th>MAC</th>
<th>DKK</th>
<th>CMM</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>RF mass (kg)</td>
<td>18.48</td>
<td>30.30</td>
<td>9.20</td>
<td>7.91</td>
<td>11.76</td>
<td>4.97</td>
<td>0.49</td>
<td>11.87</td>
</tr>
<tr>
<td>RF mass (% of total mass)</td>
<td>64</td>
<td>76</td>
<td>36</td>
<td>36</td>
<td>38</td>
<td>18</td>
<td>2</td>
<td>39</td>
</tr>
<tr>
<td>RF&gt;5 mass (% of total mass)</td>
<td>29</td>
<td>65</td>
<td>2</td>
<td>1</td>
<td>13</td>
<td>6</td>
<td>0</td>
<td>17</td>
</tr>
<tr>
<td>RF&gt;5 (% of total RF mass)</td>
<td>46</td>
<td>86</td>
<td>5</td>
<td>2</td>
<td>34</td>
<td>30</td>
<td>0</td>
<td>29</td>
</tr>
<tr>
<td>RF volume (l)</td>
<td>7.30</td>
<td>12.35</td>
<td>4.11</td>
<td>3.53</td>
<td>5.90</td>
<td>1.98</td>
<td>0.21</td>
<td>5.05</td>
</tr>
<tr>
<td>RF volume (% of total volume)</td>
<td>35</td>
<td>46</td>
<td>19</td>
<td>17</td>
<td>22</td>
<td>8</td>
<td>1</td>
<td>21</td>
</tr>
<tr>
<td>RF&gt;5 volume (% of total volume)</td>
<td>17</td>
<td>39</td>
<td>1</td>
<td>0</td>
<td>8</td>
<td>2</td>
<td>0</td>
<td>10</td>
</tr>
<tr>
<td>RF&gt;5 volume (% of RF volume)</td>
<td>49</td>
<td>84</td>
<td>5</td>
<td>2</td>
<td>34</td>
<td>30</td>
<td>0</td>
<td>29</td>
</tr>
</tbody>
</table>
The mass and volume of the four RF size classes (i.e. 0.2–2, 2–5, 5–20 and > 20 cm) from the pit samples were analyzed to ascertain their distribution, and their relative mass (Figure 4.09) and volume (Figure 4.10) contribution to the RF component of the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ calculations. The two lowest size classes (i.e. 2–20 mm and 20–50 mm) of RF are equivalent to those which theoretically could fit into the 5 cm diameter cores. The GLR site was the only site with RF in the > 20 cm size-class, and it also had the largest mass and volume of the 5–20 cm size-class. The CMM site only had RF in the 0.2–2 cm size-class.
**Figure 4.9** The mass (kg) of the rock fragments from the pit samples in four size classes.

**Figure 4.10** The volume (l) of the rock fragments from the pit samples in four size classes.
4.4.5 “Core-scaling” rock fragment mass and volume, and comparison of derived BD results with core and pit $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$.

Using the “core-scaling” method with the 0–40 cm depth mean mass and volume of the RF found in the core samples, “Scaled RF” mass and volume values were derived for the seven pit sites (Table 4.5). This method resulted in a “Scaled RF mass” range of 0.55 to 12.69 kg, with a mean (n = 7) of 7.41 kg. The “Scaled RF volume” ranged from 0.26 to 5.09 l, with the mean = 3.18 l. The “Scaled RF mass (% of actual RF mass)” ranged from 42 % (GLR site) to 112 % (CMM site) of the RF mass found in the respective pit samples, with a mean for the seven pit sites of 80 %. The “Scaled RF volume (% of actual RF volume)” ranged from 41 % to 121 % (also at the GLR and CMM sites, respectively) of the RF volume in the pits, with a mean of 81 %.

Table 4.5 The mean OD mass and volume of the core sample RF, from the seven sites sampled using both the core and pit methods, scaled up to match the total volume of their respective quantitative pits.

<table>
<thead>
<tr>
<th>Core data scaled up to pit volume:</th>
<th>CHR</th>
<th>GLR</th>
<th>WTB</th>
<th>Sites</th>
<th>WTA</th>
<th>MAC</th>
<th>DKK</th>
<th>CMM</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scaled RF mass (g)</td>
<td>10,940</td>
<td>12,689</td>
<td>8,343</td>
<td>7,041</td>
<td>7,111</td>
<td>5,227</td>
<td>550</td>
<td>7,414</td>
<td></td>
</tr>
<tr>
<td>Scaled RF mass (% of actual RF mass)</td>
<td>59</td>
<td>42</td>
<td>91</td>
<td>89</td>
<td>60</td>
<td>105</td>
<td>112</td>
<td>80</td>
<td></td>
</tr>
<tr>
<td>Scaled RF volume (cm$^3$)</td>
<td>5,088</td>
<td>5,055</td>
<td>3,405</td>
<td>2,934</td>
<td>3,203</td>
<td>2,283</td>
<td>256</td>
<td>3,175</td>
<td></td>
</tr>
<tr>
<td>Scaled RF volume (% of actual RF volume)</td>
<td>70</td>
<td>41</td>
<td>83</td>
<td>83</td>
<td>54</td>
<td>115</td>
<td>121</td>
<td>81</td>
<td></td>
</tr>
</tbody>
</table>

Using the core-scaling method with the “Scaled RF” data from Table 4.5, the scaled up $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$ values were calculated (hereafter, scaled-$\rho_{total}$, scaled-$\rho_{fe}$, scaled-$\rho_{hybrid}$).
and scaled-$\rho_{\text{hybrid}}$, respectively), and compared with their counterpart results for the cores and pits from the seven pit sites (Figure 4.11a, b, and c, respectively). As $\rho_{\text{total}}$ includes the RF mass and volume, and RF density exceeds fine earth density, it is expected that including the RF>5 would lead to the pit-$\rho_{\text{total}}$ being $\geq$ the core-$\rho_{\text{total}}$. For example, the GLR pit contained RF>5 that accounted for 86 % of the total RF mass and 76 % of the total soil sample mass, and when the core-scaling method adjustment for RF>5 was made, the scaled-$\rho_{\text{total}}$ value exceeded the core- and pit-$\rho_{\text{total}}$ values. This was the case for all the pits except DKK, which ranked 6$^{\text{th}}$ ($n = 7$) for overall RF mass and volume, but ranked 4$^{\text{th}}$ for RF>5 mass and volume, indicating a high proportion of medium and coarse (i.e. 6–60 mm) RF in the core samples. Except for the DKK site, the comparison of the scaled-$\rho_{\text{total}}$ with the core- and pit-$\rho_{\text{total}}$ results illustrates the under-representation of RF mass and volume in the core versus the pit samples.

The comparison of the $\rho_{\text{fe}}$ results (Figure 4.10b) show the effect of focusing only on the fine earth mass and volume, and thereby eliminating the influence of the RF>5 mass and volume on the soil BD calculation (Eq. 4.4). For all sites the core-$\rho_{\text{fe}}$ is $\geq$ the pit-$\rho_{\text{fe}}$. The scaled-$\rho_{\text{fe}}$ values were $\leq$ the core- and pit-$\rho_{\text{fe}}$ values for all sites except the DKK site, where the scaled-$\rho_{\text{fe}}$ was greater than the pit-$\rho_{\text{fe}}$. While the mean scaled-$\rho_{\text{fe}}$ for the seven pit sites, i.e. 0.80 g cm$^{-3}$, is 0.06 g cm$^{-3}$ lower than the pit-$\rho_{\text{fe}}$ (0.86 g cm$^{-3}$) it proves to be 33 % more accurate than the core-$\rho_{\text{fe}}$ (0.95 g cm$^{-3}$).

The $\rho_{\text{hybrid}}$ method, the soil BD method recommended by Throop et al. (2012) for SOCD calculations in stony soils led to the lowest BD values of the three methods compared in this study (Figure 4.10c). The two sites, with the lowest RF>5 (by percentage mass and volume of total pit mass and volume), i.e. WTA and CMM, had a core-$\rho_{\text{hybrid}}$ equal to the scaled-$\rho_{\text{hybrid}}$ results. Only the almost stone-free CMM site had a core-$\rho_{\text{hybrid}}$ less than the pit-$\rho_{\text{hybrid}}$. For the other six sites, the scaled-$\rho_{\text{hybrid}}$ was greater
than the pit-\(\rho_{\text{hybrid}}\). The scaled-\(\rho_{\text{hybrid}}\) (Figure 4.10c) for the three pits with the highest percentage mass and volume (i.e. CHR, GLR, and MAC) were 48, 86, and 47 % more accurate, respectively, than the core-\(\rho_{\text{hybrid}}\), when compared to the pit-\(\rho_{\text{hybrid}}\) results. The mean (n = 7) scaled-\(\rho_{\text{hybrid}}\) (0.75 g cm\(^{-3}\)) was 50 % more accurate than the mean core-\(\rho_{\text{hybrid}}\) (0.81 g cm\(^{-3}\)) when compared to the mean pit-\(\rho_{\text{hybrid}}\) (0.69 g cm\(^{-3}\)).

The comparison of the pit-\(\rho_{\text{total}}\), pit-\(\rho_{\text{fe}}\), and pit-\(\rho_{\text{hybrid}}\) values and their scaled-\(\rho_{\text{total}}\), scaled-\(\rho_{\text{fe}}\), and scaled-\(\rho_{\text{hybrid}}\) counterparts show a strong linear correlation between the respective data sets (Figure 4.12). The coefficient of determination (\(R^2\)) for the linear regression of the three core-scaling BD methods with their pit counterparts ranged from 0.85 for \(\rho_{\text{fe}}\) to greater than 0.94 for the \(\rho_{\text{total}}\) and \(\rho_{\text{hybrid}}\) methods.
Figure 4.11 Comparison of the soil bulk density (BD) for the 0–40 cm depth for the seven sites, derived using the combination of the core, pit and core-scaling sampling methods, and calculated using the a) $\rho_{\text{total}}$, (b) $\rho_{\text{te}}$, and (c) $\rho_{\text{hybrid}}$ methods.

121
Figure 4.12 The correlation of the pit $\rho_{\text{total}}$ (a) values to the $\rho_{\text{fe}}$ (b) and $\rho_{\text{hybrid}}$ (c) values calculated using the mean mass and volume of the soil fine and coarse fraction from the cores scaled up to match the pit volume, and adjusted for the mass and volume of the pit rock fragments larger than the core diameter, i.e., 5 cm.
4.4.6 Soil organic carbon concentration

The mean SOC concentration (%) for the 0–40 cm sampling depth ranged from 5.2 ±0.04 (all results are reported as mean ± SD) % at the brown podzolic GLR site to 3.3 ±0.01 % for the brown earth soil of the WTB site (Table 4.6). For the majority of the sites (n = 8/10) the highest SOC concentration was found in the uppermost sampled depth (0–10 cm), with the brown podzolic CMM and MAC sites being the exceptions, where the 10–20 cm depth had the highest SOC concentration. The SOC concentration decreased with each successive sampled depth below 10 cm for all the sites except GLR and CLY, where the 30–40 cm depth had a higher % than the overlying 20–30 cm depth. The three reforested former Sitka spruce plantation sites GLR, MAC, and CHR had the highest mean SOC of 4.6 ±0.03 % for the 0–40 cm depth. The five sites that were previously rough pasture (CLY, CMM, DKB, DKK, and DKN) had the next highest mean SOC concentration with 4.4 % ±0.01 %, followed by the two former Forest Service nursery sites (WTA and WTB) with a SOC of 3.8 ±0.01 % for the 0–40 cm depth.

From a soil type perspective the four brown podzolic sites (CHR, GLR, MAC and CMM) had the highest mean SOC concentration of 4.7 ±0.03 % for the 0–40 cm depth. The next highest mean SOC concentration for the 0–40 cm depth was for the four gley sites (CLY, DKB, DKK, and DKN) at 4.2 ±0.01 % SOC. The two brown earth sites (WTA, WTB) had the lowest SOC concentration at 3.8 ±0.01 % for the 0–40 cm depth. Of the tree genera in the study, the four sites planted with Alnus (DKN, CLY, CMM, and DKB) had the highest mean SOC concentration of 4.5 ±0.01 % for the 0–40 cm depth. The five sites planted with Eucalyptus (CHR, GLR, MAC, WTA, and WTB) had the next highest SOC of 4.3 ±0.03 % for the 0–40 cm depth, while the Acer site (DKK) had SOC of 4.0 ±0.02 % for the same depth.
Table 4.6 The estimated soil organic carbon (SOC) concentration (%) of the 10 short rotation forestry sites. The soil was sampled in 10 cm depth increments down to 40 cm depth.

<table>
<thead>
<tr>
<th>Sites</th>
<th>Depth</th>
<th>CHR</th>
<th>GLR</th>
<th>WTB</th>
<th>WTA</th>
<th>MAC</th>
<th>DKK</th>
<th>CMM</th>
<th>DKB</th>
<th>DKN</th>
<th>CLY</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0-10 cm</td>
<td>4.2</td>
<td>11.5</td>
<td>4.0</td>
<td>5.1</td>
<td>4.6</td>
<td>6.7</td>
<td>5.1</td>
<td>4.2</td>
<td>7.1</td>
<td>5.6</td>
<td>5.8</td>
</tr>
<tr>
<td></td>
<td>10-20 cm</td>
<td>4.1</td>
<td>5.1</td>
<td>3.4</td>
<td>4.5</td>
<td>9.3</td>
<td>5.1</td>
<td>6.9</td>
<td>4.1</td>
<td>6.0</td>
<td>3.5</td>
<td>5.2</td>
</tr>
<tr>
<td></td>
<td>20-30 cm</td>
<td>3.2</td>
<td>2.0</td>
<td>3.2</td>
<td>4.3</td>
<td>3.7</td>
<td>2.1</td>
<td>5.4</td>
<td>4.0</td>
<td>3.7</td>
<td>3.4</td>
<td>3.5</td>
</tr>
<tr>
<td></td>
<td>30-40 cm</td>
<td>2.8</td>
<td>2.2</td>
<td>2.5</td>
<td>3.6</td>
<td>2.2</td>
<td>2.0</td>
<td>2.5</td>
<td>3.6</td>
<td>2.9</td>
<td>3.9</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Mean:0-40 cm</td>
<td>3.6</td>
<td>5.2</td>
<td>3.3</td>
<td>4.4</td>
<td>5.0</td>
<td>4.0</td>
<td>5.0</td>
<td>4.0</td>
<td>4.9</td>
<td>4.1</td>
<td>4.3</td>
</tr>
</tbody>
</table>

4.4.7 Estimated core, pit, and core-scaling soil organic carbon density, calculating using the $\rho_{total}$, $\rho_{fe}$, $\rho_{hybrid}$ methods

The SOCD (Mg C ha$^{-1}$) for the 0–40 cm depth were calculated using the $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$ methods using:

i) the core method for all 10 SRF sites (Figure 4.13)

ii) the core method for just the seven pit excavation sites (Figure 4.14a)

iii) the pit method the seven pit excavation sites (Figure 4.14b)

iv) the core-scaling method applied to the $\rho_{total}$, $\rho_{fe}$, and $\rho_{hybrid}$ methods for the seven pit excavation sites (Figure 4.14c)

The mean SOCD for the 10 core sampled sites, and the seven pit and core-scaling method sites, was also calculated. To simplify the reporting of the SOCD results, the field sampling methods (i.e. core, pit, and core-scaling) and soil BD calculation
methods (i.e. $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$), are concatenated as follows: core-$\rho_{\text{total}}$-SOCD, core-$\rho_{\text{fc}}$-SOCD, and core-$\rho_{\text{hybrid}}$-SOCD; pit-$\rho_{\text{total}}$-SOCD, pit-$\rho_{\text{fc}}$-SOCD, and pit-$\rho_{\text{hybrid}}$-SOCD; scaled-$\rho_{\text{total}}$-SOCD, scaled-$\rho_{\text{fc}}$-SOCD, and scaled-$\rho_{\text{hybrid}}$-SOCD, respectively. The prefix “mean-” is attached when referring to the mean SOCD for all sites using any one of the field sampling and soil BD calculation combinations, e.g., the mean SOCD for the 10 core sampled sites using the $\rho_{\text{total}}$ calculation method is referred to as “mean-core-$\rho_{\text{total}}$-SOCD”.

The SOCD estimates for all 10 core sampled sites (Figure 4.13) are presented to illustrate the high disparity in the core-$\rho_{\text{total}}$, core-$\rho_{\text{fc}}$, and core-$\rho_{\text{hybrid}}$-SOCD between the sites with a moderate to high RF content (i.e. CHR, GLR, WTA, WTB, MAC, and DKK) and the low disparity for sites with low RF content (i.e. CMM, DKB, DKN, and CLY). For all sites the core-$\rho_{\text{total}}$-SOCD results (ranging from 126 to 205 Mg C ha$^{-1}$) were greater than the core-$\rho_{\text{fc}}$-SOCD results (ranging from 124 to 194 Mg C ha$^{-1}$), which were greater than the core-$\rho_{\text{hybrid}}$-SOCD results (ranging from 105 to 191 Mg C ha$^{-1}$). It was likewise across all sites for the pit-$\rho_{\text{total}}$-SOCD (ranging from 156 to 309 Mg C ha$^{-1}$) which exceeded the pit-$\rho_{\text{fc}}$-SOCD (ranging from 109 to 186 Mg C ha$^{-1}$) and pit-$\rho_{\text{hybrid}}$-SOCD results (ranging from 71 to 184 Mg C ha$^{-1}$). Also, the scaled-$\rho_{\text{total}}$-SOCD results (ranging from 157 to 342 Mg C ha$^{-1}$) were greater than the scaled-$\rho_{\text{fc}}$-SOCD and scaled-$\rho_{\text{hybrid}}$-SOCD results (ranging from 105 to 184 Mg C ha$^{-1}$, and from 80 to 184 Mg C ha$^{-1}$, respectively).

When comparing only the seven sites where the SOCD was estimated using the core, pit and core-scaling methods (i.e. CHR, GLR, WTA, WTB, MAC, DKK, and CMM) the mean-core-$\rho_{\text{total}}$-SOCD (Figure 4.14a) had the lowest mean value of 181 Mg C ha$^{-1}$, which was 25 Mg C ha$^{-1}$ lower than the mean-pit-$\rho_{\text{total}}$-SOCD (i.e. 206 Mg C ha$^{-1}$), and 42 Mg C ha$^{-1}$ lower than the mean-scaled-$\rho_{\text{total}}$-SOCD (i.e. 223 Mg C ha$^{-1}$).
In contrast, the mean-core-\(\rho_{fe}\)-SOCD (i.e. 151 Mg C ha\(^{-1}\)) was 3 and 11 Mg C ha\(^{-1}\) greater than the mean-pit-\(\rho_{fe}\)-SOCD (148 Mg C ha\(^{-1}\)) and mean-scaled-\(\rho_{fe}\)-SOCD (136 Mg C ha\(^{-1}\)), respectively. The mean-core-\(\rho_{hybrid}\)-SOCD (i.e. 132 Mg C ha\(^{-1}\)) was 13 Mg C ha\(^{-1}\) greater than the mean-pit-\(\rho_{hybrid}\)-SOCD (119 Mg C ha\(^{-1}\)), and 6 Mg C ha\(^{-1}\) greater than the mean-scaled-\(\rho_{hybrid}\)-SOCD (126 Mg C ha\(^{-1}\)).

Figure 4.13 The estimated core-, pit-, and scaled- \(\rho_{total}\), \(\rho_{fe}\), and \(\rho_{hybrid}\) SOCD for the 0–40 cm depth, for the 10 core sampled sites, with the means for the three calculation methods.
Figure 4.14a,b,c The estimated SOCD for the 0–40 cm depth using the BD calculations from the core-, pit-, and scaled- $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$, using data from the a) core method, b) pit method (b), and (c) core-scaling sampling methods.
4.5 Discussion

Bulk density is a required property in both the accounting of mineral nutrient pools and SOCD on an areal basis, and therefore its measurement accuracy is of critical importance (Maynard and Curran 2007, Jurgensen et al. 2017). The primary aim of this study was to evaluate the influence of sampling and calculation methods soil BD estimates and subsequently derived SOCD estimates. To achieve that aim, two objectives were undertaken; i) to measure soil BD using two different sampling methods (i.e. core and pit), and ii) to compare three methods for calculating soil BD, namely $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$ to elucidate the impact of RF volume on Irish SOCD estimates.

Due to sampling and cost constraints, BD measurements are often neglected in soil surveys, and are frequently omitted from national soil databases (Wiesmeier et al. 2012, Reidy et al. 2016). Furthermore the equally important recording of soil stoniness, for more accurate estimation of SOCD, is overlooked and further impairs the possibility of establishing universal methods of monitoring SOCD (Jandl et al. 2014). Soil sampling using cores allows many samples to be taken relatively quickly and easily compared to using quantitative pits, but large volume depth-based pit samples provide direct measurements of total soil mass, which are more representative of the true soil BD, RF volume, and elemental concentrations (Vadeboncoeur et al. 2012). To the best of the author’s knowledge, there are no Irish examples in the literature of SOCD estimated using BD data from quantitative pits, but there are several with BD data estimated using the core method (Black et al. 2009, Wellock et al. 2011, Wellock et al. 2014, Reidy and Bolger 2014, Clancy et al. 2015, Premrov et al. 2017, Jovani-Sancho et al. 2017, Creamer pers. comm.), with 30 cm being the maximum sampling depth for most of these studies.
Before the launch of the Irish Soil Information System (Irish SIS) database (Creamer et al. 2014), the An Foras Talúntais National Soil Survey (NSS), conducted between the 1963 and 2011, provided detailed descriptions of 560 soil profiles, but only their last county survey (County Waterford, Diamond and Sills 2011) published BD data, and only 1 out of 38 profiles was in forestry. The NSS covered almost half the 26 counties in Ireland, and the Irish SIS field survey covered the remainder. Although the Irish SIS field survey (Simo et al. 2014) added soils data from 246 soil profiles across 16 counties, 54 % of the 1028 identified horizons in those profiles could not be sampled for BD using cores due to the abundance of RF (Reidy et al. 2016). Also, only 12 of the 246 Irish SIS sites were in forests, and only one (8 % of the sites) had partial soil BD data (Creamer pers. comm.). The hampering of soil BD sampling due to high RF content is not unique to Irish forest soils and many field and laboratory methods have been used to improve the accuracy of the BD estimates (Blake and Hartge 1986, Page-Dumroese et al. 1999, Rodeghiero et al. 2010, Vadeboncoeur et al. 2012, Xu et al. 2015, Poeplau et al. 2017).

In light of the discussion above, and the discussion on soil sampling methods and minimum detectable difference (MDD) in Chapter 2, reconciling the design of a field sampling campaign with the time and effort required to meet the challenge of SOC spatial heterogeneity is a difficult issue to resolve when resources are constrained (FAO 2018). This study used 12 BD and 12 SOC concentration (%) samples, to 40 cm depth per site, which falls short of the 16 samples used by Garten and Wullschleger (1999) to ascertain a MDD of 5 Mg SOC ha\(^{-1}\), i.e., a 10–15 % change in existing SOC. It is recommended that future SOCD studies involving these sites should aim to apply sufficient resources to acquire the necessary number of samples to reach at least that level of MDD.
Although many studies have investigated different methods of sampling soil BD with a focus on accounting for RF (e.g. Lyford 1964, Cunningham and Matelski 1968, Flint and Childs 1984, Muller and Hamilton 1992, Poesen and Lavee 1994, Kulmatiski et al. 2003, Beem-Miller et al. 2016), there are much fewer studies that deal with the influence of the BD calculation methods on the estimates of SOCD or nutrient pool sizes (e.g. Throop et al. 2012, Poeplau et al. 2017). When comparing the mean soil BD for the seven sites investigated using the core and pit methods (Figure 4.7), this study found a -5, 10, and 18 % difference using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ methods, respectively. Only the mean-core-$\rho_{\text{total}}$-SOCD had a lower value than the mean-pit-$\rho_{\text{total}}$-SOCD, which reflects the greater proportion of RF found in the CHR, GLR, WTB, WTA and MAC pits compared to their respective core samples. The DKK site was the only site where the core-$\rho_{\text{total}}$ was greater than the pit-$\rho_{\text{total}}$, and it was also the only site where the “RF volume (% of total volume)” was greater in the core samples than the pit sample, i.e., 9 and 8 %, respectively. The CMM core and pit “RF volume (% of total volume)” values were equal at 1 %, and this is reflected in the identical core- and pit-$\rho_{\text{total}}$ and $\rho_{\text{fe}}$ BD results, and almost identical core- and pit-$\rho_{\text{hybrid}}$ results for that site.

Throop et al. (2012) compared the impact of using the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ variations on the core method and found that BD varied widely on three out of four soils sampled. The single exception was in a soil with only 0.05 % of the core volume occupied by the soil coarse fraction, and therefore the $\rho_{\text{total}}$, $\rho_{\text{fe}}$, and $\rho_{\text{hybrid}}$ results were virtually identical, as was found with the CMM site in this study. For the other three Throop et al. (2012) soils the use of $\rho_{\text{total}}$ resulted in a BD 17–26 % greater than $\rho_{\text{hybrid}}$, and the use of $\rho_{\text{fe}}$ led to a 11–17 % increase in BD relative to $\rho_{\text{hybrid}}$. Analysing the core samples, this study found a 20 % increase in BD between the mean (n = 7) $\rho_{\text{total}}$ (1.13 g
cm$^{-3}$) and $\rho_{fe}$ (0.95 g cm$^{-3}$), and a 24 % increase in BD between the mean $\rho_{fe}$ and $\rho_{hybrid}$ (0.81 g cm$^{-3}$).

In a study using cores and pits at three sites, and calculating the soil BD using the $\rho_{total}$ method, Lichter and Costello (1994) found their core $\rho_{total}$ results to be 3–9 % greater than their pit $\rho_{total}$ results. These authors do not mention the RF content of the soils, and while they mention the possibility of sample compaction due to core insertion, they are unable to identify clear causes for the variance in results between the two methods used. In contrast, the $\rho_{total}$ of core samples (392.7 cm$^3$) taken from a compacted surface-mine soil with high RF content by Muller and Hamilton (1992) were 12–17 % lower than the $\rho_{total}$ for their pit samples (~1000 cm$^3$). They gave two reasons for this, i) in stony soils the use of cores causes considerable soil disturbance and sifting of the RF, which led to the core samples containing significantly less RF content than the pit samples, and ii) the smaller volume of the cores was unrepresentative of the overall stony soil conditions.

Rytter (2012) found that even in arable soils with RF content exceeding just 3 % of the sample volume, the exclusion of the RF volume led to an overestimation of soil C and N stocks of 3–9 %, and RF content of around 8 % had a statistically significant impact on soil nutrient stocks. Several studies (Shipp and Matelski 1965, Aubertin 1971, Poesen and Lavee, 1994, Sauer and Logsdon 2002), attribute an increase in macropores at the interface between the fine earth fraction and RF surfaces to the presence of high RF content in the soil, which leads to a lower overall soil BD. Shipp and Matelski (1965) used this phenomenon to explain the lower soil BD found in their pit samples (as a non-biased representation of the soil RF content) when compared to the RF content and BD of their smaller clod samples.
The density of the RF found in the pit samples in this study ranged from 1.99 to 2.53 g cm\(^{-3}\) with a mean of 2.33 ±0.19 g cm\(^{-3}\). As expected, the fine earth BD (\(\rho_{fe}\)) of the pit samples had a much lower range of 0.66 to 0.97 g cm\(^{-3}\) with a mean of 0.86 ±0.11 g cm\(^{-3}\). The \(\rho_{\text{total}}\) method reflects the high density of the RF content of the core and pit samples, whereas the \(\rho_{fe}\) method reflects only the density of the soil fine fraction. Hence in soils with a high RF content, \(\rho_{\text{total}}\) is generally greater than \(\rho_{fe}\), and the method of BD calculation can have a substantial impact on SOCD estimates. The \(\rho_{\text{total}}\) method is the appropriate BD method when the aim of the study is to estimate total SOCD, e.g., where the C content of the RF, root, and soil biota pools are of interest. The \(\rho_{fe}\) method is more appropriate for SOC or nutrient stocks, where there is negligible RF content in the soil, and where soil C flux or nutrient availability are the focus of the study. The \(\rho_{\text{hybrid}}\) method recommended by Throop et al. (2012) as being closest to the true soil BD value is therefore the appropriate method in stony soils for SOC stock inventories, as it accounts for soil volume dilution due to RF content.

In skeletal soils, i.e., where the RF volume is ≥ 35 % of sample volume (Childs and Flint 1990, Poesen and Lavee 1994, Jurgensen et al. 2017) the accuracy of SOCD on an areal basis to a given depth, e.g., Mg C ha\(^{-1}\) to 40 cm depth, is highly dependent on knowing the relative volume of the soil occupied by RF (Rytter 2012, Throop et al. 2012). Without knowledge of the RF volume the soil BD is likely to be overestimated, with a similar effect on the derived SOCD estimates (Poeplau et al. 2017). Based on the core method, none of the 10 sites sampled in this study fall into the skeletal soil classification, while two of them (GLR and CHR) had RF content (by volume) ≥ 35 % when sampled using the quantitative pits, and would be classified as skeletal soils.

When comparing the mean- \(\rho_{\text{total}}\)- and \(\rho_{\text{hybrid}}\)-SOCD estimates for the core, pit, and core-scaling methods, it is interesting to note that the results for the pit and core-
scaling methods (Figures 4.13 b and c) are more closely related, with SOCD differences of 18 and 7 Mg C ha\(^{-1}\), respectively, than the SOCD estimates from the pit and core methods, with SOCD differences of 25 and 13 Mg C ha\(^{-1}\), respectively. The mean-core-\(\rho_{fe}\)-SOCD was more closely related to the mean-pit-\(\rho_{fe}\)-SOCD, with a difference of 3 Mg C ha\(^{-1}\) compared to a difference of 9 Mg C ha\(^{-1}\) between the mean-pit-\(\rho_{fe}\)-SOCD and mean-scaled-\(\rho_{fe}\)-SOCD estimates. If the pit method is the most accurate means of measuring the true soil BD and RF content, these results suggest that the core-\(\rho_{fe}\) method is next best method in low RF content soils, and the core-scaling method is useful to employ when the study involves stony soils.

4.6 Conclusions

For all soils the combination of the sampling methods (e.g. core, pit, or core-scaling) and the soil properties, i.e., SOC concentration, soil BD, RF volume, and the method of calculating the soil BD (e.g. \(\rho_{total}\), \(\rho_{fe}\), or \(\rho_{hybrid}\)) determines the SOCD to the chosen sampling depth. The results of this study demonstrate the substantial differences in the estimated SOCD that are achieved depending on the chosen combination of soil sampling and BD calculation methods. New Irish forest soil BD, RF content, and SOCD estimates derived from analysis of core and quantitative pit samples were presented, which show the disparity in results that occur when the soil BD is calculated using different methods of treating their constituent RFs. This study found that the mean SOCD for the seven pit sites, calculated using the core-\(\rho_{total}\) was 25 Mg C ha\(^{-1}\) lower than the mean SOCD calculated using the pit-\(\rho_{total}\) method, and 42 Mg C ha\(^{-1}\) lower than the mean SOCD calculated using the scaled-\(\rho_{total}\) method. Conversely, the mean SOCD for the seven pit sites, calculated using the core-\(\rho_{hybrid}\) was 13 Mg C ha\(^{-1}\) higher than the mean SOCD calculated using the pit-\(\rho_{hybrid}\) method, and 6 Mg C ha\(^{-1}\) higher than the
mean SOCD calculated using the scaled-$\rho_{\text{hybrid}}$ method. A novel method of combining core and pit sampling, called the “core-scaling” method, to expedite the incorporation of large RFs in the estimation of soil BD, was also presented. According to the literature the pit method provides the best estimate of SOCD, due to the direct and non-biased nature of the sampling, and its inherent accounting for the volume occupied by large RF in the samples. The soil BD results obtained using the core-scaling method were strongly correlated with the pit BD estimates ($R^2 > 0.84$ for the $\rho_{\text{total}}$, $\rho_{\text{fc}}$, and $\rho_{\text{hybrid}}$ methods), and gives a more accurate estimation of the SOCD than the results achieved with the core method alone. Data from the Irish SIS field survey (Simo et al. 2014) show that the BD data for more than half of the sampled horizons were omitted due to the abundance of rock fragments, making the 5 cm diameter cores unusable (Reidy et al. 2016). This concurs with the Irish geology literature which states that much of the country is covered in stony glacial till (Farrell 2016) and therefore highlights that all RF content should be considered when assessing SOCD in Irish soils. Given the prevalence of stony soils in Ireland, the use of core sampling for estimates of soil BD, which neglects to account for the relative fine earth volume dilution by large RF, may lead to overestimation of SOCD in many soils.
5. Chapter 5 - Life Cycle Assessment of the Greenhouse Gas Balance of Irish Short Rotation Forestry
5.1 Abstract

Short Rotation Forestry (SRF) has the potential to increase biomass production and contribute to the EU Renewable Energy Directive target for Ireland of 16 % renewable energy by 2020. When sustainable forest management practices are employed SRF can also help offset Ireland’s greenhouse gas (GHG) emissions from the combustion of decreasing reserves of peat and fossil fuels through their displacement in industrial and domestic heat and power plants. The Life Cycle Assessment (LCA) methodology was used to study Irish SRF biomass production, and pulpwood (7–13 cm diameter) and forest residues (brash and stumps) assortments from an Irish Sitka spruce plantation were also assessed as a reference system. Eucalyptus nitens was used as a representative tree species for the 3×10 year SRF rotations, while the biomass yield used for modelling those SRF rotations was 28 m$^3$ ha$^{-1}$ y$^{-1}$. A Sitka spruce yield class of 24 was used in the modelling of a single 30 year rotation before clearfell. The LCA attributed material and energy inputs and GHG and other environmental outputs to each phase of the SRF life cycle. The LCA also examined the GHG balance consequences of several SRF management scenarios, e.g., stem-only versus wholetre harvesting, with or without forest residues and stump biomass recovery, and the use of fertilizers over multiple rotations. The mean GHG balance of the SRF scenarios (i.e. 2441 t CO$_2$-eq ha$^{-1}$ and 1135 t CO$_2$-eq ha$^{-1}$, at the 30 % and 50 % co-firing rates, respectively) are both approximately seven times greater than the GHG balance of the Sitka spruce scenarios (i.e. 339 t CO$_2$-eq ha$^{-1}$ and 168 t CO$_2$-eq ha$^{-1}$, at the same respective co-firing rates). Hence, the mean GHG emission reductions from the SRF biomass scenarios outperform those from the Sitka spruce scenarios at both the per-MWh$_e$ and per-hectare levels.

Keywords: Short rotation, biomass, bioenergy, renewable energy, peat displacement
5.2 Introduction

The impact of fossil fuel consumption, increasing greenhouse gas (GHG) emissions and associated climate change is driving the need for more sustainable energy structures, technologies and practices to decarbonise our energy systems (SEAI, 2016a). The energy sector is responsible for 61 % (36 Mt CO$_2$-eq) of the total annual Irish GHG emissions from electricity generation with natural gas, coal and peat being the major contributors at 4.4 Mt CO$_2$-eq (41 %), 3.8 Mt CO$_2$-eq (35 %), 2.5 Mt CO$_2$-eq (23 %) respectively (Duffy et al. 2015).

From a historic low of 1 % forest cover in the early 1900’s, the current Irish land area occupied by forest has grown to approximately 11 % (653,980 ha) in 2012, of which 52.4 % (334,560 ha) is planted with Sitka spruce (Picea sitchensis (Bong. Carr.) (DAFM 2012). Sitka spruce as the dominant species in Irish forest biomass production, with an average yield class of 17 m$^3$ ha$^{-1}$ yr$^{-1}$, is the main source of roundwood to the sawmill and board processing industries (Murphy et al. 2016). Irish government policy on forestry development has recently begun focusing on how to maximise the utilisation of the previously unrecovered forest thinnings and residues, i.e., tree tops, branches, and stumps for use as bioenergy products (Kent et al. 2014). Mobilising these additional biomass streams may have potential adverse environmental impacts such as reduced SOM inputs and increased nutrient export from the forest via wholentree harvests, residue bundles and stumps, and potential SOC losses from increased soil disturbance from stump excavation (Walmsley and Godbold 2010, Moffat et al. 2011). Strömgren et al. (2013) report a 6 Mg C ha$^{-1}$ reduction in the C stock from a 25 year study of intensive Finnish forestry operations involving stump harvesting and logging residues compared to stem only harvesting. Jarvis et al. (2009) estimate that soil disturbance from windthrow, site preparation by ploughing, and stump removal may contribute to
emissions of 14–20 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$, and it could take 15 years for young plantations to return disturbed sites to a C sink status.

Renewable energy sources such as biomass for bioenergy offer an alternative to fossil fuels and can assist in mitigating atmospheric GHG emissions (Demirbas et al. 2009). In Ireland, SRF has the potential to increase biomass production for renewable energy, and contribute to the EU RED target of 16 % renewable energy and a target of 20 % reduction in GHG emissions by 2020 (Howley and Holland 2016). When sustainable forest management practices are employed SRF can also help offset Ireland’s GHG emissions from the combustion of decreasing reserves of peat and other fossil fuels through their displacement in industrial and domestic heat and power plants.

This study investigated the GHG balance of a eucalyptus ($E. nitens$) SRF plantation over three 10 year rotations through the use of Life Cycle Assessment (LCA) methodology, combined with specialised LCA software tools and databases. These tools enabled the assessment of Irish SRF plantations from initial establishment operations, through to biomass harvesting and bioenergy products end-use at a peat and biomass co-fired electricity generating plant.

The LCA attributed material and energy inputs and GHG and other environmental outputs to each phase of the SRF life cycle. The LCA also examined the GHG balance consequences of several SRF management scenarios, e.g. stem-only versus whol-tree harvesting, with or without forest residues and stump biomass recovery, and the use of fertilizers over multiple rotations.

**5.2.1 Objectives of this study:**

The aim of this LCA study was to evaluate the cradle-to-grave GHG balances of SRF biomass for bioenergy, and to compare the results to an existing forest biomass
reference system, i.e., potentially available Sitka spruce biomass assortments. The study also assessed both the SRF and Sitka spruce biomass streams GHG balance in relation to the use of fossil fuel (milled peat) at the EPL electricity generation plant.

1. The 1st objective was to use LCA methodology to assess the GHG balance of SRF biomass for bioenergy.
2. The 2nd objective was to use LCA methodology to also assess the GHG balance of Sitka spruce biomass for bioenergy as a reference system, and compare the results to those for SRF biomass.
3. The 3rd objective was to compare both the SRF and Sitka spruce biomass to a fossil fuel (milled peat) reference system, and determine their potential impact on the Irish RED target (16 % renewable energy - biomass streams) and on the 20 % Irish GHG reduction targets, both by 2020.

5.3 Materials & Methods

The methodology used to assess the GHG balance and the potential environmental impact of SRF biomass for bioenergy was Life Cycle Assessment (LCA). The study follows the LCA principles and framework of the International Standards Organization (ISO) 14040 and 14044 methodologies (ISO 2006a, 2006b, respectively). All phases of the silvicultural production and technological processing of the Sitka spruce and SRF biomass to woodchips or hogfuel (e.g. Figure 5.1), along with the provision of milled peat for combustion in a co-firing electricity generation plant, were addressed. This was achieved using experimental and reference data from literature combined with the ecoinvent database v3.2 (Ecoinvent 2016), the ISO (14040 and 14044) compliant software openLCA v.1.5 (GreenDelta, 2016), and the GROWFOR biomass yield simulation model (Broad and Lynch 2006).
**Figure 5.1** A graphic overview of the integrated harvesting, forwarding, chipping and transport to the point-of-use of three biomass assortments: stems, wholentree, and stumps (Source: METLA Forest Energy Portal).

### 5.3.1 LCA Goal and Scope Definition

The goal of this LCA study was to assess the environmental impacts of Irish SRF biomass for bioenergy usage, with the emphasis on a cradle-to-grave quantification of the GHG balance of associated emissions and sinks. The intended application of the study was to identify the main processes contributing to environmental impacts, and to provide data for future assessment of the GHG mitigation potential of using SRF biomass for bioenergy. The study also compared the production and usage of SRF biomass for bioenergy to the following two electrical energy generation related reference systems:

a) Potentially available Sitka spruce biomass assortments for bioenergy.
b) Fossil fuel (milled peat) currently used in co-firing with biomass at the Edenderry Power Ltd. (EPL) electricity generation plant.

5.3.1.1 System Boundary

The LCA system boundary for the Sitka spruce and SRF biomass covers eight scenarios. The four conventional forestry scenarios, based on Sitka spruce biomass assortments from a plantation grown over 1×30 year rotation (SS1–SS4), and the four SRF scenarios, based on the eucalyptus species *E. nitens*, grown over 3×10 year rotations (SRF1–SRF4), are shown in Figure 5.2. Each LCA scenario within this system boundary encompasses either a standalone biomass product (“product system” in LCA terms), e.g., the Sitka spruce pulpwood from clearfell harvesting (scenario SS1), or combinations of product systems (scenario SS2) which comprises Sitka spruce biomass from clearfell pulpwood, brash (forest residues from tree tops and branches), and excavated stumpwood. The second reference system, the peat scenario (Peat-0), has a separate system boundary (Figure 5.3), which reflects the different cradle-to-gate production and transport processes of the milled peat product system, but shares the same combustion and ash disposal processes as the biomass scenarios.

5.3.1.2 Product Systems and Functional Units

The specific function of each product system is to produce biomass for combustion and conversion to electricity in an energy plant co-firing biomass from renewable sources with a non-renewable fossil fuel, i.e., milled peat. To aid comparisons with other bioenergy LCA studies, two functional units were selected.
1. The first functional unit measures the environmental impact of the Sitka spruce and SRF biomass scenarios per megawatt hour of electricity (MWh\textsubscript{e}), i.e., the unit of electrical energy delivered from the co-fired fuel combustion at the EPL plant.

2. The second functional unit presents the results in terms of biomass yield (in m\textsuperscript{3} of solid ob wood biomass) per hectare (ha), to allow comparison of each scenarios GHG balance as it relates to biomass yields and land area allocation for bioenergy purposes.

The bark with its high C content of 51–66 % (Alakangas 2005) is included for bioenergy purposes, though it may have a detrimental effect on soil fertility through important nutrient export, possibly leading to the need for site application of fertilizers (Merino et al. 2005). The scenarios outlined above are evenly split between existing Sitka spruce and possible future SRF forest biomass scenarios, respectively.

The cradle-to-grave LCA for the biomass and peat scenarios were also subdivided to account for the cradle-to-gate operations within the respective biomass and peat system boundaries. The biomass and peat production and processing that takes place prior to the “Combustion” operation shown in their respective system boundaries are hereafter represented by the biomass-at-the-gate (BAG) and peat-at-the-gate (PAG) components of the LCA results. The BAG component of the LCA scenarios represents the aggregated sum of the operations starting with “Seedling Production” and ending with “Haulage” to the power plant gate. Similarly for the Peat-0 scenario LCA, the PAG component is the aggregated sum of the mechanised peat production (e.g. milling, harrowing, and ridging) and haulage by train to the power plant.
Figure 5.2 The Life Cycle Assessment (LCA) system boundary for four Sitka spruce (SS1–4, green arrows) and four SRF (SRF1–4, blue arrows) biomass assortment scenarios. The system boundary encompasses all the cradle-to-grave biomass production and processing operations that have potential environmental impacts. The Sitka spruce undergoes end-of-rotation (×1) clearfell harvesting of pulpwood (SS1–4), with optional thinning operations (e.g. SS3–4), recovery of brash and stumps (SS2, SS4), and fertilizer application after the Sitka spruce site establishment (SS4). The SRF undergoes end-of-rotation (×3) clearfell harvesting of stems (SRF1) or whole-tree assortments (SRF2–4), with optional recovery of stumps (SRF3–4) and fertilizer application after the SRF site establishment (SRF4). The LCA operations shown encompass all the material (e.g. fertilizers, machines) and energy (e.g. diesel oil, electricity) inputs, and subsequent outputs (e.g. biomass products, or greenhouse gases).
5.3.2 Life Cycle Inventory data sources and assumptions

As there are no mature commercial SRF operations currently existing in Ireland, this study had to rely on life cycle inventory (LCI) reference data from external sources for material and energy inputs and outputs. In the absence of the authors own empirical data specific to the SRF and Sitka spruce forestry operations within the system boundary, many assumptions regarding the LCI data were made. Those assumptions were based on the most appropriate Irish, British, or European analytical data from the literature, modelling tools such as GROWFOR, and the ecoinvent reference data. Several of the key assumptions that are applied throughout the LCA study are as follows:

1. Aboveground biomass harvesting is 98 % mechanized cut-to-length (CTL), 2% manual power sawing.
2. All biomass is left to season in the forest to dry to 40 % moisture content.
3. SRF/E. nitens biomass yields based on literature, solid over bark (o.b.).

4. Irish grown E.nitens basic density = 435 kg m$^{-3}$.

5. Irish grown Sitka spruce basic density = 380 kg m$^{-3}$.

6. Solid biomass to woodchip/hogfuel conversion factors are based on Irish data.

7. Transport of machinery and materials is included in each life cycle process.

8. SRF biomass yields for each rotation are the same.


10. Sitka spruce available biomass for bioenergy consists only of thinning and/or clearfell pulpwood (7–13 cm diameter), all other roundwood goes to Irish sawmills for wood products.

11. The GHG balance related to direct land use change (LUC) is included, indirect LUC is not included.

Other assumptions related to yields, solid wood to woodchip or hogfuel conversion factors, and embedded energy are listed in Table 5.1. All other assumptions related to the main life cycle processes are discussed in the succeeding paragraphs, and marked within the “Source” field of the associated tables with their citations, or the tag “This study” for items specific to this study.

The ecoinvent database is the world’s leading LCI database, and the 2016 release of version 3.2 was chosen as the best available source of data on both conventional and short rotation forest operations (Werner et al. 2007). The input and output flows from the most relevant ecoinvent processes were modified and combined with related data from Irish and international literature (Tables 5.2–5.5). By doing so, this study was tailored to reflect Irish forestry conditions and management practices.
The product, machine, infrastructure and chemical flows, many of which were parameterised within the openLCA tool, are shown in the tables, alongside the non-tailored upstream product processes and flows available from the ecoinvent database.

**Table 5.1** The assumed SRF and Sitka spruce biomass yields, by scenario, per hectare, for $3 \times 10$, and $1 \times 30$ year rotations respectively, with their associated solid-to-woodchip/hogfuel conversion factors (CF), and embedded energy values (MWh$_e$/ha).

<table>
<thead>
<tr>
<th>Yield related data and units</th>
<th>Scenarios</th>
<th>SRF1</th>
<th>SRF2</th>
<th>SRF3</th>
<th>SRF4</th>
<th>SS1</th>
<th>SS2</th>
<th>SS3</th>
<th>SS4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Solid (m$^3$/ha)</td>
<td></td>
<td>840</td>
<td>1050</td>
<td>1155</td>
<td>1155</td>
<td>18</td>
<td>39</td>
<td>80</td>
<td>101</td>
</tr>
<tr>
<td>Solid (m$^3$/ha) $-12%$ harvest loss</td>
<td></td>
<td>739</td>
<td>924</td>
<td>1029</td>
<td>1029</td>
<td>16</td>
<td>150</td>
<td>70</td>
<td>204</td>
</tr>
<tr>
<td>CF: solid to l.v.$^*$ chip/hogfuel</td>
<td></td>
<td>2.3</td>
<td>2.42</td>
<td>2.42</td>
<td>2.42</td>
<td>2.85</td>
<td>-</td>
<td>2.85</td>
<td>-</td>
</tr>
<tr>
<td>Chip/hogfuel m$^3$ (l.v.)</td>
<td></td>
<td>1700</td>
<td>2236</td>
<td>2490</td>
<td>2490</td>
<td>45</td>
<td>447</td>
<td>201</td>
<td>603</td>
</tr>
<tr>
<td>Chip/hogfuel m$^3$ (l.v.) $-5%$ chip loss</td>
<td></td>
<td>1615</td>
<td>2124</td>
<td>2366</td>
<td>2366</td>
<td>43</td>
<td>425</td>
<td>191</td>
<td>573</td>
</tr>
<tr>
<td>CF: m$^3$ (l.v.) to tonnes chip/hogfuel</td>
<td></td>
<td>3.2</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>3.6</td>
<td>2.95</td>
<td>3.6</td>
<td>4.2</td>
</tr>
<tr>
<td>Chip/hogfuel (t), 40 % MC</td>
<td></td>
<td>505</td>
<td>708</td>
<td>781</td>
<td>781</td>
<td>13</td>
<td>134</td>
<td>56</td>
<td>177</td>
</tr>
<tr>
<td>Chip/hogfuel t/MWh$_e$, 40 % MC</td>
<td></td>
<td>0.9</td>
<td>0.85</td>
<td>0.86</td>
<td>0.86</td>
<td>0.9</td>
<td>0.9</td>
<td>0.9</td>
<td>0.9</td>
</tr>
<tr>
<td>Chip/hogfuel MWh$_e$/ha, 40 % MC</td>
<td></td>
<td>561</td>
<td>833</td>
<td>914</td>
<td>914</td>
<td>14</td>
<td>149</td>
<td>62</td>
<td>197</td>
</tr>
<tr>
<td>MWh$_e$/ha: 30 % co-firing</td>
<td></td>
<td>1869</td>
<td>2777</td>
<td>3046</td>
<td>3046</td>
<td>46</td>
<td>497</td>
<td>206</td>
<td>657</td>
</tr>
<tr>
<td>MWh$_e$/ha: 50 % co-firing $^*$ l.v. = loose volume</td>
<td></td>
<td>1122</td>
<td>1666</td>
<td>1827</td>
<td>1827</td>
<td>28</td>
<td>298</td>
<td>124</td>
<td>394</td>
</tr>
</tbody>
</table>

$l.v.$ = loose volume
The starting point for the LCI was the flows associated with the production of seedlings at a nursery, the transport of materials and machines to the site, leading to the establishment of 1 ha of Sitka spruce or SRF plantation (Table 5.2). The input flows include the spot-application of herbicide, crushed gravel for building and maintaining of forest roads, digging of shallow drains, and fence posts for one side of each site. The weight of crushed gravel transported to the site for forest road building, i.e. 20 m$^2$ ha$^{-1}$ of road, includes the trucks empty return journey, and 20 % additional gravel for road maintenance over the course of the rotation. Also provided are the flows for the N, P, and K fertilizers, which are applied in one each of the Sitka spruce and SRF scenarios, and their transport to the site. The assumed round-trip transport distances for the gravel, excavator, seedlings, fence posts, and fertilizers were 50, 50, 100, 50, and 50 km, respectively, with a 30 % loading value used for the empty return leg of each journey.

**Table 5.2** Seedling production and site establishment process flows for 1 ha of forest plantation, for biomass assortments in both the Sitka spruce and SRF scenario models.

<table>
<thead>
<tr>
<th>Input Flows</th>
<th>Amount</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Herbicide: Glyphosate</td>
<td>0.06</td>
<td>kg</td>
<td>Murphy <em>et al.</em> 2014b</td>
</tr>
<tr>
<td>Gravel, crushed &amp; transported to site</td>
<td>9.36</td>
<td>t</td>
<td>Whittaker <em>et al.</em> 2011, Murphy <em>et al.</em> 2014</td>
</tr>
<tr>
<td>Tree seedlings produced at Irish nursery</td>
<td>2500</td>
<td>Item(s)</td>
<td>ecoinvent, Phillips 2004</td>
</tr>
<tr>
<td>Drain excavation with hydraulic digger</td>
<td>3.645</td>
<td>m$^3$</td>
<td>Teagasc 2012</td>
</tr>
<tr>
<td>Nitrogen fertilizer</td>
<td>350</td>
<td>kg</td>
<td>DAFM 2014</td>
</tr>
<tr>
<td>Phosphate fertilizer</td>
<td>350</td>
<td>kg</td>
<td>DAFM 2014</td>
</tr>
<tr>
<td>Potassium fertilizer</td>
<td>250</td>
<td>kg</td>
<td>DAFM 2014</td>
</tr>
<tr>
<td>Fence posts</td>
<td>0.1</td>
<td>m$^3$</td>
<td>Teagasc 2012</td>
</tr>
<tr>
<td>Excavator transport, lorry 16-32 t</td>
<td>357.5</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>Seedlings transport, lorry 3.5-7.5 t</td>
<td>81.25</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>Fence posts transport, lorry 3.5-7.5 t</td>
<td>3.77</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>Fertilizer transport, lorry 3.5-7.5 t</td>
<td>30.9</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Output Flows</th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Site established</td>
<td>1</td>
<td>ha</td>
<td>Reference Product</td>
</tr>
</tbody>
</table>
The yields for the SRF (*E.nitens*), which were derived from the literature (Thompson *et al.* 2012) and Sitka spruce, which were derived from the GROWFOR modelling tool (Table 5.3).

**Table 5.3** The short rotation forestry (SRF, represented by the *Eucalyptus nitens* species) and the Sitka spruce biomass assortment yields, for 3×10 year and 1×30 year rotations, respectively.

<table>
<thead>
<tr>
<th>E. nitens (SRF) Assortments:</th>
<th>SRF stem-only (m³ ha⁻¹)</th>
<th>SRF wholetree (m³ ha⁻¹)</th>
<th>SRF stump (m³ ha⁻¹)</th>
<th>Total SRF wholetree &amp; stump biomass (m³ ha⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clearfell rotation #1 (10 yrs)</td>
<td>280</td>
<td>350</td>
<td>35</td>
<td>385</td>
</tr>
<tr>
<td>Clearfell rotation #2 (20 yrs)</td>
<td>280</td>
<td>350</td>
<td>35</td>
<td>385</td>
</tr>
<tr>
<td>Clearfell rotation #3 (30 yrs)</td>
<td>280</td>
<td>350</td>
<td>35</td>
<td>385</td>
</tr>
<tr>
<td><strong>Total Clearfell - 3 rotations</strong></td>
<td><strong>840</strong></td>
<td><strong>1050</strong></td>
<td>105</td>
<td><strong>1155</strong></td>
</tr>
</tbody>
</table>

*Biomass losses of 12 % 246 308 18 326 739 924 92 1016

*Coillte allows for an average harvesting loss of 9%, plus a chronic access reduction of 3% (on average)*

<table>
<thead>
<tr>
<th>Sitka Spruce (SS) Assortments</th>
<th>SS totals: above &amp; belowground biomass (m³ ha⁻¹)</th>
<th>SS Roundwood (14 - &gt; 20 cm) (m³ ha⁻¹)</th>
<th>SS Clearfell Residues (m³ ha⁻¹)</th>
<th>SS Pulpwood (7 - 13 cm) (m³ ha⁻¹) + Residues</th>
</tr>
</thead>
<tbody>
<tr>
<td>SS Thin #1 (18 yrs.)</td>
<td>50</td>
<td>20</td>
<td>30</td>
<td></td>
</tr>
<tr>
<td>SS Thin #2 (22 yrs.)</td>
<td>50</td>
<td>31</td>
<td>19</td>
<td></td>
</tr>
<tr>
<td>SS Thin #3 (26 yrs.)</td>
<td>50</td>
<td>37</td>
<td>13</td>
<td></td>
</tr>
<tr>
<td>SS Thin total</td>
<td>150</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>SS Clearfell (30 yrs.)</strong></td>
<td><strong>521</strong></td>
<td><strong>503</strong></td>
<td><strong>18</strong></td>
<td></td>
</tr>
<tr>
<td>SS Thin x3 + Clearfell at 30 yrs.</td>
<td><strong>671</strong></td>
<td><strong>591</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SS clearfell brash¹</td>
<td></td>
<td></td>
<td>156</td>
<td>86</td>
</tr>
<tr>
<td>SS clearfell stump²</td>
<td></td>
<td></td>
<td>115</td>
<td>48</td>
</tr>
<tr>
<td><strong>Total biomass (m³ ha⁻¹) 1x30 yr. rotation</strong></td>
<td><strong>942</strong></td>
<td></td>
<td><strong>214</strong></td>
<td></td>
</tr>
</tbody>
</table>

¹ SS brash (available yield = 30% of clearfell ABG, only 55% removed)
² SS stump (available yield = 22% of clearfell ABG, only 42% removed)
³ SRF stump (available yield = 22.5% of wholetree, only 50% removed)
The harvesting processes (Table 5.4) for the Sitka spruce plantation were modelled using the following ecoinvent process "softwood forestry, spruce, sustainable forest management | pulpwood, softwood, measured as solid wood under bark | APOS, U - SE" as a starting point. The harvesting processes for the *E. nitens* SRF plantation were modelled using the ecoinvent process "hardwood forestry, eucalyptus ssp., sustainable forest management | roundwood, eucalyptus ssp., under bark | APOS, U - TH" as a starting point. The Sitka spruce pulpwood from the thinning and clearfell operations and the SRF stem and wholetree assortments factored in 12 % biomass losses from the harvesting and forwarding operations, per the guidelines from Coillte (COFORD 2015). Those losses are reflected in the output quantity of 0.88 m$^3$ of recovered energywood from an input of 1 m$^3$ of standing wood, as shown in the “Output flows” of Table 5.3. The ecoinvent process that formed the basis of the Sitka spruce brash bundling operation used a mass unit (kg) for the “Reference product”. For consistency with all the other Sitka spruce and SRF assortments the brash unit mass was converted to a unit volume, i.e. 0.00513 m$^3$, using a brash bundle bulk density of 195 kg m$^{-3}$ (Kent *et al.* 2014) for the Sitka spruce brash bundles.

The recovery of 55 % of the available Sitka spruce brash (Kent *et al.* 2014) and 42 % of the stumps (Murphy *et al.* 2014) per hectare is accounted for in the LCI results. These brash and stump recovery rates are relative to the prevailing ground conditions in Sitka spruce forest plantations on marginal agricultural and hillside land in Ireland. With the same planting density of 2500 trees ha$^{-1}$ used for both the Sitka spruce and SRF plantations, it was assumed that the diesel used in site-tending operations during the growing phase prior to harvesting would be similar for each 1 m$^3$ solid wood (ob).

The C (as CO$_2$) sequestration in the Sitka spruce pulpwood is based on a mean basic density of 380 kg m$^{-3}$ (Kofman 2010) and a mean C content of 46 % (Tobin and
The Sitka spruce brash C sequestration is based on a bundle bulk density of 195 kg m$^{-3}$ (Kent et al. 2014) and a mean C content of 51 % (Alakangas 2005). The C sequestration in the SRF stem and stump biomass assortments is based on the *E. nitens* basic density of 435 kg m$^{-3}$ (Thompson et al. 2012) and a mean C content of 49.4 % (ecoinvent, 2015). The *E. nitens* wholetree C sequestration is based on a basic density of 439 kg m$^{-3}$ (McKinley et al. 2000) and a mean C content of 49.4 % (ecoinvent, 2015).

There is also a difference between the Sitka spruce and SRF plantations in the treatment of the forest soil C sequestration. This is due to the Sitka spruce plantation soil being treated primarily as a C sink (Black et al. 2009), and the SRF plantation soil being treated as a C source (Keith et al. 2015). The CO$_2$ emissions due to LUC accounted for in the harvest process outputs represent soil disturbance events associated with the Sitka spruce brash and stump removals, and the detrimental effect on the soil C pool of the shorter rotation lengths of the SRF plantation. The output from each of the different biomass assortments harvesting processes is the “Recovered energywood” (Table 5.4) measured in m$^3$ (ob).
Table 5.4 A consolidated inventory of the harvesting process input and output flows for the aboveground Sitka spruce pulpwood and brash, and SRF (E. nitens) stem-only roundwood and wholletree biomass assortments, along with the belowground Sitka spruce and SRF stumpwood recovered after clearfell.

<table>
<thead>
<tr>
<th>Input Flows</th>
<th>Unit</th>
<th>SRF assortment</th>
<th>Sitka spruce assortment</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>diesel, burned in site tending</td>
<td>MJ</td>
<td>3.140</td>
<td>3.140</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>Energy, GCV, in biomass</td>
<td>MJ</td>
<td>8265.0</td>
<td>8570.0</td>
<td>Thompson et al. 2012, This study</td>
</tr>
<tr>
<td>forwarding, forwarder</td>
<td>h</td>
<td>0.10718</td>
<td>0.10718</td>
<td>Klvac &amp; Skoupy, 2009.</td>
</tr>
<tr>
<td>harvesting, forestry harvester</td>
<td>h</td>
<td>0.11442</td>
<td>0.11442</td>
<td>Klvac &amp; Skoupy, 2009.</td>
</tr>
<tr>
<td>harvesting/brash bundling</td>
<td>h</td>
<td></td>
<td>0.04207</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>stump excavation-hydraulic digger</td>
<td>h</td>
<td></td>
<td>0.04207</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>lubricating oil for harvester</td>
<td>kg</td>
<td>0.30</td>
<td>0.30</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>Site established (1 ha)</td>
<td>m²</td>
<td>25.92</td>
<td>25.92</td>
<td>This study</td>
</tr>
<tr>
<td>C as CO₂ sequestration in biomass</td>
<td>kg</td>
<td>789.00</td>
<td>796.00</td>
<td>Kofman 2010; McKinley et al. 2000; Tobin &amp; Nieuwenhuis 2007; Thompson et al. 2012</td>
</tr>
<tr>
<td>C as CO₂ sequestration in soil</td>
<td>kg</td>
<td></td>
<td>27.36</td>
<td>Keith et al. 2015</td>
</tr>
<tr>
<td>Occupation, forest, intensive</td>
<td>m²* a</td>
<td>259.22</td>
<td>259.22</td>
<td>This study</td>
</tr>
<tr>
<td>Occupation, road area</td>
<td>m²* a</td>
<td>0.5</td>
<td>0.5</td>
<td>This study</td>
</tr>
<tr>
<td>power sawing</td>
<td>h</td>
<td></td>
<td>0.05489</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>Transformation, from unspecified, non-use</td>
<td>m²</td>
<td></td>
<td>12.4000</td>
<td>This study</td>
</tr>
<tr>
<td>Transformation, to forest, intensive</td>
<td>m²</td>
<td>25.9220</td>
<td>25.9220</td>
<td>This study</td>
</tr>
<tr>
<td>Source</td>
<td>Unit</td>
<td>SRF assortment</td>
<td>Sitka spruce assortment</td>
<td>Source</td>
</tr>
<tr>
<td>--------------------------------------------</td>
<td>-----------</td>
<td>----------------</td>
<td>-------------------------</td>
<td>--------</td>
</tr>
<tr>
<td>Transformation, to road area</td>
<td>m²</td>
<td>0.0519</td>
<td>0.0519</td>
<td>0.0249</td>
</tr>
<tr>
<td>Wood biomass, standing</td>
<td>m³</td>
<td>1.0000</td>
<td>1.0000</td>
<td>1.0000</td>
</tr>
<tr>
<td>Stumpwood, in the ground</td>
<td>m³</td>
<td></td>
<td>1.0000</td>
<td></td>
</tr>
<tr>
<td><strong>Output Flows</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Recovered energywood</td>
<td>m³</td>
<td>0.880</td>
<td>0.880</td>
<td>0.880</td>
</tr>
</tbody>
</table>
| CO₂ emission from LUC                      | kg        | 180.16         | 180.16                  |        | 0.01   | 8.01   | Keith *et al.* 2015
The input and output flows of the post-harvesting forwarder and the chipper processes are shown in Tables 5.5 and 5.6. These represent the material and energy requirements for one productive machine hour (PMH), which per common practice includes work stoppages or delays shorter than 15 minutes (Klvac and Skoupy 2009, Laitila and Vääätäinen 2012, Schumeyer and Hüttl 2014), along with the associated environmental emissions. Also accounted for are the 50 and 60 km round-trips to the site by lorry of the forwarder and the chipper, respectively. These processes are a good example of the comprehensive environmental impact related flows compiled in the ecoinvent database for product, machine, and infrastructure related processes.

Table 5.5 Forwarding process flows for 1 hour of work on-site.

<table>
<thead>
<tr>
<th>Assortments</th>
<th>Input Flows</th>
<th>Amount</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>All</td>
<td>diesel, low-sulphur</td>
<td>9.24</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Forwarder</td>
<td>5.68E-05</td>
<td>Item(s)</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>All</td>
<td>lubricating oil</td>
<td>0.35432</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Forwarder transport, lorry 16-32 t</td>
<td>44.88656</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>All</td>
<td>waste mineral oil</td>
<td>-0.39369</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>Output Flows</td>
<td>Ammonia</td>
<td>0.00018</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Benzo(a)pyrene</td>
<td>2.77E-07</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Cadmium</td>
<td>9.25E-08</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Carbon dioxide, fossil</td>
<td>28.86946</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Carbon monoxide, fossil</td>
<td>0.10282</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Copper</td>
<td>1.57E-05</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Dinitrogen monoxide</td>
<td>0.00111</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Dioxins, 2,3,7,8-tetrachlorodibenzo-p</td>
<td>5.54E-13</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td><strong>forwading, forwarder</strong></td>
<td>1 h</td>
<td></td>
<td><strong>Reference Product</strong></td>
</tr>
<tr>
<td></td>
<td>Methane, fossil</td>
<td>0.00046</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Nickel</td>
<td>6.49E-07</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Nitrogen oxides</td>
<td>0.187</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>NMVOC</td>
<td>0.01851</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>PAH</td>
<td>3.10E-05</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Particulates, &lt; 2.5 um</td>
<td>0.00804</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Particulates, &gt; 10 um</td>
<td>0.00054</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Particulates, &gt; 2.5 um, and &lt; 10um</td>
<td>0.00036</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Selenium</td>
<td>9.25E-08</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>Zinc</td>
<td>9.25E-06</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
</tbody>
</table>
Table 5.6 Chipping process (at the roadside) flows for 1 hour of work on-site.

<table>
<thead>
<tr>
<th>Assortments</th>
<th>Input Flows</th>
<th>Amount</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>All</td>
<td>Chipper, mobile, diesel</td>
<td>6.67E-05</td>
<td>Item(s)</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>All</td>
<td>diesel, low-sulphur</td>
<td>5.88E+01</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>lubricating oil</td>
<td>0.9245</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Chipper transport, lorry 16-32 t</td>
<td>93.6</td>
<td>t*km</td>
<td>ecoinvent, This study</td>
</tr>
<tr>
<td>All</td>
<td>waste mineral oil</td>
<td>-1.02722</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td><strong>Output Flows</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All</td>
<td>Ammonia</td>
<td>0.00118</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Benzo(a)pyrene</td>
<td>1.76E-06</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Cadmium</td>
<td>5.89E-07</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Carbon dioxide, fossil</td>
<td>183.71472</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Carbon monoxide, fossil</td>
<td>0.65433</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Copper</td>
<td>9.99E-05</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Dinitrogen monoxide</td>
<td>0.00705</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Dioxins, 2,3,7,8-</td>
<td>3.52E-12</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td></td>
<td>tetrachlorodibenzo-p</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All</td>
<td>Methane, fossil</td>
<td>0.00197</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Nickel</td>
<td>4.13E-06</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Nitrogen oxides</td>
<td>6.53E-01</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>NMVOC</td>
<td>0.07992</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>PAH</td>
<td>0.0002</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Particulates, &lt; 2.5 um</td>
<td>2.41E-02</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Particulates, &gt; 10 um</td>
<td>0.0016</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Particulates, &gt; 2.5 um, and &lt; 10um</td>
<td>0.00107</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td>Selenium</td>
<td>5.89E-07</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
<tr>
<td>All</td>
<td><strong>wood chipping, chipper, diesel</strong></td>
<td><strong>1.00E+00</strong></td>
<td>h</td>
<td>Reference Product</td>
</tr>
<tr>
<td>All</td>
<td>Zinc</td>
<td>5.89E-05</td>
<td>kg</td>
<td>ecoinvent</td>
</tr>
</tbody>
</table>

The transport process for the chipped and shredded SRF biomass assortments from the 3 × 10 year rotations, converted from m³ to tonnes, are shown in Table 5.7. The conversion factors (CFs) used for converting solid stem, whole-tree, or stumpwood to woodchip and hogfuel, and the CFs used in converting the chip and hogfuel loose volume to tonnes, are given in Table 5.0. The transport distance for the roundtrip from the plantation site to the gates of the EPL plant is assumed to be 200 km, with a 30 % loading for the empty return leg. The area of Ireland within a 100 km radius of the EPL plant covers approximately 1/3 of the country (Figure 5.4).
Table 5.7 Transport process for biomass woodchip and hogfuel to plant gate.

<table>
<thead>
<tr>
<th>Assortments</th>
<th>Input Flows</th>
<th>Amount</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stem</td>
<td>SRF stem-only woodchips - at roadside</td>
<td>1615</td>
<td>m³</td>
<td>This study</td>
</tr>
<tr>
<td>Stump</td>
<td>SRF stump hogfuel - at roadside</td>
<td>241</td>
<td>m³</td>
<td>This study</td>
</tr>
<tr>
<td>Wholetree</td>
<td>SRF wholetree woodchips - at roadside</td>
<td>2124</td>
<td>m³</td>
<td>This study</td>
</tr>
<tr>
<td>All</td>
<td>Transport energywood, lorry 16-32 t</td>
<td>83589</td>
<td>t*km</td>
<td>This study</td>
</tr>
</tbody>
</table>

Output Flows

| All         | Energywood at the gate, after chipping | 1286   | t     | Reference product |

Figure 5.4 The area of existing forestry and areas with soil types suitable to afforestation in Ireland, within 100 km radius of the Edenderry Power Ltd. peat and biomass co-firing electricity generation plant (based on map from Schulte et al. 2016).
5.3.2.1 Sensitivity Analysis

Sensitivity analyses of some key parameters with high uncertainty within the LCA system boundary were incorporated into the SRF and Sitka spruce biomass life cycles. Specifically, those parameters were:

- Assortments and associated yields of potentially available SRF and Sitka spruce biomass for bioenergy. Individual assortments and combinations of assortments were analysed using different scenarios. NB: The percentage of recoverable biomass, e.g., stem wood or whole tree, forest residues (brash) and stumps is generally dependent on site conditions and nutrient balances.

- Co-firing rates and exclusive fuel usage. The LCA scenarios were altered to account for two peat and biomass co-firing rates of 30 and 50 %, along with exclusive (100 %) peat and biomass usage at the EPL plant.

- Fertilizer applications. The impact of fertilizer use was analysed in scenarios SS-4 and SRF-4, using the maximum permissible aerial application of N (350 kg), P (350 kg), and K (250 kg) per hectare (DAFM 2014a), once per rotation.

5.3.3 LCIA methodology, and its modification

In the LCIA phase of the study, the ReCiPe 2008 (Goedkoop et al. 2013) Midpoint (H, for hierarchist) method was chosen to conduct the characterisation of each scenarios climate change related flows. The characterisation assigns the mass unit (kg CO$_2$-eq) and GWP factor (e.g. CO$_2$ = 1, CH$_4$ = 25) to each relevant flow, and enables the quantification the total GWP environmental impact of each process. The ReCiPe method was chosen due to it being an updated and more comprehensive method than its predecessors, i.e., CML 2001 (Guinée et al. 2002) and Eco-Indicator 99 (Goedkoop et al. 2013). The ReCiPe method was then modified using the openLCA software by
adding two negative value (GWP “Factors” of −1) C related flows which were present in CML 2001 but not in ReCiPe, i.e., “Carbon dioxide; resource/unspecified”, and “Carbon dioxide, in air; resource/in air”. Those flows were added to isolate and uniquely identify the GHG balance impacts of soil C sinks associated with LUC to Sitka spruce plantations, and the biomass C sequestration (BCS) associated with all biomass scenarios. Three more flows were added, i.e., “Carbon dioxide, in air - LUC CO\textsubscript{2} emission”, “Carbon dioxide, in air - peat combustion CO\textsubscript{2} emission”, and “Carbon dioxide, in air - wood combustion CO\textsubscript{2} emission”, which were also necessary to compartmentalise the GHG balance impacts of SRF related LUC, and peat and wood combustion, respectively.

5.4 Results

The aim of this research chapter was to assess the GHG balance of indigenous SRF biomass used for bioenergy (LCA scenarios SRF1–4) as a substitute for fossil fuel (peat) at the EPL co-firing electricity generating plant. The reference systems used in the LCA are the peat-only scenario (Peat-0) and the potentially available assortments of biomass for bioenergy from Sitka spruce plantations, i.e. scenarios SS1–4. The cradle-to-grave, cradle-to-gate, and gate-to-grave LCA results were determined for all the SRF, Sitka spruce, and peat scenarios, in kg CO\textsubscript{2}-eq MWh\textsuperscript{−1} and t CO\textsubscript{2}-eq ha\textsuperscript{−1}.

The study focused on quantifying the GHG balance of the scenarios at the following biomass-for-peat substitution percentages:

1. 0 % (i.e. the 100 % peat reference system)
2. 30 % (the 2015 EPL plant ratio of 70 % peat to 30 % biomass)
3. 50 % (the 2020 targeted EPL plant ratio of 50 % peat and 50 % biomass)
4. 100 % (i.e. a theoretical future use of 100 % biomass at the EPL plant)
The results are presented by scenario and peat substitution percentage, per MWh<sub>e</sub> and per hectare, using four GHG balance hierarchies within the overall life cycle, as follows:

i. Overall cradle-to-grave GHG balance

ii. Cradle-to-gate and gate-to-grave GHG balance

iii. Disaggregated biomass and peat cradle-to-gate GHG balance

iv. GHG balance of the biomass and peat processing and transport processes

Hereafter, the numeric suffix to the peat, Sitka spruce, and SRF scenarios (e.g. Peat-0, SS1-30, SRF1-50, etc.) represent the percentage of biomass in the co-firing fuel mix. The 100 % peat (Peat-0), 100 % Sitka spruce (SS4-100) and 100 % SRF (SRF4-100) biomass scenarios represent theoretical operations at the EPL plant. The purpose of including the exclusive peat and wood biomass scenarios in the results is to provide a comparative fossil fuel baseline value, and a potential future all-biomass “reference point” at either end of the 30 % and 50 % co-firing ratios.

5.4.1 Cradle-to-grave greenhouse gas balances

The cradle-to-grave LCA of the net GHG balance from four Sitka spruce and four SRF biomass scenarios co-fired with peat, were quantified. The GHG balance results in kg CO<sub>2</sub>-eq MWh<sub>e</sub><sup>−1</sup> and t CO<sub>2</sub>-eq ha<sup>−1</sup> of plantation (Figures 5.5 and 5.7, respectively), i.e., the primary and secondary LCA functional units, respectively, are presented in the following four peat-substitution or co-firing percentages: 0, 30, 50, and 100.

The cradle-to-grave GHG balance of the Peat-0 scenario is 1217 kg of CO<sub>2</sub>-eq MWh<sub>e</sub><sup>−1</sup>, which is almost six times greater than the GHG balance of the SRF4-100 scenario, i.e. 216 kg of CO<sub>2</sub>-eq MWh<sub>e</sub><sup>−1</sup> (Figure 5.5). In contrast the GHG balance of
the Peat-0 scenario is three times the value for SS4-100 scenario, at 394 kg CO₂-eq MWhₑ⁻¹. The GHG balance MWhₑ⁻¹ results for the SRF scenarios SRF1-30–SRF4-30 show respective increasing values within a narrow 3 % range, from 889 to 916 kg CO₂-eq. The range also increases stepwise across the SRF1-50–SRF4-50 scenarios, from 670 to 716 kg CO₂-eq MWhₑ⁻¹. At over 6 %, the SRF1-50–SRF4-50 range is double the SRF1-30–SRF4-30 range. The GHG balances for the SS1-30–SS4-30 and SS1-50–SS4-50 scenarios have wider ranges of 11 % and 21 %, from 882 to 987 kg CO₂-eq MWhₑ⁻¹, and from 659 to 833 kg CO₂-eq MWhₑ⁻¹, respectively.

The mean value of the SRF1-30–SRF4-30 range is 908 kg CO₂-eq MWhₑ⁻¹, whereas the mean for the SRF1-50–SRF4-50 scenarios is 23 % lower at 701 kg CO₂-eq MWhₑ⁻¹. When comparing the latter mean value to the SRF4-100 scenario GHG balance of 216 kg CO₂-eq MWhₑ⁻¹ there is a 70 % decrease in the GHG balance for an associated 50 % increase in SRF biomass used. The Sitka spruce scenarios SS1-30–SS4-30 have a mean GHG balance MWhₑ⁻¹ of 936 kg CO₂-eq compared with a mean of 749 kg CO₂-eq for the SS1-50–SS4-50 scenarios, a difference of 20 % which is in line with the percentage increase in the Sitka spruce biomass used. In contrast to the SRF biomass results above, the difference between the mean GHG balances for 50 % and 100 % peat substitution by Sitka spruce biomass matches the percentage increase (i.e. 50 %) in the consumed Sitka spruce biomass.
Figure 5.5 The GHG balance (kg CO₂-eq) per MWhₑ of the cradle-to-grave LCA of the peat, Sitka spruce, and SRF biomass scenarios, at 0, 30, 50, and 100 % substitution of peat at the combustion phase of the life cycle. Subst.: substitution.

The GHG balance per hectare results (Figure 5.7) for the Sitka spruce and SRF biomass scenarios, paint a very different picture of their respective environmental impacts, primarily due to their substantially different biomass yields and energy potential per hectare (Table 5.0 and Figure 5.6, respectively). Despite the large difference in the absolute GHG balance per hectare for the Sitka spruce and SRF scenarios, the ratio of ODT ha⁻¹ to t CO₂-eq ha⁻¹ for SS1-30–SS4-30 and SRF1-30–SRF4-30 have relatively narrow ranges, between 5.4–6.1 ODT ha⁻¹ and 5.5–5.9 ODT ha⁻¹, respectively, with the mean for both ranges = 5.8 ODT ha⁻¹. The same ratios for SS1-50–SS4-50 and SRF1-50–SRF4-50 scenarios are also within relatively narrow ranges, i.e., 2.4–3.1 and 2.5–2.8, with means of 2.8 and 2.7 respectively.
Figure 5.6 The estimated energy potential from the woodchip and hogfuel from the Sitka spruce and SRF biomass produced in each scenario, converted to MWh_e ha^{-1}, at 40 % moisture content (MC).

The contrast between the Peat-0 and the SS4-100 and SRF4-100 scenarios GHG balances per hectare (Figure 5.7) is indicative of the renewable biomass “carbon neutrality” argument for sustainable bioenergy versus energy generation from fossil fuels. At 14,055 t CO_2-eq ha^{-1} the Peat-0 GHG balance is over 70 times greater than the SRF4-100 value (197 t CO_2-eq ha^{-1}), and almost 185 times greater than the SS4-100 value (78 t CO_2-eq ha^{-1}). The mean GHG balance for the SRF1-30–SRF4-30 scenarios is 2441 t CO_2-eq ha^{-1}, which is more than double the mean (1134 t CO_2-eq ha^{-1}) for the SRF1-50–SRF4-50 scenarios. The mean GHG balance for the SS1-30–SS4-30 scenarios is 339 t CO_2-eq ha^{-1}, which is double the mean (168 t CO_2-eq ha^{-1}) for the SS1-50–SS4-50 scenarios.
5.4.2 Cradle-to-gate and gate-to-grave greenhouse gas balances

The cradle-to-grave GHG balances by scenario shown above represent the consolidation of the GHG balances of the processes accounted for within the LCA system boundary. The GHG balances per MWh of all the assessed scenarios were disaggregated into six components (Figure 5.8), i.e., “biomass-at-the-gate” (BAG), “peat-at-the-gate” (PAG), “power station” (PS) infrastructure, “peat combustion” (PC), “biomass combustion” (BC), and “ash disposal” (AD). Those six GHG balance components represent both net GHG sinks, e.g., BAG, and emissions, e.g., BC. They can also be categorised into cradle-to-gate (i.e. BAG and PAG) and gate-to-grave (i.e. PS, PC, BC, AD) components of the total GHG balance for each LCA scenario. The positive values of GHG emissions
from the combined PC, BC, PS, and AD components exceed the negative GHG balance values for the BAG components, giving the positive overall GHG balance values.

The PC component of the Peat-0 scenario, at 1173 kg CO$_2$-eq MWh$^{-1}$, accounts for over 96 % of the total GHG balance MWh$^{-1}$. The sum of three of the six GHG balance components, BAG, PC, and BC, comprise 96–97 % of the total GHG balance for all the Sitka spruce and SRF co-firing scenarios at both the 30 % and 50 % peat substitution levels, and 98–99 % of the total GHG balance for the two 100 % peat substitution scenarios. The negative BAG values for all the Sitka spruce scenarios (range: −539 to −145 kg CO$_2$-eq MWh$^{-1}$; mean: −292 kg CO$_2$-eq MWh$^{-1}$) and SRF scenarios (range: −439 to −221 kg CO$_2$-eq MWh$^{-1}$; mean: −311 kg CO$_2$-eq MWh$^{-1}$), reflects the substantial quantities of C sequestered in the biomass on arrival at the EPL power plant. The variations in the BAG MWh$^{-1}$ values across the Sitka spruce and SRF scenarios also reflect the varying GHG emissions from the mechanised forest management and transport operations, the details of which are dealt with later in the Results section.

The per-hectare view of those same six components (Figure 5.9) again reflects the assumed peat yield over 30 years and the assumed Sitka Spruce and SRF biomass yields by scenario over their respective rotations. For scenarios SS1-30–SS4-30 the mean percentage split of the biomass to peat combustion (BC:PC) GHG balance, which is a product of the quantity, the C content, and net calorific value (NCV) of each fuel used, is 35:65 %. The same mean percentage split value for scenarios SS1-50–SS4-50 is 19:81 %, which demonstrates the GHG mitigation potential of increasing the substitution of peat with biomass in the bioenergy sector.

The GHG balance of the PC component of the Peat-0 scenario, which represents the burning of the milled peat yield from one hectare of Irish peatland per year over a 30
year period, is estimated to be 13,545 t CO₂-eq. That PC value is four times greater than the combined PC and BC components of the SRF4-30 scenario (3358 t CO₂-eq ha⁻¹), and seven times greater than the same components for the SRF4-50 scenario (1930 t CO₂-eq ha⁻¹), over the same period. The lower quantities of available Sitka spruce biomass per hectare over a 30 year rotation means the PC component of the Peat-0 scenario is almost 19 times greater than the combined PC and BC components of the SS4-30 scenario (721 t CO₂-eq ha⁻¹), and almost 33 times greater than the same components for the SS4-50 scenario (412 t CO₂-eq ha⁻¹).
Figure 5.8 The GHG balance (kg CO2-eq) per MWhe of the cradle-to-gate (BAG, PAG) and gate-to-grave processes (PS, PC, BC, AD) at the EPL power-station. BAG: biomass-at-the-gate, PAG: peat-at-the-gate, PS: power station infrastructure, PC: peat combustion, BC: biomass combustion, AD: ash disposal. The data labels in the figure are for the three main contributors to the net GHG balance, i.e., BAG, PC, and BC.
Figure 5.9 The GHG balance (t CO$_2$-eq) per hectare of the cradle-to-gate (BAG, PAG) and gate-to-grave processes (PS, PC, BC, AD) at the EPL power-station. BAG: biomass-at-the-gate, PAG: peat-at-the-gate, PS: power station infrastructure, PC: peat combustion, BC: biomass combustion, AD: ash disposal. The data labels in the figure are for the three main contributors to the net GHG balance, i.e., BAG, PC, and BC.
5.4.3 Disaggregated biomass and peat cradle-to-gate greenhouse gas balances

The biomass-at-the-gate (BAG) values from Figures 5.8 and 5.9 are further disaggregated into three components, namely, “biomass land-use change” (BLUC), “biomass C sequestration” (BCS), and “biomass processing and transport” (BP&T) in Figures 5.10 and 5.11, per MWh\(_e\) and per hectare, respectively. Likewise the PAG values are disaggregated into “peat land-use change” (PLUC) and “peat processing and transport” (PP&T). The most influential factor in the difference between the peat and biomass scenarios is the accounting of the BCS component of their cradle-to-gate GHG balances. The BCS differences between the Sitka spruce and the SRF scenarios are attributable to a combination of factors, e.g., the variation in the respective basic densities and C densities of the biomass assortments in each scenario, along with the solid-wood to woodchip bulk density conversion factors. The Sitka spruce biomass has both a lower assumed basic density (380 kg m\(^{-3}\)) and C density (46.0 %) than the \textit{E. nitens}, i.e. 435 kg m\(^{-3}\) and 49.4 %, respectively, used in the SRF modelling.

The mean GHG balance of the BCS components (per MWh\(_e\)) for the SS1-30–SS4-30 and SS1-50–SS4-50 scenarios are −196 and −327 kg CO\(_2\)-eq MWh\(_e\)^\(-1\), respectively. Those values are 38 % less than the same mean values for the SRF1-30–SRF4-30 and SRF1-50–SRF4-50 scenarios of −315 and −524 kg CO\(_2\)-eq MWh\(_e\)^\(-1\), respectively. The accumulated effect of the varying biomass assortment densities and solid-to-bulk conversion factors is most evident in the 43 % difference in the BCS values of the SS4-100 (−571 kg CO\(_2\)-eq MWh\(_e\)^\(-1\)) and SRF4-100 (−1005 kg CO\(_2\)-eq MWh\(_e\)^\(-1\)) scenarios.

The net calorific value of the peat is constant for all scenarios and therefore the quantity of peat required only changes based on the peat substitution percentage of the
scenarios. Hence, regardless of biomass type used, the PLUC and PP&T components of the GHG balance $\text{MWh}_{\text{e}}^{-1}$ remain constant in each of the 30 % scenarios, and likewise for the 50 % scenarios. For the SS1-30–SS4-30 and SRF1-30–SRF4-30 scenarios the PLUC and PP&T components are 8.1 and 17.3 kg CO$_2$-eq $\text{MWh}_{\text{e}}^{-1}$, respectively. The same components of the SS1-50–SS4-50 and SRF1-50–SRF4-50 scenarios are 5.8 and 12.3 kg CO$_2$-eq $\text{MWh}_{\text{e}}^{-1}$, respectively. The PLUC component of the Peat-0 scenario is 11.5 kg CO$_2$-eq $\text{MWh}_{\text{e}}^{-1}$, while the PP&T component is 24.7 kg CO$_2$-eq $\text{MWh}_{\text{e}}^{-1}$.

The biomass yield per hectare for each of the Sitka spruce and the SRF scenarios are the same regardless of the respective peat substitution percentage for the scenario. Hence, there is no difference in the BCS results per hectare (Figure 5.11) for the SS1-30–SS4-30 and SS1-50–SS4-50 scenarios (mean values for both = −60 t CO$_2$-eq ha$^{-1}$), or the SRF1-30–SRF4-30 and SRF1-50–SRF4-50 scenarios (mean values for both = −834 t CO$_2$-eq ha$^{-1}$).

The results are likewise for the BLUC per hectare across the 30 % and 50 % co-firing rates for the Sitka spruce (mean values for both = −2.8 t CO$_2$-eq ha$^{-1}$) and the SRF (mean values for both = 189 t CO$_2$-eq ha$^{-1}$) scenarios. Also, comparing the SS4-100 and SRF4-100 scenarios BLUC values, both per $\text{MWh}_{\text{e}}$ (−26 and 228 kg CO$_2$-eq $\text{MWh}_{\text{e}}^{-1}$, respectively) and per hectare (−5 and 208 t CO$_2$-eq ha$^{-1}$, respectively), highlight the differences in their respective LUC soil C accounting, where the BLUC of the 3×10 year SRF is a CO$_2$ emissions source, while the longer 30 year rotation Sitka spruce BLUC represents a soil C sink. The PLUC per hectare of the Peat-0 scenario is 133 t CO$_2$-eq ha$^{-1}$, while the PP&T per hectare is 285 t CO$_2$-eq ha$^{-1}$.
Figure 5.10 The GHG balance (kg CO$_2$-eq) per MWh$_e$ of the aggregated biomass land-use change (BLUC), biomass carbon sequestration (BCS), biomass processing and transport (BP&T), peat land-use change (PLUC), and peat processing and transport (PP&T) operations of each scenario.

NB: i) The data label values are rounded for display purposes, ii) The Sitka spruce BLUC values are shown just below the x-axis, iii) The Sitka spruce and SRF BCS values are shown below their bars.
Figure 5.11 The GHG balance (t CO₂-eq) per hectare of the aggregated biomass land-use change (BLUC), biomass carbon sequestration (BCS), biomass processing and transport (BP&T), peat land-use change (PLUC), and peat processing and transport (PP&T) operations of each scenario. The data label values are rounded for display purposes.
5.4.4 Greenhouse gas balances of the biomass and peat processing and transport processes

The consolidated GHG balance results of the mechanised cradle-to-gate peat and forestry management operations for all the scenarios are in the range of 32.26 (SS2-50) to 41.64 (SRF4-100) kg CO$_2$-eq MWh$_e$^{-1} (Figure 5.12). Excluding the peat production and peat transport components from all the Sitka spruce and SRF biomass scenarios, the majority of the GHG emissions are from the biomass transport and harvesting processes. The transport processes for the Sitka spruce SS1-30–SS4-30 and SS1-50–SS4-50 account for 2.92 and 4.87 kg CO$_2$-eq MWh$_e$^{-1}, respectively, while the mean GHG emissions from the harvesting processes for the same scenarios account for 2.35 and 3.91 kg CO$_2$-eq MWh$_e$^{-1}, respectively. Similarly for the SRF1-30–SRF4-30 and SRF1-50–SRF4-50 scenarios, the mean GHG emissions from the transport processes account for 2.81 and 4.69 kg CO$_2$-eq MWh$_e$^{-1}, respectively, while the mean GHG emissions from the harvesting processes for the same scenarios account for 2.78 and 4.63 kg CO$_2$-eq MWh$_e$^{-1}, respectively.

The remaining forestry processes, i.e., forwarding, chipping, and site establishment (which includes seedling production), make up the remainder of the respectively decreasing GHG emission contributions. The site establishment GHG emissions for all the non-fertilized Sitka spruce scenarios, i.e., SS1-30–SS3-30, and SS1-50–SS3-50, range from 0.09–0.11 kg CO$_2$-eq, and from 0.15–0.18 kg CO$_2$-eq, respectively. Likewise, for all the non-fertilised SRF scenarios, i.e., SRF1-30–SRF3-30, and SRF1-50–SRF3-50, the site establishment GHG emissions have a narrow range from 0.23–0.27 kg CO$_2$-eq MWh$_e$^{-1}, and from 0.38–0.45 kg CO$_2$-eq MWh$_e$^{-1}, respectively. Regardless of the peat substitution level or the tree species, the site establishment result differences between the scenarios is mainly attributable to the basic
density and calorific value of their respective biomass assortments, and the proportion of the site area required to supply the biomass to produce the one MWh\textsubscript{e} functional unit. The only difference between the SS3 and SS4 scenarios, and between the SRF3 and SRF4 scenarios, is the use of fertilizer in the site establishment process, with 93 % of the SS4 and 94 % of the SRF4 scenarios site establishment GHG emissions attributable to the fertilizer production and transport.

Figure 5.12 The GHG balance MWh\textsubscript{e}^{-1} of the SRF and Sitka spruce biomass production and transport phases of the 30 and 50 % peat substitution levels for all LCA scenarios. SE: site establishment, H: harvesting, F: forwarding, C: chipping, T: transport, PP: peat production, PT: peat transport. NB: The data label values are for the H, T, and PP components, and are rounded for display purposes.
The per-hectare GHG balance of the mechanised cradle-to-gate peat and forestry management operations (Figure 5.13) are very much indicative of the respective assumptions applied to the Sitka spruce and SRF biomass yields. The far greater biomass available per hectare from the SRF scenarios result in mean GHG balances of 27 t CO$_2$-eq ha$^{-1}$ for both the SRF1-30–SRF4-30 and SRF1-50–SRF4-50 scenarios, which far exceeds the mean value of 3.3 t CO$_2$-eq ha$^{-1}$ for the SS1-30–SS3-30, and SS1-50–SS3-50 scenarios.

**Figure 5.13** The GHG balance ha$^{-1}$ of the SRF and Sitka spruce biomass production and transport phases of the 30 and 50 % peat substitution levels for all LCA scenarios. SE: site establishment, H: harvesting, F: forwarding, C: chipping, T: transport, PP: peat production, PT: peat transport. NB: The data label values are for the H, T, and PP components, and are rounded for display purposes.
5.4.5 Potential annual greenhouse gas emission reductions

Given an annual output of 980 gigawatt hours of electricity (GWh$_e$) from the EPL plant (Bord na Mona 2018b), this study shows that in comparison to the peat-only scenario (i.e. Peat-0, as measured at the per-megawatt-hour level), the SS4 scenario at 30 and 50 % co-firing could potentially deliver GHG emission reductions of 0.24 and 0.40 Mt CO$_2$-eq yr$^{-1}$, respectively. Those Sitka spruce emission reductions in Mt CO$_2$-eq yr$^{-1}$ equate to annual reductions in GHG emissions of 20 and 34 %, respectively (Figure 5.14), in relation to the Peat-0 scenario.

Making the same comparison for the SRF4 scenario, 30 and 50 % co-firing could potentially deliver GHG emission reductions of 0.29 and 0.49 Mt CO$_2$-eq yr$^{-1}$, respectively. Those SRF4 emission reductions in Mt CO$_2$-eq yr$^{-1}$ equate to annual reductions of emissions from the Peat-0 scenario of 25 and 41 %, respectively. Under the SS4-100 and SRF4-100 scenarios (100 % Sitka spruce and SRF biomass, respectively) the potential GHG emission reductions are 0.81 and 0.98 Mt CO$_2$-eq yr$^{-1}$, respectively. The 100 % Sitka spruce and SRF biomass emission reductions in Mt CO$_2$-eq yr$^{-1}$ equate to annual reductions in GHG emissions from the Peat-0 scenario of 68 and 82 %, respectively, in relation to the Peat-0 scenario.
Figure 5.14 Potential GHG emissions percentage reduction (per MWh\text{e}^{-1} basis) by scenario from co-firing peat and biomass for a full year of the Edenderry power plant operation, compared to exclusive peat usage, i.e., the Peat-0 scenario.
5.5 Discussion

5.5.1 Research aims and previous Irish LCA studies

This LCA study evaluated the cradle-to-grave GHG balances of indigenous Irish SRF biomass for bioenergy, and compared the results to a conventional forest biomass reference system, i.e., potentially available indigenous Sitka spruce assortments. The LCI data, assumptions, and parameters focused on the use of *E. nitens* in Irish SRF plantations as fast growing source of biomass for bioenergy, and the use of indigenous Sitka spruce and milled peat in electricity generation at the EPL plant. The LCI data models and parameters used in this study are readily adaptable to the physical, chemical and biological attributes of any other tree species that may be seemed suitable to SRF management practices.

The study also assessed the GHG balance of both the SRF and Sitka spruce biomass scenarios relative to the use of milled peat (fossil fuel) at the EPL plant. To the best of the author’s knowledge there are no published LCA studies of Irish SRF biomass for bioenergy. Previous Irish LCA studies have evaluated the GHG emissions of biomass for bioenergy from Sitka spruce plantations (Murphy *et al.* 2014a, Fitzpatrick 2016), from a Miscanthus bioenergy crop (Styles and Jones 2008b, Murphy *et al.* 2013), a short rotation coppice willow plantation (Styles and Jones 2007b, 2008a; Murphy *et al.* 2014b), and from peat harvesting (Murphy *et al.* 2015).

5.5.2 LCA scope and system boundaries

The LCA system boundaries for those Irish studies vary from cradle-to-gate (Murphy *et al.* 2014a, 2014b; Styles and Jones 2007a) to cradle-to-grave (Murphy *et al.* 2015, 2016; Styles and Jones 2008b). Using ash recycling (e.g. in concrete production) versus disposal as an example of significantly different impacts, Muench and Guenther (2013)
stated the importance of including the end-of-life process in bioenergy LCAs. Also, following their recommendation the system boundaries for the Sitka spruce and SRF scenarios in this LCA study (Figure 5.2) were harmonised to improve the comparability of the results. Both biomass streams were derived from very similar forest management processes, with just a few divergences in their life cycles and the assortments produced. For example, the Sitka spruce pulpwood thinning and brash bundling are harvesting operations that do not occur in the SRF biomass life cycle.

In a review of 28 forest biomass LCA studies Klein et al. (2015) categorised 25 different cradle-to-gate processes. There were a maximum of 13 considered processes in any one of those studies, and a mean of nine processes considered. Only 14 % of those 28 studies considered a cradle-to-grave approach to the LCA system boundary. This LCA study used LCI data for 16 of the 25 considered cradle-to-gate processes analysed by Klein et al. (2015). This study also included data on biomass and soil C gains and losses, power station build and maintenance, and ash disposal, i.e., data which are often neglected in bioenergy LCAs (Muench and Guenther 2013), placing this LCA at the high end of comprehensiveness for such studies.

5.5.3 Study limitations and uncertainty analysis

The limitations of the study and the issue of uncertainty are linked in several ways through the need for assumptions regarding fundamental LCI data items, and based on data from related literature and databases. The Sitka spruce and SRF biomass assortments within their respective scenarios were chosen to reflect the current uncertainties around how power plants like EPL will fulfil its growing biomass feedstock demands. The dominance of Sitka spruce in Irish forestry makes the associated pulpwood, brash, and stumps the most likely large-scale source of feedstock
in the near-term, but the mobilisation of that feedstock is uncertain due to cost and environmental factors such as nutrient cycling (Murphy et al. 2016). The study was further limited by the lack of Irish data on mature SRF plantation yields and their effect on soil C stocks over multiple rotations, and SRF biomass for bioenergy operations in Ireland. Future Irish biomass LCA studies that could potentially provide key field data related to those limitations, and thereby replace the aforementioned assumed data, will make those LCAs stronger. Data assumptions, e.g., the GHG balances of peat and biomass LUC, biomass C sequestration, or fertilizer use, and their inherent uncertainties are carried into the modelling of each scenarios life cycle, though the magnitude and range of those sinks or sources is transparent in the presented results. Also, although the study focused solely on the climate change impact of the scenarios, the LCIA method used (ReCiPe) produced results for several other environmental impacts (e.g. eutrophication and acidification potentials), but these were deemed beyond the scope of this LCA.

5.5.4 Biomass and peat greenhouse gas balances per MWh$_e$

The GHG balance of producing and transporting Irish Sitka spruce and SRF biomass, and combusting it exclusively, and at 30 % and 50 % co-firing rates with milled peat (fossil fuel) was determined. At the 30 % co-firing rate, the mean GHG emission reductions per MWh$_e$, compared to the peat-only scenario (i.e. 1217 kg CO$_2$-eq MWh$_e^{-1}$), for the SRF and Sitka spruce scenarios are 25 % and 23 %, respectively, and 42 % and 38 %, respectively, at the 50 % co-firing rate. While there is a large GHG balance disparity (per MWh$_e$) between the peat and biomass scenarios, there is only a 2 % and 4 % difference between the mean GHG balances for the Sitka spruce and SRF biomass scenarios at 30 and 50 % co-firing, respectively.
5.5.5 Peat greenhouse gas balance per MWh_e

The GHG balance MWh_e\(^{-1}\) for the Peat-0 scenario in this study (1217 kg CO\(_2\)-eq) is almost 3 % higher than the 1183 kg CO\(_2\)-eq reported by Bord Na Mona (2018), the operator of the EPL plant, and 5 % higher than the 1150 kg CO\(_2\)-eq reported by Styles and Jones (2007c), and Murphy et al. (2015b) for a smaller 100 MWh_e peat plant. Bord na Mona do not breakdown their GHG balance MWh_e\(^{-1}\) components from peat, or separately report indirect GHG emissions from the drainage of the peat extraction area (Bord na Mona 2018), which Styles and Jones (2007c) estimated to be 44 kg CO\(_2\)-eq MWh_e\(^{-1}\). This study estimated the LUC (from pristine bog to peat harvesting) related GHG emissions to be 11.5, 8.1, and 5.8 kg CO\(_2\)-eq MWh_e\(^{-1}\) at the 100, 50, and 30 % peat substitution levels, respectively. The largest contributing factor to the GHG balance differences between this study and the Styles and Jones (2007c) values are the LUC emission factors used. This study used the Wilson et al. (2015) value of 1.7 (±0.47) t CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) which is much lower than the IPCC (2013) Wetlands Supplement Tier 1 emission factor (2.8 ±1.7 t CO\(_2\)-eq ha\(^{-1}\) yr\(^{-1}\)), and the 2.09 t CO\(_2\) ha\(^{-1}\) yr\(^{-1}\) (using CO\(_2\) emissions only, excluding CH\(_4\) and DOC - in line with SRF LUC) used by Murphy et al. (2015b).

5.5.6 Peat and biomass co-firing greenhouse gas balance per MWh_e

The GHG balance MWh_e\(^{-1}\) of the Sitka spruce 30 % co-firing scenarios are in a range of 879-984 kg CO\(_2\)-eq MWh_e\(^{-1}\), with a mean of 933 kg CO\(_2\)-eq MWh_e\(^{-1}\), which is 20 % higher than the mean value of 744 kg CO\(_2\)-eq MWh_e\(^{-1}\) for their 50 % co-firing scenarios. The impact of the extra 20 % Sitka spruce for peat substitution in lowering the GHG balance of the electricity generation is mirrored in the mean values for the 30 and 50 % co-firing rates. The 23 % difference in the mean GHG balance MWh_e\(^{-1}\)
between the SRF 30 and 50 % co-firing scenarios (i.e. 905 and 697 kg CO2-eq MWh\(_e\)^{-1}, respectively) also reflects the lower GHG emissions from the 20 % increase in biomass versus peat combustion.

5.5.7 Greenhouse gas balances per hectare

When assessed on a per-hectare basis, the study also highlights the large gap in the GHG emissions reduction from using SRF biomass versus the potentially available biomass from conventional Sitka spruce plantations. The mean GHG balance of the SRF scenarios (i.e. 2441 t CO\(_2\)-eq ha\(^{-1}\) and 1134 t CO\(_2\)-eq ha\(^{-1}\), at the 30 % and 50 % co-firing rates, respectively) are both approximately seven times greater than the GHG balance of the Sitka spruce scenarios (i.e. 339 t CO\(_2\)-eq ha\(^{-1}\) and 168 t CO\(_2\)-eq ha\(^{-1}\), at the same respective co-firing rates). Hence, the mean GHG emission reductions from the SRF biomass scenarios outperform those from the Sitka spruce scenarios at both the per-MWh\(_e\) and per-hectare levels. The main factors contributing to the SRF outperformance are the differences in the BCS and BLUC values for each biomass stream, as detailed below.

5.5.8 Impact of biomass and peat land use change

Though there is only a 2 % and 5 % difference between the mean GHG emission reductions for the SRF and SS biomass scenarios at the ‘per MWh\(_e\)’ level for 30 % and 50 % co-firing (Figure 4.16), respectively, the overall GHG balances for each scenario conceal considerable underlying differences in the components of the disaggregated cradle-to-gate GHG balances. The main differences are derived from the LUC and biomass C sequestration (BCS) components. For example, whereas LUC associated with the drainage and industrial peat harvesting of formerly pristine peatlands is
generally accepted as a strong source of GHG emissions (Wilson et al. 2018, Ciais et al. 2013, Strack 2008), there is not the same consistency when assessing LUC related to forest plantations.

In a global study Paul et al. (2002) found that conventional forestry established on agricultural land initially lost soil C and remained in deficit for at least 10 years, followed by a slow recovery and maximised accumulation over longer (20–50 years) rotations. Black et al. (2009) found that Irish Sitka spruce plantations on former grassland surface-water gley soils sequestered 0.2 Mg C ha$^{-1}$ yr$^{-1}$, which this study proportionally allocated to the bioenergy feedstock over a 30 year rotation.

The negative (−) biomass land-use change (BLUC) values used in the Sitka spruce scenarios of this study reflect the benefit of the longer 30 year rotation length of those plantations, where the limited removal of forest residues and less frequent soil disturbance contributes to the accumulation of the soil C stock (Achat et al. 2015). Conversely, based on the limited paired-plot data from Keith et al. (2015), this study assumed an agriculture to forestry LUC soil C penalty for E. nitens SRF plantations of 1.89 t C ha$^{-1}$ yr$^{-1}$, also proportionally allocated to the bioenergy feedstock over three 10 year rotations. The more substantial biomass removals per hectare associated with the intensive wholetree harvesting of the SRF2–SRF4 scenarios, and greater soil disturbance of the 10 year SRF rotations, contributes to the decline in soil C stocks (Achat et al. 2015, Keith et al. 2015).

5.5.9 Impact of intensive harvesting and related fertilizer use

This LCA study used data from a UK study (Keith et al. 2015) of SRF related LUC, including four transitions to Eucalyptus plantations which 6–8 years after site establishment showed a loss of soil C of 1.41–2.36 Mg C ha$^{-1}$ yr$^{-1}$. Due to the young
age and limited number of the plantations studied Keith et al. (2015) concluded that it was not possible to say whether the soil C losses would persist as the stands got older. Studies by Walmsley and Godbold 2010 and Moffat et al. (2011) discuss the risks of, and strong evidence for, potential soil C loss and increased CO$_2$ emissions due to extensive soil disturbance events associated with stump harvesting. Data from Strömgren et al. (2013) showed a soil C loss of 6 Mg C ha$^{-1}$ 25 years after stump and residue harvesting, which supports the application of a soil C penalty when addressing these biomass streams in future LCA studies.

Apart from the potential impact on soil C stocks, the intensive harvesting and nutrient export associated with the SS2, SS4, SRF3 and SRF4 scenarios generally leads to a decrease in forest soil nutrient availability (Paré and Thiffault 2016, Achat et al. 2015, Vadeboncouer et al. 2014). To counteract those nutrient losses SRF plantations designed and managed solely to maximise biomass production for bioenergy need to consider the impact of nutrient replacement via fertilizer application. The application of the maximum allowable N (granulated Urea, 46 %), P (granulated Rock Phosphate, 12 %), and K (Muriate of Potash, 50 %) fertilizer (i.e. 350, 350, and 250 kg ha$^{-1}$, respectively) once per rotation in the SS4 and SRF4 scenarios adds approximately 3 and 7 kg CO$_2$-eq MWh$_{e}^{-1}$, respectively, to the GHG balance values of the non-fertilised scenarios of both biomass supply streams. Those values are within the emissions range of 7 kg CO$_2$-eq MWh$_{e}^{-1}$ calculated by Wihersaari (2005) for the production of spruce woodchip for a CHP plant in Finland.
5.5.10 The influence of tree species carbon and basic density on biomass carbon sequestration

The greater biomass C sequestration (BCS) values per MWh_e (Figure 4.10) for the SRF scenarios in comparison to the Sitka spruce scenarios are also reflective of the physical and chemical differences of their respective bioenergy feedstocks. The three percent difference in the mean GHG balance MWh_{e}^{-1} for the Sitka spruce and SRF 30 % co-firing scenarios (i.e. 933 and 905 kg CO2-eq MWh_{e}^{-1}, respectively) is primarily due to the higher basic density and C density of the *E. nitens* compared to the Sitka spruce biomass, i.e., less SRF biomass is combusted to meet the calorific requirement for generating 1 MWh_e at the EPL plant. Those species attributes, allied to fast growth and high-nutrient use efficiency, are important factors to consider when selecting SRF trees species for bioenergy plantations (Guo *et al.* 2006).

5.5.11 Potential short rotation forestry contribution to EU and Irish greenhouse gas targets

Due to economic growth and expansion of the agricultural sector, Ireland is currently projected to achieve only a 1 % reduction of its 2005 GHG emissions level, compared to the EU target of 20 % by 2020 (EPA 2018). Greenhouse gas emissions from Irish power generation are expected to grow strongly up to 2025 due to the expansion of co-firing of peat and biomass (EPA 2018). The projected 2020 and 2025 GHG emissions from energy industries, factoring in additional measures as outlined in Government renewable and energy efficiency policies, are expected to be 11.08 and 12.17 Mt CO_{2}-eq (EPA 2018), respectively.

In 2015 the contribution from renewable energy sources to Ireland’s total primary energy requirements (TPER) had reached 8.8 % of the 16 % EU RED target.
Also in 2015 the EPL plant reached 30 % co-firing and peat accounted for 5.5 % of the TPER, while biomass contributed 2 % (SEAI 2016b). The EPL plant supplies almost one third, 120/370 MWh_e (Murphy et al. 2015), or 1.83 % of the TPER from peat. That implies that if EPL reaches 50 % co-firing by 2020, then the TPER from peat will reduce by 0.37 % to 5.13%, and subsequently to 3.67 % of the TPER if it reaches exclusive (100 %) biomass combustion by 2030. The corollary of that calculation is that the biomass contribution to the Irish TPER will increase by 0.37 % to 2.37 % and by 1.83 % to 3.83 % by 2020 and 2030, respectively, and bring Ireland closer to reaching the EU mandated renewable energy targets.

5.6 Conclusions

This study adds to the research on the environmental impact of using biomass for bioenergy by applying cradle-to-grave LCA methodology to determine the GHG balance of SRF in Ireland. Using several scenarios, the SRF results were compared to LCAs of Sitka spruce and milled-peat reference systems. The LCA results focused on assessing each biomass scenarios climate change mitigation potential in light of national targets. Defining the appropriate system boundaries and functional units play a major part in how the LCA results can be interpreted. Widespread variations in biomass LCA system boundaries and functional units are two of the principal reasons for the challenges that arise when comparing biomass for bioenergy LCAs. When viewed from a “per MWh_e” perspective, the cradle-to-grave LCA of co-firing SRF and Sitka spruce biomass scenarios demonstrates their similar potential to mitigate GHG emissions from peat usage at the EPL plant. When assessed at the 100 % biomass level the considerable differences in their respective GHG balance components were revealed. The main differences between the SRF and Sitka spruce scenarios are related to the biomass C
sequestration (BCS) and LUC components of their respective GHG balances. These findings highlight the need to consider tree species that maximise biomass and C density, and implement management practices that minimise soil C loss in SRF silviculture. The disparity in the potentially available biomass yields per hectare, and the associated SRF and Sitka spruce GHG mitigation potentials, underlines the need for greater mobilisation of under-utilised land to dedicated SRF biomass for bioenergy. The need for that mobilisation is urgent if Ireland is to meaningfully address its targets for renewable energy under the EU RED for 2020 and beyond. This study has also shown that increasing co-firing, and eventually reaching full displacement of peat in Irish electricity generation with sustainable SRF or conventional forestry biomass, can play a significant part in mitigating Irish GHG emissions, in line with EU RED commitments.
6. Chapter 6 - Synthesis and recommendations
6.1 General discussion and synthesis

It is Irish Government policy to increase national forest estate from the current 11 % of the land area to 18 %, i.e., from approximately 750 thousand ha to 1.2 million ha (DAFM 2014b). Their intention is to achieve this target by the middle of this century, and in so doing sustainably meet the increasing demand for wood products and wood fibre for energy. Another aspect of the targeted level of afforestation is the need to enhance C sequestration and comply with the EU RED targets related to renewable energy and GHG emission reductions by 2020 and beyond (DAFM 2014b). The EU RED targets are enshrined in legally-binding commitments, and with new UNFCCC targets on C and GHG neutrality extending to 2050, forestry expansion and its associated challenges have now made LULUCF a priority issue in Ireland (Farrelly and Gallagher 2015a).

In a survey of 79 unenclosed Irish sites (mean area 124 ha) carried out to identify land with potential for afforestation grant aid, Farrelly and Gallagher (2015b) found that 52 % were classified as blanket peat or lithosols, and deemed to have extremely limited potential for agriculture or forestry. The study also found that 34 % of the sites had soils with moderate potential for forestry (i.e. peaty gleys, peaty podzols, and basin peats), while a further 14 % were classified as having soils very suitable to a range of forest species (i.e. brown earths, brown podzolics, gleys, and podzols). Their findings support the conclusion in Chapter 3 regarding further LUC associated with the expansion of forestry into podzols and peaty podzols, which could have potentially adverse impacts on the stability of their SOCD.

In the event the Government overcomes the economic and social obstacles, and re-energisises currently unsuccessful grant-aid incentives to bring productive unenclosed land into the forest estate (Farrelly and Gallagher 2015a), it will be necessary to
improve the accuracy of the SOCD estimates for these soils (Byrne and Black 2003).

This study found mean SOCD estimates for afforested podzols and peaty podzols of 132 Mg C ha\(^{-1}\) and 304 Mg C ha\(^{-1}\), respectively. That SOCD disparity and the adverse effect of a low signal-to-noise ratio in estimating soil C changes in highly organic soils using soil sampling (Baker and Griffis 2005), demonstrates the need to disaggregate their treatment in soil surveys. The high SOCD estimates of the peaty podzols in this study, which are within the lower range of Irish peatlands (240–3070 Mg C ha\(^{-1}\), Tomlinson 2005), suggests the need for specific EFs for SOCD values above a threshold of 200 Mg C ha\(^{-1}\), as used by Smith et al. (2006) in isolating organic soils in their SOC modelling work.

The issue of designing and resourcing field sampling campaigns that address the issue of MDD (Garten and Wullschleger 1999, FAO 2018) of SOCD change per hectare was discussed in Chapters 2 and 4. It is recommended that future field campaign sampling designs for SOCD inventories and SOCD change detection be implemented with sufficient time and resources to ascertain a MDD of at least 5 Mg SOC ha\(^{-1}\), i.e., 10-15 % change in existing SOC. Other sampling design considerations that accommodate power analysis of geo-spatial statistics would also enhance the accuracy and usefulness of the data.

To add to the challenges of measuring and monitoring the impact of increasing afforestation related LUC on national SOCD values, there is also a need to ensure that the appropriate BD sampling and calculation methods are applied by soil type (Throop et al. 2012, Poeplau et al. 2017, Jurgensen et al. 2017). Geological reports (Farrell 2016) along with the published data from the Irish SIS field survey (Reidy et al. 2016) and from Chapter 4 of this study, attest to the prevalence of RFs in many Irish soils.

This study found that the use of the pit excavation method of sampling soil BD on sites
with RF contents of >3 % (by volume) led to lower BD and lower SOCD estimates compared to those derived from the core method. When using the recommended $\rho_{\text{hybrid}}$ BD calculation method (Throop et al. 2012) for the seven sites sampled using both cores and pits, showed the mean pit BD results were 9 % lower than the mean core results. Those mean BD values, when combined with their respective SOC % estimates translated into a mean pit SOCD value for the seven sites that was 10 % lower than the core derived SOCD. The results of this study emphasise the need for choosing the appropriate BD sampling and calculation methods for the soils being analysed.

Life cycle assessment is an important tool in determining the environmental impacts of producing biomass for bioenergy, and in aiding the development of policies to maximise the potential benefits of bioenergy (Bruton et al. 2017). This study showed the GHG emissions mitigation potential of using SRF biomass and conventional forest harvest residues to displace the use of milled-peat at the EPL power plant. When assessing the Irish options for meeting its overall electrical energy demands and maintaining a nominal level of energy security, while simultaneously partially fulfilling its renewable energy and climate change mitigation related commitments, there is currently only one scenario that addresses all four issues. That scenario is the displacement of milled-peat as the fuel at the EPL plant. Currently it is the only power generation plant equipped for co-firing with biomass, with the potential for exclusive biomass combustion.

In line with the Irish Government targets to increase the energy share from renewable sources and reduce GHG emissions, there are two other Irish peat burning stations (namely, West Offaly at Shannonbridge, and Lough Ree at Lanesborough) in the Irish midlands which, starting in 2019, are also due to convert to biomass co-firing. Those plants which have similar capacity to the EPL plant, are operated by the semi-
state owned “ESB Power Generation” and are also expected to transition to biomass-only power plants by 2030. This study shows that, in relation to the peat-only scenario (Peat-0) at the EPL plant, the Sitka spruce and SRF 50 % co-firing (SS4-50 and SRF4-50) and the 100 % biomass scenarios (SS4-100 and SRF4-100) could potentially deliver GHG emission (per MWh$_e$) reductions of 0.40, 0.49, and 0.98 Mt CO$_2$-eq, respectively, on the projected 2020 energy emissions. Those potential emission reductions, from the EPL plant alone, equate to 3.6, 4.4, and 8.9 %, respectively, of the 11.08 Mt CO$_2$-eq energy industries emissions projected by the EPA for 2020, with greater reductions possible if co-firing has also commenced at the ESB Power Generation plants.

The ESB owned coal-powered Moneypoint plant in County Clare is Ireland’s largest power station and is currently responsible for providing 20 % (approximately 7 million MWh$_e$ per year) of the country’s electricity demand (ESB 2018). The Irish Government, under the National Development Plan 2018–2017, has announced that the plant is due to end the burning of coal at Moneypoint by 2025 (DCCAE 2018). Moneypoint has the potential for conversion to co-firing or exclusive power generation using biomass, though the issues of sustainable supply and cost need to be solved (Energy Institute 2018). As yet there is no decision on its future fuel source, and potential solutions are much debated and urgently required (Deane et al. 2013, BW Energy 2014, Energy Institute 2018). Some fuel options such as gas, biomass, or nuclear, are deemed either currently too costly, or socially and environmentally untenable (Energy Institute 2018).

### 6.2 Recommendations for further research

The research conducted for this thesis has raised some issues and questions that I would like to recommend for further investigations and studies. In Chapter 1 the study initially
focused on developing baseline data for the podzol group of soils. In the analysis of the
data from that work the need to more clearly disaggregate the methods used for
measuring changes in the SOCD of podzols and peaty podzols was proposed due to the
large difference in the mean SOCD estimates. Given the mean SOCD values found in
the peaty podzols were within the SOCD range of published data on Irish peatland soils
(histolsols), it was recommended that specific EFs for these organo-mineral soils are
developed, to more accurately determine how these soils react to afforestation and
related LUC.

Due to the laborious and time-consuming nature of the work, the accurate
measurement soil BD data has often been overlooked in past soil surveys. There is an
ongoing need for faster, more efficient means of more accurately sampling soil BD,
particularly in stony soils. The usefulness of the data provided by the Irish SIS could be
enhanced by the addition of forest soil BD data, and to depths of greater than the 30 cm
that is commonly found in the literature. The data and experience gained from the soil
BD study in Chapter 2 leads to my recommendation that more field and laboratory
testing of the novel pit excavation and core-scaling methods that were used to
determining the BD of soils with high RF contents. Further trials and data based around
that work may deliver a more productive system of determining soil BD to support
development of SOCD estimates and scales up to regional and national level.

A further recommendation is to expand on the cradle-to-grave LCA data
provided in Chapter 5. To support policy and incentives for the development of Irish
SRF biomass for bioenergy, the development is recommended of scenarios for several
more fast growing trees for species suited to the Irish climate, and to incorporate further
important environmental impacts, e.g. eutrophication and acidification potentials. Life
cycle assessment is heavily dependent on data that is specific to each aspect of a product
or services life cycle, hence there is often a reliance on assumptions based on data from literature or proprietary databases. The primary way to overcome this drawback is to fund further collection and assessment of field data related to local operations, parameters, and conditions. As mentioned in the Chapter 5 Introduction and section 5.5.9, the impact of intensive biomass harvesting, involving wholletree, brash and stump removal on SOCD, particularly on the high C stocks of organo-mineral soils (e.g. peaty podzols and gleys) needs to be further developed in LCA studies. The work of Jarvis et al. (2009) and Vanguelova et al. (2017) are recommended as a starting point for that undertaking.
7. Chapter 7 - References


Byrne, K.A. (2010) 'The role of plantation forestry in Ireland in the mitigation of greenhouse gas emissions', *Irish Forestry*.


Cannell, M. (1999) 'Growing trees to sequester carbon in the UK: answers to some


Conant, R.T., Paustian, K. and Elliott, E.T. (2001) 'Grassland Management and


DAFM (2014b) Ireland, Department of Agriculture, Food and the Marine. ‘Forests, products and people. Ireland’s forest policy – a renewed vision’. Dublin.


and productivity of Sitka spruce (Picea sitchensis) in Ireland in relation to site, soil and climatic factors', *Irish Forestry*.


GreenDelta (2016) ‘The video explaining how to create flows, processes, product systems and projects has been improved and is now available online’, GreenDelta, Berlin, Germany. [online], available: http://www.openlca.org/the-


Unit Working Group III, Potsdam Institute for Climate Impact Research (PIK),
Cambridge University Press, USA.

Gas Inventories: Wetlands’. Eds. Hiraishi, T., Krug, T., Tanabe, K., Srivastava,
N., Baasansuren, J., Fukuda, M. and Troxler, T.G., IPCC, Switzerland.

Groups I, II and III to the Fifth Assessment Report of the Intergovernmental
Panel on Climate Change’, [Core Writing Team, R.K. Pachauri and L.A. Meyer
(eds.)]. IPCC, Geneva, Switzerland, 151 pp.

IPCC (2018) ‘Global Warming of 1.5 °C. IPCC special report’. [online], available:
http://www.ipcc.ch/report/sr15/

Working-Group WRB, FAO, Rome.

IUSS (2014) ‘World Reference Base for Soil Resources – WRB Working Group,
International Soil Classification System for Naming Soils and Creating Legends
for Soil Maps’. World Soil Resources Reports No. 106. FAO, Rome.

Development Strategy, Scion.

Geological Institute.

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson,
D.W., Minkkinen, K. and Byrne, K.A. (2007) ‘How strongly can forest
management influence soil carbon sequestration?’, Geoderma, 137(3–4), 253-

Jandl, R., Rodeghiero, M., Martinez, C., Cotrufo, M.F., Bampa, F., van Wesemael, B.,
organic carbon monitoring’, Science of the Total Environment, 468-469(C), 376-
383, available: http://dx.doi.org/10.1016/j.scitotenv.2013.08.026.

Agriculture, Ecosystems & Environment, 104(3), 399-417, available:
http://dx.doi.org/10.1016/j.agee.2004.01.040.

and exchange of energy and greenhouse gases’, In: Combating Climate Change
– A Role for UK Forests. An Assessment of the Potential of the UK’s Trees and
Woodlands to Mitigate and Adapt to Climate Change, (eds Read DJ, Freer-Smith
PH, Morison JIL, Hanley N, West CC, Snowdon P), pp. 21–47. TSO,
Edinburgh.


209


Lal, R. (2004b) 'Soil Carbon Sequestration Impacts on Global Climate Change and Food Security', 304(5677), 1623-1627.


Lundmark, T., Bergh, J., Hofer, P., Lundstrom, A., Nordin, A., Poudel, B.C., Sathre, R.,


Mehler, K., Schöning, I. and Berli, M. (2014) 'The Importance of Rock Fragments...


Murphy, F., Devlin, G. and McDonnell, K. (2014b) 'Forest biomass supply chains in


Purser and Lynch (2012), ‘GROWFOR, Dynamic yield models used in Irish Forestry”. *COFORD Connects* #20. COFORD.


Schweier, J., Schnitzler, J.P. and Becker, G. (2016) 'Selected environmental impacts of the technical production of wood chips from poplar short rotation coppice on


SEAI (2016a) 'Ireland’s Energy Targets Progress, Ambition & Impacts', Sustainable Energy Authority of Ireland, Ireland.


Smith, P. (2004a) 'How long before a change in soil organic carbon can be detected?', *Global Change Biology*, 10(11), 1878-1883.


8. Chapter 8 - Appendices
### Appendix 1.1 – WRB Keys to Reference Soil Groups

<table>
<thead>
<tr>
<th>1. Soils with thick organic layers:</th>
<th>Histosols</th>
</tr>
</thead>
<tbody>
<tr>
<td>2. Soils with strong human influence</td>
<td>Anthrosols</td>
</tr>
<tr>
<td>Soils with long and intensive agricultural use:</td>
<td></td>
</tr>
<tr>
<td>Soils containing many artefacts:</td>
<td>Technosols</td>
</tr>
<tr>
<td>3. Soils with limited rooting due to shallow permafrost or stoniness</td>
<td></td>
</tr>
<tr>
<td>Ice-affected soils:</td>
<td>Cryosols</td>
</tr>
<tr>
<td>Shallow or extremely gravelly soils:</td>
<td>Leptosols</td>
</tr>
<tr>
<td>4. Soils influenced by water</td>
<td></td>
</tr>
<tr>
<td>Alternating wet-dry conditions, rich in swelling clays:</td>
<td>Vertisols</td>
</tr>
<tr>
<td>Floodplains, tidal marshes:</td>
<td>Fluvisols</td>
</tr>
<tr>
<td>Alkaline soils:</td>
<td>Solonetzes</td>
</tr>
<tr>
<td>Salt enrichment upon evaporation:</td>
<td>Solonchaks</td>
</tr>
<tr>
<td>Groundwater affected soils:</td>
<td>Gleysols</td>
</tr>
<tr>
<td>5. Soils set by Fe/Al chemistry</td>
<td></td>
</tr>
<tr>
<td>Allophane or Al-humus complexes:</td>
<td>Andosols</td>
</tr>
<tr>
<td>Chelivation and chelivation:</td>
<td>Podzols</td>
</tr>
<tr>
<td>Accumulation of Fe under hydromorphic conditions:</td>
<td>Plinthosols</td>
</tr>
<tr>
<td>Low-activity clay, P fixation, strongly structured:</td>
<td>Nitosols</td>
</tr>
<tr>
<td>Dominance of kaolinite and sesquioxide:</td>
<td>Ferralsols</td>
</tr>
<tr>
<td>6. Soils with stagnating water</td>
<td></td>
</tr>
<tr>
<td>Abrupt textural discontinuity:</td>
<td>Planosols</td>
</tr>
<tr>
<td>Structural or moderate textural discontinuity:</td>
<td>Stagnosols</td>
</tr>
<tr>
<td>7. Accumulation of organic matter, high base status</td>
<td></td>
</tr>
<tr>
<td>Typically mollic:</td>
<td>Chernozems</td>
</tr>
<tr>
<td>Transition to drier climate:</td>
<td>Kastanozems</td>
</tr>
<tr>
<td>Transition to more humid climate:</td>
<td>Phaeozems</td>
</tr>
<tr>
<td>8. Accumulation of less soluble salts or non-saline substances</td>
<td></td>
</tr>
<tr>
<td>Gypsum:</td>
<td>Gypsisols</td>
</tr>
<tr>
<td>Silicon:</td>
<td>Durisols</td>
</tr>
<tr>
<td>Calcium carbonate:</td>
<td>Calcisols</td>
</tr>
<tr>
<td>9. Soils with a clay-enriched subsoil</td>
<td></td>
</tr>
<tr>
<td>Allochthonous:</td>
<td>Allochvisols</td>
</tr>
<tr>
<td>Low base status, high-activity clay:</td>
<td>Alisols</td>
</tr>
<tr>
<td>Low base status, low-activity clay:</td>
<td>Acrisols</td>
</tr>
<tr>
<td>High base status, high-activity clay:</td>
<td>Luvisols</td>
</tr>
<tr>
<td>High base status, low-activity clay:</td>
<td>Lixisols</td>
</tr>
<tr>
<td>10. Relatively young soils or soils with little or no profile development</td>
<td></td>
</tr>
<tr>
<td>With an acidic dark topsoil:</td>
<td>Umbrisols</td>
</tr>
<tr>
<td>Sandy soils:</td>
<td>Arenosols</td>
</tr>
<tr>
<td>Moderately developed soils:</td>
<td>Cambisols</td>
</tr>
<tr>
<td>Soils with no significant profile development:</td>
<td>Regosols</td>
</tr>
</tbody>
</table>

(Source: IUSS, 2007)
Appendix 1.2 – FAO subordinate characteristics within master horizons

<table>
<thead>
<tr>
<th>Suffix</th>
<th>Short description</th>
<th>Used for</th>
</tr>
</thead>
<tbody>
<tr>
<td>a</td>
<td>Highly decomposed organic material</td>
<td>H and O horizons</td>
</tr>
<tr>
<td>b</td>
<td>Buried genetic horizon</td>
<td>mineral horizons, not cryoturbated</td>
</tr>
<tr>
<td>c</td>
<td>Concretions or nodules</td>
<td>mineral horizons</td>
</tr>
<tr>
<td>c</td>
<td>Coprogenous earth</td>
<td>L horizon</td>
</tr>
<tr>
<td>d</td>
<td>Dense layer (physically root restrictive)</td>
<td>mineral horizons, not with m</td>
</tr>
<tr>
<td>d</td>
<td>Distomaceous earth</td>
<td>L horizon</td>
</tr>
<tr>
<td>e</td>
<td>Moderately decomposed organic material</td>
<td>H and O horizons</td>
</tr>
<tr>
<td>f</td>
<td>Frozen soil</td>
<td>I and R horizons</td>
</tr>
<tr>
<td>g</td>
<td>Stagnic conditions</td>
<td>no restriction</td>
</tr>
<tr>
<td>h</td>
<td>Accumulation of organic matter</td>
<td>mineral horizons</td>
</tr>
<tr>
<td>i</td>
<td>Slickensides</td>
<td>mineral horizons</td>
</tr>
<tr>
<td>i</td>
<td>Slightly decomposed organic material</td>
<td>H and O horizons</td>
</tr>
<tr>
<td>j</td>
<td>Jarosite accumulation</td>
<td>no restriction</td>
</tr>
<tr>
<td>k</td>
<td>Accumulation of pedogenetic carbonates</td>
<td>no restriction</td>
</tr>
<tr>
<td>l</td>
<td>Capillary fringe mottling (glying)</td>
<td>no restriction</td>
</tr>
<tr>
<td>m</td>
<td>Strong cementation or induration (pedogenetic, massive)</td>
<td>mineral horizons</td>
</tr>
<tr>
<td>m</td>
<td>Marl</td>
<td>L horizon</td>
</tr>
<tr>
<td>n</td>
<td>Pedogenetic accumulation of exchangeable sodium</td>
<td>no restriction</td>
</tr>
<tr>
<td>o</td>
<td>Residual accumulation of sesquioxides (pedogenetic)</td>
<td>no restriction</td>
</tr>
<tr>
<td>p</td>
<td>Ploughing or other human disturbance</td>
<td>no restriction, E, B or C as Ap</td>
</tr>
<tr>
<td>q</td>
<td>Accumulation of pedogenetic silica</td>
<td>no restriction</td>
</tr>
<tr>
<td>r</td>
<td>Strong reduction</td>
<td>no restriction</td>
</tr>
<tr>
<td>s</td>
<td>Illuvial accumulation of sesquioxides</td>
<td>B horizons</td>
</tr>
<tr>
<td>t</td>
<td>Illuvial accumulation of silicate clay</td>
<td>B and C horizons</td>
</tr>
<tr>
<td>u</td>
<td>Urban and other human-made materials</td>
<td>H, O, A, E, B and C horizons</td>
</tr>
<tr>
<td>v</td>
<td>Occurrence of plinthe</td>
<td>no restriction</td>
</tr>
<tr>
<td>w</td>
<td>Development of colour or structure</td>
<td>B horizons</td>
</tr>
<tr>
<td>x</td>
<td>Fragipan characteristics</td>
<td>no restriction</td>
</tr>
<tr>
<td>y</td>
<td>Pedogenetic accumulation of gypsum</td>
<td>no restriction</td>
</tr>
<tr>
<td>z</td>
<td>Pedogenetic accumulation of salts more soluble than gypsum</td>
<td>no restriction</td>
</tr>
<tr>
<td>@</td>
<td>Evidence of cryoturbation</td>
<td>no restriction</td>
</tr>
</tbody>
</table>

(Source: FAO, 2006)
Appendix 1.3 – FAO Soil Description - Stone

<table>
<thead>
<tr>
<th>Abundance of rock fragments and artefacts, by volume</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>N None</td>
<td>0</td>
</tr>
<tr>
<td>V Very few</td>
<td>0–2</td>
</tr>
<tr>
<td>F Few</td>
<td>2–5</td>
</tr>
<tr>
<td>C Common</td>
<td>5–15</td>
</tr>
<tr>
<td>M Many</td>
<td>15–40</td>
</tr>
<tr>
<td>A Abundant</td>
<td>40–80</td>
</tr>
<tr>
<td>D Dominant</td>
<td>&gt; 80</td>
</tr>
<tr>
<td>S Stone line any content, but concentrated at a distinct depth of a horizon</td>
<td></td>
</tr>
</tbody>
</table>

(Source: FAO, 2006)

Charts for estimating proportions of coarse fragments and mottles

(Source: FAO, 2006)
### Classification of rock fragments and artefacts

<table>
<thead>
<tr>
<th>Rock fragments</th>
<th>(mm)</th>
<th>Artifacts</th>
<th>(mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>Fine gravel</td>
<td>2.5</td>
<td>Very fine artefacts</td>
</tr>
<tr>
<td>M</td>
<td>Medium gravel</td>
<td>6–20</td>
<td>Fine artefacts</td>
</tr>
<tr>
<td>C</td>
<td>Coarse gravel</td>
<td>20–60</td>
<td>Medium artefacts</td>
</tr>
<tr>
<td>S</td>
<td>Stones</td>
<td>60–200</td>
<td>Coarse artefacts</td>
</tr>
<tr>
<td>B</td>
<td>Boulders</td>
<td>200–600</td>
<td></td>
</tr>
<tr>
<td>L</td>
<td>Large boulders</td>
<td>&gt; 600</td>
<td></td>
</tr>
</tbody>
</table>

**Combination of classes**
- FM: Fine and medium gravel/artefacts
- MC: Medium and coarse gravel/artefacts
- CS: Coarse gravel and stones
- SB: Stones and boulders
- BL: Boulders and large boulders

(Source: FAO, 2006)

### Appendix 1.4 – FAO Soil Description - Roots

#### Classification of the diameter of roots

<table>
<thead>
<tr>
<th>Diameter</th>
<th>mm</th>
</tr>
</thead>
<tbody>
<tr>
<td>VF</td>
<td>Very fine</td>
</tr>
<tr>
<td>F</td>
<td>Fine</td>
</tr>
<tr>
<td>M</td>
<td>Medium</td>
</tr>
<tr>
<td>C</td>
<td>Coarse</td>
</tr>
</tbody>
</table>

*Note: Additional codes are: FF, very fine and fine; FM, fine and medium; and MC, medium and coarse.*

(Source: FAO, 2006)

#### Classification of the abundance of roots

<table>
<thead>
<tr>
<th>Abundance</th>
<th>&lt; 2 mm</th>
<th>&gt; 2 mm</th>
</tr>
</thead>
<tbody>
<tr>
<td>N</td>
<td>None</td>
<td></td>
</tr>
<tr>
<td>V</td>
<td>Very few</td>
<td>1–20</td>
</tr>
<tr>
<td>F</td>
<td>Few</td>
<td>20–50</td>
</tr>
<tr>
<td>C</td>
<td>Common</td>
<td>50–200</td>
</tr>
<tr>
<td>M</td>
<td>Many</td>
<td>&gt; 200</td>
</tr>
</tbody>
</table>

(Source: FAO, 2006)
Appendix 1.5- Measuring rock fragment bulk density using the water displacement method
8.1 For Appendices 3.1 to 5.4 please see attached CD

8.1.1 A3.1 – Chapter 3 Results table (as published)

8.1.2 A4.1 – Chapter 4: Core and pit excavation data

8.1.3 A4.2 – Chapter 4: SOCD data

8.1.4 A5.1 – Chapter 5: LCA scenario data, per MWhe

8.1.5 A5.2 – Chapter 5: LCA scenario data, per hectare

8.1.6 A5.3 – Chapter 5: LCI data sources (numbers coded in parentheses)

8.1.7 A5.4 – Chapter 5: LCA literature (coded as per A5.3)