Carbon Sequestration and Irish Forest Ecosystems

Edited by Kevin G. Black and Edward P. Farrell

COFORD, National Council for Forest Research and Development
Arena House
Arena Road
Sandyford
Dublin 18
Ireland
Tel: + 353 1 2130725
Fax: + 353 1 2130611

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Foreword

Ireland's rapidly expanding economy has, over the period since 1990, pushed emissions of greenhouse gases to well over the level the country signed up to in the Kyoto Protocol. Government policies to counteract this trend are rightly focused on reducing consumption of fossil fuels, changes in agricultural practices and a range of other measures, including the use of wood energy.

Along with reductions in emissions arising the areas outlined forests have a well defined role in mitigating climate change, as a land use that sequesters and stores atmospheric carbon dioxide, the main greenhouse gas, as a provider of energy efficient wood materials, and as a source of clean, renewable wood fuel.

Carbon sequestration was the focus of the work funded by COFORD in the CARBiFOR project, carried at University College Dublin, and reported here. Using a range of approaches, the team has done valuable work in updating estimates of carbon uptake in the main forest type – Sitka spruce, across a range of age classes. Innovative and challenging research has also been undertaken on ecosystem level uptakes and fluxes of greenhouse gases - work that has been linked to international efforts in gaining a deeper understanding of how forest ecosystems interact with the global climate. New information and insights have also been gained on forest soils and their carbon dynamics.

Linking research to policy development is a key task of nationally funded R&D programmes. CARBiFOR has been of great value in this arena: in providing input to guide the development of afforestation policy and, of course, in the climate change area itself, including the emissions trading scheme. The work has been instrumental in providing estimates of annual sequestration rates over the period 2008-2012. Without CARBiFOR, estimates of changes in forest carbon stocks would be on a far weaker footing, which would have direct economic consequences for Ireland in terms of purchases of carbon credits.

Expertise development is another important task of national R&D funding. CARBiFOR has proven to an excellent investment from this perspective - not only have team members achieved post-graduate qualifications and published extensively, but they have also been part of the team that has developed the national forest carbon reporting system to an international standard. They have also been active in developing land use reporting processes under the Intergovernmental Panel on Climate Change.

Forests have an important role to play in future climate change polices. At the global level the main need is to reduce tropical deforestation, and the greenhouse gases that arise when these forests are converted to food production. How forests adopt to climate change is another important area for the future – as is the role that forests may have in helping adaptation strategies, such as flood control.

Continued national investment is required in researching climate change mitigation and adaptation by forest ecosystems, and in building reporting systems that will add to the work described in this important report.

Eugene Hendrick

David Nevins
Chairman

Eugene Hendrick
Director

Brollach

Mar gheall ar fhorbairt ghasta gheilleagar na hÉireann thar an tréimhse ó 1990, tá astuithe de gháis cheaptha teasa á mbrú go maith thar an leibhéal ar shínigh an tír dó i bPrótacal Kyoto. Mar is cóir, tá polasaithe an Rialtais leis an treocht seo a chealú dírithe ar thomhailt na mbreoslaí iontaise a laghdú, athruithe a dhéanamh ar chleachtais talmhaíochta agus raon bearta eile, lena n-áirítear fuinneamh adhmaid a úsáid.

Chomh maith le laghduithe ar astuithe ag teacht ó na rudaí thuasluaite, tá ról soiléir ag foraoisí maidir le hathrú aeráide a mhaolú, cionn is gur úsáid thalaimh iad a leithlisíonn agus a stórálann dé-ocsaíd charbóin atmasféarach, an príomhghás ceaptha teasa, a chuireann ábhair adhmaid atá tíosach ar fhuinneamh ar fáil, agus atá ina foinse breosla adhmaid, breosla atá glan inathnuaite.

Bhí leithlisiú carbóin mar fhócas ag an obair a mhaoinigh COFORD sa tionscadal CARBiFOR, a rinneadh ag Coláiste Ollscoile Bhaile Átha Cliath, agus a thuairiscítear anseo. Ag úsáid réimse leathan modhanna oibre, tá obair luachmhar déanta ag an fhoireann le meastacháin ar thógáil charbóin a nuashonrú sa phríomhchineál foraoise – is é sin an sprús Sitceach, thar réimse d’aicmí aoise. Tá taighde dúshlánach nuálaíoch déanta chomh maith ar thógáil agus floscanna de gháis theach ghloine ar leibhéal an éiceachórais - obair atá nasctha le hiarrachtaí idirnáisiúnta chun tuiscint níos doimhne a bhaint amach faoin dóigh a mbíonn éiceachórais fhoraoise ag idirghníomhú leis an aeráid dhomhanda. Fuarthas eolas agus léargais úra chomh maith faoi ithir fhoraoise agus an dinimic charbóin a bhaineann léi.

Ar cheann de na tascanna is tábhachtaí atá ag na cláir thaighde agus forbartha a mhaoinítear go náisiúnta, tá an taighde a nascadh le forbairt pholasaí. Bhí CARBiFOR an-luachmhar maidir leis seo: ag soláthar ionchuir leis an pholasaí coilltithe a stiúradh, agus ar ndóigh, i limistéar athraithe aeráide é féin, lena n-áirítear scéim trádála na n-astuithe. Bhí an obair seo ina shiocair le meastacháin a chur ar fáil de rátaí bliantúla leithlisithe thar an tréimhse 2008-2012. Murach CARBiFOR, bheadh bunús i bhfad níos laige faoi mheastacháin d’athruithe ar stoic charbóin fhoraoise, rud a mbeadh torthaí díreacha eacnamaíocha acu d’Éirinn maidir le sochair charbóin a cheannach.

Feidhm thábhachtach eile atá ag an mhaoiniú náisiúnta ar thaighde agus fhorbairt ná scileanna a fhorbairt. Chruthaigh CARBiFOR gur infheistíocht den scoth a bhí ann ar an dóigh seo – ní hamháin gur ghnóthaigh baill den fhoireann cáilíochtaí iarchéime agus go bhfuil cuid mhór foilsithe acu, bhí siad ina gcuid den fhoireann a d’fhorbair córas tuairiscithe carbóin na bhforaoisí náisiúnta go caighdeán idirnáisiúnta. Bhí siad gníomhach fosta i bpróisis thuairiscithe úsáid thalún a fhorbairt faoin Phainéal Idir-rialtais ar Athrú Aeráide.

Tá ról tábhachtach le himirt ag foraoisí i bpolasaithe amach anseo maidir le hathrú aeráide. Ar an leibhéal domhanda, is é an gá is mó ná dífhoraoisiú trópaiceach a laghdú, chomh maith leis na gáis ceaptha teasa a dhéantar nuair a thiontaítear na foraoisí seo chun bia a tháirgeadh. Rudaí tábhachtacha eile don am le teacht ná an dóigh a gcuireann foraoisí iad féin in oiriúint don athrú aeráide, agus an ról a thiocfaí a bheith ag foraoisí i straitéisí oiriúnaithe, amhail tuilte a smachtú.

Tá gá le hinfheistíocht náisiúnta leanúnach a dhéanamh i dtaighde ar mhaolú athraithe aeráide agus oiriúnú ag éiceachórais fhoraoise, agus i gcórais thuairiscithe a thógáil a chuirfidh leis an obair a ndéantar cur síos air sa tuairisc thábhachtach seo.

David Nevins
Cathaoirleach

Eugene Hendrick
Stiúrthóir

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Executive Summary

Under the agreed terms of the Kyoto Protocol, carbon sequestration by forests may be used to offset greenhouse gas emissions. In the Irish context, carbon sequestration which is eligible for Kyoto compliance purposes refers to afforestation activities since 1990. The primary objective of the CARBiFOR project was to provide data to enable the level of such offsets to be determined. It provided data on carbon (C) stocks and sequestration rates, spanning the entire life cycle of the main forest type in Ireland, for national reporting.

Based on inventory, eddy covariance and other physiological methods, the average C sequestration rate over the entire life cycle of a selected chronosequence (10 to 50 year old stands) was estimated to be 4 to 8 t C ha^{-1} yr^{-1}. This value is higher than a previously reported average estimate of 3.36 t C ha^{-1} yr^{-1} (Kilbride et al. 1999). The higher average sequestration rate reported in this study was due to the use of improved biomass models, particularly for younger Sitka spruce stands. Afforested wet mineral soils also represent a significant C sink of 0.05 to 1 t C ha^{-1} yr^{-1}. However, changes in soil C stocks may be more difficult to measure in highly organic soils, such as peats, because of the high background soil C.

The information available from the CARBiFOR project has now been used to improve the CARBWARE model to report national forest C stock changes. The new estimated annual C sequestrations rates for Irish forests are equivalent to 1.56 to 2.39 Mt CO_2 yr^{-1} for the first commitment period 2008-2012 (assuming an annual afforestation rate of 14000 ha), which is higher than the previously reported CARBWARE values for the same period (1.54 Mt yr^{-1}, ICF/Byrne Ó Cléirigh report 2004). Assuming a business-as-usual scenario, it is estimated that the contribution of Article 3.3 forests for this period may account for ~ 20% of the required reduction in national emissions for Ireland to meet its Kyoto target. The CARBWARE model is constantly being improved as new research information and national inventory data become available.

Although these results are useful for providing a baseline for the carbon sequestration potential of the dominant forest species in Ireland, more information on C stocks and fluxes for different soil types and associated species is required. Furthermore, the potential for integrating the work reported here as well as future work with the ongoing National Forest Inventory, in order to improve the national C accounting system, should be explored and implemented.

More work is required to facilitate the development of comprehensive C balance models that can be used to simulate the effect of stand age, disturbance and climate change scenarios on forest C fluxes. The development of these models would be aimed at end users, such as forest managers and policy makers, to aid in management decisions, reporting procedures to the UNFCCC and climate change risk assessments. Beyond changes associated with stand age, there is a need to incorporate an analysis of the effects of disturbances, particularly those related to management practices (e.g. thinning) and land-use change, such as the transition from grassland to forest.

Achoimre Fheidhmeannach

Faoi théarmaí a aontaíodh de réir Phrótacal Kyoto, is féidir leithlisiú carbóin ag foraoisí a úsáid chun astuithe de ghás ceaptha teasa a chúiteamh. I gcomhthéacs na hÉireann, is é an leithlisiú carbóin atá incháilithe chun críche cloí le Kyoto ná gníomhaíochtaí coilltithe ó 1990. Ba í príomhchuspóir thionscadal CARBiFOR ná sonraí a chur ar fáil le leibhéal an chúitimh den sórt sin a chinntiú. Chuir sé sonraí ar fáil faoi stoic charbóin (C) agus rátaí leithlisithe, a mhaireann thar thimthriall iomlán saoil an phríomhchineáil fhoraoise in Éirinn, faoi choinne tuairiscithe náisiúnta.

Bunaithe ar fhardal, comhathraitheas guairneáin agus modhanna fiseolaíochta eile, measadh gurbh é gnáthráta leithlisithe carbóin thar thimthriall iomlán de chróinisheicheamh áirithe (clampaí 10 go 50 bliain) ná 4 go dtí 8 t C ha^{-1} yr^{-1}. Tá an luach seo níos airde ná an meánmheastachán a tuairiscíodh roimhe seo de 3.36 t C ha^{-1} yr^{-1} (Kilbride et al. 1999). Bhí an meánráta leithlisithe ní b'airde a tuairiscíodh sa staidéar seo le cur síos d'úsáid samhlacha bithmhaise níos fearr, go háirithe do chlampaí níos óige de sprús Sitceach. Is féidir le hithir fhliuch mhianra bheith ina ndoirteal substaintiúil carbóin de 0.05 go 1 t C ha^{-1} yr^{-1}. Bíodh sin mar atá, is féidir le hathruithe i stoic charbóin ithreach a bheith níos deacra a thomhas in ithir an-orgánach, mar mhóin, mar gheall ar an leibhéal ard carbóin sa chúlra san ithir féin.

Tá úsáid bainte as an tionscadal CARBiFOR anois leis an tsamhail CARBWARE a fheabhsú le hathruithe stoic C i bhforaoisí náisiúnta a thuairisciú. Is ionann na rátaí bliantúla measta leithlisithe C d'fhoraoisí na hÉireann ná 1.56 go 2.39 Mt CO_2 yr^{-1} don chéad tréimhse thiomantais 2008-2012 (ag glacadh leis go bhfuil ráta bliantúil coilltithe de 14000 heicteár ann). Tá sé seo níos airde ná na luacha CARBWARE a tuairiscíodh roimhe seo don tréimhse chéanna (1.54 Mt yr^{-1}, ICF/Byrne Ó Cléirigh - tuairisc 2004). Ag glacadh leis nach dtarlóidh mórathrú ar bith, meastar gurb ionann foraoisí Alt 3.3 ná 20% den laghdú ar astuithe a bheadh de dhíth le sprioc na hÉireann faoi Kyoto a bhaint amach. Tá múnla CARBWARE á fheabhsú go leanúnach de réir mar a chuirtear níos mó faisnéise agus sonraí fardail náisiúnta ar fáil.

Cé go bhfuil na torthaí seo úsáideach sa dóigh is go gcuireann siad bunlíne ar fáil faoi acmhainn leithlisithe carbóin na príomhspéiceas foraoise in Éirinn, tá níos mó faisnéise de dhíth ar stoic agus ar fhloisc charbóin do chineálacha difriúla ithreach agus na speicis a bhaineann leo. Lena chois sin, ba chóir na féidearthachtaí atá ann a fhiosrú agus a chur i bhfeidhm leis an obair a thuairiscítear anseo agus obair san am le teacht a chomhlánú leis an Fhardal leanúnach Náisiúnta Foraoise, d'fhonn córas cuntasaíochta náisiúnta carbóin a fheabhsú.

Tá tuilleadh oibre de dhíth chun forbairt samhlacha cuimsitheacha de chothramaíocht charbóin a éascú, samhlacha arbh fhéidir a úsáid le héifeachtaí aois an chlampa, cur isteach agus cásanna athraithe aeráide ar fhloscanna carbóin fhoraoise a ionsamhlú. Bheadh forbairt na samhlacha seo bunaithe ar úsáideoirí deiridh, leithéidí bainisteoirí foraoise agus lucht déanta polasaithe, le cuidiú leo ina gcinntí bainistíochta, modhanna oibre tuairiscithe leis an UNFCCC agus le measúnuithe riosca athraithe aeráide. Taobh amuigh d'athruithe a bhaineann le haois an chlampa, tá gá le hanailís ar éifeachtaí cur isteach a ionchorprú, go háirithe iad siúd a bhaineann le nósanna bainistíochta agus athrú ar úsáid talaimh (m.sh. caolú), amhail an t-athrú ó thalamh féaraigh go foraois.

List of abbreviations and terms

a	annum
ABEF	above-ground biomass expansion factor
BEF	biomass expansion factor
cm	centimetre (10^{-2} m)
C	Carbon
CAI	current annual increment
CF	carbon fraction
COP	Conference of the Parties
D	basic wood density
D_a	above-ground annual litter fall
D_b	below-ground detritus
d	day
dbh	diameter at breast height (1.3 m)
F-disturb	disturbance related-changes
GHG	green house gas
GPP	gross primary productivity (or forest photosynthesis)
h	height
ha	hectare
IPCC	International Panel and Climate Change
LULUCF	Land-Use, Land-Use Change and Forestry
MDV	mean diurnal variation
m	metre
mg	milligram (10^{-3} g)
NBP	net biome productivity
NEP	net ecosystem productivity
NPP	net primary productivity
ppmv	parts per million by volume
Pg	Petagram (10^{15} g)
r^2	coefficient of determination
R	respiration
R_H	heterotrophic respiration
R_A	autotrophic respitation
R_{TOT}	total soil respiration
R:S	root to shoot ratio
SEE	standard error of estimate
SOM	soil organic matter
TBCA	total below-ground carbon allocation
Tg	Teragram (10^{12} g equivalent to a ton (t))
UNFCCC	United Nations Framework Convention on Climate Change
w/v	weigh to volume
w/w	weight to weight
y	year
YC	yield class (index representing potential maximum mean annual increment)
ΔAGB	annual change in above-ground dead plant material
ΔB	annual change in living biomass
ΔC	change in carbon stock
Δ_e	ecosystem isotope discrimination
‰	parts per thousand
$\delta^{13}C$	^{13}C isotope discrimination relative to ^{14}C

Chapter 1

INTRODUCTION

Kenneth A. Byrne and Kevin G. Black

CLIMATE CHANGE AND POLICY

Weather and climate are intimately linked to life on Earth. They are essential for food production and health as well as being closely linked to ecosystem functioning and sustainability, infrastructure, and property. Many consider the prospect of human-induced climate change to be one of the foremost environmental issues facing the Earth. There is growing scientific evidence that human activities are altering the climate (IPCC 2001) and this has been attributed to increasing atmospheric levels of the so-called greenhouse gases (GHG) because of their ability to absorb and re-emit infrared radiation and so raise the temperature near the Earth's surface. The principal greenhouse gases are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), and since 1750 their atmospheric concentrations have increased by 31, 151 and 17% respectively. About 75% of human-induced CO_2 emissions are derived from fossil fuel burning with the remainder due to land-use change, particularly deforestation. CH_4 is mainly produced in anaerobic environments such as wetland soils, rice paddies, landfills and the ruminant gut. The principal emitters of N_2O are agricultural soils, cattle feed lots and the chemical industry. A proper understanding of the cycling of greenhouse gases at global, regional and national level is a vital prerequisite to attempts to mitigate, respond to and predict the effects of climate change.

The United Nations Framework Convention on Climate Change (UNFCCC), agreed at the Earth Summit in Rio de Janeiro in 1992, was the first major attempt to deal with climate change. While it did not set targets for the reduction of greenhouse gas emissions, Parties to it (including Ireland) are required to develop, publish, update and make available national inventories of GHG emission by sources and removals by sinks. Forests are one sector for which an inventory is required. This was followed in 1997 by the Kyoto Protocol. Its main features are as follows:

- 1990 is the base year against which all emission reductions are calculated.
- Developed countries (so-called Annex I) committed to reduce annual GHG emissions to 5.2% below 1990 levels by the first commitment period of 2008-2012.
- The European Union committed itself to a reduction of 8%. This burden is shared between member states and under this agreement Ireland is committed to limiting its GHG emissions to 13% above 1990 levels by 2008-2012.
- The Protocol could only enter into force when ratified by at least 55% of Annex I countries, which cumulatively represent at least 55% of global GHG emissions.
- The Protocol made provision for the use of carbon (C) sequestration by land-use, land-use change and forestry (LULUCF) as a means to offset GHG emissions.

The principal articles of the Kyoto Protocol that refer to forestry are 3.3 and 3.4. Article 3.3 refers to net changes in greenhouse gas emissions by sources and removals by sinks resulting from direct human-induced afforestation, reforestation and deforestation which have taken place since 1990; Article 3.4 refers to additional human-induced activities in the agriculture, land-use change and forestry sectors. The rules for the implementation of the Kyoto Protocol were agreed at the Seventh Conference of the Parties (COP7) at Marrakesh in November 2001. Under this agreement, there is no limit to the amount of credits a Party may accrue from Article 3.3 while limits have been placed on the amount of credits which can be obtained from forest management under Article 3.4; for Ireland this limit is set at 50,000 t C yr^{-1} during the first commitment period, i.e. 2008-2012. Furthermore all developed countries are required to account for C stock changes in:

- above-ground biomass,
- below-ground biomass,
- litter,
- deadwood and
- soil organic carbon.

THE DYNAMICS OF FOREST CARBON STORES

The balance between gross primary productivity (GPP or forest photosynthesis) and respiration from living and/or dead material determines the net sequestration rate by a forest ecosystem (Figure 1.1). Generally, about half of GPP is respired by plants to provide energy for growth and maintenance (Waring and Running 1998). Net primary productivity (NPP) is the net C gain by trees (equivalent to biomass increment) and equals the difference between GPP and autotrophic (plant) respiration (Figure 1.1).

Plants lose C through several pathways including litterfall and root exudation. The soil organic matter (SOM) C pool is the largest and most significant storage pool in the forest ecosystem. An estimated 247-286 Pg C are stored in boreal forest soils Schlesinger 2000). In addition, the soils are considered to be a significant sink for atmospheric C, accounting for as much as 662 Tg C yr^{-1}. The ability of forest soils to sequester C is due to the deposition and accumulation of a resistant slowly decomposable C pool with a half-life 1000 + years. The size of the SOM pool represents a balance between the organic matter input as litter of various types and the rate of decomposition of dead material (heterotrophic respiration). Decomposition is determined largely by climate, the chemical quality of the litter input and the decomposer organisms present in the system.

The changes in the sequestration potential of a forest at different ages can be determined by natural or disturbance related-perturbations in GPP, NPP and respiration. Disturbance related-changes (F-disturb) include various management practices such as planting, thinning and harvest.

Forest inventory data, together with additional knowledge on biomass conversion factors, can be

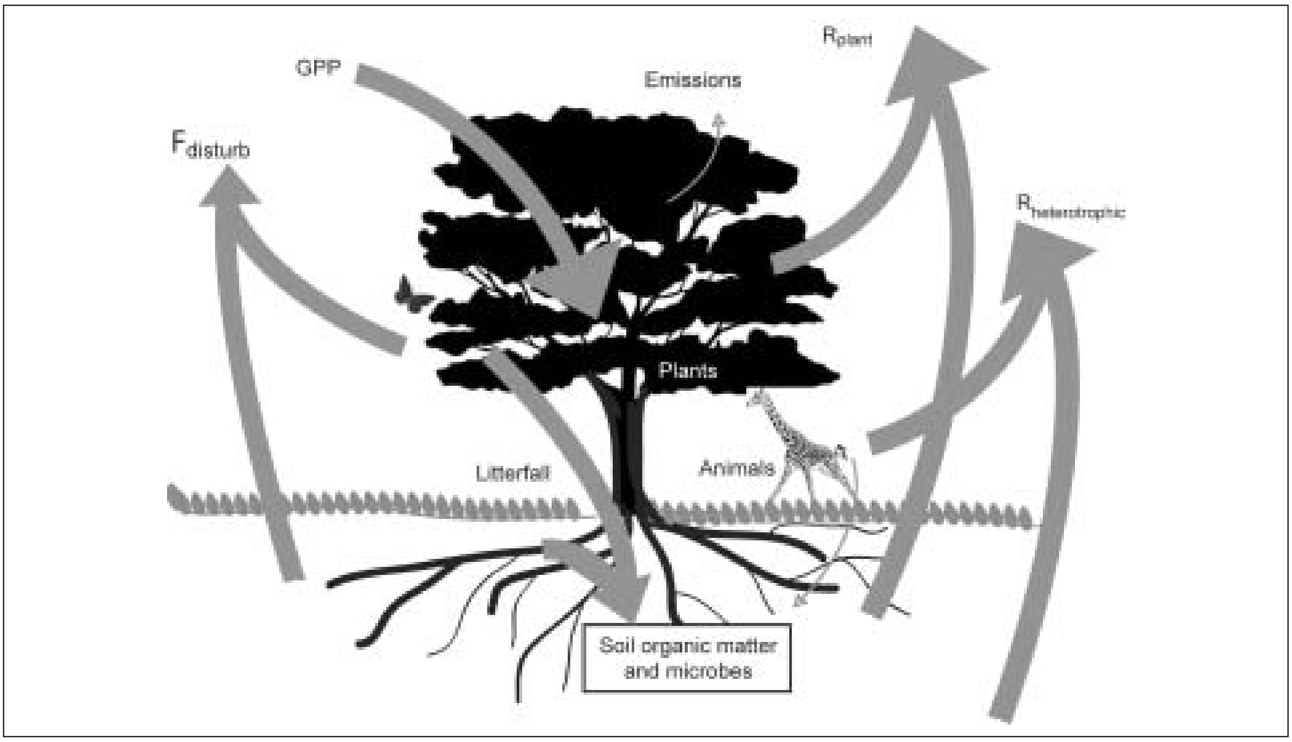

Figure 1.1: An overview of the major C fluxes of a forest ecosystem (see text for a description of these processes (taken from Chapin et al. 2002).

used to estimate NPP and this forms the framework for estimating C sequestration rates for national reporting. However, additional information on respiratory, soil stock changes and disturbance related changes are required to compile an ecosystem C sequestration balance. The innovative approach adopted by CARBiFOR was to assess C sequestration rates using both inventory and ecosystem-based methodologies.

DEVELOPMENT OF A CARBON ACCOUNTING SYSTEM

Estimation of the extent to which forests sequester C in the mid to long term is hindered by a high degree of spatial heterogeneity and temporal variability. Therefore, carbon accounting procedures rely heavily on the establishment of national forest inventory programmes, which can capture the change in C stocks over time. Guidance on the development of inventories is provided by the Revised 1996 Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories (Houghton et al. 1997) and the Intergovernmental Panel on Climate Change Good Practice Guidance for Land-use, Land-Use Change and Forestry (Penman et al. 2003). These guidelines have raised a demand for country-specific biomass and growth data, and measurements of inputs and losses from forest soils, which can be used to calculate the potential of forests to mitigate the anthropogenic increase of atmospheric CO_2 concentrations. It is also important that these inventory methods, may be suitable for C accounting reports, but these methods should be cross-validated using a 'full C accounting' procedure to estimate C flux losses associated with disturbance and land-use change, which can not be determined using conventional biomass or soil C stock estimates. From a 'full carbon accounting' perspective, ecosystem flux measurements by eddy covariance (Aubinet et al. 2002) can be used to verify, at a stand or local scale, reported changes in carbon stocks by providing data that can be compared with the activities, emission factors or default values used to compile carbon budgets for a particular LULUCF category.

AIMS AND OBJECTIVES

The primary objective of the CARBiFOR project was to provide data to determine carbon stocks and sequestration rates spanning the entire life cycle of the major forest type (i.e. Sitka spruce) in Ireland. The project cluster was designed to provide improved estimates, based on micrometeorological and inventory methods, of above- and below-ground, litter, deadwood and soil carbon stocks, and sequestration rates for first rotation Sitka spruce stand on wet mineral soils. Each working group had specific objectives and these are described in the succeeding chapters.

The specific aims of the CARBiFOR project were:

- To account for post 1990 afforestation activities in terms of the location and quantity of carbon sequestered or emitted for the major Irish forest type (Article 3.3)
- Provide the basis for estimating the contribution of the Land-use Change and Forestry Sectors to Ireland's net greenhouse gas emissions in the first and subsequent commitment periods of the Protocol
- Improve the basis of 'annual' inventory reporting for Land-use Change and Forestry Sectors by enabling verification of estimates and establishment of criteria, standards and acceptable confidence bounds
- Collect better information about the carbon stocks in live biomass and litter associated with the major forest type in Ireland and its CO_2 sequestration or emission time course.

THE CHRONOSEQUENCE

The core sites representing the Sitka spruce chronosequence (reconstructed historical age distribution) were located in different Coillte forests in the Portlaoise area (Table 1.1). Selected stands (9 to 45 year old trees) are representative of the typical yield class (18-24) for Sitka spruce growing on wet mineral soils in Ireland (Table 1.2). An additional 14 year old-stand, with a yield class of 24 was used for eddy covariance measurements and validation of inventory-based estimates. A grassland site was also selected to represent a non-

Table 1.1: Location and management history of core CARBiFOR chronosequence sites.

Site Name/ Code	**Planting date**	**Compartment ID**	**Geo-reference Position**	**Previous Land Management**	**Cultivation and Management**	**Thinning History (dates)**
Baunoge (B9)	1993	72018 K-2	52°55’ N, 7°14’ W	Grass/rush pasture	Ripped (1 m depth with ball and chain), surface drains across rip lines, no fertiliser.	N.a.
Clontycoe (C14)	1988	78010 E-1	52°56’ N, 7°15’ W	Poor fertilised marginal grassland	Ripped (1 m depth with ball and chain), surface drains across rip lines, no fertiliser.	N.a.
Dooary (D14)	1988	78012 R-1	52°57’ N, 7°15’ W	Grassland, regularly fertilised	Ripped (1 m depth with ball and chain), surface drains across rip lines, no fertiliser.	first due in ’06.
Glenbarrow (G25)	1978	77166 O-1	53° 8’ N, 7°27’ W	Rough grassland	Mouldboard ploughed. Poor establishment (80%).	first ‘98
Dooary (D30)	1972	77971 A-8	52°57’ N, 7°16’ W	Grass pasture	Mole drained, shallow ploughed, no fertiliser.	’91, ’95, ’99, ’03.(20%)
Cullenagh (C45)	1957	77982 M-1	52°57’ N, 7°15’ W	Rough grassland	Mouldboard ploughed, no fertiliser.	mid ‘70s, ’81, late ‘80s, mid ‘90s, ’01 and felled in ’04.

Table 1.2: Description and biometric characteristics of core Coillte stands in the CARBiFOR chronosequence (2002).

Forest Code	**Age**	**Stocking density**	**DBH**	**Height**	**Yield class**	**Basal area**	**Timber volume***	**Leaf area index**	**Crown to Height ratio**
	Years	stem ha^{-1}	cm	m		m^2 ha^{-1}	m^3 ha^{-1}	m^2 m^{-2}	
B9	9	2333	6	3.5	20-24	6.3	60.4	4.3	0.08
C14	14	2533	13	7.3	20	34.0	173.6	7.8	0.13
D14	14	2467	16	9.5	24	52.2	290.0	8.4	0.41
G25	25	1250	22	14.3	16	48.9	363.4	5.7	0.56
D30	30	1033	24	16.8	22-24	60.7	500.3	5.6	0.59
C45	45	767	32	26.7	22	65.3	835.3	4.8	0.62

forested site, to assess changes in C sequestration associated with land-use change.

Additional sites were selected for individual aspects of the project and these are highlighted in the working group chapters. A grassland site (G0) was also selected to represent an non-forested stand, to assess changes in C sequestration associated with land-use change.

Site D14 was selected as the main site, where all measurements were made and ecosystem models were parameterised with measured values. Measurements at this site included eddy covariance, inventory surveys, soil C, soil respiration, litterfall, biomass data and isotope tracer studies.

The soils in these sites are wet mineral gleys (Table 1.3), with a clay content of ca. 30-50%,

Table 1.3: Soil Description and characteristics of core sites (adapted from Green et al. submitted).

Forest Code	Age years	Sand %	Silt %	Clay %	pH	P[a] (mg/l Available)	K[b] (mg/l Available)	N (% w/w)	Bulk Density kg m^{-3}
G0	0	23	41	36	5.3	11.07	63.54	0.40	0.918
D10	10	37	32	30	4.8	8.46	92.30	0.38	0.950
D14	14	9	38	53	5.0	9.11	78.06	0.29	0.998
D30	30	14	39	47	4.3	8.20	90.60	0.45	1.002
C45	45	20	50	30	4.3	6.39	53.14	0.48	1.063

[a] Available Phosphorus-Sodium Bicarbonate Extractable – "Olsens" – reported as mg/l dry basis

[b] Available Potassium – Ammonium Nitrate Extractable – reported as mg/l day basis

except for the C45 site where there was a more sandy soil. An additional 47 year old stand, with a higher clay content, was selected at the Dooary forest (D47) for C analysis. However, the stand was felled before any biometric measurements were recorded.

THE CARBIFOR TEAM

The CARBiFOR project (Carbon Sequestration in Irish Forest Ecosystems, October 2001 – June 2005) was co-ordinated by Prof. Edward P. Farrell and organised into four working groups as follows:

Working Group 1: Ecosystem Processes

Prof. Bruce Osborne, Department of Botany, UCD

Dr Kevin Black, Department of Botany, UCD

Collaborators:

Dr Deborah Hemming, Weisman Institute, Israel; and CarboEurope, C isotope studies

Prof. M Jones, Trinity College, Grassland and agricultural GHG fluxes

Working Group 2: Soil Carbon Fluxes

Prof. Edward Farrell, Forest Ecosystem Research Group, Department of Environmental Resource Management, UCD

Dr Kenneth Byrne, Forest Ecosystem Research Group, Department of Environmental Resource Management, UCD (now at University College Cork)

Dr Gustavo Saiz, Forest Ecosystem Research Group, Department of Environmental Resource Management, UCD

Working Group 3: Soil Carbon Stocks

Prof. Tom Bolger, Department of Zoology, UCD

Brian Reidy, Department of Zoology, UCD

Dr Olaf Schmidt, Department of Environmental Resource Management, UCD

Dr Jens Dyckmans, Department of Environmental Resource Management, UCD

Working Group 4: Biomass Carbon Stocks and Expansion Factors

Prof. Maarten Nieuwenhuis, Department of Crop Science, Horticulture and Forestry, UCD

Dr Brian Tobin, Department of Crop Science, Horticulture and Forestry, UCD

Prof. John Gardiner, Department of Crop Science, Horticulture and Forestry, UCD

Chapter 2

SOIL CARBON FLUXES

Gustavo Saiz, Kenneth A. Byrne and Edward P. Farrell

OBJECTIVES

The aim of Working Group 2 was the study of soil respiration over the Sitka spruce (*Picea sitchensis* (Bong.) Carr.) chronosequence in Co Laois (see Tables 1.1 and 1.2). The research focused on the quantification and understanding of the factors that drive C emissions from soils, which are essential for the understanding of forest ecosystem C balance as a whole.

The specific objectives of the study were to:

- Investigate the influence of climatic variables on soil respiration;
- Investigate the effect of stand age on soil respiration;
- Estimate mean annual soil respiration rates for each stand age;
- Estimate fine root biomass;
- Estimate of heterotrophic and autotrophic soil respiration rates;
- Quantify the temporal variability of soil respiration;
- Quantify the spatial variability of soil respiration.

Supplementary objectives were:

- Provision of data to Working Group 1 for the construction of empirical and mechanistic models to be used to derive net ecosystem productivity, net primary productivity and gross primary productivity.
- Provision of soil respiration and fine root biomass data to Working Group 3 for calibration of the CENTURY model.

RATIONALE FOR THE STUDY

An ecosystem acts as a C source when gross primary productivity surpasses autotrophic and heterotrophic respiration (see Figure 1.1; Klopatek 2002). Soil respiration is the primary path by which CO_2 fixed by land plants returns to the atmosphere (Raich and Schlesinger 1992, Rustad et al. 2000, Schlesinger and Andrews 2000). In forests, total ecosystem respiration (i.e. loss of CO_2) tends to be dominated by soil respiration (Valentini et al. 2000).

The production of CO_2 in the soil profile is primarily the result of autotrophic respiration by live roots (allowing their growth and maintenance), associated mycorrhizae, and heterotrophic respiration by microbes oxidizing plant detritus, root exudates and humified organic matter (Boone et al. 1998, Buchmann 2000, Hanson et al. 2000).

The factors that control heterotrophic and autotrophic respiration determine the contribution of each component to the total soil respiration. Temperature, together with soil moisture content, is the main factor controlling soil respiration (Raich and Schlesinger 1992, Kirschbaum 1995, Davidson et al. 1998, Fang and Moncrieff 2001). Moderate, low or high soil water contents may limit soil respiration. These environmental factors affect root respiration and soil organic matter decomposition differently. Research has shown that the contribution of autotrophic and heterotrophic respiration to total soil respiration varies seasonally (Hanson et al. 2000).

In addition to temperature and moisture, root respiration is strongly influenced by plant photosynthetic activity (Högberg et al. 2001, Kuzyakov and Cheng 2001). This light-dependant process may co-vary with temperature, which in some field studies may have been the factor responsible for the different temperature sensitivity values reported for autotrophic and heterotrophic respiration (Bååth and Wallander 2003). Roots not only make a direct contribution to heterotrophic respiration, but also affect the respiratory activity of heterotrophic organisms by exuding C-rich substances and by altering the soil physical and chemical environment (Kuzyakov and Cheng 2001).

An important area of research is aimed at focusing on the impact of forest age on soil CO_2 efflux (Irvine and Law 2002). Stand soil respiration is dependant upon both the amount of fine roots and quality of soil C pools. Soil respiration can be expected to differ according to the structure and age of the stand (Klopatek 2002). It is therefore important to understand the processes that determine total soil respiration in order to predict how each component may react to alterations of the soil environment (Kirschbaum 1995, Boone et al. 1998, Ohashi et al. 2000, Fierer et al. 2003).

High spatial and temporal variation of soil respiration has been reported for a number of ecosystems (Thierron and Laudelout 1996, Davidson et al. 1998, Epron et al. 1999b). An understanding of this variability is a vital component of efforts to understand and quantify the C balance in forest ecosystems.

METHODS

Site Characteristics

The selected study sites were first rotation Sitka spruce plantations established on former unmanaged grassland (afforestation sites). The stands were characterised by a nearly absolute absence of understory or herbaceous vegetation after reaching canopy closure. The stands made up a chronosequence ranging from 10 to 47 years (Table 2.1), with the oldest stand being mature for harvest. Three sites per stand age were used, giving an overall total of twelve sites. All the stands were within 5 km of each other.

For each stand age, one site was used to study total and heterotrophic soil respiration (hereafter referred to as core sites). For each stand age two additional sites were used to study total respiration (hereafter referred to as reference sites). The core sites were common to all CARBiFOR working groups and facilitated collaboration between working groups (Tables 1.1, 1.2 and 1.3). The two additional sites per stand age had the same characteristics for forest management, tree age, soil type, and tree establishment as their core site.

Soil types were classified as low humic (mineral) gleys in the 10, 15, and 31 year old stands, and as gleyic brown earth in the 47 year old stands, which seemed to have better drainage due to their location on sloping terrain (Image 1).

The younger sites (10 and 15 years old stands) had been established along ripped lines 1 m deep and 2 m apart. There were surface drains across the ripped lines at 50 m intervals. Prior to the establishment of the 31 and 47 year old stands the sites were ploughed at 1.7 m intervals, following the contour lines of the slope. In these two older plantations, 3-4 regular thinnings had already taken place (Table 1.1). Neither fertilisation nor drainage works had been carried out since tree establishment in any of the stands. All the plots selected for the present study had reached canopy closure. Since no thinning had been carried out in either the 10 or 15 year old plantations, their stand densities were higher when compared to the older stands (Table 2.1).

Table 2.1: Characteristics of the Sitka spruce stands over which measurement of total and heterotrophic soil respiration took place (core sites). pH, nitrogen and phosphorous (pooled samples) measured for A1 soil horizon (n=15).

Site	Age (years)	Stem Density (trees ha^{-1})	Basal Area (m^2 ha^{-1})	Yield Class	pH	Nitrogen (g kg^{-1})	Phosphorous (mg l^{-1} available)
Dooary 1	10	2300	5.3	16-20	4.52	0.627	11.2
Dooary 2	15	2366	34.4	20-24	4.77	0.418	11.4
Dooary 3	31	1083	54.0	16-20	4.07	0.599	9.4
Cullenagh	47	730	57.0	16-20	4.01	nd	8.8

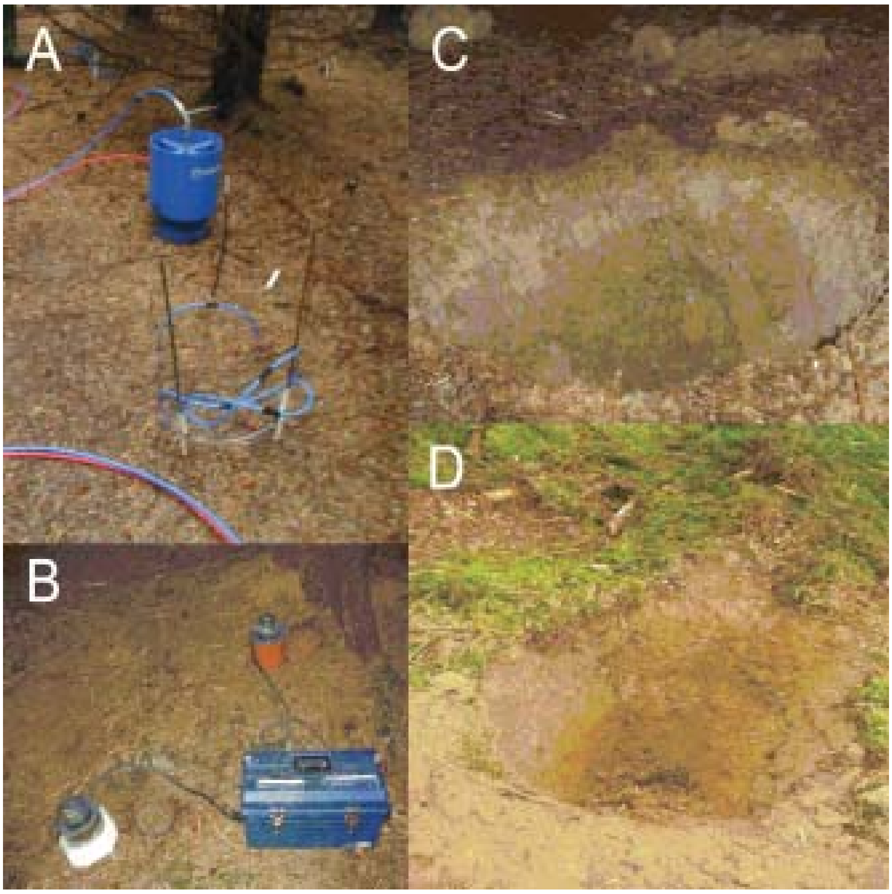

Image 1: The open top automated (A) and manual (B) respiration chambers used to measure soil CO_2 fluxes. Soils were typical wet mineral gleys (C) at all of the sites except for the 47 year old stand, where the soil was characterised as a gleyic brown earth (D).

Experimental design

Soil respiration measurements were carried out in three forest stands each for the age classes 10, 15, 31 and 47[1] years. For each site, a stratified random sampling design was used taking into account disturbance during site preparation and distance to the nearest tree. Trees were randomly chosen from within 30 x 30 m plots. The study plots were placed at least 20 m apart from stand discontinuities or its boundaries.

A series of 30 PVC circular collars (16 cm internal diameter) per stand were inserted into the soil to an average depth of 1.5 cm for measurements of total soil CO_2 fluxes. Collars set at this depth were stable and caused minimal disturbance to shallow fine roots. For the estimation of heterotrophic respiration, at each of the core sites, 30 stainless steel pipes (16 cm internal diameter) were driven into the soil to a depth of 30 cm to kill tree roots. After the freshly killed fine roots had decomposed, it was assumed that the soil respiration within the cores were derived from the decomposition of organic matter only (Ewel et al. 1986b, Bowden et al. 1993). In this study, a stabilisation period of eight months was allowed after the collars were installed, before the first measurements were taken. Subsequently, a correction was made to account for the overestimation of heterotrophic respiration produced by the decomposition of trenched roots. Root density within the pipes was assumed to be the same as the average calculated for each stand. To calculate the flux resultant from roots left within the pipes, a decay constant of 0.2 $year^{-1}$ was applied, based on Silver and Miya (2001).

Soil CO_2 flux measurements

Soil respiration measurements were carried out during 2003 at all the core sites on a monthly basis. Reference sites were sampled bi-monthly during the same year. At the core sites, soil respiration was

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[1] *The 47 year old site is the same as the 45 year old stand in the previous chapter. Soil respiration measurements were taken two yeas after the sites were initially characterised.*

measured at each paired sampling point using two portable infrared gas analysers connected to soil respiration chambers having a headspace volume of 2250 cm^3 (EGM-4 and SRC-1; PP Systems, Hitchin, UK). The system was calibrated before each sampling day against CO_2 with a nominal concentration of 409 ppmv. Measurements of total (RTOT) and heterotrophic (RH) respiration were carried out simultaneously.

Whilst the importance of diurnal variation in soil CO_2 efflux has been recognised in agricultural research (Parkin and Kaspari 2003), this variation is less important in heavily shaded forested areas (Davidson et al. 2000). Furthermore, it has been previously suggested that there are no significant diurnal changes in soil respiration under closed canopies (Epron et al. 1999a). In order to minimise the influence of any diurnal variation in soil respiration, measurements were made between 10 am and 4 pm. On two occasions, 24 hours measurements were carried out. The average values obtained for the hours 10 am to 4 pm did not differ significantly from the other 18 hours period in both days where the 24 hours measurements were taken.

Continuous total soil respiration was also measured, using an automated open-top chamber system based on the design of Fang and Moncrieff (1998), with a parallel 4-way solenoid control system and data logger (Goulden and Crill 1997). Continuous chamber respiration measurements were taken every half hour for 120 days to assess the sampling resolution of the soil respiration model (Black et al. 2005).

Soil temperature and moisture measurements

Soil temperature at 2, 6 and 10 cm depths was measured adjacent to each collar (220 K temperature meter, Jenway, Essex, UK). In addition, soil temperature was continuously monitored in the 15 year old stand using soil temperature probes that were vertically inserted into the soil to different depths, from ground level to 30 cm (Campbell Scientific Ltd., Shepshed, UK). The system was set up to record 30 minute average temperatures. Site-specific temperatures for each of the core and reference sites were estimated by means of linear regressions performed between the temperature taken at the time of measurements in the different stands and the ones simultaneously recorded by the automatic system. Given the physical proximity among all the stands, regressions were highly correlated (data not shown).

Soil moisture content in the top 6 cm of every collar was determined using a moisture probe (ThetaProbe ML2x, Delta-T Devices, Cambridge, UK). A soil-specific calibration of the instrument based on the relationship between the measured dielectric constant of the soil and its volumetric moisture content was performed to theoretically achieve an accuracy of ±1% (Delta-T Devices soil-specific calibration protocol).

Relationships between soil temperature and soil respiration

A two parametric exponential function was used to describe the relationships between soil CO_2 fluxes and soil temperature. The formula is:

$$y = a\, e^{b\, T}$$

where y is the measured soil CO_2 efflux rate, T is the measured soil temperature, a and b are the fitted parameters obtained using non-linear regressions with SigmaPlot V.8.02 software.

Q_{10} is an index used to explain the sensitivity of soil respiration to temperature. The Q_{10} values were calculated as:

$$Q_{10} = e^{10\, b} \qquad (1)$$

Standard error for Q_{10} was calculated as $Q_{10} \cdot 10 \cdot$ s.e.(b).

Root biomass

In May 2003, between 7 and 15 soil cores per stand were taken using a root auger (4 cm radius) up to a depth of 30 cm. Soil samples were immediately stored at 4ºC and processed within four days of their collection. For this, fine roots were washed and sieved to separate them from soil mineral particles. Roots were sorted into three diameter classes (<1 mm, 1-2 mm and 2-5 mm). Finally, washed roots were weighed after being oven-dried at 70º for 48 hours to determine fine root biomass. The carbon content and nitrogen concentration of roots <1 mm were determined in a Vario El-III C/N analyzer.

Statistical analysis

Based on results of tests on normal distribution of respiration rates or other parameters (Kolmogorov-Smirnov test) either parametric t-test or the non-parametric Mann-Witney test was used to identify differences between sites. Correlation analyses were used to examine relationships between soil respiration rates and soil temperature and soil moisture. The SPSS 12.0 software package (SPSS Inc., United States) was used for all statistical analyses.

RESULTS

Influence of soil temperature and soil water content on soil respiration

Consistently for all stand ages, maximum soil temperatures coincided with minimum soil water contents during the summer (Figure 2.1). In contrast, minimum soil temperatures occurred when soil water contents were highest, except for stands located on a flat terrain (10 and 15 year old), where there was a peak in soil moisture after a period of sustained rainfall in late July. No significant differences in mean annual soil temperature rates taken at a depth of 2 cm were found among stands (P>0.05). Volumetric soil moisture contents determined in the top 6 cm of the soil profiles averaged about 36%, and showed no significant differences between three of the four stands over the course of the year (P>0.05). However, there was an exception with the 31 year old stand, where soil water content was significantly lower in comparison to the rest of the stands. Soil temperature and soil volumetric water content were not significantly different within stands of the same age (P>0.05).

In all stand ages, total soil respiration (R_{TOT}) showed pronounced seasonal variations, which paralleled changes in soil temperature (Figure 2.1). R_{TOT} values were at their lowest during winter time with a minimum CO_2 flux rate of 24.3 ± 3.1 mg C m^{-2} h^{-1} measured in the 47 year old stand. By contrast, values of soil respiration peaked in late July or early August in all the stands, with the maximum rate recorded of 220.2 ± 16.7 mg C m^{-2} h^{-1} in the 10 year old stand. Subsequently, soil respiration rates followed a steady decrease towards the end of the year, with the exception of sites where low soil moisture content during late summer acted as a limiting factor for soil CO_2 efflux. In such cases, there was a drop in R_{TOT}, reaching a rate as low as 17.3 ± 2.5 mg C m^{-2} h^{-1} at the 31 year old stand (Figure 2.1). The limitation of soil respiration rates due to low values of soil moisture was obvious in all plantations when the soil water content approached 20%, and lasted until the soil water deficit was replenished by rainfall.

For each stand age, an exponential model was fitted with soil respiration rates and soil temperatures at depths of 2, 6 and 10 cm. The temperature depth used in the regressions was 2 cm because it produced the best fit for the models among all depths (Figure 2.2). Models fitted for the core sites explained more of the annual variation in those stands where moisture deficits were minimal (e.g. 10, 15 and 47 year old stands). Q_{10} based estimates explained 79% of the annual variation of R_{TOT} in the 47 year old stand, but only 56% in the 31 year old stand (Table 2.4). The estimated Q_{10} values ranged from 3.5 (15 year old) to 4.6 (10 year old). The average Q_{10} value was calculated by pooling the core stands together was 3.8. Q_{10} values calculated for 6 and 10 cm depth were slightly higher and had lower regression fits (data not shown).

Spatial heterogeneity in similar aged stands

Soil respiration rates measured at the core sites were compared to the ones obtained at their reference stand ages by means of paired t-tests (SPSS Inc., USA), (Table 2.2). There were no significant differences (P>0.05) among core sites and their correspondent references in the 10, 31, and 47 year old stands. In the 15 year old stands, while the references sites had higher respiration rates than the core site, only one of them was

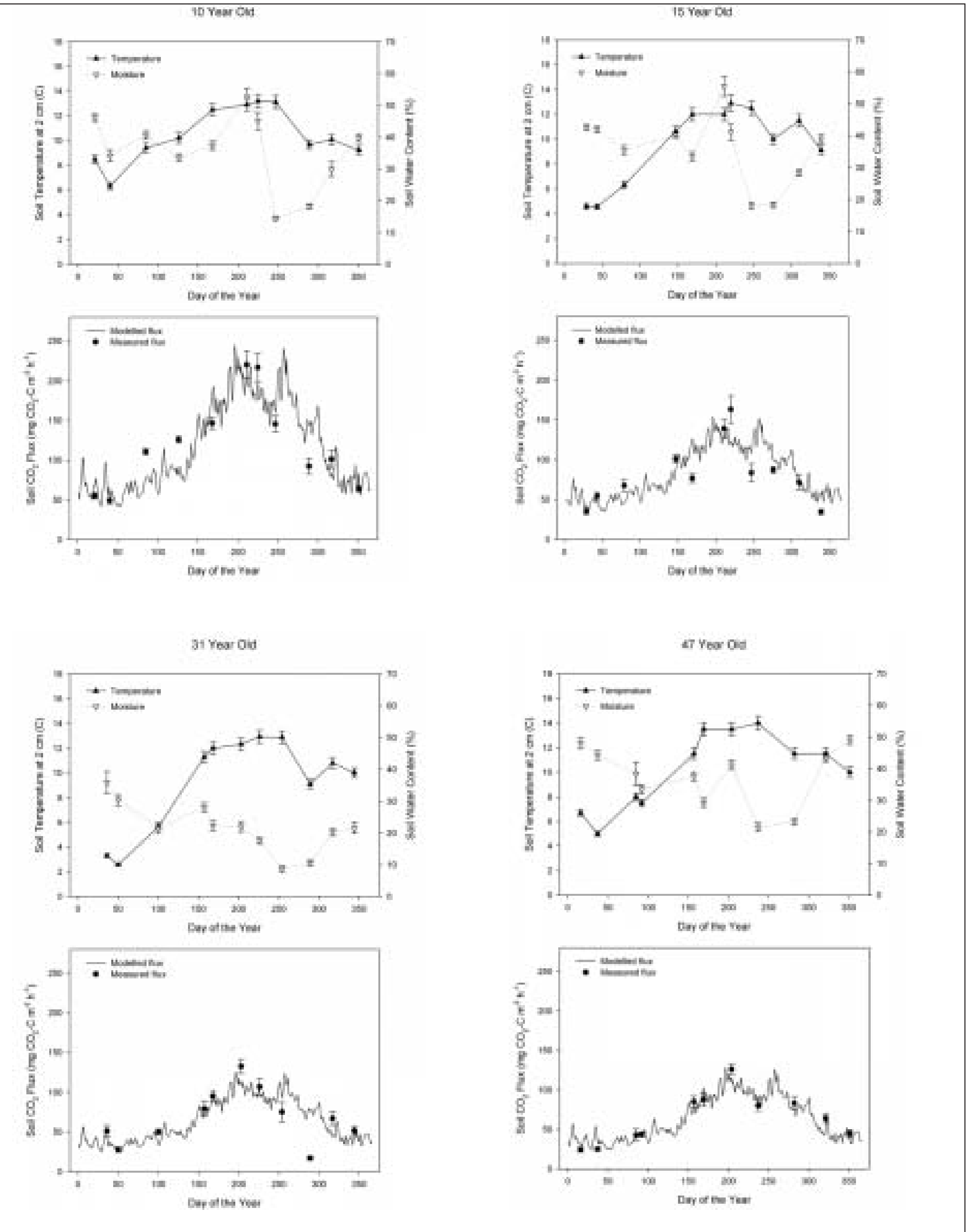


Figure 2.1: Seasonal variation of soil CO$_2$ flux, soil temperature and soil water content in the different stand ages. Each measured value for temperature, water content and soil CO$_2$ flux is the mean of 30 measurements. Error bars are standard errors of the means. The continuous line represents the modelled flux based on the exponential functions fitted for each stand (Table 2.4).

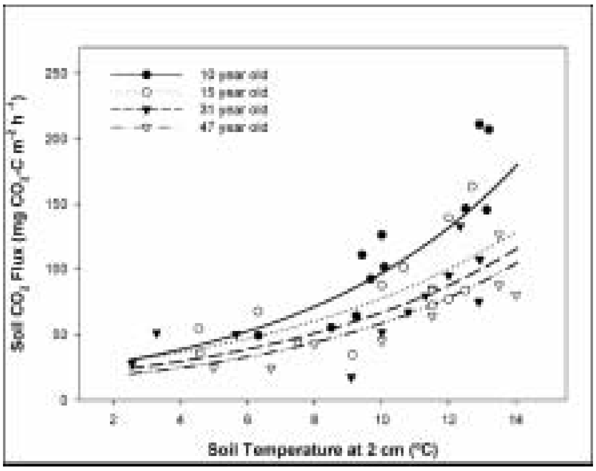


Figure 2.2: Relationships between soil CO_2 fluxes and soil temperature measured at 2 cm for the core Sitka spruce stands. Each point is the mean of 30 sampling points made per sampling day. The exponential functions are shown in Table 2.4.

marginally different from its core site (P=0.042). The other reference was not significantly different from its core site (P=0.075). The 15 year old references were not significantly different from each other (P=0.580).

Age related changes in soil respiration

There was a decrease in RTOT with stand age (Table 2.3). All the core stand ages were compared by means of a paired t-test. The youngest stand (10 year old) with a mean soil respiration rate of 111.1 ± 16.4 mg C m^{-2} h^{-1} was significantly higher (P<0.05) than the 15, 31, and 47 year old stands. The mean soil respiration rates of the older stands (> 10 years-old) were similar (P>0.05) (Table 2.3), which may indicate a steady state in soil respiration process.

Table 2.2: Mean soil respiration rates (mg CO_2-C m^{-2} h^{-1}) and standard errors for core and reference sites in each stand age. Different letters within the same rows denote significant different soil respiration rates (P< 0.05).

		Sites		
Stand age	**n**	**Core**	**Reference B**	**Reference C**
10 year old	6	132.4±22.4^a	124.2±26.6^a	124.1±28.2^a
15 year old	6	80.8±18.1^a	109.8±17.8^{ab}	136.2±39.9^b
31 year old	6	59.5±17.8^a	81.0±24.5^a	69.8±21.7^a
47 year old	6	60.1±14.5^a	65.5±12.3^a	54.9±17.7^a

Annual soil respiration rates

Estimations of annual RTOT were achieved with the temperature-based models fitted for each stand. Daily mean soil temperatures were used to simulate daily soil respiration. Mean modelled daily values were then summed to obtain the annual estimate of soil respiration in each stand. Those values ranged from 991, 686, 556, and 564 g C m^{-2} for the 10, 15, 31, and 47 year old stands respectively (Table 2.4). There was close agreement between predicted and observed soil CO_2 efflux in the different stand ages as shown in Figure 2.1, with R^2 values of 0.75; 0.70; 0.57, and 0.92 for the 10, 15, 31, and 47 year old stands respectively. A further estimate of annual R_{TOT} at each stand age was calculated by interpolating measured soil respiration between sampling dates for every day of the year to obtain the annual value. Interpolated respiration rates

Table 2.3: Mean soil respiration rates (mg CO_2-C m^{-2} h^{-1}) measured over 2003 for each treatment in the four core sites of the chronosequence. Means and standard errors for the different stand ages are based on n=11 sampling dates in which an average of 30 points per treatment were sampled. Different letters within the same treatment denote significantly different soil respiration rates (paired t-tests, α=0.05).

Treatment	n	10 year old	15 year old	31 year old	47 year old
Total	11	111.1 ± 16.4 a	83.3 ± 13.2 b	68.7 ± 10.3 b	68.8 ± 11.1 b
Heterotrophic	11	48.5 ± 5.3 a	36.6 ± 3.5 b	30.1 ± 5.5 b	35.1 ± 5.2 b
Roots	11	62.6 ± 11.2 a	48.7± 7.8 b	36.7 ± 6.0 b	32.9 ± 5.1 b

produced similar results as the modelled ones (Table 2.4). These interpolated annual values were 1013, 692, 559, and 577 g C m^{-2} for the 10, 15, 31, and 47 year old stands respectively (Table 2.4).

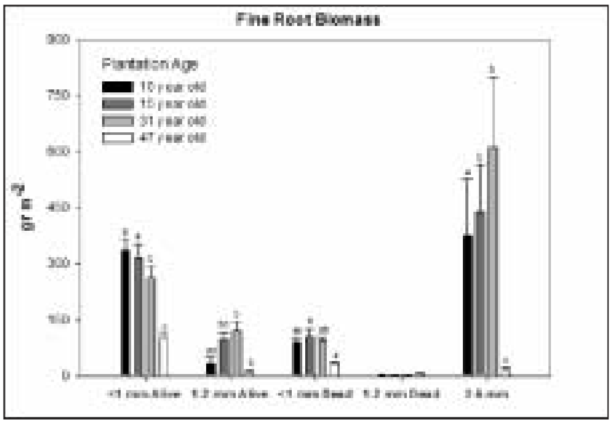


Figure 2.3: Categories of fine root biomass in the different stand ages. Different letters represent significant stand age differences, where P<0.05. Roots were sampled to a depth of 30 cm.

Fine root biomass

Fine root biomass varied over the chronosequence, and did so following trends depending on the diameter of the roots (Figure 2.3). There was a decreasing trend in root biomass <1 mm diameter over the chronosequence (Figure 2.3). The oldest site (47 year old) had significantly lower fine root biomass in all categories <5 mm, but showed the highest biomass of roots >5 mm. Fine roots were concentrated in the surface horizons, especially in the litter-humus layer. Nitrogen concentration in roots <1 mm showed no significant differences among stand ages (data not shown).

Annual variation of soil respiration components

a) Heterotrophic soil respiration (R_H)

R_H rates ranged from 79.1 ± 7.7 mg C m^{-2} h^{-1} measured at the 10 year old stand during mid summer, to 6.5 ± 1.3 mg C m^{-2} h^{-1} recorded in the 31 year old stand (Figure 2.4). Mean R_H rates ranged from 48.5 ± 5.3 mg C m^{-2} h^{-1} in the 10 year old stand to 30.1 ± 5.5 mg C m^{-2} h^{-1} observed in the 31 year old site. The youngest stand (10 year old) had significantly higher rates of R_H (P<0.05) when compared to all the other forest ages (Table 2.3).

b) Autotrophic respiration (R_A)

Root or autotrophic respiration (R_A) was calculated as the difference in daily mean respiration rates between total and heterotrophic plots. Autotrophic respiration includes respiration from live roots, mycorrhizae and the decomposition of exudates

Table 2.4: Relationship between all soil respiration rates (mg CO_2 m^{-2} h^{-1}) and soil temperature (°C) measured at a 2 cm depth (n = 11). All r^2 values and partial coefficients (i.e. *a* and *b*) for the relationship ($y = a \times e^{b \times T}$) were significant at P < 0.0001. Values in parenthesis next to the Q_{10} values represent a standard deviation. Values in parenthesis represent the modelled annual estimated corrected for decomposition of trenched roots in R_H component.

Stand age	R component	r^2	a	b	Q_{10}	Modelled	Interpolated
10 years	R_{TOT}	0.77	20.8	0.15	4.6 (1.5)	991	1013
	R_H	0.72	13.7	0.12	3.3 (0.9)	403 (434)	439
	R_A	0.77	9.4	0.09	5.6 (2.1)	584 (553)	574
15 years	R_{TOT}	0.66	21.9	0.13	3.5 (1.1)	686	692
	R_H	0.71	14.0	0.09	2.4 (0.5)	297 (329)	328
	R_A	0.67	13.6	0.15	3.5 (1.6)	427 (395)	387
31 years	R_{TOT}	0.56	17.1	0.14	3.9 (1.9)	556	559
	R_H	0.27	11.0	0.11	2.9 (1.9)	240 (267)	263
	R_A	0.66	8.8	0.14	4.0 (1.8)	322 (295)	299
47 years	R_{TOT}	0.79	13.8	0.14	4.2 (1.4)	564	577
	R_H	0.7	9.4	0.12	3.3 (1.1)	284 (294)	296
	R_A	0.7	5.5	0.16	5.1 (2.7)	286 (276)	237

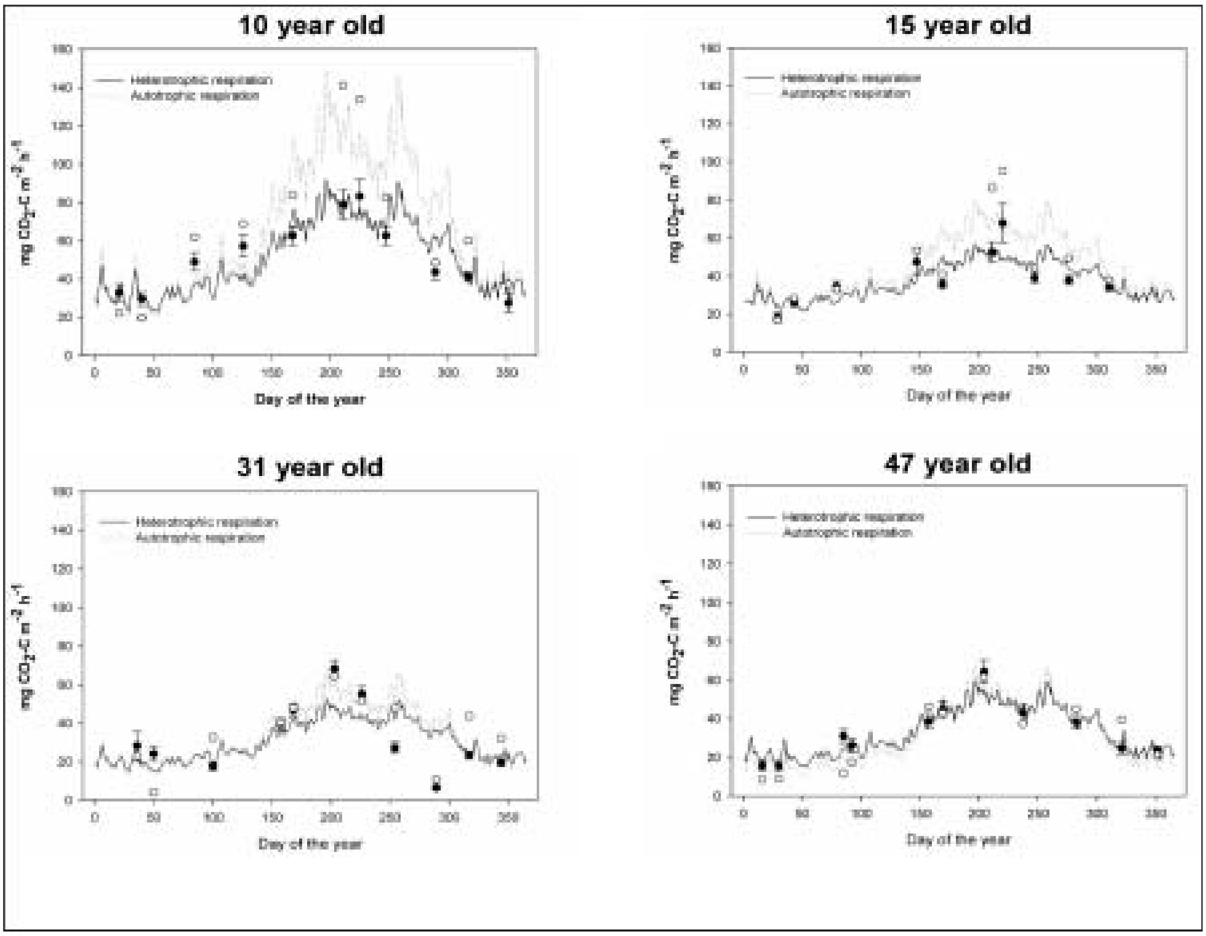


Figure 2.4: Seasonal variation of heterotrophic and autotrophic soil CO_2 flux measured in the different stand ages. Each measured heterotrophic CO_2 flux is the mean of 30 measurements per sampling date. Error bars are standard errors of the means. The lines represent the modelled fluxes based on the exponential functions fitted for each stand and treatment (see Table 2.4).

from the roots. In all the stands, R_A rates peaked around mid summer (day of year 209), and reached minimum values during winter. The maximum calculated rate was 141.1 mg C m^{-2} h^{-1} at the 10 year old stand. The lowest rate in R_A was 4.1 mg C m^{-2} h^{-1} and this occurred at the 31 year old stand at the same time as lowest recorded mean daily soil temperature at 2 cm (2.1ºC) for all sites during 2003.

The measured annual mean R_A for the 10 year old stand was 62.6 ± 11.2 mg C m^{-2} h^{-1}, and was significantly higher ($P<0.05$) than R_A rates obtained for the older stands (Table 2.3). Measured mean annual R_A rate was lowest (32.9 ± 5.1 mg C m^{-2} h^{-1}) at the 47 year old stand.

The contribution R_A to R_{TOT} varied throughout the year as shown in Figure 2.4. Based on the variation in the modelled components of soil respiration, R_A contributed most to the total soil respiration during the growing season (May to October). R_A accounted for 64% of the total flux at the 10 year old stand in late July. During winter months R_H and R_A contributed equally (each at ca. 50% of R_{TOT}).

Continuous soil respiration measurements

Comparisons of the model estimates with the automated soil respiration measurements also suggested that the temporal resolution of the soil respiration model could be improved by accounting for the influence of rainfall events (Figure 2.5). This is manifested by the large fluctuations in measured soil respiration rates during and after major rainfall events, which were not evident from the predicted soil respiration data (Figure 2.5; Black et al. 2005).

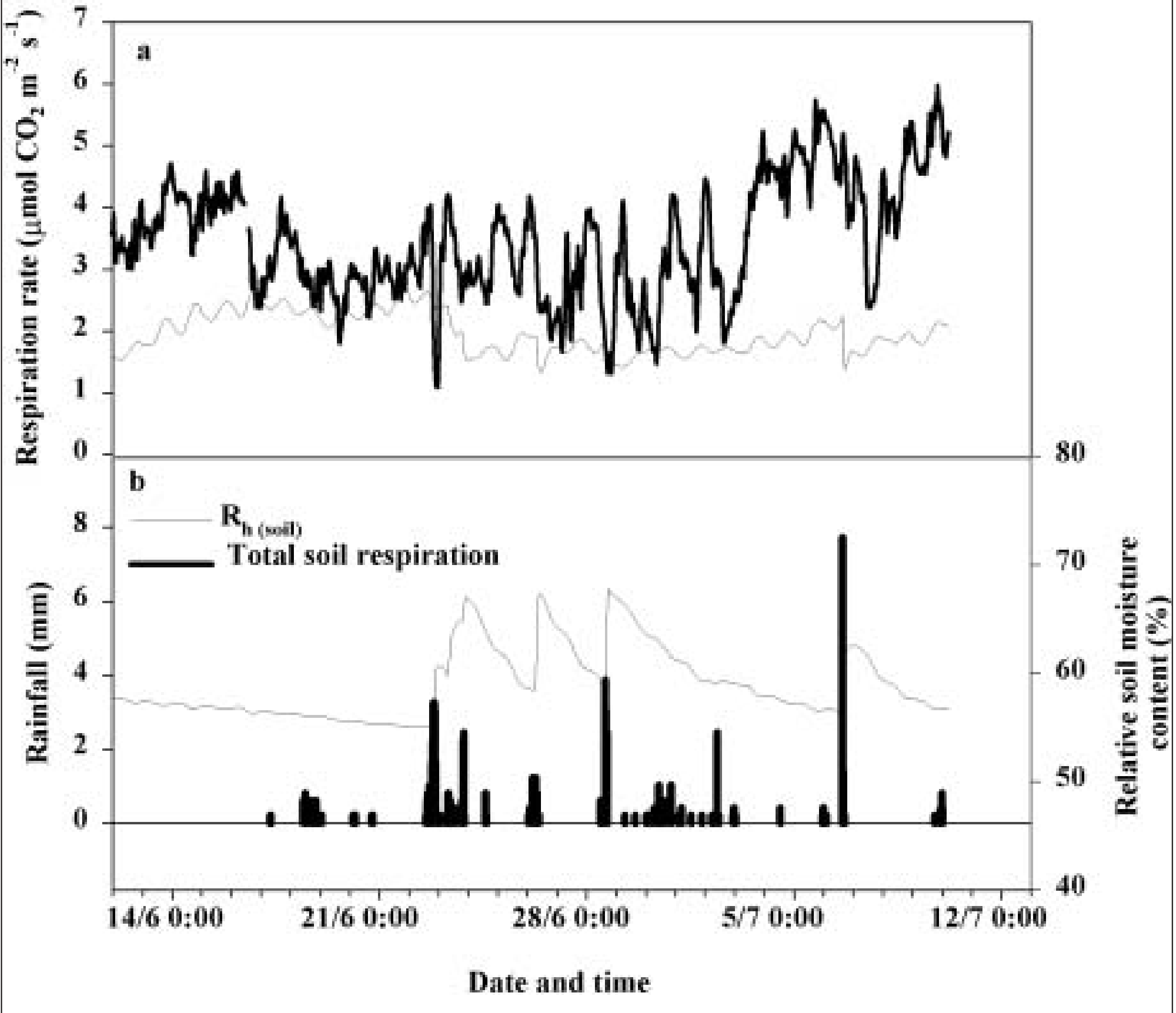


Figure 2.5: A comparison between predicted soil heterotrophic respiration ($R_{h\ (soil)}$) and measured total soil respiration rates (a), and changes in relative soil moisture content (line plot) and rainfall (histograms) during June and July (b).

CONCLUSIONS

The results show that afforestation with Sitka spruce on former semi-managed grassland leads to a high soil efflux for the first 10 years followed by a decline at canopy closure and maturation. Soil respiration rates reach steady state in the later stages prior to felling.

Annual soil respiration rates were modelled by means of temperature derived functions. Total annual soil respiration rates were 991, 686, 556, and 564 g C m^{-2} for the 10, 15, 31, and 47 year old stands respectively. Our reported values are considerabley higher than those reported for afforested blanket peats, (Byrne and Farrell 2005), where soil CO_2 emissions varied from 170 to 260 g C m^{-2} for 3 to 39 year old Sitka spruce stands. Clearly the variation in CO_2 emissions from forests on different soils requires further investigation.

Autotrophic respiration showed a decreasing trend with stand age, which is explained by a decrease in fine root biomass <1 mm diameter over the chronosequence. This difference was not explained by fine root nitrogen concentrations. Heterotrophic respiration paralleled soil organic carbon inputs (litter and fine root) over the chronosequence (see Chapter 3). Heterotrophic respiration decreased with stand age, but slightly increased again in the oldest stand (47 year old) as a result of accumulated above-ground litter and root inputs. Soil carbon effluxes in the older stands reached a steady state, as was evident from the similar respiration rates in the 31 and 47 year old stands.

Higher soil respiration rates in the youngest stand may be a result of both the larger availability of soil organic matter, and the greater activity of its root system compared to the more mature stands. The relative contribution of root respiration (R_A) to total CO_2 efflux decreased with stand age. The decrease in the relative contribution of autotrophic respiration to total soil CO_2 efflux from 59.3% in the youngest stands to 49.7% in the oldest stand was explained by the higher activity of the root system in younger stands.

The results show that the use of models to predict annual mean soil respiration in forest ecosystems may underestimate soil respiration rates of young afforested ecosystems if a stand age factor is not included. Whilst the proportion of heterotrophic to total soil respiration (Figures 2.2 and 2.3) appears to be consistent with previous reports (Hanson et al. 2000) and the heterotrophic soil model estimates appear to be as robust as those reported by Ehman et al. (2002) and Curtis et al. (2002), there is a need to develop mechanistic process-based models. Based on the observation that soil temperature and moisture only accounted for 61% of the observed variation in soil respiration in the current study (Black et al. 2005), empirical models may fail to fully capture temporal variability in respiration. In addition, the sharp decrease in soil C efflux during and after rainfall events highlights the importance of accounting for transport processes through the soil profile, such as the diffusion of CO_2 out of, and O_2 into, the soil column (Fang and Moncrieff 1999, Moncrieff and Fang 1999).

Site productivity may confound the effects of stand age on soil respiration and may influence the time for a young spruce plantation to change from a carbon source to a carbon sink by means of reducing soil respiration and increasing carbon storage in biomass. However, other issues such as enhancement of N_2O emissions may require further careful evaluation when establishing plantations with the objective of sequestering carbon. Further research over longer periods and on different ecosystems is required to validate and assess the importance of stand age on afforestation sites.

CHAPTER 3

SOIL STOCKS AND SOIL PROCESSES

Brian Reidy, Jens Dyckmans, Olaf Schmidt and Tom Bolger

A) SOIL C STOCKS

Introduction

The soil carbon pool is the largest and most significant carbon storage pool in the forest ecosystem. An estimated 247-286 Pg C are stored in the soil of boreal forests (Schlesinger 1984, 1991). In addition, the system is considered to be a significant sink for atmospheric C, accounting for as much as 662 Tg C yr^{-1} (Apps et al. 1993). The ability of forest soils to sequester C is due to the deposition and accumulation of a resistant slowly decomposable C pool with a half-life 1000 + years. In CARBiFOR this pool was estimated to represent approximately 65% of the soil organic matter (SOM) in the sites studied.

The size of the soil organic matter pool represents a balance between the organic matter input as litter of various types and the rate of decomposition. Litterfall is positively correlated with NPP (Vilà et al. 2004) while the rate of decomposition is determined largely by climate, the chemical quality of the litter input and the decomposer organisms present in the system. However, there is one important additional factor: soil texture. All other things being equal, soils with a high clay content are known to contain more SOM. This arises because of the physical interactions between the clay particles and the SOM, which provide a high degree of protection for the organic matter (McInerney and Bolger 2000).

Objectives

Litter inputs and soil carbon stocks were estimated by Working Group 3. An attempt was then made to use these figures to model sequestration in a particular forest type.

The specific objectives were:

- To measure the temporal variation, quantity and quality of litter input within the stands that make up the chronosequence
- To measure the carbon stocks in the soils of the chronosequence and to partition the stocks into microbial, structural and passive pools.
- Integrate the results of this work package and those of the other Working Groups into the CENTURY model to describe the 15 year old site.
- Use basic soil carbon data from other forested sites to test validity of the model predictions.

Methods

The soil is described as being associated with the Raheenduff Imperfectly Drained Phase according to Soils of County Laois, from the national soil survey of Ireland. Coillte surveys of the forested area indicates that 90% of the soil is gley with 10% brown earth/podsol mixture. Mean sand, silt and clay contents of the sites in the chronosequence are 22%, 36% and 42% respectively, pH is 4.6 and bulk density is 1.018 kg m^{-3} (Carly Green pers. comm.) These figures mirror those described in the national soil survey.

Litter input was measured every month using litterfall collectors at each site in the chronosequence. The collectors were plastic bins with a 350 cm diameter opening and 10 were placed in each site. In the case of grassland (G0), estimates were made using the harvest technique of Sims et al. (1978). The same technique was used to estimate the increase in grass litter due to canopy closure in the 10 year old site.

The carbon, nitrogen and lignin contents of the litter inputs were measured twice at each site of the chronosequence. The carbon and nitrogen concentrations were measured using a CHN analyser. The lignin content was estimated following the methods of Allen (1989).

Total C content of the soils at each site was estimated for each of the organic and upper mineral horizons based on loss on ignition, which was calibrated using a CHN analysis. Sub-samples of fresh soil were removed and microbial biomass C estimated using the chloroform fumigation technique (Vance et al. 1987). Light and heavy fractions were separated following the methods of Compton and Boone (2002). The light fraction is believed to be representative of the slow pool plus the microbial biomass, while the heavy material is primarily the resistant/passive pool. Woody debris larger than 1 cm in diameter was removed and the soil was then sieved through a 5.6 mm mesh and dried at 80°C. The sub-sample was then floated in a solution of sodium meta-tungstate, 10 g per litre of water (which has a density of 1.75). The resulting heavy and light material were dried, weighed, ground and mixed thoroughly and the carbon and nitrogen content was estimated using a CHN analyser. The total stock was estimated by multiplying the carbon and nitrogen concentrations by the weight of the samples (which was area based).

The CENTURY Model was originally developed for grassland systems in North America and has been developed and used extensively in recent years to model forest systems. An analysis of the applicability of the CENTURY model, version 4, to predict the carbon stocks of the stands under our site conditions was carried out. To measure SOM pools equivalent to those used in the CENTURY model, the heavy and light fractions mentioned above were used according to Christiansen (1992). An analysis of the applicability of the model was performed to predict the carbon stocks of forests under the conditions prevalent in Ireland and the results tested against measured stocks in a variety of sites.

Results

Litter inputs

Litter input was measured at four forested sites in the chronosequence. Monthly litter C inputs tended to be greatest in the March/April period of the year associated with bud burst (Figure 3.1). The input was greatest in the 15 year old site as canopy closure had resulted in all the needles of the lower branches dying and no thinning had occurred. The largest inputs (4.04 t C ha^{-1} a^{-1}) have been recorded in the 15 year old site (Figure 3.2). The annual litter input values for the 10 year old stand (D10, Table 1.1 and Figure 3.1) does not include the dead grass C input due to land-use change from a grassland to a closed forest canopy. This represented an additional C input of ca. 3 t C ha^{-1} yr^{-1}, based on a decrease in measured grassland biomass of 1.5 t C ha^{-1} yr^{-1} and a shoot to root ratio of 0.5. The annual litterfall was closely related to changes in leaf area

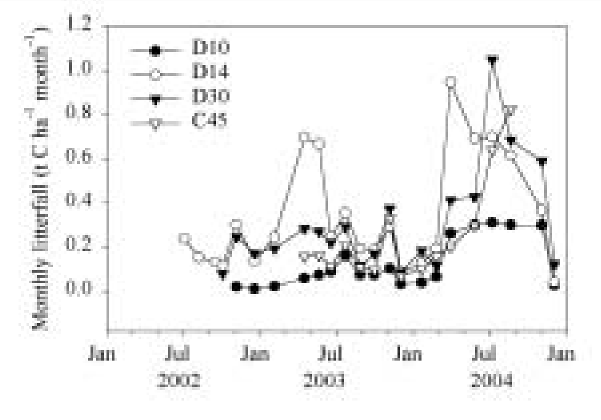


Figure 3.1: Monthly litterfall, at the 10 (D10), 14 (D14), 30 (D30), and 45 (C45) year old afforested sites, between July 2002 and December 2004. For information on the site characteristics refer to Tables 1.1, 1.2 and 1.3.

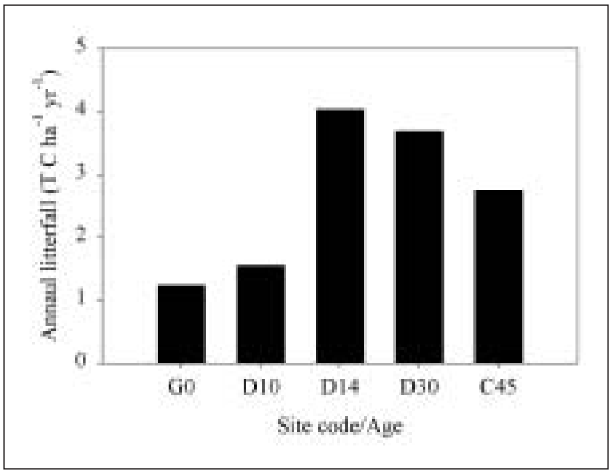


Figure 3.2: Annual litterfall, at the grassland (G0) and the 10 (D10), 14 (D14), 30 (D30), and 45 (C45) year old afforested sites, for the year 2004. For information on the site characteristics refer to Tables 1.1, 1.2 and 1.3.

index and net primary production (NPP) over the chronosequence (see Table 1.2, and Chapters 5 and 6).

The carbon, nitrogen and lignin contents of the litter inputs for the four afforested sites did not vary significantly (Table 3.1). The grassland litter (G0) was seen to be more decomposable due to its low lignin content which was ~25% as distinct from >40% in spruce needles.

Table 3.1: Quality of litter inputted into the five sites of the chronosequence, % C, N and lignin.

Age	C	N	Lignin
0 (G0)	43.14	1.02	24.91
10 (D10)	47.12	1.14	41.79
14 (D14)	47.58	1.37	41.69
30 (D30)	47.94	1.10	40.10
45 (C45)	47.62	1.06	41.41

Litter C Pools

The standing litter in each site increased with age. There was an accumulation on the forest floor especially with the onset of canopy closure where the rate of decomposition was less than accumulation. In the 10 year old site (D10) there was still living understorey vegetation and an open canopy, which partially explains the lack of accumulation at this site. The oldest site (D47) showed greatest variation due in part to standing water and patches of bare soil where no litter accumulated (Figure 3.3). The accumulation of litter at the 45 year old site (C45) was lower, when compared to the 47 (D47) year old site. However, the topography of this site (C45) was sloped with a gradient of 10%. In addition, the soils from the C45 site had higher sand and lower clay contents (Table 1.3).

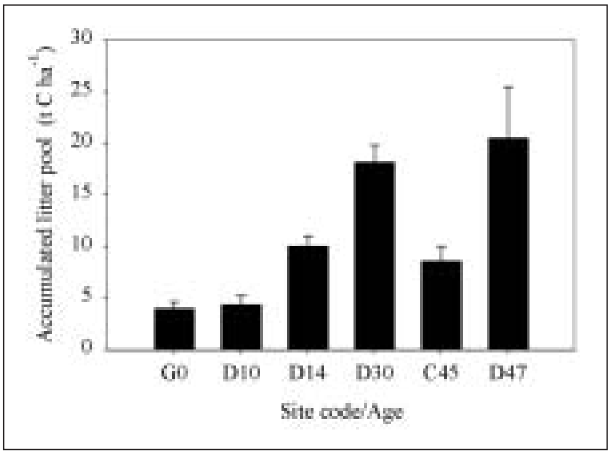


Figure 3.3: Accumulated or standing litter at the grassland (G0) and the 10 (D10), 14 (D14), 30 (D30), 45 (C45) and 47 (D47) year old afforested sites. For information on the site characteristics refer to Tables 1.1, 1.2 and 1.3.

Soil C pools

The stocks of carbon in the soil varied considerably between the closed canopy sites in the chronosequence but generally accumulated with time and were always greater than 100 t C ha^{-1} (Figure 3.4). Generally, the increase in soil C stocks over time varied between 0.2 and 2.3 t C ha^{-1} yr^{-1}. The lower C stock and stock changes occurred in the 30 (D30) and 45 (C45) year old stands (Figure 3.4). These stands had sloping topography and lower clay and soil moisture contents. In contrast, the higher accumulation of soil C in the 10 (D10), 14 (D14) and 47 (D47) year old stands may be associated with the flat topography and higher soil moisture contents (see Chapters 1 and 2).

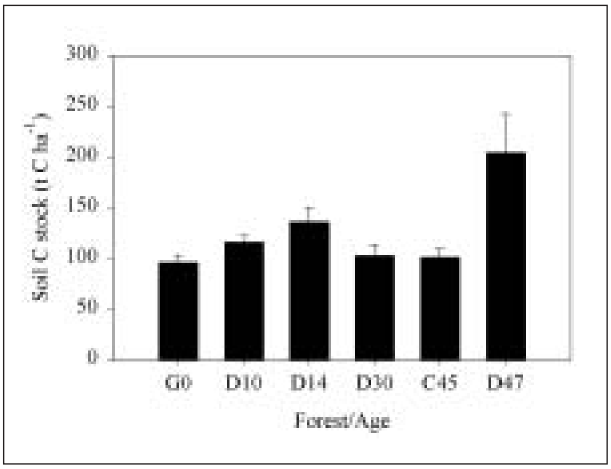


Figure 3.4: Total soil carbon stocks (t C ha-1) in mineral and organic layers at the grassland (G0) and the 10 (D10), 14 (D14), 30 (D30), 45 (C45) and 47 (D47) year old afforested sites. For information on the site characteristics refer to Tables 1.1, 1.2 and 1.3.

The total soil organic matter (SOM) was predominantly comprised of a fraction (highly resistant C pool) of the total soil C pool increased over the chronosequence whereas the light fraction (degradable labile pool) did not increase, but fluctuated between 18 and 36 t C ha^{-1}.

Modelling

Using our data on heavy, light and microbial fraction analysis, estimates were made of the active, slow and passive carbon pools for the model. All other necessary parameters for the 15 year old site were input from data gathered by other working groups or estimates from the literature. The CenW (Kirschbaum and Paul, 2002) model was parameterised using information from the chronosequence and estimates of t C ha^{-1} soil organic carbon generated for a 50 year period. These are illustrated in Figure 3.5 with the measured stocks from the sites. The values give reasonable estimates for four of the stands but grossly overestimate the stocks at 30 and 45 years. The CenW model was based on the CENTURY model (Parton et al. 1987).

Recent versions of CENTURY (e.g. CENTURY 4.0) have been extended to include forest systems. Preliminary runs of this model, with only minor adaptations of default values, give apparently better predictions of the stocks in a range of older sites sampled in the midlands (Figure 3.6). This is the case even though the current parameterisation of the model is crude and will require more work.

Discussion

The measured temporal variation in our site (Figure 3.1) showed litterfall reaching its peak in the April/May period of the year for the younger stands (0.97 t C ha^{-1} yr^{-1} April 2004, D14) and the June/July period for the older stands (1.05 t C ha^{-1} yr^{-1} July 2004, D30). This was comparable to the estimate of 1.1 t C ha^{-1} yr^{-1} in a 34 year old site in June/July in Co Wicklow reported by Carey and Farrell (1978). The total inputs of 4.43, 2.02 and 1.93 t C ha^{-1} yr^{-1} in 34, 39 and 45 year old stands respectively reported in that paper are also similar amounts measured in our 30 and 45 sites which were 3.7 and 2.8 t C ha^{-1} yr^{-1} respectively (Figure 3.2). In contrast, 30–34 year old stands in Denmark had a much lower average input of 1.76 t C ha^{-1} yr^{-1} for five years 1989-1994 (Pedersen and Bille-Hansen 1995).

The C and N concentrations in the Dooary litter were 47.5% C and 1.20% N which are comparable with the 48.4% C and 1.27% N reported by Carey and Farrell (1978). Mean lignin content is 41.2% for the forested sites (Table 3.1), which is at the

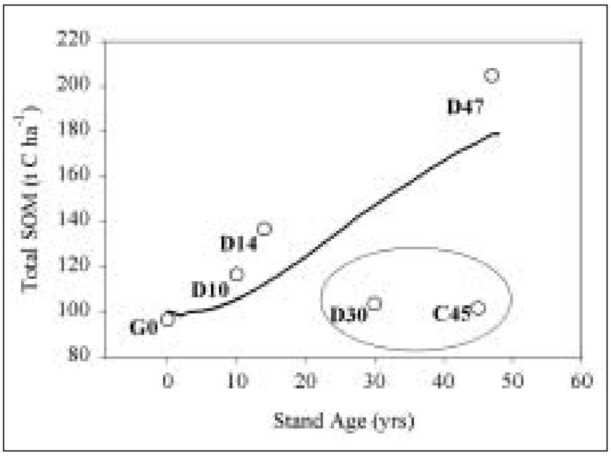


Figure 3.5: Modelled (solid line) and measured (labelled symbols) total soil organic matter (SOM) in different aged stands. The CENW model was parameterised using the chronosequence light and heavy SOM pools across the chronosequence. The highlighted outliers represent sites with sloping topography.

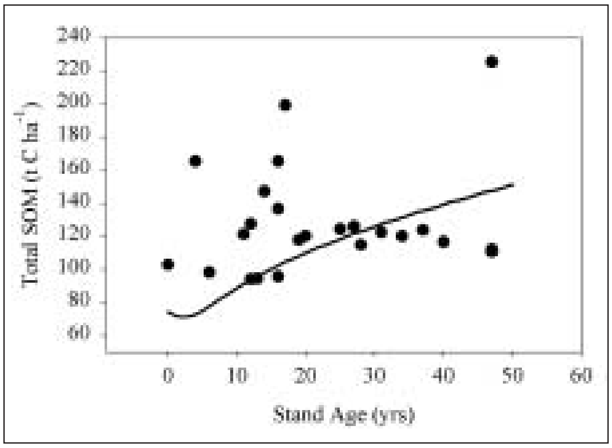


Figure 3.6: Modelled (solid line) and measured (black symbols) total soil organic matter (SOM) in mineral soils from different aged stands. This CENTURY 4.0 model was parameterised using the chronosequence light and heavy SOM pools across the chronosequence.

high end of the spectrum of values reported in the literature. This ranges from 35-36% in Norway spruce litter in southern Sweden (Berg 2000, Lundmark-Thelin and Johansson 1997), to 40.47% lignin in fir in Italy (Rutigliano et al. 1996).

Total soil carbon stocks ranged from 120-200 t C ha^{-1}. There are no studies of carbon stocks on similar soils with the same species; however, similar ranges were noted in Sitka spruce on a peaty gley in north east England which 140 t C ha^{-1} in a 40 year old stand and 250 t C ha-1 for 30 year old reforested stand (Zerva et al. 2005). Also, 143-164 t C ha^{-1} were reported in a 32 year old stand of *Pinus radiata* on a volcanic soil in New Zealand (Oliver et al. 2004). A Norway spruce plantation on glacial clay soils had 209 t C ha^{-1} in sites aged 55-60 years (Berg et al. 2001).

The stocks for the first 20 years in the sites assessed were very variable (Figure 3.6). This may be due to site preparation for afforestation, which disturbs the soil carbon content by exposing lower soil layers to mixing with upper soil layers. The variation of the sites from the model may also be due to higher planting densities and different thinning regimes than that of North America, where the model was developed.

B) SOIL PROCESSES – STABLE ISOTOPES

Objectives

The objective of this research was to investigate critical pathways and processes controlling C storage in Irish forest soils using stable isotope techniques. Three specific topics were addressed experimentally by the stable isotope sub-group of Working Group 3:

- The role of photoautotrophic soil micro-organisms as a C source in temperate forest soils was investigated.
- Simple and cost-effective tracer techniques were developed to investigate the role of soil macrofauna species in soil C dynamics.
- The decomposition of isotopically labelled (^{13}C and ^{15}N) beech litter as influenced by N availability was studied in a long term decomposition experiment with special reference to the role of the lignin fraction and to the lignin-N complexes during decomposition.

Methods

Stable isotope techniques measure ratios of stable isotope pairs (e.g. $^{13}C/^{12}C$, $^{15}N/^{14}N$), providing a powerful tool to trace and quantify C flows in soil–plant systems. In CARBiFOR, the isotope experiments used enriched ^{13}C tracer techniques to quantify:

- the flow of CO_2-C through soil micro-organisms into soil;
- the C flow through soil animals;
- the fate of C from decomposing beech litter.

Isotopic analyses were carried out by continuous flow isotope ratio mass spectrometry (CF-IRMS) at the Scottish Crop Research Institute, Dundee, UK, and the Centre for Stable Isotope Research and Analysis, University of Goettingen, Germany.

C fixation by photoautotrophic soil micro-organisms

Forest soils were collected, transferred into air-tight glass jars and incubated in the light in a plant growth chamber for 21 days. Isotopically labelled (^{13}C)-CO_2 was injected in two pulses into the jars, which were then incubated in the dark (to estimate non-photosynthetic C fixation) or in the light for 12 days. The soil with the microbial crust was either left intact or mixed manually to simulate soil disturbance that distributes superficial organic matter. Then, representatives of three important soil animal groups were introduced:

- the springtail *Ceratophysella denticulata* (Hexapoda: Collembola), a common surface-dwelling species that is known to feed on soil algae;
- the slug *Deroceras reticulatum* (Mollusca: Agriolimacidae), both adults and newly hatched individuals that are believed to graze on soil algae;
- the earthworm *Allolobophora chlorotica* (Annelida: Lumbricidae), a common endogeic, geophagous species that lives in the mineral layers of forest soils.

Invertebrates lived freely in the jars for seven days before being sampled. They were analysed without gut content to quantify assimilated C, rather than ingested C, derived from atmospheric CO_2 in the absence of higher plants.

C flow through soil fauna

Two series of experiments were conducted to develop novel rapid and cost-effective methods for dual-labelling soil animals with isotopic tracers, ^{13}C and ^{15}N. First, terrestrial slugs, *Deroceras reticulatum* (Mollusca: Agriolimacidae), were labelled by providing them with a mixture of uniformly ^{13}C labelled [U-$^{13}C_6$]-glucose and ^{15}N enriched lettuce grown for 28 days in a greenhouse, in a base of wheatbran. Food was offered on glass slides to individual slugs and consumption measured. Stability of ^{13}C enrichments in the moistened food source was tested after 4, 8, 24, 48 and 192 hours. Slugs were analysed without gut content after 15 days. Second, an even simpler and more rapid method that circumvents the production of labelled plant material was developed for labelling soil-feeding earthworms simultaneously with ^{13}C and ^{15}N. Soil was amended with $^{15}NH_4^+$ and incubated for 7 days. Then, [U-$^{13}C_6$]-glucose was added and *Aporrectodea caliginosa* (Annelida: Lumbricidae), a common endogeic, geophagous species that lives in the mineral layers of forest soils, were provided individually with this soil for four days. Loss of the tracers was monitored for 21 days post-labelling.

Fate of C from decomposing beech litter

In June 2002, a long term incubation laboratory experiment was set up to investigate the decomposition of isotopically labelled (^{13}C and ^{15}N) beech litter as influenced by N availability with special reference to the role of the lignin fraction and to the lignin-N complexes during decomposition. Three treatments were established as follows:

i) labelled litter mixed with sand,
ii) labelled litter mixed with forest mineral soil, and
iii) forest mineral soil without litter addition.

In 50 ml syringes, 18 g soil or 12 g of fine sand were mixed with 2 g of labelled beech litter (–48‰ $\delta^{13}C$, 7 atom% ^{15}N). Sand treatments were supplied with an inoculum obtained from an aqueous extraction of soil. All treatments are maintained in the dark in an incubator at 15ºC with and without N addition (3 mmol N per pot bimonthly during the first 12 month of incubation) for 18 months. As control, half of each pot was sampled directly before incubation.

The experiment was sampled destructively in January 2004 and samples are being stored for analysis. Methodological work on the extraction of the recalcitrant lignin pool showed that existing lignin extraction methods were unsuitable because they introduce non-systematic isotopic fractionation between ^{14}N and ^{15}N. Methodological development is continuing at the University of Goettingen and the incubated beech samples will be analysed when a suitably reliable extraction method for the recalcitrant lignin pool has been developed.

Results

C fixation by photoautotrophic soil micro-organisms

Microbial mats and crusts are known to act as C (and N) input routes into soils in extreme environments. These experiments tested the hypothesis that photoautotrophic soil micro-organisms can act – independently of higher plants – as a C source in temperate forest soil food webs. In the dark, non-photosynthetic fixation of $^{13}CO_2$ in soil was negligible and no ^{13}C was assimilated by earthworms (Figure 3.7). In the light, significant amounts of ^{13}C were fixed photosynthetically by micro-organisms, as evidenced by high ^{13}C values of the bulk soil organic matter. This ^{13}C was also assimilated by earthworms, *A. chlorotica*, independent of whether the soil was left intact or mixed (Figure 3.7). The proportion of new C fixed during the incubation from CO_2 on the total soil C was 0.07% and 0.15% in the undisturbed and mixed soil, respectively. Earthworms derived 2.1% and 0.6% of their total body C from this microbial source in only 7 days (Figure 3.7). Springtails, which are much smaller and thus have faster tissue

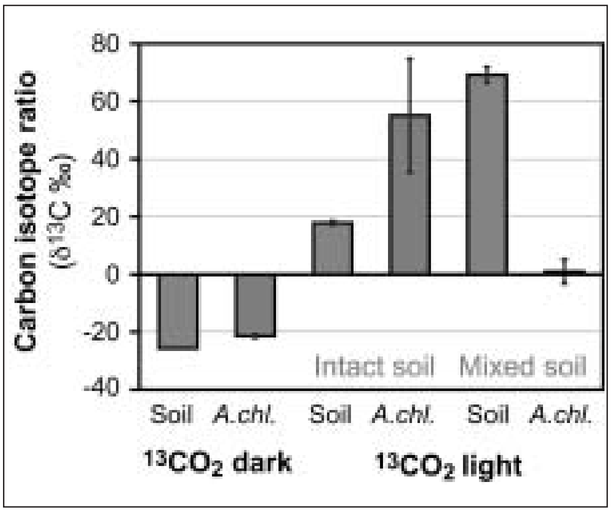


Figure 3.7: Assimilation of microbially fixed C by earthworms. Mean ±1 SD (n=5).

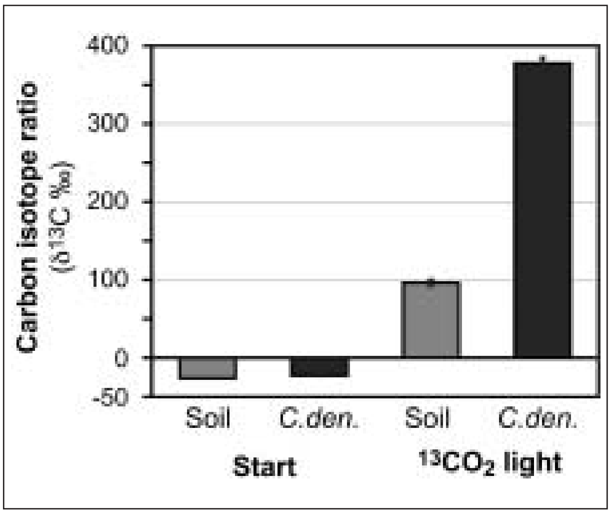


Figure 3.8: Assimilation of microbially fixed C by collembola. Mean ±1 SD (n=5).

turnover rates than earthworms, were very highly labelled after 7 days feeding on soil algal mats, deriving 16.8% of their total body C from this source (Figure 3.8).

Neither adult nor newly hatched slugs, D. reticulatum, assimilated any labelled C derived from CO_2 in these experiments (data not shown).

C flow through soil fauna

Some invertebrate soil animals are known to be keystone species that play important roles in the cycling of C and related elements in forest soils. They affect litter decomposition and C storage and release both directly (through feeding, digestion and casting) and indirectly (mainly through modifying the physical soil environment). Isotope tracer techniques provide a powerful tool to quantify the direct involvement in soil animals in C cycling. The objective of this research was to develop methods to label important forest soil animals with ^{13}C and ^{15}N for experimental purposes and estimate the flow of C and N through them.

The novel labelling technique was successful for enriching slugs simultaneously with ^{13}C and ^{15}N (Figure 3.9). The loss of ^{13}C from labelled food through microbial respiration was negligible (data not shown) and the feeding regime (restricted or ad libitum) did not have a significant effect on tissue enrichments. Assimilation efficiencies were high, 24.2% for ^{13}C and 27.4% for ^{15}N. This method can be used to devise protocols for the production of large numbers of slugs with desired single or dual isotopic enrichments.

In the experiments with earthworms (*A. caliginosa*), very high enrichments of 0.41 atom% excess ^{13}C and 0.69 atom%excess ^{15}N in whole-body tissues were achieved after only 4 days labelling. Further, labelling efficiencies (percentage of applied tracer retained in tissues) were high, 10.3% for ^{13}C and 15.8% for ^{15}N. The time course of enrichments in body tissue and cutaneous mucus (Figure 3.10) strongly suggests that earthworms possess one slow and one very fast C and N pool. Estimated half-lives for C were 0.56 and 27.5 days for these pools.

From the collaboration with groups in Germany the following results have been obtained:

- The N store utilisation in beech was shown not to be affected by atmospheric [CO_2]. Together with earlier findings that N uptake in beech does not increase with increased atmospheric [CO_2], this may indicate that the C sequestration effects of increased atmospheric [CO_2] will level out due to N limitation in the long term. In contrast we could show that the N_2 fixation (but not N uptake form soil) of black locust will increase with increasing atmospheric [CO_2]. This suggests that N_2-fixing trees will be at a competitive advantage with non-fixing trees.

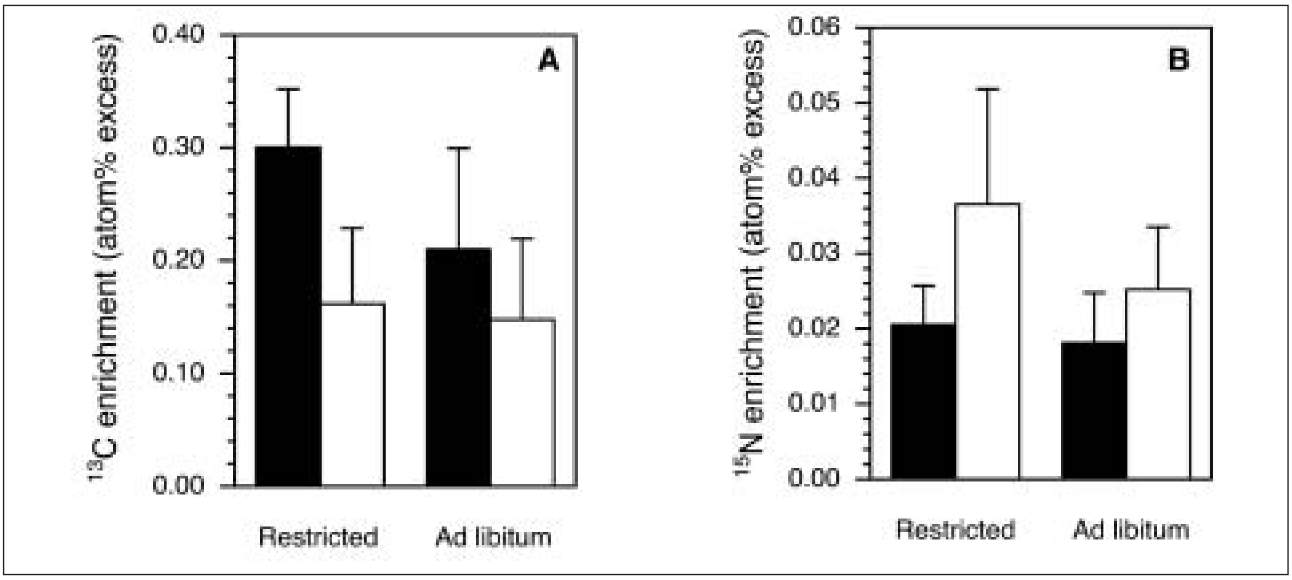


Figure 3.9: Isotopic enrichment in (A) ^{13}C and (B) ^{15}N in whole-body tissue (black columns) and cutaneous mucus (white columns) in slugs on two feeding regimes. Mean ± 1 SD (n=4).

- The decomposition of freshly added plant litter is favoured by, but not dependent on, N availability and the presence of soil. The results indicate that the succession of litter degrading micro-organisms is enhanced by the presence of soil.
- C mineralization in soil is strongly inhibited by anoxic conditions and the onset of alternative reduction processes (NO_3^-, Fe^{3+}, Mn^{3+}, CH_4 formation) can maintain C mineralization at a much reduced level. However, after re-aeration, C mineralization quickly regains the normal level.

Conclusions

An experiment was devised to trace microbial C inputs into soil and soil fauna. It was shown that C fixed from CO_2 by photosynthetic soil micro-organisms is used as a food source by important soil animals (collembola and earthworms). Photosynthetic soil micro-organisms can act as a ‘niche’ route of C input into soil food webs in temperate forests that exist independent of higher plants. Results suggest further that this C input route could be of ecological importance because it represents a very active C pool that is grazed selectively by soil animals which in turn play functional roles in litter decomposition and C dynamics. Further research is required to establish

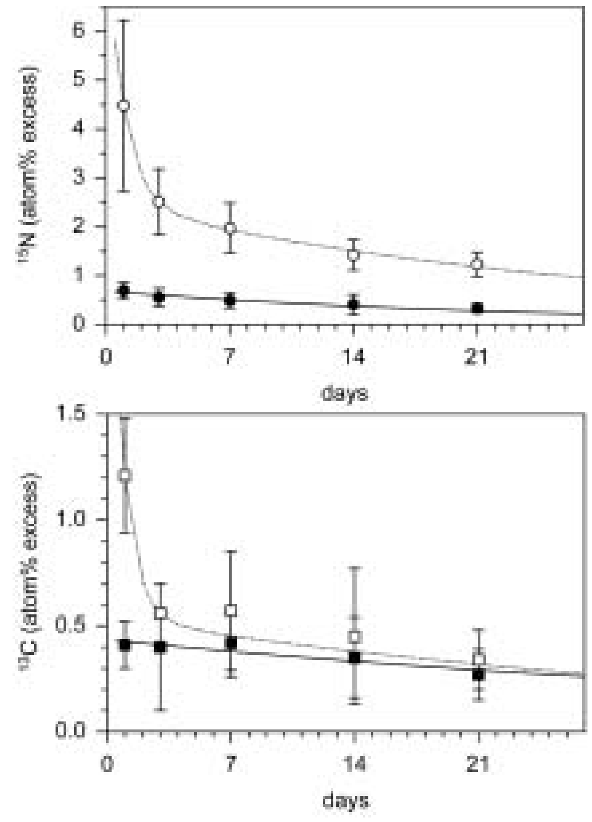


Figure 3.10: Isotopic enrichment and time course of the loss of ^{15}N (upper panel) and ^{13}C (lower panel) from isotopically enriched earthworms: whole-body tissue (closed symbols) and cutaneous mucus (open symbols). Mean ± 1 SD (n=8).

the significance of such microbial C inputs in forest soils, especially in open forest stands or cleared sites.

Novel rapid and cost-effective methods were developed for labelling soil macrofauna with isotopic tracers. These methods can be adapted to devise pulse-labelling or repeated-labelling protocols for the production of large numbers of soil macrofauna species with desired isotopic enrichments for experimental purposes. They provide new tools for research on C dynamics in forest soils.

Chapter 4

BIOMASS EXPANSION FACTORS AND FOREST BIOMASS C STORES

Brian Tobin, Kevin G. Black, Maarten Nieuwenhuis and John Gardiner

INTRODUCTION

There are many uncertainties in estimates of forest biomass C pools, such as the amount of forest biomass (Schroeder et al. 1997, Brown et al. 1999), the appropriate biomass expansion factors (BEFs, Brown 2002), and values for biomass density and carbon fraction (Lowe et al. 2000). Studies on the sink capacity of Irish forests are limited. It is estimated that the average rate of carbon sequestration by Irish forests is approximately 3.36 t C ha^{-1} yr^{-1} (Kilbride et al. 1999), based on the model developed by Dewar and Cannell (1992). However, this estimate of the sink capacity of Irish forests is based on a single biomass expansion factor (BEF) value of 1.3 t t^{-1} for all species, age and yield classes (Kilbride et al. 1999) and ignores the below-ground component. Since the allocation of biomass between different forest components is dependent on stand density, forest species and nutritional status, current national estimates of sink capacity would be improved using classified age and species-specific BEF values that include the below-ground fraction.

In addition to their importance for compiling C inventories in forests, biomass estimations are relevant for studying biogeochemical cycles and understanding variations in structural and functional attributes of forest ecosystems across a wide range of environmental conditions and silvicultural practices. Whilst many ecological studies have used allometric algorithms to predict forest biomass (Cannell 1984, Kauppi et al. 1992, Brown et al. 1999, Schroeder et al. 1997, Nelson et al. 1999), the accuracy of estimating standing biomass through the development of empirical models is reduced due to numerous factors. First, few universal species-specific allometric equations have been developed that are applicable at different regional scales or appropriate for different environmental conditions. Secondly, since most models are parameterised using national inventory data, estimates are generally limited to older age classes. Consequently, the application of models to younger stands can lead to large errors when predicting forest biomass. The prediction of C storage by younger forests, in particular, is important with regard to Article 3.3 of the Kyoto Protocol where potential mitigation of carbon dioxide emissions only includes afforested sites planted since 1990.

In accordance with the IPCC good practice guidance, assessments of the biomass carbon (C) pool in forests can be calculated from national forest inventory timber volume data using biomass expansion factors (BEF) and root-to-shoot ratios (R:S). Generally, BEFs are multiplication factors, which are used to expand merchantable growing stock biomass to account for non-merchantable biomass components (needles, branches, bark, stump, roots; Brown 2002, Schoene, 2002). Or more practically, when used with conversion factors, BEFs and root to shoot ratios (R:S) convert readily available estimates of merchantable stemwood volumes (from forest inventories) to total biomass carbon values that can then be used to estimate carbon budgets (Fukuda et al. 2003). For the purposes of C reporting, however, the IPCC defined basic equation (Penman et al. 2003) is:

$$\text{Carbon mass (Mg C ha-1)} = [V * D * ABEF]*(1+R:S) * CF \tag{2}$$

where V is the merchantable stemwood volume (m^3 ha^{-1}), D is the stemwood basic density (Mg m^{-3}), ABEF is the above-ground biomass expansion factor (t t^{-1}), R:S is the ratio of below-ground to above-ground biomass and CF is the carbon fraction of dry biomass (Mg C Mg^{-1} dry mass)

By definition, ABEF may not be suitable for calculation of C storage in forest ecosystems because of the exclusion of younger trees (dbh<7 cm). To incorporate the younger stands of a chronosequence into current ABEF estimates, ABEF has to be redefined as the ratio of above-ground biomass to growing stem volume (Schroeder et al. 1997). In this study, we propose that below-ground biomass should also be included in the BEF calculations to enable the determination of total biomass from growing stem biomass. This method could potentially result in a smaller error in estimation of C increment because of large variations in R:S as stands develop. We do, however, also report on ABEF, based on timber and total stem values, for comparison purposes.

Because BEF, R:S and D are known to vary with forest type, growing conditions, forest management, crop density, age and climate, there is a need for the determination of conversion coefficients and factors specific to Irish conditions.

OBJECTIVES

The objectives of this Working Group were to assess biomass C stocks in the major Irish forest type (i.e. managed monocultures of Sitka spruce) planted on mineral soils (the predominant soil type afforested since 1990, so-called “Kyoto forests”), and estimate a series of nationally pertinent BEFs and to quantify the changes in BEF over a rotation under normal commercial stand development. This new information can be used to greatly improve current biomass and C-store estimates for Irish forests and to assist in national carbon accounting processes. Specific objectives were:

- Determination of the above- and below-ground biomass of an age sequence of Sitka spruce stands;
- Estimation of basic stemwood density for Sitka spruce;
- Estimation of the carbon content in dry biomass, which is common over all tree compartments;
- Development of allometric functions for total biomass and biomass components, which can be used to estimate C stock changes based on inventory data;
- Estimation of the coarse wood debris C pool due to harvest residue. These include brash, stumps, roots and un-removed timber.

METHODS

A chronosequence, consisting of five even-aged monospecies stands (9, 14, 25, 30 and 45 years old, (see Table 1.1), representing the typical commercial rotation of Sitka spruce in Irish forest conditions was identified (Table 4.1). Stands on mineral soils were chosen, as this best represents the majority of better quality sites supporting post-1990 plantations of Sitka spruce in Ireland. Accordingly, six stands were located in the midlands of Ireland, in Co Laois, each on a wet surface-water mineral gley soil. Efforts were made to standardise the site conditions as far as possible (i.e. stands growing under homogenous environmental and management conditions) in terms of topography, exposure and drainage.

Tree sampling and harvest

A total of 37 trees were harvested from across the chronosequence and destructively sampled. Sample trees were cut at ground level, and their dbh and height were measured. They were divided into their component parts (roots, stem, dead branches and live branches). The point along the stem where it was deemed that live branches predominated (i.e. where >75% of the whorl’s branches were alive/photosynthetically active) was marked, and all branches originating from the stem below this point were considered dead. These components were weighed in the field with a portable spring balance (precision 0.1 kg, see Image 2). To determine moisture content of the tree components sub-samples were taken from the components and dried to constant weight at 70oC, and fresh to dry weight ratios were used to calculate the dry weight of each tree component (Mund et al. 2002). Values in Table 4.2 are displayed as a percentage of the wet weight. Sub-samples of branches were dried and separated into branch-wood and needles, and the subsequent dry weight ratios of needles to branch-wood were used to separate the dry weight of the foliage into its two constituent fractions.

To estimate stem volume, discs were taken at points every 10% of total tree height along the stem beginning at ground level. Tree stem diameters were calculated from two diameter measurements made in perpendicular directions per disc (Drexhage et al. 1999, Mund et al. 2002). Two bark thicknesses per diameter were also measured and averaged in a similar way.

Excavation was carried out within a square of side 2 m centred on the rootstock, and to a depth sufficient to recover all live tree roots with a diameter ≥ 2 mm, both belonging to the sample tree and its neighbours. Roots were cleaned using compressed air and then sub-sampled according to diameter classes to establish wet weight/dry weight ratios (see Image 3). The size of the excavation square was increased to 3 x 3 m in the oldest two stands to account for the lower stocking density. A correction factor was applied at plot level to the excavated root data to ensure the soil area excavated represented the stand tree density.

Basic wood density, normally defined as the ratio of oven dry weight of timber divided by its green volume (Porté et al. 2002), was determined using the stem discs already used for measuring diameters. Volume was measured using the water displacement method, described and used by Olesen (1971), O'Sullivan (1976) and Woodcock and Shrier (2003).

To determine the C fraction (CF), sample material taken from all tree components was oven dried at 70oC and ground using a hammer flail mill (screen size: 1.0 mm. Culatti DFH 48, Glen Creston Ltd, UK) and analysed for %C in a LECO SC-144DR elemental analyser.

Allometric models

Total, above- and below-ground biomass, foliar biomass, stem and timber biomass, were estimated using algorithms based on harvested data and biometric measures such as dbh, top height, crown height and width. The following were used as indicators of goodness of fit for the regression models: r^2 and F-value of the regression, the slope and the standard error of estimate (SEE) of the linear relationship of the regressions between predicted and measured data.

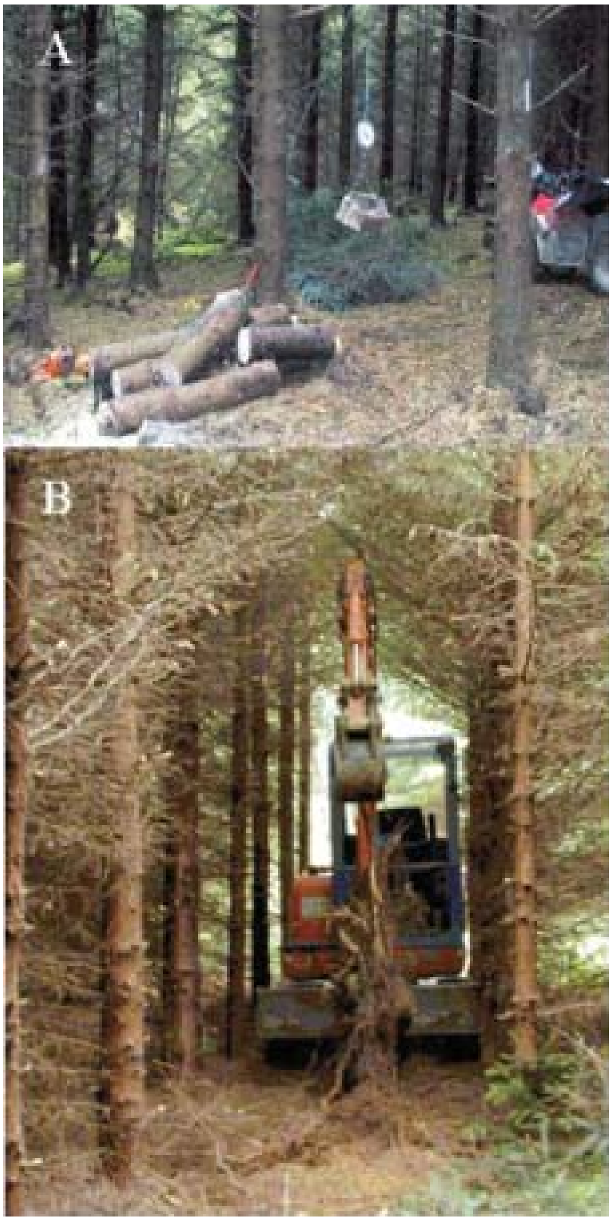


Image 2: Sub-sampling and weighing of stem sections (A) and excavation of roots at the Dooary 15 year old stand (B).

Expansion and conversion factors

A range of age-related BEFs was calculated for each stage in the chronosequence using biomass functions and survey data from 2002.

$$ABEF = \frac{AbovegroundBiomass}{TimberBiomass} \tag{3}$$

$$ABEF = \frac{AbovegroundBiomass}{StemBiomass} \tag{4}$$

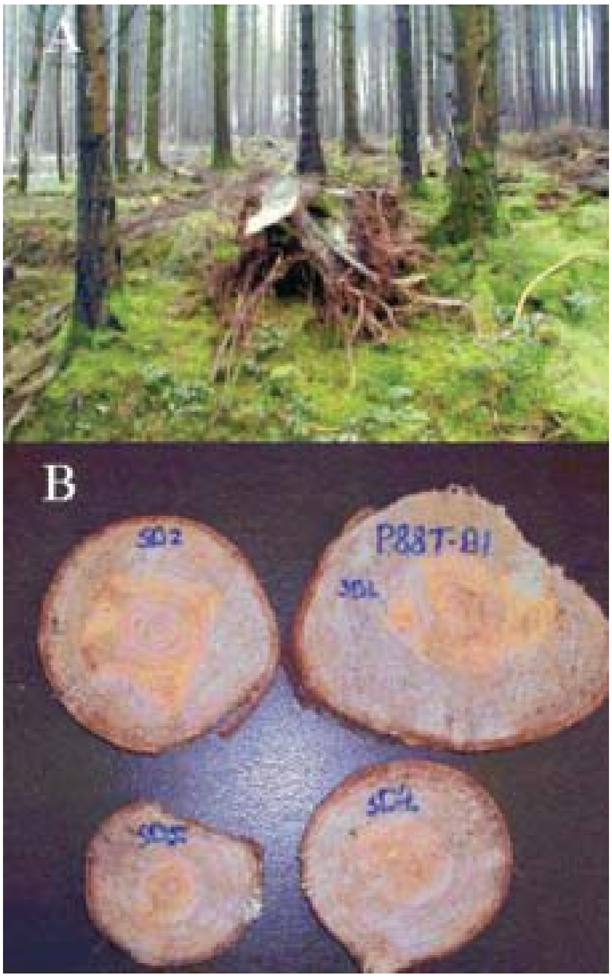


Image 3: Excavated root system from the 31 year old stand (A) and stem discs (B) from trees harvested at the 45, 30, 15 and 10 year old stands.

$$BEF = \frac{TotalBiomass}{TimberBiomass} \tag{5}$$

$$BEF = \frac{TotalBiomass}{StemBiomass} \tag{6}$$

Above-ground biomass included all living biomass attached to the tree from ground level to terminal bud, including bark, live branches and needles. Total biomass included below-ground biomass, including root >2mm diameter. Dead branches still attached to the stem were considered as necromass, and in line with the IPCC reporting process (Penman et al. 2003), were included in the C litter pool inputs.

RESULTS

Carbon and moisture content, and timber density

The fraction with the lowest moisture content was deadwood. The average percentage moisture content across all age classes was 22.47%. The moisture content of all live tissues followed a similar trend over the forest lifecycle.

A large variation in the basic density of stemwood biomass (D) was measured across different sites in this study, reflecting a similarly high variability, as shown in a recent study on the relative differences in density within Sitka spruce provenances in Ireland (Treacy et al. 2000). However, a decreasing trend in D with age was noted across the chronosequence. Some variation may be accounted for by the varying ratio of sapwood/heartwood at different positions in the stem, where discs were sampled. For a nodal area with a large number of branch knots, the overall basic density would have been greater than if the disc had been taken from a more knot-free or internodal area of the stem. The basic density values arising from this study concur well with those found by Ward and Gardiner (1976). The lower part of the chronosequence basic density range (values from sites G25 to C45) was also similar to the UK range of 343-399 kg m^{-3} reported by Savill (1992). When data from the youngest stand are omitted, the overall chronosequence mean became 387 kg m^{-3} which is very close to the value of 370 kg m^{-3} that Ireland has used in a preliminary reporting process (Löwe et al. 2000, Gallagher et al. 2004).

The mean C fraction (CF), was the mean C% of individual tree component samples. This appeared to be quite stable throughout the chronosequence, with no significant changes with tree size, age or productivity (Table 4.2).

The primary reason for higher BEFs in younger crops is variations in the allocation between foliage and timber production (Figure 4.1). In the youngest stand (B9) foliage accounted for 33% of the total biomass, compared to 7% in the oldest stand (C45). In contrast, allocation to timber biomass was higher in older, compared to younger, stands as expected. The proportion of non-timber to timber

Table 4.1: Plot mean moisture content (%) and standard deviation (SD) of different tree components across the chronosequence.

Code	**B9**	**C14**	**D14**	**G25**	**D30**	**C45**
Stand age	**9**	**14**	**14**	**25**	**30**	**45**
Stemwood	48.54	58.90	61.00	54.39	51.62	47.71
SD	*2.87*	*5.35*	*4.98*	*3.89*	*3.02*	*3.43*
Stem bark	45.20	52.77	62.66	57.26	59.00	52.42
SD	*2.75*	*6.19*	*0.96*	*2.58*	*2.56*	*3.75*
Deadwood	0*	27.45	18.72	21.90	16.86	27.43
SD		*3.26*	*3.85*	*5.33*	*1.24*	*8.39*
Roots	51.64	56.69	53.37	53.03	50.89	51.32
SD	*2.88*	*6.18*	*2.14*	*2.07*	*3.2*	*1.96*
Foliage	46.77	55.83	60.80	61.47	58.67	49.67
SD	*2.1*	*3.83*	*3.11*	*1.61*	*3.68*	*3.46*

* There was no deadwood on the 9 year old trees.

stem was higher in the younger crops, reflecting a higher allocation to sapwood to support foliage production in younger stands. The allocation of C into roots was higher in the younger than the older stands. Deadwood (dead branches still attached to the stem), is an important input to the forest ecosystem litter pool, and after the onset of branch senescence (between B9 and D14 second chronosequence stages) this fluctuated between 3 and 12% of total biomass. Bark biomass was found to be more consistent and accounted for about 10% of total biomass at all sites.

Table 4.2: Conversion factors from CARBiFOR chronosequence data. D is expressed in kg m-3 and CF the mean% C of samples of dry biomass from whole trees. Data are expressed as the mean ± the 95% confidence interval.

Code	**Stand Age**	**D**	**CF**
	Yrs	**Kg m^{-3}**	**% C**
B9	9	475.0 ± 17.10	46.0 ± 0.57
C14	14	412.8 ± 53.50	45.9 ± 0.41
D14	14	375.3 ± 40.85	45.5 ± 0.48
G25	25	389.3 ± 43.33	45.1 ± 0.48
D30	30	378.5 ± 57.20	45.8 ± 0.35
C45	45	365.2 ± 19.25	46.5 ± 0.84

Biomass allocation at the tree level

Allometric relationships

Models for biomass displayed a tendency for slight overestimation in the youngest trees, which was mostly overcome by the model predicting total biomass or biomass components for trees <7 cm dbh and >7cm dbh (Table 4.3). All models were tested for systematic bias based on residual analysis of observed and predicted values. These models were used to scale up the tree harvest data to account for variations in the population when estimating stand level BEF or biomass.

Foliar biomass (FB) models were also developed to assess changes in leaf area and model changes in litterfall over the chronosequence. Whilst the relationship between sapwood area and foliar biomass has been suggested to provide the best fit (Turner et al. 2000), sapwood area is not usually measured in traditional forest inventories. Therefore, we used an alternative function based on dbh and additional crown measured characteristics (Tobin et al. in press):

$$\mathrm{FB} = a \times \exp^{((\mathrm{DBH} \times \mathrm{C_o} \times \mathrm{HC}) \times b)} \qquad (7)$$

where a (3.36) and b (0.0043) are specific coefficients, HC is the live crown depth (m) as determined from measured total tree heights and crown ratios (Table 4.1) and $\mathrm{C_o}$ is canopy openness, based on the formula described by (Leech 1984):

$$C_o = \pi \times w^2 \times n/_{10000} \qquad (7a)$$

This calculation of canopy openness is based on crown radius (w) and the number of trees per ha, assuming trees are regularly spaced. The term for w was derived from the linear relationship:

$$w = 0.0456 \times DBH + 0.693 \qquad (7b)$$

which was derived from inventory measurements taken in 2002.

The biomass of foliage litter was estimated using stand FB, calculated using the inventory data and the best-fit FB algorithm, based on an equation described by (Law et al. 2001):

$$LAI = \frac{(M_{lit} \times (1 + F_{abs}) \times SLA)}{F_t} \qquad (8)$$

where LAI is the leaf area index of the stand, M_{lit} is the annual foliage litterfall (t ha^{-1} yr^{-1}), SLA is the half surface specific leaf area of needles, annual foliage turnover rate ($F_t = 0.20$, Norman and Jarvis 1975) and the fractional mass loss on abscission ($F_{abs} = 0.1$, Law et al. 2001). Equation 6 was rearranged to solve for M_{lit}:

$$M_{lit} = FB \times (\frac{n}{10000}) \times \frac{F_t}{1 + F_{abs}} \qquad (9)$$

where n is the number of trees per ha, used to express the formula as a function of FB (measured foliar biomass), rather than LAI. SLA is then not required in this formula (Tobin et al. in press).

Needle litter accounted for 86 to 100% of the total annual litterfall (August 2003 to July 2004, see Chapter 3), for all of the sites investigated. The total litterfall in the younger Dooary site was 6.71 t ha-1 yr-1 in 2002/3, compared to 7.27 t ha-1 yr-1 in 2003/4 (data not shown). There was no general agreement between measured annual needle litterfall and predicted needle litterfall, based on FB estimates and a needle turnover rate of 20% from Norman and Jarvis (1975). However, when the needle turnover was reduced from 5 years to 2.5, and the corresponding turnover rate increased from 20 to 40%, the predicted needle litterfall agreed very closely with the measured values.

Expansion and conversion factors

The BEFs, based on total and timber biomass, varied from 44 t t^{-1}, in the youngest, to 1.63 t t^{-1} in oldest stand (Table 4.4; Figure 4.2). The timber-based BEF values for the younger stands decreased by 2 to 3 fold over a two year period (Figure 4.2).

According to the IPCC definition, the timber-based BEFs are not suitable for the calculation of C storage in forests since they cannot account for younger stands below the size of merchantable timber (Black et al. 2004). This problem can be overcome when stem-based BEFs are used (Table 4.4; Figure 4.2) and there is a smaller variation in younger stands.

Generally, stand BEF decreased with age and was constant after ~ 30 yrs, resembling the trends reported by Lehtonen et al. (2004), Levy et al. (2004) and Fukuda et al. (2003). This trend was also seen in the decreasing ratio of below-ground/above-ground (R:S), similar to that found by Helmisaari et al. (2002).

The advantage of estimating BEF based on basal area rather than age (Figure 4.2), is that biometric based estimates do, theoretically, account for differences in tree development (Black et al. 2004). For example, BEF values calculated for 14 year old stands were different for yield classes 18- 20 and 20-24 (Figure 4.2), due to the fact that the higher yield class stand may be developmentally older than a stand with a lower yield class. Therefore, if the allometric relationship covers the biomass range for all stands in a chronosequence, the prediction of biomass or BEF would not be influenced by the yield class, but by the characteristics that underlie tree allometry.

Biomass C stocks

The total stand biomass was calculated using the developed allometric equations and survey data collected in 2002 (Table 4.3). Average total stand dry biomass ranged from 22.8 t ha^{-1} on the Baunoge site (mean dbh 6 cm) to 231.4 t ha^{-1} at Cullenagh (mean dbh 32 cm) (Figure 4.3A). Tree biomass increased at a higher rate after the first thinning cycle, when compared to stand-based estimates. This was associated with thinning events

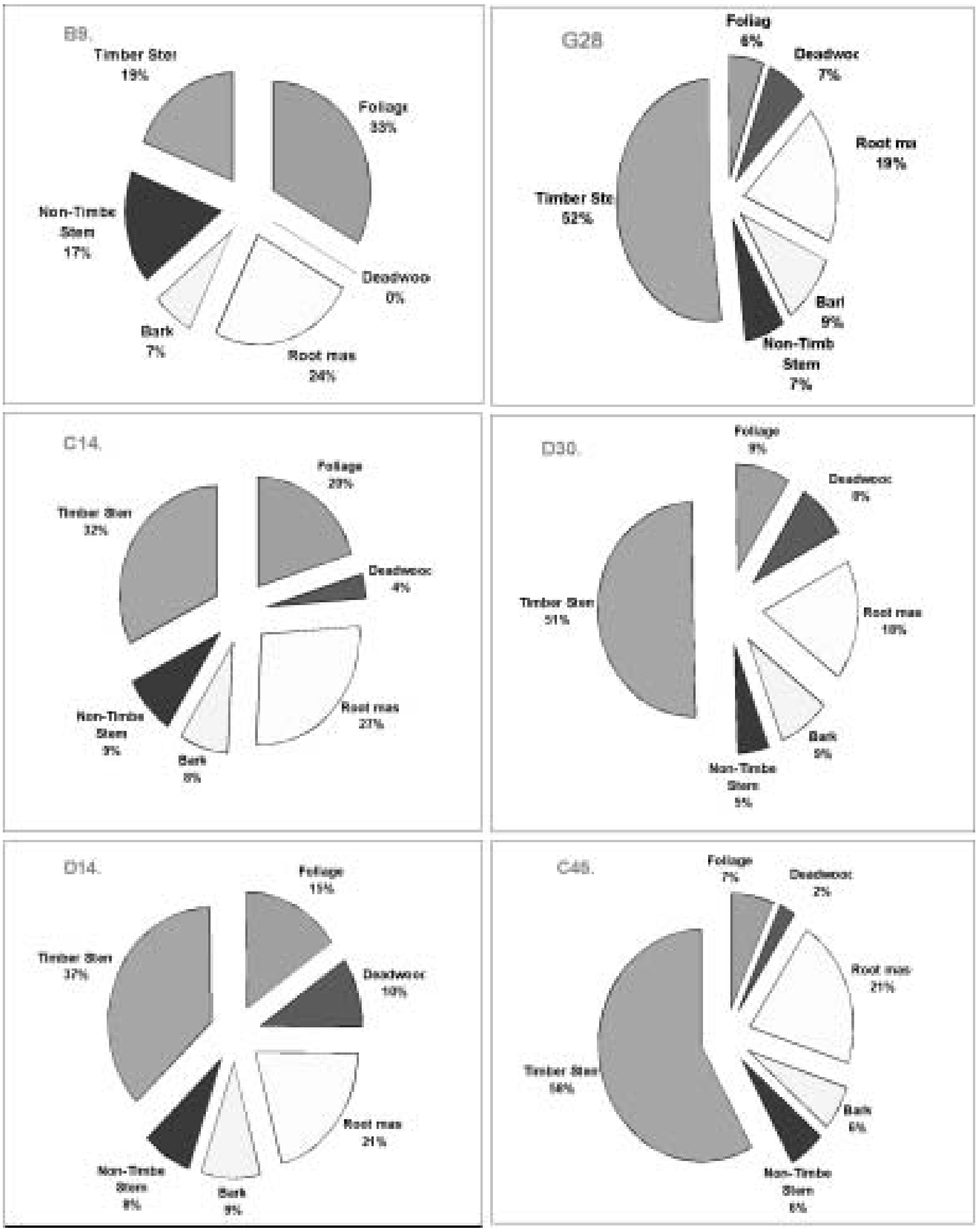


Figure 4.1: Age related changes in carbon allocation into different biomass components.

in older crops. Therefore, current annual increment in biomass for the different stands was calculated using the tree growth curve function (Figure 4.3B) and converted to stand increment using stocking densities (Figure 4.4).

Biomass increment

Biomass increment is the largest component of net primary production (NPP) and can be calculated using annual timber volume increment and BEFs. However, this technique would only be only applicable to stands where there is a measurable increment in timber volume (i.e. in older stands). Biomass functions were used to calculate the change in biomass over two years, based on surveys of the chronosequence in 2002 and 2004. The calculated current annual increment (CAI) was highest at canopy closure followed by a decline after the first thinning cycle (Figure 4.4). There was generally a good agreement between calculated biomass CAI and measured increment,

Table 4.3: Coefficients for various biomass and component biomass algorithms derived from harvested tree data. Correlation coefficients (r^2), the slope (a) of the linear relationship and the unsigned standard error of estimate (%SEE) were obtained from linear regression between predicted and observed values for all dbh distributions.

Parameter	**Algorithm**	**Coefficient value**	**Slope α**	r^2	**%SEE**
Total biomass <7 cm dbh	$a \times (dbh \times h)^b$	a = 0.138 b = 1.54	0.99	0.98	10.2
>7 cm dbh	$a \times (dbh \times h)^b$	a = 0.234 b = 1.177			
Above-ground biomass <7 cm dbh	$a \times (dbh \times h)^b$	a = 0.119 b = 1.47	0.98	0.99	8.0
>7 cm dbh	$a \times (dbh \times h)^b$	a = 0.120 b = 1.23			
Stem biomass <7 cm dbh	$a \times (ba \times h)^b$	a = 89.26 b = 0.691	1.02	0.97	16.3
>7 cm dbh	$a \times (ba \times h)^b$	a = 191.8 b = 0.973			
Timber biomass <7 cm dbh	$a \times (ba \times h)^b$	a = 4.28 b = 0.61	0.89	0.91	20.3
>7 cm dbh	$a \times (ba \times h)^b$	a = 173.1 b = 1.17			

All coefficients and r^2 were significant at $P < 0.05$.

Table 4.4: Age related changes in above-ground (ABEF) and total biomass (BEF) biomass expansion factors and root to shoot ratios (R) across the chronosequence. Values represent a mean and standard error.

Forest/Code	**BEF**		**ABEF**		**R:S**
	Timber	**Stem**	**Timber**	**Stem**	
B9	44.3 (7.5)	3.4 (0.2)	14.9 (2.9)	2.6 (0.3)	0.32 (0.008)
C14	8.9 (2.4)	2.5 (0.3)	3.8 (1.2)	1.8 (0.2)	0.31 (0.007)
D14	5.6 (1.8)	2.3 (0.05)	1.8 (0.3)	1.7 (0.08)	0.28 (0.014)
G25	2.4 (0.2)	1.8 (0.04)	1.7 (0.1)	1.4 (0.06)	0.25 (0.01)
D30	2.05 (0.08)	1.75 (0.05)	1.5 (0.05)	1.35 (0.05)	0.24 (0.01)
C45	1.63 (0.06)	1.7 (0.06)	1.1 (0.05)	1.25 (0.06)	0.22 (0.03)

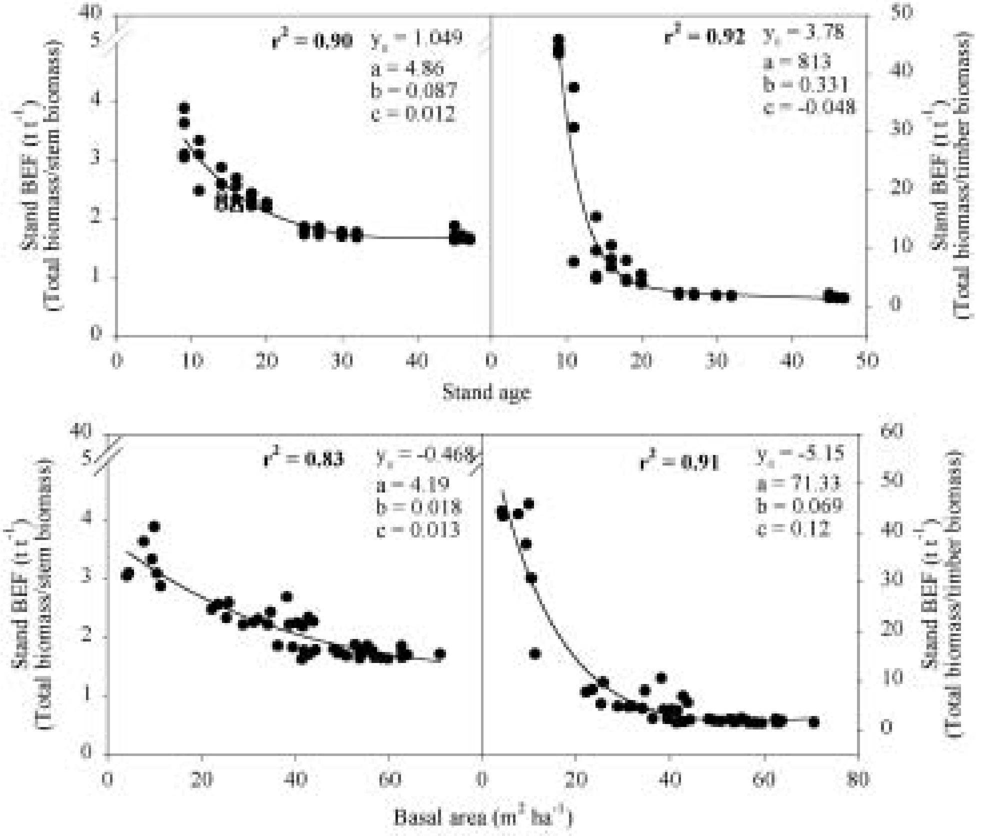


Figure 4.2: The relationships between stand age (normalised to a YC22) and timber or stem-based BEFs (Top panels). The bottom panel shows the relationship between BEF and basal area. The curves line interpolated through the symbols is based on an exponential decay function y = yo + a x exp$^{(-bx)}$ + *cx*. The white triangle symbols (top left panel) represent the stand biomass for the main Dooary stand (D14), which was rescaled to 16 yrs old to normalise the data for differences in yield class.

suggesting that this method can be used to estimate biomass changes over time. The interpolation of the data over the chronosequence was complicated by the different YC over the chronosequence. Although this could be overcome by normalising the age class to a YC 22, the biomass increment method can lead to a under or over estimation of increment as evident from the differences in the predicted and measured biomass increment in the 14 year old stand (Figure 4.4).

Coarse woody debris and harvest residue pools

The coarse woody C pool, which represents timber and dead plant material residues left on site after harvesting or thinning operations, was only assessed in thinned sites. This preliminary investigation suggests that brash, remaining stump and root and un-removed timber C residues can vary between 5 and 27 t C ha^{-1}depending on stand age, the number of thinning cycles and thinning intensity (Table 4.5). The analysis also suggests that the additional decomposition of this harvest residue could lead to a loss of C equivalent to 0.08 to 1 t C ha^{-1} yr^{-1}.

CONCLUSIONS

The chronosequence approach in identifying a time series of forest sites of similar conditions has been a useful tool in studying the biomass C sequestration and storage dynamics of Irish forests. However, inherent differences in YC over the chronosequence hinder the interpolation of the data, which can lead to errors in the estimation of

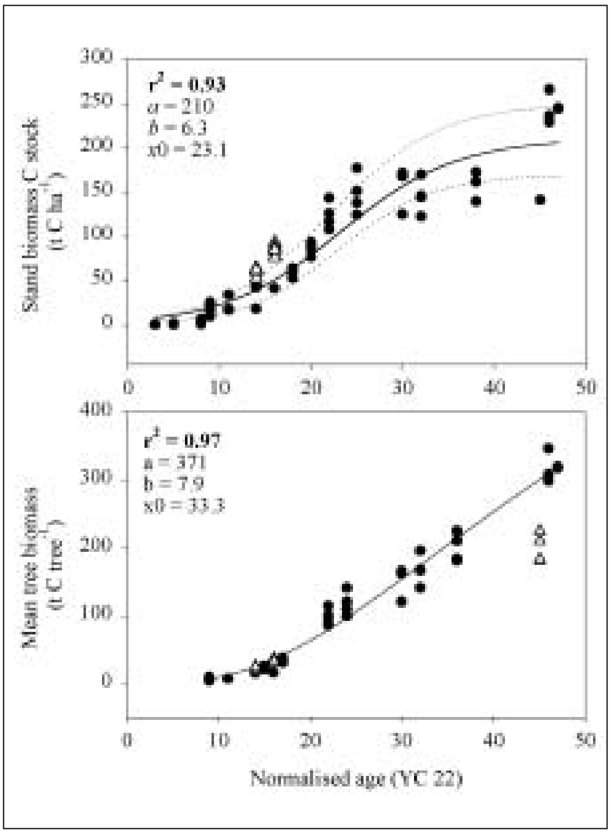


Figure 4.3: Stand (A) and mean tree (B) biomass stocks across the chronosequence. Error bars express the 95% confidence intervals. The white triangle symbols represent the stand biomass for the main Dooary stand (D14), which was rescaled to 16 yrs old to normalise the data for differences in yield class, based on the sigmoidal function y = a/(1+exp(-(x-x0)/b)). Additional biomass data from a related study in Mayo, including younger Sitka spruce trees, and values from the Irish literature were included in this analysis to cover the complete age class range.

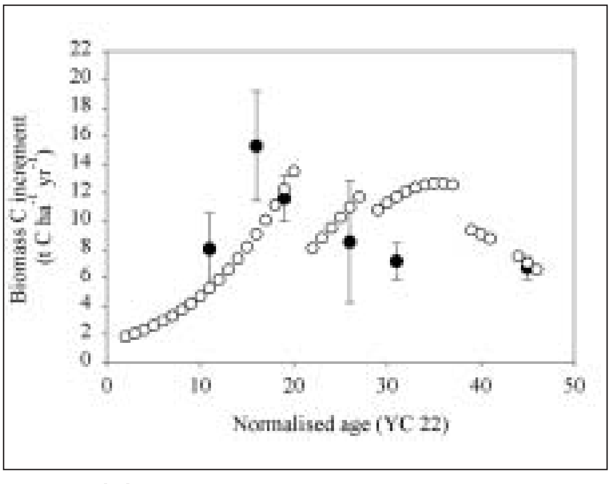


Figure 4.4: Predicted annual biomass increment (white symbols, based on the sigmoidal function in Figure 4.3b, and measured (black symbols) biomass increment across the chronosequence. Age class was rescaled to represent YC22 across the chronosequence.

age related changes in C stock and CAI in biomass.

The BEF range calculated for the CARBiFOR chronosequence (Black et al. 2004.) clearly demonstrates the value of the use of age-defined classes in C-store estimation. The higher BEF values in younger stands reflect large amounts of non-timber biomass. A shortcoming of the BEF range calculated here is that it does not include understorey or surrounding vegetation.

As expected, the IPCC defined BEFs, based on minimum diameter timber volumes, proved impractical for use in forests of sub-minimum dbh dimensions, where variation and uncertainty was very high. Stemwood basic density (D) displayed a high degree of variation but once the value from the youngest site was excluded, use of the value used by Löwe et al. (2000) was validated. The carbon fraction (CF) varied between tree component parts but when expressed on a whole tree basis was not significantly different from the IPCC default value of 50% (Mathews 1993, Penman et al. 2003).

Although the species-specific allometric equations developed here performed well for the chronosequence as a whole, more age/developmental stage specific equations for younger forests are required. The results from this project meet the objectives set out and add greatly to the information available for Sitka spruce, although, as confidence intervals were quite large, further data will strengthen the results. However, all the data presented are species-specific, and pertain to crops of YC22 on wet mineral soils. More research is needed to establish more fully the extent of changes due to YC. There are little available data for any other coniferous species, and no published data for biomass distributions in semi-mature and mature broadleaf crops, so these areas are obvious candidates for further research.

Table 4.5: Harvest residue stock and losses (t C ha^{-1}) in thinned stands. The decomposition of the total harvest residue was calculated using a mass loss decomposition function for Engelman spruce (Laiho and Prescott 1999). All C stock and decomposition data represents cumulative value since the first thinning cycle. Refer to Table 1.1 for thinning dates.

C Pools	**Age/Code**		
	G25	**D30**	**C45**
Brash, stump and root	4.39 ± 2.6	25.3 ± 4.3	25.1 ± 9.1
Coarse woody debris (>7cm)	0.52 ± 0.31	1.38 ± 0.33	1.46 ± 0.65
Total stock	**4.91± 2.98**	**26.72 ± 3.88**	**26.79 ± 4.95**
C losses			
Harvest residue decomposition	**0.71 ± 0.32**	**15.21 ± 0.7**	**23.17 ± 3.73**

Chapter 5

ECOSYSTEM PROCESSES

Kevin G. Black, Brian Tobin and Bruce Osborne

INTRODUCTION

The net exchange of carbon (C) or sequestration by a forest ecosystem over a given period of time is termed net ecosystem productivity (NEP). NEP captures a variety of processes and feedbacks associated with C cycling between the atmosphere, vegetation and soil pools (Figure 5.1).

Eddy covariance is a micro-meteorological technique that directly measures the exchange of CO_2 or net ecosystem productivity (NEP) between the atmosphere and a forest canopy. A major drawback of this technique is an inability to cover an entire age sequence or replicated stands on different soil types, because of the large expense, lack of suitable topography and intensive maintenance requirements. However, detailed analyses at selected sites do provide insight into how ecosystem processes, such as canopy photosynthesis (GPP), may respond to climatic or anthropogenically-induced changes. The provision of long term eddy covariance data also facilitates the development of comprehensive carbon balance models that can be used to simulate the effects of stand age, disturbance and climate change scenarios on forest carbon fluxes. This aspect of the project formed a large part of the work package and is described in detail here.

Despite advances in micro-meteorological technology, national C accounting methodologies for reporting to the International Panel on Climate Change and the Kyoto Protocol rely on forest inventory data and biometric approaches. However, these require independent validation. Theoretically, eddy covariance and inventory-based NEP measures (i.e. NPP - heterotrophic respiration) are independent assessments of annual carbon sequestration rates with unrelated errors. Therefore, such intercomparisons provide information on the causes of interannual and site-specific variations in NEP (Curtis et al. 2002, Falge et al. 2002). Flux-based measurements also provide information on emission factors, which cannot be detected when conventional inventory methods are used. In this section, we also report on a comparison of inventory and eddy covariance

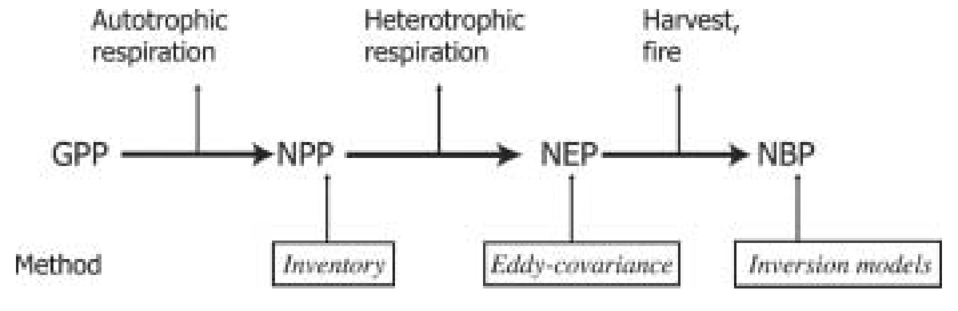


Figure 5.1: The relationship between different components of ecosystem exchange and associated observation methods (from Ehman et al. 2002). Gross primary productivity (GPP) is the C pool taken up during photosynthesis. Net primary productivity (NPP) is the biomass increment and other lost biomass pools, such as herbivory. Net ecosystem productivity (NEP) represents the net uptake (positive) or loss (negative value) for the entire ecosystem. Net biome productivity (NBP) represents the total uptake or loss of C over for a particular land-use type, including flux changes associated with harvest, thinning or fires.

procedures for assessing NEP of a closed canopy Sitka spruce stand over a two year period (Black et al. 2005) and analysis of the sources of uncertainties and errors associated with the different approaches.

OBJECTIVES

Since the eddy covariance component of the project was limited to one site only, we concentrated of three major themes:

- Assessments of annual and interannual variations in NEP of a closed canopy forest, prior to the first thinning.
- Cross-validation with inventory-based assessments for reporting to the UNFCCC and Kyoto Protocol.
- Characterisation of component fluxes and development of ecosystem models to predict changes in NEP over the entire chronosequence.

METHODS

Study site

The site used was a 14 year old Sitka spruce forest growing on a wet mineral soil located in Co Laois in the Irish midlands (ca. 52º 57’ N, 7º 15’ W, altitude of 260 m, site D14 in Table 1.1). The 30 year mean annual temperature for this area was 9.3ºC, with a mean rainfall of 850 mm. The site was previously a semi-managed grassland, now planted at a density of ca. 2500 stem ha^{-1}, with a current leaf area index of 8.7 m^2 m^{-2} (Black et al. 2004). This site was selected for numerous technical and policy related reasons:

- The area (>30 ha) and flat topography of the site was ideal for eddy covariance measurements.
- In accordance with article 3.3 of the Kyoto Protocol, which relate to afforested stands planted since 1990-2008, it was more suitable to investigate the sequestration rates in stands less than 18 years old.
- In terms of carbon accounting in Irish forests, age classes between 12 and 20 years old would represent the higher proportion of the total area covered by Sitka spruce.
- According to research in Scotland it was suggested that Sitka spruce annual sequestration rates do not increase significantly after 14 years.

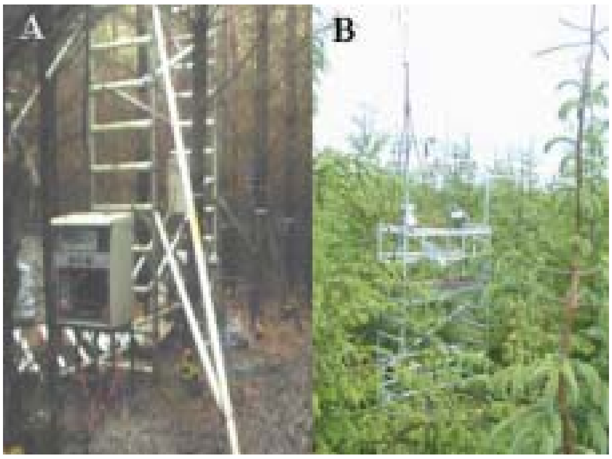


Image 4: A view from the bottom (A) and top (B) of the canopy showing the eddy covariance and other meteorological equipment at the Dooary site.

Meteorological measurements

Eddy covariance measurements of NEP were made from February 2002 to February 2005 using the EdiSol system described in detail by Moncrieff et al. (1997). Fluxes of sensible heat, water vapour, CO$_2$ and friction velocity (u-star) were calculated for 30 minute periods using the EdiSol software (Moncrieff et al. 1997). For detailed information on eddy covariance methodology, gap-filling procedures, footprint and uncertainty analysis refer to Black et al. (2005).

Annual surveys

The estimation of the change in living biomass (ΔB) was based on the relationships between tree biomass, stem diameter at 1.3 m (dbh, cm) and height (h, m), using an annual survey conducted in four 100 m^2 plots in the vicinity of the flux tower. Band dendrometers were installed to measure annual changes in the circumference of all of the trees in the sample plots. Tree height was estimated to the closest cm, in February 2002, 2003 and 2004, using a laser hypsometer (Laser Technology Inc., Colorado, USA).

Development of biomass models and estimation of NPP

Biomass, dbh, and tree height data were used to derive total living and dead above-ground and below-ground biomass (phytomass and necromass, Equation 10) and total phytomass functions (Equation 11) for individual trees across the entire chronosequence (Black et al. 2005).

Total dead and living biomass = 0.1403 (dbh × h)$^{1.23}$ ($r^2 = 0.97$) (10)

Total living biomass = 0.2667 (dbh × h)1.13 ($r^2 = 0.98$) (11)

The change in living biomass (ΔB) was based on the relationship between dbh, height and total living biomass only (i.e. phytomass, Equation 11), to account for potential changes in standing dead biomass (ΔAGD) over time. The total and living biomass models were validated against an independent Sitka spruce data set (Black et al. 2004 and biomass data presented in Chapter 4) to assess the model error associated with the estimation of stand biomass. The % standard error of estimate (SEE) of the regression line (measured versus predicted) was used to estimate the uncertainty range associated with the biomass estimates (Black et al. 2004, 2005; see Tables 5.1 and 5.2).

Above-ground litterfall (D_a) was collected every month over the two year period from ten 25-litre plastic buckets, randomly located within one 30 x 30 m plot. Litter was separated into green leaf, dead leaf, bud scale and woody components prior to oven drying to a constant mass. The green leaf and bud scale components of litterfall (D_a) were used in the NPP estimates (see Tab1e 5.1).

Below-ground detritus production (D_b) was determined as the product of fine root biomass and the annual fine root turnover ratio (see Chapter 2). The change in annual soil and associated litter C (ΔC_{soil}) content was estimated as the mean difference in soil C content of the non-unforested grassland (time zero) and 15 year old (flux tower site) sites in the selected chronosequence (see Chapter 3). A total of 15 soil cores (10 cm diameter with a penetration depth of 30 cm) were collected over an area of 900 m^2 during the 2003/4 season for C and N analysis using a C-N analyser (Leco CSN-1000, Leco Corp., St Joseph, MI, USA).

Uncertainty analysis for inventory estimates

Potential uncertainties associated with the sampling, model and measurement errors of the inventory procedures were examined. Sampling errors, in this case, represent the variability in the estimate due to measuring a subset of the population. Model errors, such as those associated with allometric functions or respiration models, were estimated using %SEE from regression analysis of observed and predicted values (see Table 5.2). The only measurement errors assessed were the tolerances for dbh, using dendrometers (1 mm), and height (0.05 m). The total standard errors for the inventory methods (σNEP_{eco} or $\sigma NEP_{\Delta C}$) were estimated as:

$$\sigma^2\ NEP^{eco} = \sigma^2\ \Delta B + \sigma^2 D_a + \sigma^2 D_b + \sigma^2 R_{H\ (soil)} + \sigma^2 R_{H\ (AGD)} \qquad (12)$$

or

$$\sigma^2\ NEP_{\Delta C} = \sigma^2 \Delta C_b + \sigma^2 \Delta C_{dead} + \sigma^2 \Delta C_{soil} \qquad (13)$$

Ecosystem models

A carbon, energy, nitrogen, water (CENW) model was used to estimate NEP over the entire chronosequence, based on parameters derived from the survey, harvest and physiological experiments. The model was validated against biomass, volume and eddy covariance data collected between 2002 and 2004.

RESULTS

Eddy covariance measurements and quality control

The site provided a fetch (representative area of measurement) of 590 m in the SW direction and 1420 m in the S direction, which were the dominant wind directions over the 2002 to 2005 measurement period. The fetch was 250 m in the N

Table 5.1: An intercomparison of inventory (NEP_{eco}, $NEP_{\Delta C}$) and eddy covariance-based estimates (NEP) of net ecosystem productivity in a Sitka spruce forest for 2002/3 and 2003/4. Values represent a mean and range, based on the SE at 95% confidence interval for individual components and different C sequestration estimates. (Taken directly from Black et al. 2005)

Carbon store	**Symbol or formula**	**Mean and range for 95% confidence limits (t C ha^{-1} yr^{-1})**	
		2002/3	**2003/4**
Living biomass increment	ΔB	10.51 (8.91 – 12.11)	10.30 (8.1 – 12.5)
Dead branch biomass increment	ΔAGD	1.37 (1.29 – 1.44)	1.71 (1.62 – 1.83)
Litterfall, green shoots and bud scales	D_a	0.19 (0.17 – 0.21)	0.22 (0.15 – 0.29)
Net fine root production[a]	D_b	0.43 (0.35 – 0.51)	0.43 (0.35 – 0.51)
Herbivory	H	nd	nd
Volatile organic compound emissions	VOC	nd	nd
NPP	$\Delta B + D_a + D_b + H + VOC + \Delta AGD$	12.49 (10.89 – 14.10)	12.65 (10.37 – 14.92)
Heterotrophic soil respiration	$R_{H\ (soil)}$	2.85 (2.55 – 3.14)	3.18 (2.56 – 3.80)
Heterotrophic wood debris respiration	$R_{H\ (AGD)}$	0.09 (0.08 – 0.10)	0.09 (0.08 – 0.11)
Herbivore respiration	$R_{H\ (consumer)}$	nd	nd
Total heterotrophic respiration[b]	R_N	2.94 (2.62 – 3.32)	3.27 (2.65 – 3.89)
Total living biomass increment	$\Delta C_b = \Delta B$	10.51 (8.91 – 12.11)	
Dead branch increment	$\Delta C_{dead} = \Delta AGD$	1.37 (1.29 – 1.44)	
Soil C change [c]	ΔC_{soil}	1.15 (0.70)	
Ecological inventory	$NEP_{eco} = NPP - R_h$	9.52 (7.89 – 11.16)	9.38 (7.30 – 11.44))
Stock change inventory[d]	$NEP_{\Delta C} = \Delta C_b + \Delta C_{dead} + \Delta C_{soil}$	13.02 (11.30 – 14.75)	nd
Eddy covariance	NEP	8.90 (8.39 – 9.44)	8.31 (7.69 – 8.91)
NEP_{eco} to -NEE ratio	NEP_{eco} /NEP	1.07	1.13
$NEP_{\Delta C}$ to -NEE ratio	$NEP_{\Delta C}$/ NEP	1.46	

[a] Fine root production (total – dead fine roots) was determined for 1 year only (2003/4).

[b] Heterotrophic estimates were based on validated models

[c] Soil C stock changes were based on a chronosequence comparison

[d] When fine root C increment was included in the ΔC_b estimate, $NEP_{\Delta C}$ = 13.46 (± 1.79) t C ha^{-1} yr^{-1}

and 189 m in the E direction, but these wind directions were less frequent. Based on the footprint analysis, a total of 1.2% of all the half hourly data were discarded when more than 10% of the measured flux came from outside the experimental site area. A total of 10% of the half-hourly flux data for 2002/3 were discarded due to unsuitable atmospheric conditions with another 16% of the data missing due to instrument and/or power failure. In 2003/4, slightly less data were discarded (8.5%), but there were more missing data (21%), primarily due to an 18-day gap caused by instrument failure.

Artificial gaps in the remaining, measured, data set were created to evaluate the performance of the mean diurnal variation (MDV) and non-linear regression gap filling procedures. The mean percentage absolute error for the combined gap-filled data over the two year period was 9.8%. The annual absolute error associated with the gap filling procedures was considerably less because of the

Table 5.2: Summary of the sources of error and uncertainty for NEPeco estimates during 2002/3 (taken from Black et al. 2005).

Parameter	Source of error	Unit	Mean	Absolute SE	% SE or SEE[a]	% Cum. error[b]	% Total cum. error
	Model	kg tree^{-1}	8.272		7.97		
	Carbon content	g g^{-1}	0.490	0.004	0.87		
	Tree increment	kg tree^{-1} yr^{-1}	8.771	1.050	12.00		
	Planting density	tree ha^{-1}	2 445	117	4.81		
ΔB		t C ha^{-1} yr^{-1}	10.511	1.599[c]		15.22	
	Model	kg tree^{-1}	1.149		2.54		
	Carbon content	g g^{-1}	0.490	0.004	0.87		
	ADW increment	kg tree^{-1} yr^{-1}	1.145	0.011	0.98		
	Planting density	tree ha^{-1}	2445	117	4.81		
ΔAGD		t C ha^{-1} yr^{-1}	1.371	0.0767[c]		5.60	
	Carbon content	g g^{-1}	0.480	0.002	0.57		
	Needles per m^2	kg tree^{-1} yr^{-1}	0.039	0.004	10.31		
D_a		t C ha^{-1} yr^{-1}	0.191	0.019[c]		10.46	
	Carbon content	g g^{-1}	0.485	0.003	0.73		
	Roots per m^2	kg tree^{-1} yr^{-1}	0.089	0.017	18.56		
D_b		t C ha^{-1} yr^{-1}	0.432	0.080[c]		18.57	
NPP = ΔB+ΔAGD+D_a+D_b		t C ha^{-1} yr^{-1}	12.494	1.602[d]			12.83
$R_{H\ (soil)}$	Model	t C ha^{-1} yr^{-1}	2.855	0.294	10.30		
$R_{H\ AGD}$	Model	t C ha^{-1} yr^{-1}	0.094	0.005	5.12		
	AGD	t C ha^{-1} yr^{-1}	4.650	0.099	2.15		
R_{het}		t C ha^{-1} yr^{-1}	2.973	0.347[c]		11.7	
NEP_{eco} = NPP- R_{het}		t C ha^{-1} yr^{-1}	9.52	1.630[d]			16.39

[a] The standard error (SE) was used for measured data and the standard error of estimate (SEE) for the modelled data;

[b] The cumulative error was determined using squared the sum of the %SE or SEE, where %cum. error = sqrt(n_1^2 + n_2^2 …n_x) ;

[c] The absolute cumulative S.E for each parameter = mean x (% cum. error/100);

[d] The cumulative error for NPP, R_{het} and NEP_{eco} was determined directly from the squared sum of the SE of component parameters.

low frequency of missing data over the whole period.

The overall error of uncertainty was calculated for each measured or gap-filled half-hourly flux, using either an absolute gap filling model standard error of estimate (SEE), or the SEE of the long term energy balance regression, expressed as a percentage. The mean annual errors of the NEP estimate were ca. 7% of the cumulative flux over the periods 2002 to 2005.

Annual fluxes

Seasonal variations in NEP, GPP and total ecosystem respiration were associated with changes in temperature and light levels (Figures 5.2 and 5.3). The decline in the cumulative NEP during the winter months was associated with a net daily loss of C (Figure 5.2), primarily due to longer nights and a lower daily insolation (Figure 5.3). Whilst there was a loss of C from the system over the winter months, there was still a significant uptake of C during the day, suggesting that photosynthesis occurred throughout the year.

The net ecosystem productivity (NEP) or net annual uptake of C for the stand varied from 8.9 t C ha^{-1} yr^{-1}, in 2002/3, to 7.1 t C ha^{-1} yr^{-1}, and 2004/5 (Figure 5.2). The widespread drought in Europe, between August and October 2003, was evident at the site in Dooary, where there was a 18% reduction in rainfall, compared to the 30 year mean values for the same period. The slight reduction in annual NEP in 2003, compared to 2002, may be associated with the lower rainfall over this period. However, the general decline in NEP from 2002/3 to 2004/5 (8.9 to 7.1 t C ha^{-1} y^{-}

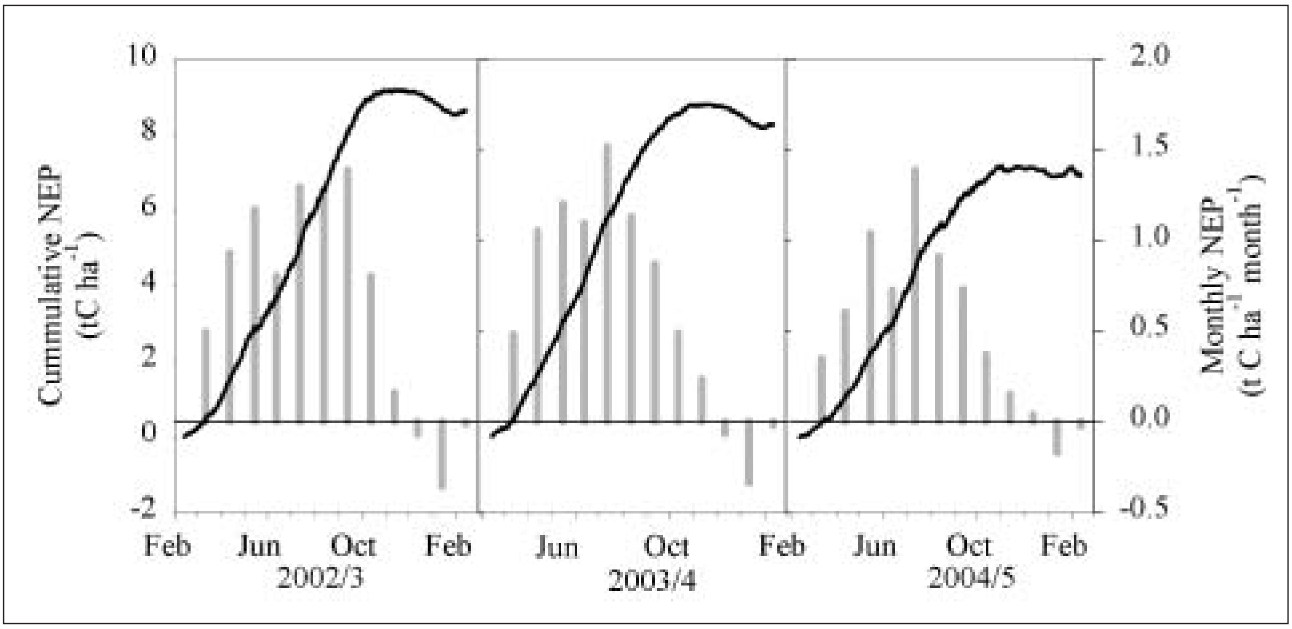


Figure 5.2: Cumulative annual (solid line) and monthly (histogram) net ecosystem productivity (NEP) of C between the atmosphere and a Sitka spruce canopy from February 2002 to February 2005. A positive NEP value represents an uptake of C and negative values represent a loss of C from the ecosystem. (Black et al. in prep.)

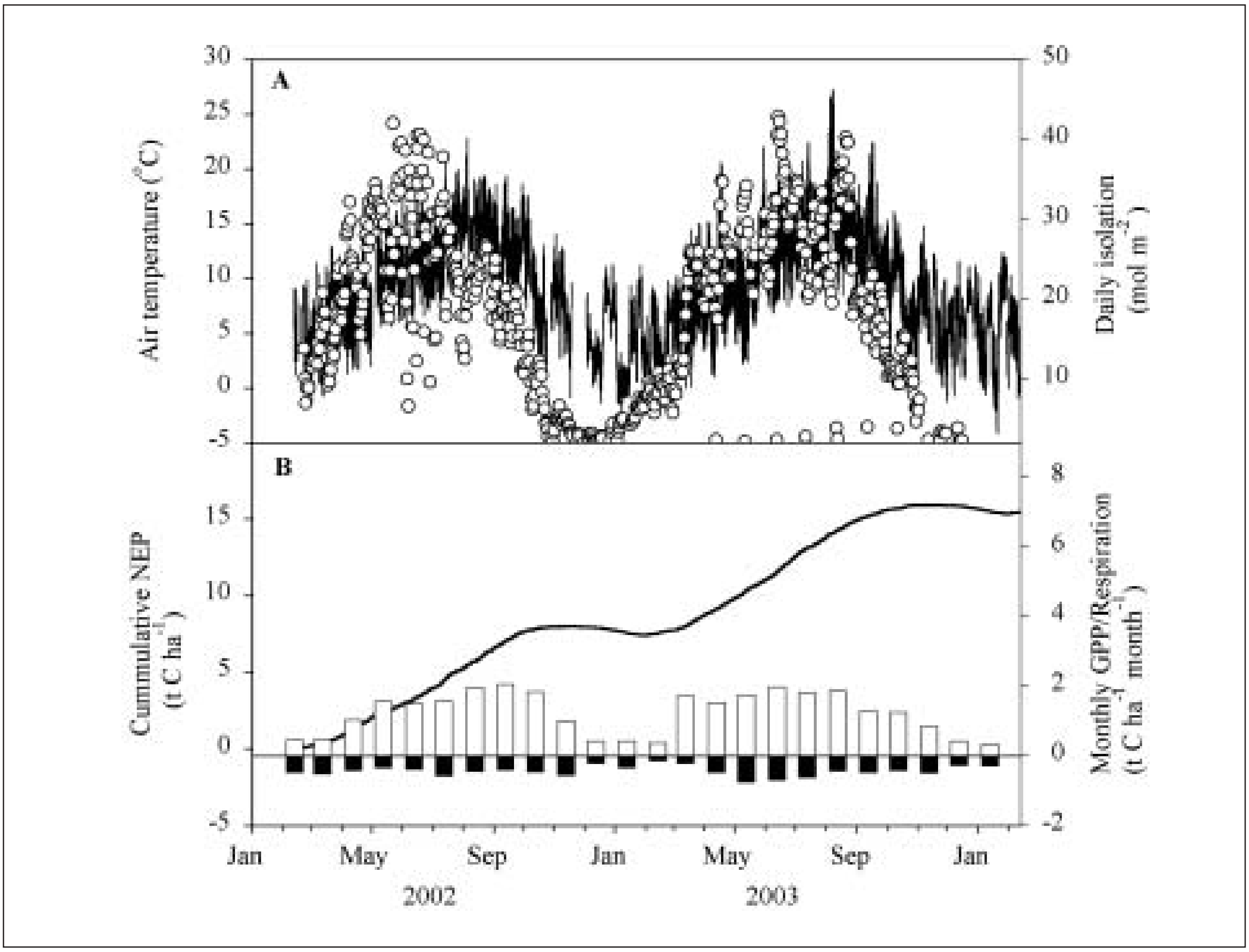


Figure 5.3: Annual and seasonal variations in air temperature (solid black line, a) and daily insolation (white symbol, a), cumulative NEP (solid line, b), monthly GPP (white histograms, b) and monthly ecosystem respiration (black histograms, b) (Black et al. in prep.).

^{1}), may be associated with canopy closure and the onset of needle senescence at the bottom of the canopy, resulting in less light capture and a reduction in GPP. The concomitant increased needle litterfall in 2003 and 2004 (see Chapter 4) may have also resulted in a higher heterotrophic respiratory flux and a reduction in annual NEP after canopy closure.

Eddy covariance estimates are higher than previously reported NEP ranges (-2 to 7.9 t C ha^{-1} yr^{-1}) for FLUXNET (Baldocchi et al. 2001, Falge et al. 2002) and EUROFLUX (Aubinet et al. 2002, Kolari et al. 2004) conifer forests. This may be associated with the higher productivity of Sitka spruce when grown in wet mineral soils in Ireland (Joyce and OCarroll 2002), in part due to the continuous uptake of C throughout the year under these mild climatic conditions (Figures 5.2 and 5.3). To our knowledge, the current study represents the only assessment performed on a first generation conifer stand, where NEP may be higher than similar-aged second rotation stands due to minimal decay from harvest residues.

Cross-validation of inventory and eddy covariance based estimates of NEP

Theoretical considerations

Long-term eddy covariance monitoring networks, such as CarboEurope-IP, have been established to assess biosphere-atmosphere net ecosystem productivity (NEP) across a range of land-use categories. However, flux-tower assessments of NEP need to be independently validated against bottom-up approaches before the data can be scaled up to the land-use level with confidence. Stand level C stock changes are generally assessed using either ecological approaches (NEP_{eco}), based on the estimation of net C balance from heterotrophic and autotrophic processes (Curtis et al. 2002, Ehman et al. 2002), or mass-balance ($NEP_{\Delta C}$) estimates of annual C gains and losses (Kolari et al. 2004, Curtis et al. 2002). The NEPeco approach includes an assessment of annual CO_2 losses from the ecosystem, via heterotrophic respiration as well as the net gain from autotrophic processes (Curtis et al. 2002, Ehman et al. 2002):

$$NEP_{eco} = NPP - R_h \qquad (14)$$

where NPP is net primary productivity and R_H is heterotrophic respiration due to microbial decomposition of soil organic C, litter, above-ground detritus (AGD) and respiration by consumers:

$$R_H(\text{total}) = R_{H\ (\text{soil})} + R_{H\ (\text{AGD})} + R_{H\ (\text{consumers})} \qquad (15)$$

Generally, C stored as AGD includes decaying stumps and branches, which may be a particularly significant component in second rotation forests (Kolari et al. 2004) and natural unmanaged stands. In young unthinned afforested stands, the residual decay due to harvest or successional losses is small and AGD, in this case, can generally be defined as all dead branches that are attached to the tree.

Inventory-based estimates of NPP can be calculated as:

$$NPP = \Delta B + \Delta AGD + D_a + D_b + H + VOC \qquad (16)$$

where ΔB is the change in living biomass, ΔAGD is the annual change in dead material still attached to the tree, D_a is above-ground litterfall measured as the annual loss of live shoots or branches, D_b is below-ground detritus, defined as all C transferred from roots and mycorrhizae to the soil through exudation and mortality, and H is related to herbivore removal of biomass. Losses associated with the emission of volatile organic compounds (VOCs), such as monoterpenes, are not commonly accounted for and assumed to be small.

The more direct mass-balance inventory approach for estimating NEP ($NEP_{\Delta C}$) is based on the change in C stocks in live (ΔC_b) and dead (ΔC_{dead}) biomass, and soil (ΔC_{soil}), as described by Kolari et al. (2004):

$$NEP_{\Delta C} = \Delta C_b + \Delta C + \Delta C_{soil} \qquad (17)$$

As ΔC_{soil} is generally very small it is difficult to detect against the considerably higher background soil C stock. An additional difficulty arises from the high spatial heterogeneity of soil C stocks.

Intercomparison of different methodolgies

Based on eddy covariance measurements (NEP), the uptake of C by the forest was 8.9 and 8.3 t C ha^{-1} yr^{-1} for 2002/3 and 2003/4, respectively. The extent of any difference between eddy covariance and inventory-based assessments of NEP varied depending on the type of inventory procedure used. Although the inventory-based NEPeco estimates were 7 and 13% higher than the cumulative NEP fluxes, this was not significant due to the large cumulative errors associated with biometric, ecophysiological and modelled estimates. However, the $NEP_{\Delta C}$ estimate was significantly higher (ca. 40%), when compared to the NEP_{eco} and -NEE measures (Table 5.1). When the fine root C increment was included in the ΔC_b estimate ($\Delta B + D_b$), the difference between $NEP_{\Delta C}$ and the other NEP assessment was even larger (41 to 51%, Table 5.1).

A comparison of inventory-based approaches with eddy covariance assessments of NEP, from this and other studies (Kolari et al. 2004, Curtis et al. 2002, Ehman et al. 2002), shows that there may be discrepancies between the different estimates of NEP (Figure 5.4). Whilst there is often a good agreement between NEP_{eco} and $NEP_{\Delta C}$ estimates (Figure 5.4a, Curtis et al. 2002), inclusion of annual fine root C increment in the $NEP_{\Delta C}$ assessment results in a systematic overestimation of annual C storage, when compared to NEP_{eco} and NEP estimates (Figure 5.4 a, b). Clearly, the fine root C increment represents a significant C stock change, which cannot be excluded from living $NEP_{\Delta C}$ assessments. Curtis et al. (2002) report annual fine root increments ranging from 1.4 to 4.9 t C ha^{-1} yr^{-1}, but excluded this C pool from their ΔC_b estimate. The $NEP_{\Delta C}$ values reported by Kolari et al. (2004) did not include fine root or ΔC_{soil} estimates. We suggest that the overestimation of NEP, using the $NEP_{\Delta C}$ method, is associated with unaccounted soil decomposition losses (Kolari et al. 2004). Alternatives to the chronosequence approach, such as the application of soil carbon balance models, may provide a better insight into how soil C stocks may change over time.

Ecosystem models

Partitioning of ecosystem processes

Based on the C stock estimates presented in the previous chapters, and the component flux analysis presented above, we were able to provide a closed C cycle for the experimental site in Dooary (Figure 5.5). This analysis suggests that ca. 30% of the C gained via gross primary productivity (GPP) is lost due to autotrophic respiration (Figure 5.5). The remaining fixed C in the NPP (essentially biomass increment) pool is either allocated to above and below-ground biomass (64%) and soil components (litter and root turnover ca. 2%) or lost due heterotrophic decomposition (34%). Although the soil C pool represents the largest pool of C, the estimated rate of soil sequestration (ca. 1-2 t C ha^{-1} yr^{-1}) was smaller than the amount of C allocated to tree biomass (10.5 tC ha^{-1} yr^{-1}, Table 5.1, Figure 5.5).

Parameterisation of validation of CENW

The CENW model links flows of carbon, energy (solar radiation), nutrients and water in trees and soil organic matter. Simulated tree growth depends on plant attributes, the size of plant C and N pools, environmental factors, and the total amount and turnover rates of soil organic matter, which drives mineralisation of N. The model was validated against experimental data obtained from the chronosequence study, presented here and in the previous chapters. A 50 year old climate data set, from the Kilkenny observatory, was used to derive the environmental inputs. Figure 5.6 shows a reasonable agreement between the modelled and measured total biomass over the 50 year period. There was generally also good agreement between the predicted and measured NPP, NEP and heterotrophic respiration data (Figure 5.7). However, the model may be over estimating NPP and heterotrophic respiration in older stands (Figure 5.7).

This preliminary modelling exercise suggests that the C sequestration potential of Sitka spruce forests growing on wet mineral soils is highest at ca. 5 to 8 years after afforestation, followed by a decline to ca. 3 t C ha^{-1} y^{-1} before harvest (Figure 5.7). Based on our eddy covariance (NEP) results and data from a similar project in other EU states (Kowalski et al. 2004), 8 t C ha^{-1} yr^{-1} would be the upper limit of annual NEP for Sitka spruce stands, under temperate conditions and current

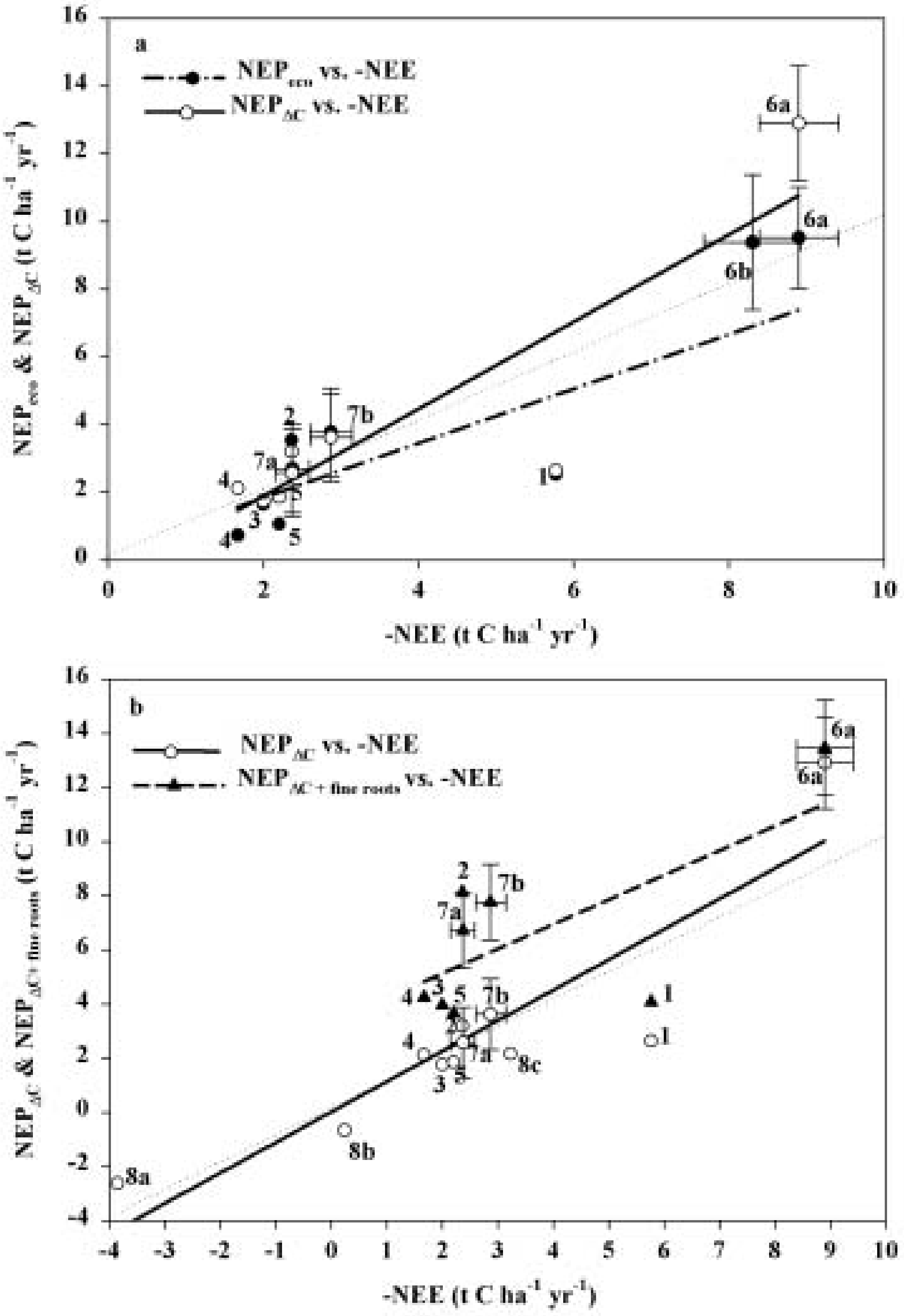


Figure 5.4: Different inventory-based estimates of NEP compared to eddy covariance (-NEE) measures for various forest sites across Europe and the U.S.A. The solid and dashed lines represent the linear relationships between NEP_{eco} and -NEE (a), $NEP_{\Delta C}$ and -NEE (a & b), and $NEP_{\Delta C+\text{fine roots}}$ and -NEE (b). The dotted line represents the 1:1 slope of the relationship. Different numbered symbols represent data from different sites. Sites 1 to 5 are deciduous forests in the U.S.A. (Curtis et al. 2002) Walker Branch, Tennnesse (1), Morgan Monroe State Forest, Indiana (2), Harvard Forest, Massachusetts (3), University of Michigan Biological Station, Michigan (4), and Willow Creek, Wisconsin (5). Site 6 is a Sitka spruce forest in Co Laois, Ireland, based on 2002/3 (6a) and 2003/4 (6b) data. Site 7 is also for Morgan Monroe State Forest, Indiana (see 2), but based on different estimates for 1998 (7a) and 1999 (7b) reported by Ehman et al. (2002). Sites 8a, 8b and 8c are 0(a), 12(b) and 75(c) year old second rotation Norway Spruce stands in Southern Finland (Kolari et al. 2004). The error bars represent SE of -NEE, NEP_{eco} and $NEP_{\Delta C}$ means for sites 6 and 7. The errors associated with NEP_{eco} and $NEP_{\Delta C}$ for site 7 were determined from the literature, using the same procedure outlined in Table 5.2.

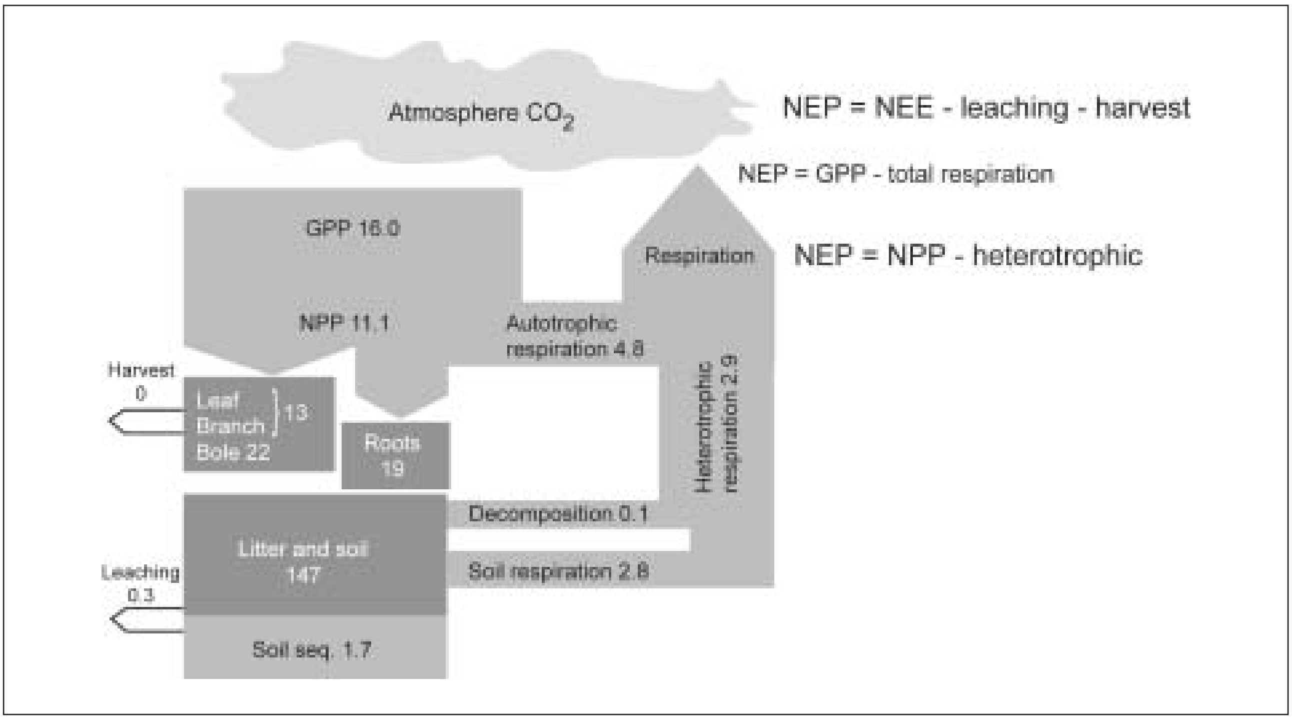


Figure 5.5: Carbon fluxes and stocks in a Sitka spruce forest located in Co Laios. The values in the yellow boxes and arrows represent the annual fluxes (t C ha^{-1} yr^{-1}) of various ecosystem components. The values shown in the brown boxes represent biomass and soil C stocks (t C ha^{-1}). Abbreviations: NEP is net ecosystem productivity (equivalent to C sequestration rate); GPP is gross primary productivity (annual canopy photosynthesis); NPP is the net primary production (annual biomass increment).

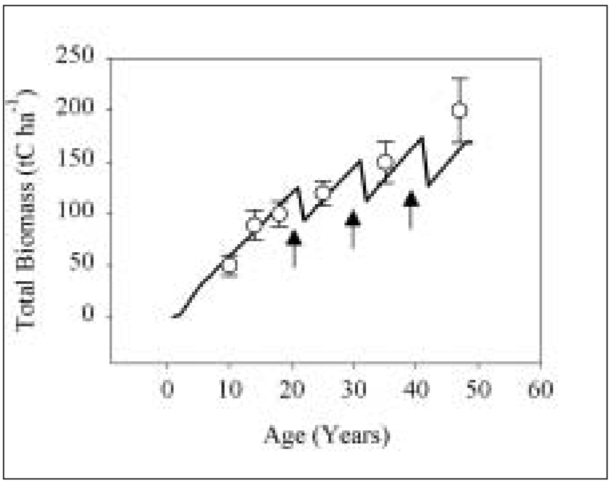


Figure 5.6: Predicted (solid line) and measured (open symbols) total biomass stock for the Sitka spruce chronosequence. The arrows indicate when the stands were thinned, assuming a removal of 33% of the standing volume for each thinning cycle. The predicted values were simulated using the CENW model (Black et al. in prep.)

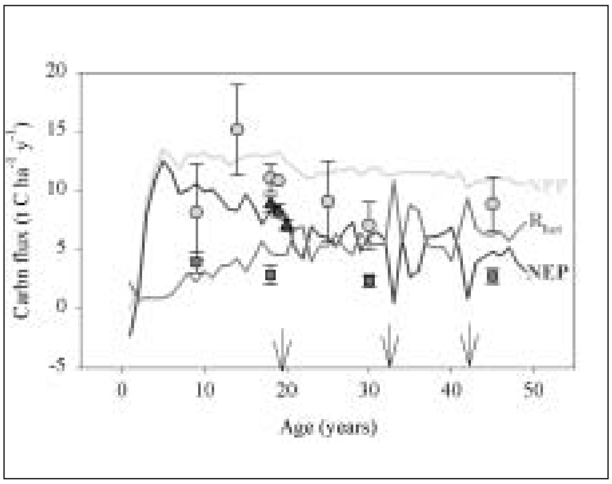


Figure 5.7: Predicted (solid lines) and measured (symbols) changes in net primary productivity (NPP, green), net ecosystem productivity (NEP, blue) and heterotrophic respiration (RH, red) over the chronosequence. The arrows indicate when the stands were thinned, assuming a removal of 33% of the standing volume for each thinning cycle. The predicted values were simulated using the CENW model (Black et al. in prep.).

management regimes. Although the CENW model predicts that NEP may be higher in younger stands, this needs to be validated with eddy covariance measurements.

It is also evident from this analysis that both the C stocks and sequestration rates (NEP) are strongly influenced by management events, such as thinning (Figures 5.6 and 5.7). Clearly the influence of thinning and land-use change on C sequestration in Sitka spruce forests requires further examination.

INTEGRATION OF CHORONSEQUENCE DATA

We were able to compile a closed C balance for some sites within the chronosequence, based on collected and modelled data from the different working groups (Chapters 2, 3, 4). Figure 5.8 shows the completed C cycles over the different aged stands.

- Gross primary productivity (GPP, gross uptake of CO_2) was estimated using the MAESTRA model, which was validated against the eddy covariance data from the flux tower site (15 years, Figure 5.8). GPP was not estimated for the grass vegetation in the younger stands.
- Heterotrophic respiration (R_H) for each site was obtained from the modelled soils data (see Chapter 2) and decomposition data from the harvest residue in older stands (see Chapter 4).

$$R_H = R_{H\,(soil)} + R_{H(cwd)} \qquad (18)$$

Autotrophic respiration (R_A, respiration by the tree) was estimated using the soil autotrophic respiration modelled data (R_A (soil), Chapter 2) and the above-ground (foliage and stem growth and maintenance respiration, (R_A (foliage and stem), based on the outputs from the MAESTRA model.

$$R_A = R_A\text{ (soil)} + R_A\text{ (foliage and stem)} \qquad (19)$$

Therefore $R_{TOT} = R_H + R_A$ (20)

(herbivory and VOC production was assumed to be negligible)

- The increment in biomass into the foliage (Fol), stem (Stem) and root ((Root), coarse and fine roots) was obtained from harvested tree data (see Chapters 2 and 4). The change in biomass in the grassland component of the open forest canopy (10 year old site) was estimated from monthly biomass samples in 2003 (see Chapter 3).
- The input of litter (Litter) and accumulation of C in soils (Soil) was obtained from the results shown in Chapter 3. The theoretical increase of total below-ground C allocation (TBCA) was calculated based on the estimation as described by Davidson et al. 2000:

$$TBCA = Fs + Rh(cwd) + Fe - (Litter + HR) + (soil + Root) \qquad (21)$$

where Fs is soil respiration, Fe is leaching and run off (assumed to be negligible), Litter is annual litter fall (Chapter 3), HR is the harvest residue including stumps, roots and brash (annual mean after 20 yrs, see Chapter 4), Root is the annual coarse and fine root biomass increment (Chapters 2 and 4) and soil is the annual soil C storage based on the difference in soil C between the current year and G0 (the non-forested site).

The net ecosystem productivity or annual C sequestration rate (NEP), which excludes harvest removals, was calculated as:

$$NEP = NPP - Rh \qquad (22)$$

where NPP is net primary productivity or annual increment of the Fol, Stem and Root biomass components. The calculated NEP was validated against the measured NEP (mNEP) for 2003.

The chronosequence approach, together with modelling and flux measurements (Figure 5.8), shows that afforested stands are a C sink at 10 years, and possibly earlier, followed by a further increase to a maximum of ~ 8 t C ha^{-1} yr^{-1} before the first thinning cycle. The lower NEP and GPP in younger crops, prior to canopy closure, is associated with a lower leaf area index and reduced light capture together with a higher respiratory loss, such as those associated with autotrophic respiration and decomposition of grassland vegetation in the 10 yr-old forest. The higher accumulation of soil C in younger stands is associated with a higher input of C, from litter and non-forest vegetation, and a relatively lower C loss via heterotrophic respiration from soils. This is consistent with the decrease in the ratio of TBCA to

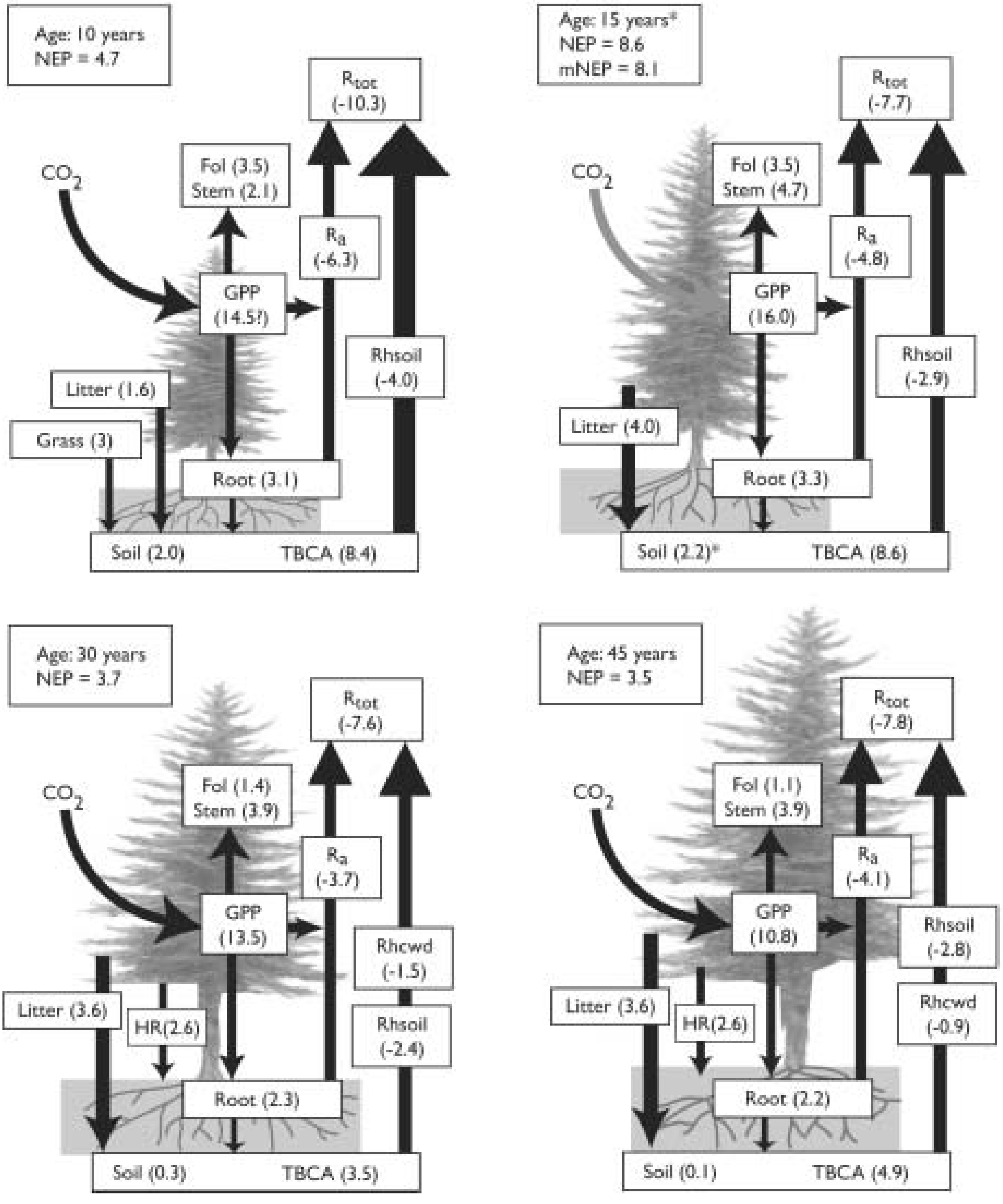


Figure 5.8: Carbon budget for different aged forests in 2003/4 expressed in t C per ha per year. GPP is gross primary productivity (gross uptake of CO_2), NEP is calculated net ecosystem productivity or annual C sequestration rate, mNEP is the measured NEP value, Fol, Stem and Root is the annual biomass increment into foliage, stem and roots, Grass is the change in grassland biomass in the 10 year old stand, Rh is heterotrophic from soils (Rh soil) and harvest residue (Rh cwd), Ra is autotrophic respiration from roots, foliage, branches and stems, Litter is annual litterfall, HR is the harvest residue including roots, stumps and brash, Soil is the measured soil C stock increment and TBCA is the theoretical soil C allocation (equation 21). * The 15 yr old stand was rescaled to an 18 year old stands with a YC of 22 to account for differences in YC over the selected chronosequence.

Rh from 2.1, in the young stand, to 1.3 in the oldest stand. The decomposition losses are increased due to thinning residues, which result in an additional C source of ~1 t C ha^{-1} yr^{-1}. The higher turnover and decomposition of litter and root exudates in the older stands in the chronosequence may, however, be associated with lower soil moisture content on sloped stands. Alternatively more soil C can be lost due to erosion and runoff in sloped stands.

NEP declined from ~8 t C ha^{-1} yr^{-1}, at closed canopy, to ~ 3.5 t C ha^{-1} yr^{-1} in older, thinned stands. These findings are consistent with the results from other studies which show the NEP of Sitka spruce stands in the UK varies from ~ 7 t C ha^{-1} yr^{-1} at canopy closure to ~ 3 t C ha^{-1} yr^{-1} in older stands (Kowalski et al. 2004). The age related decline in NEP in older stands is due to two major factors: a) a decrease in GPP, and b) an increase in maintenance respiration. A reduction in GPP in older stands is due to thinning, which effectively reduces the leaf area index and light capture by the stand. Light capture is further reduced by an increased allocation of C into non-photosynthetic tissue, such as timber. It has also been suggested that hydraulic constants also limit GPP in older stands. The increase in maintenance respiration, a component of autotrophic respiration, is due to a higher allocation of biomass in to non-photosynthetic tissue.

CONCLUSIONS

Given CARBiFOR's emphasis on the chronosequence approach, it was possible to determine the changes in biomass and soil carbon stocks over time and to relate these to the observed fluxes measured by eddy covariance for a dominant afforested soil type. Although these results are useful for providing a baseline for carbon sequestration potential for the dominant forest species in Ireland, Sitka spruce, when grown on wet mineral soils, more information on carbon stocks and fluxes for stands on different soil types, as well as other forest species is required. Clearly, a critical evaluation of biomass and soil carbon stocks both for different species and for stands grown on a range of soils is required for countrywide reporting to the IPCC on LULUCF. In addition, validated estimates for biomass increment, particularly for younger forests (age 3 to 20 years) are also required. Given the emphasis on young forests for reporting sequestration changes for the first commitment period, more analysis on afforested sites covering the period from planting to the first thinning cycle is required.

The provision of one eddy covariance tower as well as a short-term commitment to funding was a major limitation to the ecosystem component of the CARBiFOR project because this prevented a comprehensive characterisation of water and carbon dioxide fluxes over the chronosequence. As initially proposed, future investigations would require the use of at least two towers: a fixed tower used as a reference, and a second system employed to "roam" among different sites, in order to provide adequate spatial coverage of CO_2 and water fluxed across a chronosequence. The provision of long term eddy covariance data would also facilitate the development of comprehensive carbon balance models that can be used to simulate the effect of stand age, disturbance and climate change scenarios on forest carbon fluxes. Beyond changes associated with stand age, there is a need to incorporate an analysis of the effect of disturbance, particularly those related to management practices (e.g. thinning) and land-use change, such as the transition from grassland to forest.

Whilst standardised carbon inventory methodologies have been developed by the IPCC (see Houghton et al. 1997), there is a clear need for studies to reflect national circumstances. Inventory methods, based on species and region-specific biomass models, may be suitable for carbon accounting reports for the first commitment period. However, it is envisaged that 'full carbon accounting' would be required beyond the first commitment period to estimate flux losses associated with disturbance and land-use change, which can not be determined using conventional biomass or soil carbon stock estimates.

Chapter 6

RELATED STUDIES

BACKGROUND

National and international collaboration with other research groups working on C sequestration issues was an important component of the CARBiFOR project. Members of the group also participated in COST action groups, such as E21, and European Commission: Programme on Environment and Sustainable Development (CarboInvent Project). Collaboration with Prof. John Grace from Edinburgh University, who reviewed the project in 2003, was also established. Prof. Grace also heads the Forest group of the CarboEurope Project. Some aspects of the work was subsequently integrated into the CarboEurope project as an associate site. This collaboration involved the sharing of eddy covariance data across aproximately 50 sites over Europe.

The EPA-funded CC (Celtic Carbon) flux network, shared between UCD, TCD and UCC, was established to investigate C fluxes over grassland, bogs and cereal crops in Ireland. Collaboration with this network facilitated a comparison of ecosystem fluxes over various land-use types.

This chapter briefly outlines the major developments, resulting from the above-mentioned collaborations, which were not initially proposed at the outset of the project.

A) Biomass stocks in young Sitka spruce and lodgepole pine stands

Brian Tobin and Carly Green

INTRODUCTION

A study was conducted in Co Mayo to investigate the biomass stocks of trees, which had yet to reach merchantable timber volume, i.e. trees of dbh <7 cm. Such trees are often omitted from national forest inventories as they are not yet of significance from the forest industry perspective. However, with the carbon (C) reporting requirements under Article 3.3 of the Kyoto Protocol they have gained increased significance as areas afforested post-1990 should be reported during the first commitment period (2008-2012). Given that Ireland experiences significant afforestation rates, a potentially large proportion of land eligible for reporting under Article 3.3 may consist of such “small trees”. The objective of this study was to develop specific biomass functions for trees less than merchantable timber volume <7 cm dbh, which related tree height to total dry biomass. Two species of commercial importance were included in the study, lodgepole pine (*P. contorta* spp. *contorta*) and Sitka spruce. C stock estimates of both above- and below-ground tree components were investigated. Results from this study will be included in Green et al. (in prep.).

METHODS

Five commercial plantations of young (i.e. less than merchantable timber volume) Sitka spruce and lodgepole pine, planted in intimate mixtures on blanket peat in Co Mayo, were selected, ranging from 3 to 12 years. Three plots were randomly located at each site and an inventory of height and dbh (where tree height exceeded 1.3 m) was undertaken. Fifteen trees from each species were selected for destructive sampling from the range of five sampling sites. A tree closest to the tree of mean height and two others, one standard deviation either side of the mean, were selected from each species within each site.

The above-ground portions of the sample trees were separated into their component parts (stem and foliage, i.e. branches and needles) and sampled according to the methodology described in Green et al. (2005 in press). Roots were excavated from a square measuring 2 x 2 m, to a diameter of 2 mm and to the maximum depth at which they occurred (after Tobin and Nieuwenhuis, submitted).

The dry biomass of each tree was determined and, with the measured tree height, was used to develop biomass functions of the form:

$$LN(DB) = \alpha LN(h) + \beta \qquad (23)$$

where:

DB = dry biomass (kg), h = tree height (m) and α and β are parameters.

Correction factors (cf) for converting log-transformed predicted values back to arithmetic units were calculated as the antilog of one half of the sample variance (Sprugel 1983).

Total tree (above-and below-ground) C stocks were developed for each site within the chronosequence using the site inventory and biomass functions developed.

RESULTS

The height of sampled trees varied between 0.67 and 4.18 m for Sitka spruce and between 0.62 and 3.85 m for lodgepole across the five stands. The average percentage biomass of each component was similar for each species, save for the root and stemwood components (Table 6.1). Roots represented 6% more for Sitka spruce than lodgepole pine and stemwood represented 4% greater in lodgepole pine than Sitka spruce.

A strong relationship was established between tree height and biomass, regardless of tree age, for both species investigated. Furthermore it was found that the slopes and intercepts of equations for total biomass (TB), above-ground biomass (AGB) and below-ground biomass (BGB) were not significantly different between the two conifer species studied, therefore the sample tree data sets were combined to develop one general equation for the two species (Table 6.2).

The tree C stocks were estimated to range from 0.6 tC ha-1 to 17.6 tC ha-1 across the five sites.

DISCUSSION

A combination of national forest inventory (NFI) commitment to measuring small tree variables (i.e. such as height) and the development of species-specific allometric relationships will ensure an improved level of understanding of the importance of small trees in the national C budget, especially for countries with large-scale afforestation activities since 1990 for reporting to the Kyoto Protocol during the first commitment period.

For countries like Ireland, with policy initiatives stimulating significant annual land-use change to forestry (i.e. afforestation), accounting for such lands under Article 3.3 may prove very important to offsetting the liability of not reaching its emissions target under the Kyoto Protocol.

In recognition of this, Ireland's new NFI incorporates the measurement of height for small trees (less than 7 cm dbh). This information, in combination with the development of nationally specific tree (above- and below-ground) data for major tree species in Ireland will result in good practice estimates of C stocks (Penman et al. 2003)

Table 6.1: Percentage distribution of small tree component biomass.

Biomass component	Sitka spruce	Lodgepole pine
Foliage	46	48
Roots	34	28
Stem	20	24

Table 6.2: Equations for combined species of the form $\ln(y) = \alpha\ln(x) - \beta$.

To estimate:	α	β	SEE	cf	R^2
Total Biomass	2.7935	0.3562	0.0476	1.0011	0.926
Above-ground	2.8037	0.7180	0.0540	1.0015	0.912
Below-ground	2.9914	1.6696	0.0732	1.0027	0.876

with reduced associated uncertainty for post 1990 national forest tree biomass. Results from this study suggest that at the early stages of development (sub-merchantable dimensions), it is possible to combine the two species for the purposes of C stock estimates. This may prove very beneficial as an alternative to the BEF method for calculating C stocks of younger forests. The application of BEFs to small trees presents many difficulties as well as large errors because at this stage, within only a few years, the BEF values reduce markedly (see Chapter 4; Tobin and Nieuwenhuis in preparation). The use of allometric functions should reduce the uncertainties of estimates in such young forests (Black et al. 2004).

ACKNOWLEDGEMENTS

Funding for this work was provided by the Irish National Council for Forest Research and Development (COFORD), the European Commission: Programme Environment and Sustainable Development (CarboInvent Project: contract No. EVK2-2002-00157) and the FORSEE project. Many thanks to Marina Conway for invaluable help in liaising with farm foresters, and to those who willingly allowed access to their property and the removal of trees. Also to Paul Gardiner for assisting with fieldwork.

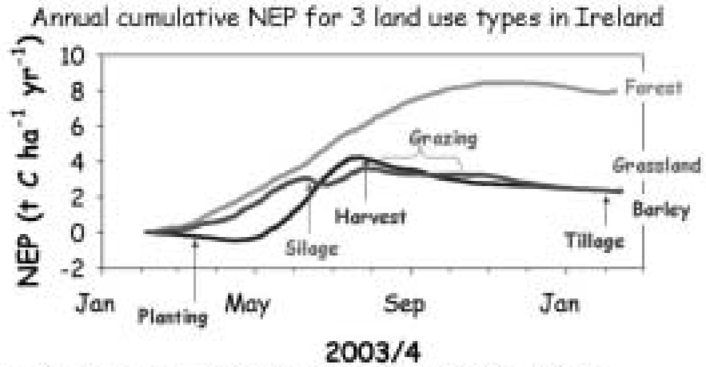
B) A comparison of CO_2 fluxes from three Irish land-use categories: Arable, Grassland and Forestry

Kevin G. Black, Phill Davis, Mike Jones and Bruce Osborne

Different land-use categories have different carbon sequestration potentials and differing significances in relation to the Kyoto Protocol. With the aim of understanding the processes driving the biogeochemical cycles of major ecosystems in Ireland, carbon fluxes were measured from three different land-use categories. The estimation of a base line C flux level for land-use types being converted to forestry is important from an Article 3.3 perspective. This information will provide the basis of calculating C losses or gains associated with land-use change. The net ecosystem productivity (NEP, C sequestration) was 2.3 and 2.1 t C ha^{-1} yr^{-1} for the grassland and barley crop, respectively (Figure 6.1). For the same period the forest site had a NEP of 8.3 t C ha^{-1} yr^{-1} (Figure 6.1). Only the forest site had no carbon export. There was a removal of silage and grain from the grassland and barley sites, which resulted in a net loss of C from the ecosystem.

The barley and the grass site are within a few hundred metres of each other and therefore experience the same weather conditions. The forest site however, is 25 km away and is approximately 200 m higher though there is little effect on the daily irradiance and mean daily temperature, two factors which may affect the NEP. In the future this collaboration will also allow the assessment of C flux associated with land-use change from grasslands or cereal crops to forestry.

More information is required on the losses associated after the transition from agricultural or grassland systems to forests. These include C flux changes due to cultivation prior to planting trees and vegetation changes up to canopy closure. The magnitude and sign (i.e. source or sink) of the C flux will depend on the previous land-use type, soil type and planting regime.



NEP = NEE - harvest- lateral transport (grazing), therefore;
Forest NEP = 8 t C ha^{-1} yr^{-1} **(sink)**

Grassland NEP = 2.3- 3.6 (silage) - 0.1 (grazing) = -1.4 t C ha^{-1} yr^{-1} **(source)**

Barley NEP = 2.1- 3.2 (grain only) = -1.1 t C ha^{-1} yr^{-1} **(source)**

Figure 6.1: Cumulative NEP in the various land-use categories in Ireland. The arrows indicate land-use disturbance events.

C) Continental-scale δ^{13}C and δ^{18}O isotope signatures of forest ecosystems: Results from a European network

Deborah Hemming and Kevin G. Black

INTRODUCTION

Stable carbon and oxygen isotopic compositions (δ^{13}C and δ^{18}O, respectively) of atmospheric CO_2 have proven to be powerful tools for examining the relative contribution of key sources and sinks to the total atmospheric [CO_2] at local, regional and global scales (Lloyd and Farquhar 1994). This is because processes associated with the exchange of CO_2 involve discrimination against the heavier ^{13}C and ^{18}O isotopes, which can be detected in the isotopic measurements of atmospheric CO_2. Because isotopic mass balance is conserved within the earth-atmosphere-ocean system, a specific fractionation in a destination component will result in an equal and opposite fractionation in a source component.

The are still significant uncertainties and lack of experimental data associated with the use of stable isotopes, such as in the estimation of terrestrial ecosystem discrimination, and the isotopic disequilibrium effect (Fung et al. 1997). Scholze et al. (2003), for example, demonstrated that time trends in annual net atmosphere-biosphere flux estimates could vary by ~0.5 PgC yr^{-1}, depending on whether or not environmentally forced changes in ecosystem discrimination are included. Given that global estimates of the total land ecosystem sink are of the order of 1.5 PgC yr^{-1}, and a variation of 0.025 ‰ (parts per thousand) in annual mean atmospheric δ^{13}C translates to a global net terrestrial flux of about 1.0 PgC, depending on the Δ_e values used (Ciais et al. 1995), more observations and better constraint of De could significantly reduce the error in carbon sink estimations.

While much effort has been invested in small-scale isotopic studies, only recently have applications at the ecosystem scale been a major focus (Buchmann and Kaplan 2001 and references therein). Further, relatively little data are available to develop isotopic perspectives at the regional or continental scale (Tans et al. 1996; http://www.gcte-focus1.org/basin.html.). Recently, attempts have been made to collect air samples from tall towers (Bakwin et al. 1995) and aircraft in order to obtain large-scale isotopic data across ecosystem boundary layers.

During 2001/2, 34 individual sampling campaigns were carried out between the July and December at 13 sites across Europe and one site in Israel (Figure 6.2), within the framework of the CarboEuroflux (Work Package 5) project. For each campaign a suite of air, plant and soil samples were collected. Here, we examine large-scale spatial and temporal trends in air sample derived δ_R and Δ_e values, the δ^{13}C of leaf, stem and soil organic matter and the δ^{18}O of leaf, stem and soil waters, and we compare these with corresponding flux tower ecophysiological and meteorological measurements. Finally, the data are used to provide an experimentally-based estimate of regional scale Δ_e and show its link to underlying ecophysiological processes.

SUMMARY OF RESULTS

The flux-weighted average Δ_e for all the sampling campaigns was 16.4 ‰ (where Δ_e is defined here as the deviation of δ^{13}C of night-time ecosystem respiration from that of background atmospheric δ^{13}C). This compares well with regional-scale discrimination estimates based on air samples collected at two European NOAA-CMDL

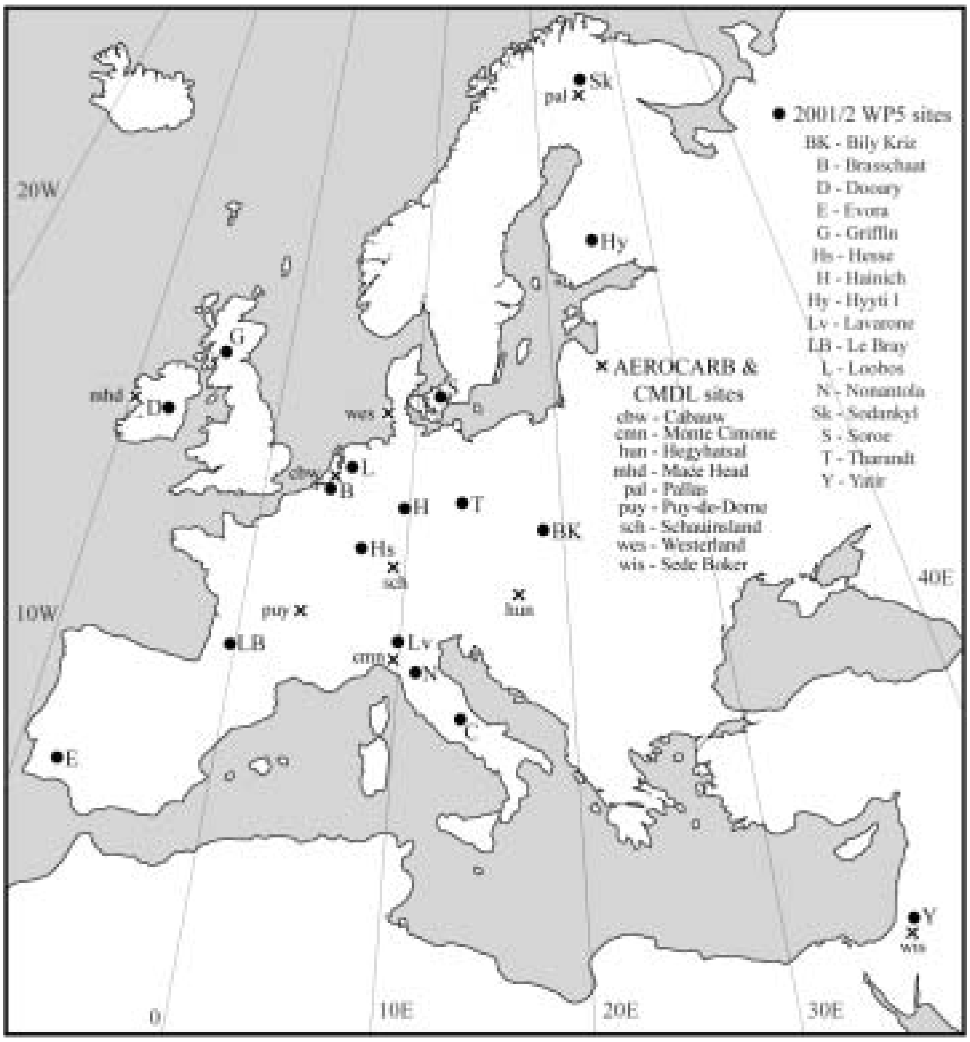


Figure 6.2: Stable isotope network (WP5) sites for both 2001 and 2002 seasons (black circles). Also shown are the AEROCARB and NOAA/CMDL atmospheric [CO_2] monitoring sites for which data is used in this study (crosses) (Taken from Hemming et al. (2005).

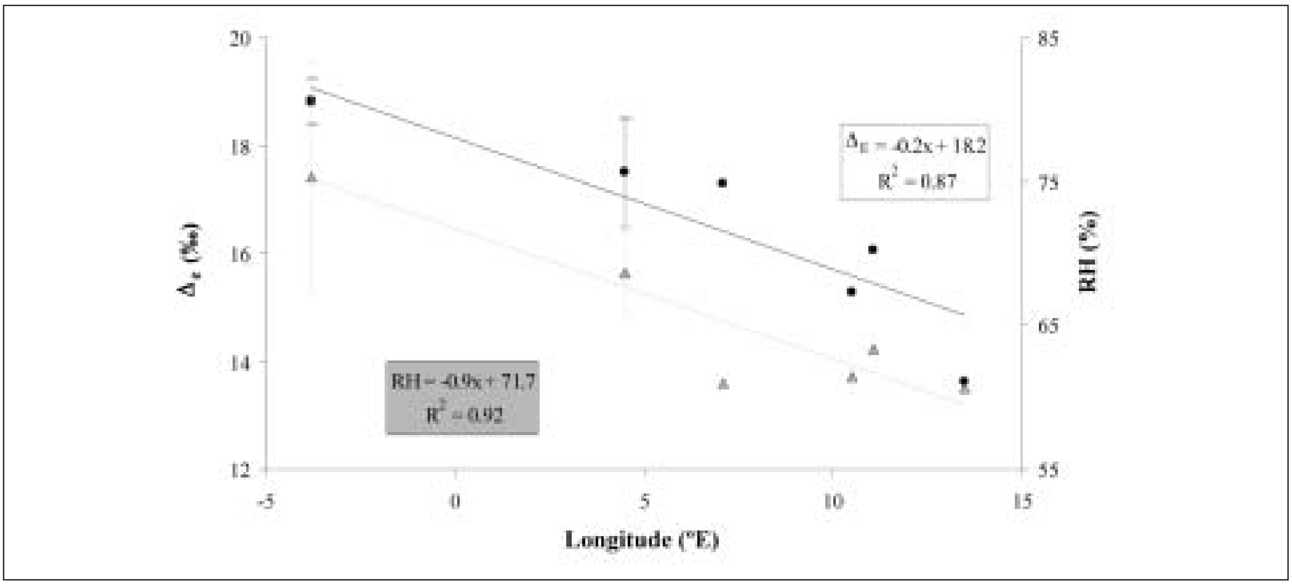


Figure 6.3: Relationship between degrees longitude and both ecosystem discrimination, Δ_e (black circles) and daily average (06:00-18:00) relative humidity, R_H (grey triangles), between July and August 2002. Note that at sites with multiple observations during this period each Δ_e and R_H value is an average of all the observations at the site, and the SD of the observations is shown by the error bars. (Taken from Hemming et al. (2005).

background stations, including Mace Head in Ireland. It was further noted that relationships between ecosystem isotope discrimination (Δ_e) and the 24-hour average net ecosystem CO_2 exchange, NEP, were bimodal. A positive linear relationship was observed between NEP and Δ_e for periods of low net CO_2 uptake. However, no trend was observed for NEP values below 2.0 mmol m^{-2} s^{-1} (negative values indicate net ecosystem uptake), De fluctuated around a mean of 16.5±1.7 ‰, where these values accounted for 81.3% of the total seasonal NEP. The relationships between Δ_e and NEE were linked with physiological adjustments, indicating a close association between leaf level discrimination and observed Δ_e at the continental scale.

Δ_e values displayed a significant relationship with longitude (“continental effect”) during the period of maximum NEE, and with morning relative humidity for up to 3 days prior to sampling (Figure 6.2). Between July and October, the average $\delta^{13}C$ of ecosystem respiration, δ_R, was ~1.5 ‰ more enriched than the $\delta^{13}C$ of corresponding plant and soil organic matter, indicating that the $\delta^{13}C$ of these organic components, which likely represents early season high photosynthetic discrimination, could not be combined with Δ_e to partition NEE into source CO_2 contributions.

CONCLUSIONS

The results of this network provide the first experimentally-based estimates of ecosystem discrimination at the European continent scale. They can be used to validate large-scale model estimates of Δ_e and they provide a scaling link between ecosystem measurements of Δ_e and those based on background atmospheric measurements. These results clearly demonstrate the ability of stable isotopes to record large-scale patterns of ecosystem ecophysiology, despite considerable inter-site variability, including the newly identified relationships between the $\delta^{18}O_{LW}$ and ecosystem water fluxes.

The stable isotope measurements are routinely being measured at the flux tower site in Dooary and this can be integrated with other Irish land-use ecosystem flux data from the CC flux network if more funding becomes available.

Chapter 7

IMPLICATIONS FOR NATIONAL REPORTING

Kevin G. Black, Kenneth A. Byrne and Gerhardt Gallagher

A) SUMMARY OF MAJOR FINDINGS

Based on inventory and other physiological methods, the average C sequestration rate over the entire life cycle of the selected chronosequence was estimated to be 4 to 8 t C ha^{-1} yr^{-1}. This value is higher than a previously reported average estimate of approximately 3.36 t C ha^{-1} yr^{-1} (Kilbride et al. 1999), based on the model developed by Dewar and Cannell (1992). The higher average sequestration rate reported in this study was due to the inclusion of soil C stocks and the use of improved biomass models. Our results suggest that the BEF value used to calculate carbon stocks for Irish forests was underestimated in the report by Kilbride et al. (1999), particularly for younger Sitka spruce stands. The use of the single BEF value of 1.3 may result in an ~ 2 to 40-fold underestimation of current carbon stocks for sites afforested since 1990.

The chronosequence approach can be used to determine the changes in biomass and soil carbon stocks over time. Afforested wet mineral soils represent a significant C sink of 0.05 to 1 t C ha^{-1} yr^{-1}. However, changes in soil C stocks may be more difficult to measure in highly organic soils, such as peats or podsols, because of the high background soil C. In a study of the effect of afforestation on soil CO_2 emissions in blanket peat, Byrne and Farrell (2005) found that soil CO_2 emissions were ~1.7 t C ha yr^{-1} in a three year old crop of Sitka spruce. CO_2 emissions were ~1.0-1.4 t C ha yr^{-1} in lodgepole pine and 2.6 t C ha yr^{-1} in 39 year old Sitka spruce. In peaty gley soils afforested with Sitka spruce, the soil C decrease from 250 t C ha^{-1} in grassland site to 150 t ha^{-1} in a similar site afforested 40 years previously, suggesting that peaty soils may be a source of C (Zerva et al. 2005).

The soil water properties and soil type may be an important determinant of organic matter decomposition rates in forest soils. In wet mineral soils, decomposition may be lower due to the inhibition of respiration at high soil water contents. In contrast, more C may be lost via decomposition processes in dry or well-drained gley soils with a sloping topography.

The estimated annual NEP values (8 t C ha^{-1} yr^{-1}) for the 14 year old Dooary site is similar, but slightly higher, than reported values for a Sitka spruce stand of a similar age, but grown on a podsol in Scotland (Kowalski et al. 2004). It is evident from our results and reports from a similar project in Scotland (Kowalski et al. 2004), that 8 t C ha^{-1} yr^{-1} would be the upper limit of annual NEP for Sitka spruce stands, under current climatic conditions and management regimes.

Inter annual variations in NEP, for the 14 year old Dooary stand, were associated with changes in temperature and light levels. The lower NEP during the winter months was primarily due to the short day length, and lower temperature to a smaller extent.

Changes in the potential sequestration rates in younger or older stands are associated with differences in leaf area index (LAI), absorption of light by the canopy and changes in the conversion efficiency of absorbed light into fixed C. Therefore, changes in current management, planting and thinning practices may potentially increase C sequestration of afforested stands. It has been reported that increasing rotation length does increase the C sequestration in European forests (Kaipainen et al. 2004).

Beyond changes with stand age, there is a need to incorporate an analysis of the effect of disturbance, particularly those related to management practices (e.g. thinning), and land-use change such as a transition from grassland to forest.

It has been shown that C losses associated with clearfelling may be equivalent to the sum of C sequestered over the first 20 years of the second forest cycle following replanting (Kowalski et al. 2004). These losses may be minimised by partial clearfelling and replanting, continuous cover forestry or minimum soil disturbance following clearfelling.

The provision of long term eddy covariance data on different age class and forest types would facilitate the development of comprehensive carbon balance models that can be used to simulate the effect of stand age, disturbance and climate change scenarios on forest carbon fluxes, to aid in management decisions and reporting procedures to the UNFCCC and climate change risk assessment. More importantly, direct flux methods can be used as a tool to validate outputs from flux and growth models and provide better estimates on inter annual variability of ecosystem carbon balance, once installed on a more permanent basis.

Collaboration with CarboEurope on ecosystem isotope discrimination studies show that isotope mixing model approaches do provide the possibility to estimate total land-use fluxes using a top–down approach. However, more work in this area is required to assess temporal and spatial variation in ecosystem discrimination.

B) CONTRIBUTION TO NATIONAL REPORTING TO THE UNFCCC AND KYOTO PROTOCOL

The requirement for reporting on greenhouse gas emissions by sources and removals in the land-use change and forestry sector as outlined in the IPCC Good Practice Guidance for Land-use, Land-use Change and Forestry (IPCC GPG) means that the current accounting procedures need to be modified. In the past, the CARBWARE model, developed by Gallagher and Hendrick (Gallagher et al. 2004), has been used to express carbon storage from 1990 on in terms of biomass over ground annually by species for each year for UNFCCC reporting and by total biomass for conifer and broadleaved species by age category for the Kyoto reporting period 2008-2012.

CARBWARE is a forest model which uses the species and age categories identified by FIPS for the base year 1995 and grows them forward and backward to give total forest stock for any one year using yield models and data for afforestation, reforestation and felling. It also identifies afforestation going back to 1983 and can provide annual or accumulated forest tree biomass in two species and two age classes using estimates of volume increment. These data are converted to carbon, taking into account species carbon content, basic density and a biomass expansion factor to covert timber volume to total tree biomass. In the past conservative BEF values (1.7) were used to convert timber volume increment to biomass increment for Article 3.3 reporting. The information available from the CARBiFOR has now been used to improve the CARBWARE model.

These improvements include:

- Development of CAI growth models, which replace the BEF based estimates of biomass C stock changes.
- Reporting C stock changes in five different pools as outlined by the IPCC good practice guidance (see Chapter 1).

Methodology

National afforestated area since 1990

The area of new forests, planted since 1990, was obtained from FIPS and Forest service planting records. These area estimates are currently being verified using IFORIS data, which will greatly improve the uncertainty in our estimates. Future C sequestration rates were calculated using three afforestation rates (i.e. 3000, 14000 and 20000 ha per annum).

Whilst this model does not utilize National Forest Inventory data, it is envisaged that this information will provide valuable information of the age, yield class, species and soil type distributions and underlying structure of the forest estate in the future. Therefore the current CARBWARE estimates are subject to a large degree of uncertainty until NFI data are used.

Current annual increment biomass model

A theoretical biomass growth curve based on age and yield class distribution (Figure 7.1) was simulated using top height, DBH and stocking, from the Forestry Commission Yield tables) and biomass functions, developed from the CARBiFOR project (Black et al., 2004; 2005). Current annual increment (CAI) was calculated using the cumulative biomass growth curve functions, as outlined by Montieth (2000) and Hargreaves et al. (2005).

A mean national YC of 16 was selected for conifers.

Broadleaf CAI curves were simulated using above biomass functions for Beech (Bartelik 1977) and generalized below ground functions for broadleaves ((Brown et al. 2002). The biometric inputs (DBH etc.) for the broadleaf CAI curves were obtained from the Forestry Commission models assuming a national YC mean of 4 and a spacing of 1.2 m. There were no thinnings for the broadleaf forests for the first 35 years of the rotation (i.e. up to 2025).

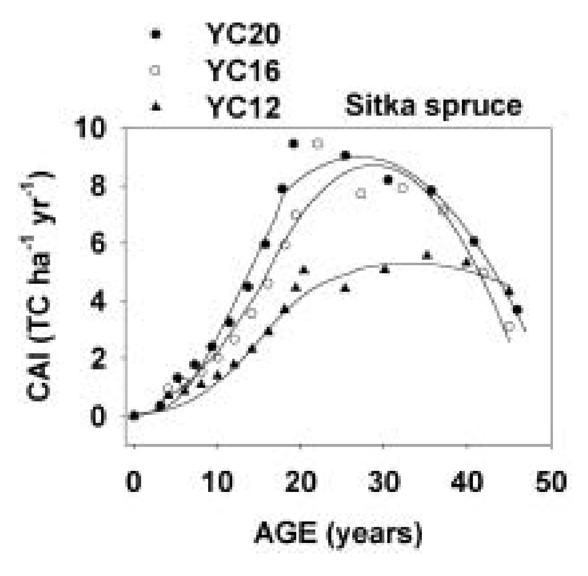


Figure 7.1: Modelled (solid line) and calculated current annual increment growth curves for Sitka spruce. Modelled were developed using Forestry Commission tables: spacing of 2 m and intermediate thinning on a 5 year cycle (Edwards and Christie 1981).

Land use change emission factors

Previously, CARBWARE assumed that 11.93 t C ha^{-1} was lost due to land use change and this was subtracted from the C increment once off. This value was based on emissions from peat soils, but was applied to all soils in the previous version of CARBWARE.

Based on new information (Twomey Forest Service personal communication; Hargreaves et al. 2005) we now only apply an emission factor to peat soils at a rate of 4 t C ha^{-1} for the first 4 years. The total area under peat soils was 56 % of the afforested area in 1990 decreasing to 29.9 % in 2003 (assumed 29.9% from 2003 to 2022).

We assumed that all other soil types are not a source (in fact they may even be sinks according to our research, see Chapter 3).

Assumptions for forest management

Thinning and harvest for conifer crops were assumed to be in accordance with a YC 16 Sitka spruce intermediately thinned stands, originally planted at a spacing of 2 m. The harvest age, at maximum annual timber volume increment was 45 years.

For the broadleaf crops we assumed that half of the recommended thinning volumes (Beech YC 4, Forestry Commission tables) were removed to account for thinning species like ash and no thinning for beech or oak over the 45 year model simulation.

C pool calculations

To facilitate IPCC reporting of the other pools the following modifications have been made to the CARBWARE model

- **Biomass pools:** Total Biomass was converted to above and below ground (BG) based on Table 4 total biomass and root to shoot ratios.
  - < 17 yrs, Ratio = 0.3

  - • > 17 yrs 0.2 (Default in IPCC GPG tables would be 0.2)
- **Harvest:** In the past, the total biomass was removed from the forest. We have now only removed the timber fraction of the biomass C stock.
  - • The timber removed is based on the BEF used, but in reverse this time i.e. Timber = harvest biomass /1.9 (BEF < 17 yr old forests = 1.9)
  - • The remaining root biomass (i.e. root harvest residue (RHR) left behind after harvest, which is now the dead wood pool, is subtracted from below ground biomass (i.e. (BG) = BG - RHR). The non-removed timber residue is not included in this calculation because there are still uncertainties regarding the estimation of the CWD pool.
- **Litter and brash:** Annual litter (L) and brash (Br) left behind after harvest are now included in the new CARBWARE model.
  - • Litter (L) for <17 yrs = (AG) x 9.6/100 x 0.2 (9.6 is the % leaf biomass and 0.2 is leaf turnover rate, 5 years = 0.2, (Black et al. in prep).
  - • L for > 17yrs = (AG) x 4/100 x 0.2.
  - • For the broadleaves the turn over rated of leaves was 1 (i.e. life span of 1 year).
  - • Brash (Br) is calculated as 11 % of above ground (AG) biomass of harvested pool (i.e. HARVEST x 0.8 x 0.11), where 0.8 is the shoot to root ratio for > 17 yrs old stands.

Results

Table 7.1 shows projected CO_2 sequestration (Mt) over the first three commitment periods using three different afforestation scenarios based on current research information.

The new estimated annual C sequestrations rates for Irish forests are equivalent to 1.56 to 2.39 Mt CO_2 per yr-1 for the first commitment period (14000 ha scenario), which is higher than the previously reported CARBWARE values for the same period (1.54 Mt yr^{-1}, ICF/Byrne Ó Cléirigh report 2004). Assuming a CO_2 trading price of 20 euro per tonne, the potential return on investment derived from improvements to the model, based on outputs from CARBiFOR research, would be in the region of ~54.4 million euro for the first commitment period.

Table 7.1: Projected CO_2 sequestration (Mt) over the first three commitment periods using three different afforestation scenarios based on current research information.

	Scenarios		
	3000 ha	14000 ha	20000ha
2008	1.66	1.56	1.51
2009	1.90	1.84	1.81
2010	2.15	2.13	2.13
2011	2.41	2.45	2.47
2012	2.29	2.39	2.44
	10.42	10.37	10.35
2013	2.44	2.62	2.71
2014	2.73	2.99	3.13
2015	2.97	3.33	3.53
2016	3.10	3.57	3.83
2107	2.81	3.42	3.74
	14.04	15.93	16.95
2018	2.95	3.70	4.11
2019	3.38	4.29	4.79
2020	3.51	4.59	5.18
2021	3.55	4.82	5.51
2022	3.09	4.56	5.37
	16.49	21.97	24.95
Total	40.95	48.27	52.26

C) POLICY RELATED ISSUES

As signatories to the Kyoto Protocol, European countries (i.e. the EU15 that were members before May 2004) are required to reduce their collective greenhouse gas (GHG) emissions by at least 8% below 1990 levels by the first commitment period 2008 to 2012. Under the EU burden sharing agreement, Ireland's target is to limit its growth in emissions to 13% above 1990 levels, which is equivalent to a total of 60.365 Mt of CO_2 per annum. The most recent and comprehensive forecast of emissions over the first Kyoto commitment period were prepared by ICF/Byrne Ó Cléirigh and published in February 2004. Given the rapid growth of the Irish economy in the past decade annual CO_2 equivalent emissions for Ireland in 2001 was 70.018 Mt. It is suggested that, in the absence of emissions trading and offsets by the forestry sector in terms of Article 3.3, the Kyoto target can be overshot by ~ 9 Mt yr^{-1}. Under the agreed terms of the protocol carbon sequestration by forests may be used to offset greenhouse gas emissions. For the first commitment period carbon sequestration by these 'Kyoto' (Article 3.3) forests include deforestation, reforestation and afforestation activities since 1990. Assuming the business-as-usual scenario and an annual afforestation rate of 14000 ha (average C sequestration rate of 2.06 Mt yr^{-1} for the first commitment period), it is estimated that the contribution of Article 3.3 forests may account for ~ 20% of the required reduction in national emissions for Ireland to meet its Kyoto target.

> *It should be noted that the CARBWARE model is continuously being improved, based on new research information and NFI data. It is possible that the approach and parameters used in CARBWARE may have changed by the time this report is published.*

D) FUTURE DIRECTIONS

Given CARBiFORs emphasis on the chronosequence approach, it was be possible to determine the changes in biomass and soil carbon stocks over time and to relate those to the observed fluxes measured by eddy covariance for a dominant afforested soil type. Although these results are useful for providing a baseline for carbon sequestration potential for the dominant forest species in Ireland, i.e. Sitka spruce planted on wet mineral soils, more information on carbon stocks and fluxes for different soil types, and other associated species is required. Clearly, a critical evaluation of biomass and soil carbon stocks for different species and other soils is required for the reporting to the UNFCCC and Kyoto Protocol on LULUCF. In addition, validated estimates for biomass increment, particularly for younger forests (age 3 to 20 years) are required. Given the emphasis on young forests for reporting sequestration changes for the first commitment period, future projects should focus on afforested sites from planting to the first thinning cycle. Furthermore, the potential for integrating the work reported here as well as future work with the ongoing National Forest Inventory in order to improve the national C accounting system should be explored and implemented.

The provision of one eddy covariance tower as well as a short-term commitment to funding was a major limitation to the CARBiFOR project because this could not provide enough information to characterise the fluxes of water and carbon dioxide over the chronosequence. As initially proposed, it will be important to have at least two towers: a fixed tower used as a reference, and a second system employed to "roam" among different sites, to provide for spatial coverage. In addition future work on CO_2 fluxes would be greatly strengthened by the inclusion of automated chamber systems for continuous measurement of soil CO_2 fluxes and resources to study fine root and soil C processes. Also, with regard to flux measurements, attention will need to be given to sampling, at least for short periods, of duplicate age classes on different soils to characterise stand spatial variability. This aspect also calls for extended funding as well as a second eddy system and tower. The provision of additional and long term eddy covariance data would also facilitate the development of comprehensive carbon balance models that can be used to simulate the effect of stand age, disturbance and climate change scenarios on forest carbon fluxes. The development of these models would be aimed at end users, such as forest managers and policy makers, to aid in management decisions and reporting procedures to the UNFCCC and climate change risk assessment. Beyond changes associated with stand age, there is a need to incorporate an analysis of the effect of disturbance, particularly those related to management practices (e.g. thinning), and land-use change such as the transition from grassland to forest.

CARBiFOR has not only opened the door to the potential of Irish forests to sequester C but has also highlighted the information gaps that remain. Much remains to be done before we have a comprehensive understanding of the role of Irish forests in mitigating GHG emissions. Future efforts should not only focus on C sequestration in soils and biomass but also on the role of forest biomass as a renewable energy source and its relationship to forest productivity and C sequestration.

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