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PHYSICAL AND BIOECONOMIC ANALYSIS OF ECOSYSTEM SERVICES FROM A SILVOPASTURE SYSTEM

A dissertation submitted in partial fulfilment of the requirements for the
degree of Doctor of Philosophy (Ph.D.) in Agroforestry, Bangor University

By

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Submitted in April 2017

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ABSTRACT

The aim of this study was to evaluate some of the physical and bioeconomic potentials of a silvopastoral agroforestry system with focus on the Henfaes Silvopastoral Systems Experimental Farm (SSEF) of Bangor University, North Wales.

The study reviewed research studies written on the SSEF from 1992 to 2012; assessed changes in pasture species composition and abundance since establishment; developed allometric equations for the estimation of aboveground biomass (AGB), carbon (C) stock and carbon dioxide (CO₂) emission potentials of red alder (*Alnus rubra* Bong); studied the effect of tree/solar radiation on pasture productivity and quality; and conducted bioeconomic analysis to compare treeless pasture/livestock, forestry, and agroforestry scenarios.

Review of the research studies show that as far as can be determined 66 research studies were conducted on ecosystem services of the UK's Silvopastoral National Network Experiment (SNNE) and temperate Europe during the period 1992 - 2012. These papers were sourced mainly from the Henfaes SSEF, the UK's SNNE, other UK and, other European research sites. The studied ecosystem services dealt with provisioning services (40%), regulating services (13%), and supporting services (47%). The scientific domains addressed include timber or wood-fuel potential (20%), pasture/livestock management (20%), biodiversity (20%), carbon sequestration (13%), water management (15%), and soils (12%).

The response of pasture species to thinning varied. The percentage composition by weight of the sown species declined, while that of the grass weeds and the forb weeds increased

slightly one year after thinning (2013 – 2014) compared to the adjacent open pastures. The change was not statistically significant. The understory pasture species composition, abundance and diversity changed significantly 20 years (1992 – 2012) after the establishment of the Silvopastoral National Network Experiment at Henfaes. Generally, pasture on the three red alder blocks was found to be largely grass weeds (46-48%) followed by forbs or broadleaf weeds while the sown species declined significantly.

In 2012, 20 years after field planting, the mean AGB were found to vary from 130 kg tree⁻¹ (26 Mg ha⁻¹) to 246 kg tree⁻¹ (49 Mg ha⁻¹) in poor form and good form red alder trees, respectively, based on a stocking density of 200 stems ha⁻¹. Mean C stock was 65 kg C tree⁻¹ (13 Mg C ha⁻¹) in poor form trees and 123 kg C tree⁻¹ (25 Mg C ha⁻¹) in good form trees. Mean CO₂ potential was 237 kg CO₂ tree⁻¹ (48 Mg CO₂ ha⁻¹) in poor form trees and 450 kg CO₂ tree⁻¹ (90 Mg CO₂ ha⁻¹) in good form trees.

Pasture productivity increased significantly with increasing solar transmission, and with increasing distance from each grazing exclusion cage to the nearest alder tree. Concentration and availability of CP, ADF, NDF and ME were greater in the *with-leaves* than in the *without-leaves* growing seasons in response to variation of photoperiod (the duration of sunshine/day length) in the United Kingdom.

The bioeconomic analysis considered three land-use plausible scenarios ('forestry', 'pasture / livestock' and 'agroforestry') and found that, in the absence of grants/subsidies, none were viable. However, application of grants/subsidies, at the baseline assumptions, revealed that forestry was the most viable option with the highest net present value and annual equivalent value, followed by pasture/livestock and agroforestry options.

DEDICATION

This dissertation is dedicated to my beloved late parents, Aaron Onunkwo Nworji and Rosaline Ekemma Nworji, for their invaluable contribution towards my home upbringing achievements. Also, to my beloved wife, Susan Barredo Nworji, for her encouragement, understanding, patience, moral and material support and readiness to share difficulties that I encountered in the course of my study. Similarly, it is dedicated to my sons, Michael Chukwuemeka Barredo Nworji and Augustine Chinedu Barredo Nworji, and to my daughters, Susan Ngozika Barredo Nworji and Angela Ijeoma Barredo Nworji, for their endurance, concern and prayers. Thank you all very much indeed and may the Almighty God bless us all abundantly.

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Engaging in this research has been one of the most momentous academic challenges I have ever encountered in my life. The completion of this dissertation would not have been possible without the encouragement, assistance and guidance of the following people and organisation to whom I owe my sincerest gratitude.

Foremost, I owe my deepest gratitude to my supervisors, Dr. James Walmsley and Dr. Mark Rayment, for their sustained guidance, wisdom, thoroughness, support, patience and commitment during the writing of this dissertation despite their many other academic commitments. To the members of my supervisory committee assessment, Professor Andrew Pullin and Dr. Robert Brook for their professionalism and unbiased critical assessment of my research works. I am also very grateful to Professors John Healey and Morag McDonald, Drs. Neal Hockley, Tim Pagella, Christine Cahalan, Hussain Omed, Fergus Sinclair, Zewge Teklehaimanot, and Mr. James Brockington for their motivation, assistance and encouragement. To other staff and fellow students of the School of Environment, Natural Resources and Geography (SENRGY) at Bangor University who played significant roles during my studies.

My special thanks will go to Ian Harris, Mrs. Llinos Hughes, Mark Hughes, and all other staff at the Bangor University research farm at Henfaes (the site of the Silvopastoral National Network Experiment) for providing me with useful information on the management of the research farm and for their support, assistance and patience during my data collection process and laboratory works. To Mrs. Helen Simpson for providing me with the necessary materials and equipment for field data collection as well as for her invaluable assistance with

data preparation for analysis in the laboratory. To Mr. Edward Ingram for his useful suggestions.

My greatest thanks go to the Farm Woodland Forum for awarding me the Lynton Incoll Memorial Scholarship that enabled me to present my research paper at the 2014 Farm Woodland Forum annual meeting in Devon. I am equally grateful to all the researchers, staff, students and volunteers who have contributed to the huge body of knowledge that has been generated by the Henfaes Silvopastoral National Network Experiment.

I am deeply indebted to the Nigerian Tertiary Education Trust Fund (TETFUND) for sponsoring this research under its Academic Staff Training and Development Programme as well as to the Management of Chukwuemeka Odumegwu Ojukwu University (COOU) (Former Anambra State University), Anambra State, Nigeria for granting me the leave to undertake this research programme. It very important to give special recognition to the Vice Chancellor of COOU, Professor Fidelis Okafor and the Dean of the Faculty of Agriculture, Dr. Uzuegbunan, and the erstwhile Dean of the Faculty of Agriculture, Professor Nnabuife, for their unflinching support and inspiration.

My gratitude also goes to all my colleagues and friends, and other nice helpful people, too numerous to mention here, for their immense contributions to the success of this work. My deepest gratitude and appreciations are also due my spouse, Susan Barredo Nworji, and children, Ngozi, Chukwuemeka, Chinedu and Angela, for their moral, material and financial support as well as for their continuous inspiration, encouragement understanding and patience.

Finally, I give gratitude to the Lord for giving me enough spirit and strength to surmount the various obstacles encountered in making the completion of this manuscript a reality.

ABOUT THE AUTHOR

My name is Michael Jide Nworji, popularly known among my close acquaintances as the “Commander-In-Chief” (much to my embarrassment). I hail from the South-Eastern region (Anambra State) of Nigeria, where I grew up.

I attended the University of the Philippines, Los Banos, Laguna and the De La Salle Araneta University, Malabon Manila, where I received my BSc in Forestry and MSc in Forest Resources Economics and Management, respectively. I have been a member of the Nigerian Forestry Association since the 1990s.

Since the completion of my postgraduate studies in the Philippines, I have lived and worked as a Forest Management Consultant in an array of locations from Manila in Luzon, to Tacloban City in Leyte, Philippines, to Anambra State in Nigeria. I have worked in various capacities as Agroforestry Management Consultant, Watershed Research/Development Officer, Horticulturist, Resource Person, Facilitator, Instructor and Trainer, and possess several years of forestry experience in Nigeria and Philippines.

I specialize in Forest Resources Economics and Management and Agroforestry, and have skills in Project Development, Planning, Management and Administration, and Project Monitoring and Evaluation, plus expertise in Hydroponics Nursery Greenhouse, Plant Propagation, Horticulture, and Urban Forestry.

I switched from forest management consultancy to University lecturing because of my keen aspiration to advance my career in the academic circle where there are greater prospects and

more challenging and interesting work as well as my desire to devote my knowledge, trainings and skills to the service of my country as University Lecturer.

University lecturing enabled me to secure the Nigerian Academic Staff Training and Development Grant to pursue PhD programme in Agroforestry in Bangor University, Gwynedd, United Kingdom where I conducted research on ecosystem services provided by silvopastoral agroforestry systems.

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LIST OF ACRONYMS

ADF:	Acid detergent fibre
AGB:	Aboveground biomass
AIC:	Akaike Information Criteria
AICc:	Akaike Information Criterion with correction
BPS:	Basic Payment Scheme
BR:	Bifurcation ratio
C:	Carbon
CA:	Crown area
CF:	Correction factor
cm:	Centimetre
cm ² :	square centimetre
CO ² :	Carbon dioxide
CP:	Crude protein
C ₂ H ₄ :	Ethylene
<i>D'</i> :	Simpson's diversity index
DBH:	Diameter at breast height
DM:	Dry matter
DSF:	Direct site factor
dwt ⁻¹ :	Unit dry weight
E':	Shannon-Wiener evenness index
EH:	Shannon equitability index
EURAF:	European Agroforestry Federation
FAO:	Food and Agriculture Organisation of the United Nations
FNdfa:	Fraction of nitrogen derived from the atmosphere

g:	gramme
GF:	Gap fraction
GHG:	Greenhouse gas
GPS:	Global positioning system
GSF:	Global site factor
H':	Shannon-Wiener diversity index
ha:	Hectare
HT:	Tree height
IPCC:	Intergovernmental Panel on Climate Change
ISF:	Indirect site factor
LAI:	Leaf area index
m:	Metre
m ² :	Square metre
m ³ :	Cubic metre
mol:	Mole- amount of substance
ME:	Metabolisable energy
mg:	milligramm
MJ:	Megajoule
NDF:	Neutral detergent fibre
NIRS:	Near infrared reflectance spectroscopy
PAR:	Photosynthetically active radiation
R':	Species richness
SNNE:	Silvopastoral National Network Experiment
SSEF:	Henfaes's Silvopastoral Systems Experimental Farm
μmol:	Micrmole

UN: United Nations
VIF: Variance inflation factor
WD: Wood density

Chapter 1 : PHYSICAL AND BIOECONOMIC ANALYSIS OF ECOSYSTEM SERVICES FROM A SILVOPASTURE SYSTEM

1.1. INTRODUCTION

There are differences between countries in the approach of their farmers, private and public tree planting programmes, and Government subsidised schemes to the application of agroforestry. Crucial and well-defined agroforestry research programmes have often been compromised or subsequently discarded because of conflicting policy or funding priorities. This problem appears to be more marked and rampant in larger nations where the enactment and administration of such long term programmes requires considerable foresight, liaison and persistence by committed individuals. Generally, there is a changing perception of agroforestry as the wide ranging and long-term benefits are brought to the fore.

Agroforestry is a collective term for a land-use pattern in which woody perennials (trees, shrubs, palms, bamboo, etc.) are grown in association with herbaceous plants (crops, pasture) or livestock, in a spatial arrangement or temporal sequence (rotation), or both. There are both ecological and economic interactions between the trees and other components of the systems. Agroforestry system practices have been defined by different authors (Nair, 1993) as practices which involve “*the deliberate integration of trees with agricultural crops and/or livestock either simultaneously or sequentially on the same unit of land*”. A complementary definition is given by the World Agroforestry Center (WAC) (Leakey, 1996, 1997) as a “*Dynamic, ecologically based, natural*

resources management system that, through the integration of trees on farms and in the agricultural landscape, diversifies and sustains production for increased social, economic and environmental benefits for land users at all levels.”

Within the agroforestry concept, silvopastoral systems are those where trees are combined with forage and livestock production on the same land management unit. Silvopastoral systems are deliberately designed and managed to produce high-value wood products (such as sawlog) in the long-term while providing short-term annual economic benefit from a livestock component through the management of forage or an annual crop component. The tree component provides shade and shelter for livestock and forage, reducing stress and potentially increasing forage production. Spatial and temporal interactions among trees, forages and livestock components, when properly managed, can enhance overall productivity compared to conventional livestock alone or timber investments, while providing both regular income from livestock and/or non-timber forest products and intermittent income from timber sales (Arbuckle, 2009). Furthermore, silvopastoral systems have been shown to have the potential to enhance biodiversity by increasing the structural and species diversity of landscapes that were previously one-dimensional grassland (McAdam *et al.*, 2007). Based on the context of the definitions, agroforestry can further be regarded as an intervention and silvopastoral systems as the link between the system components of trees and livestock (Devendra and Ibrahim, 2004). Integrating trees, forage, and livestock creates a land management system that can produce marketable products while at the same time maintain long-term productivity. Economic risk is reduced because the system produces multiple products, most of which have an established market. Though uncertainties like natural disasters (fire, wind blow), pests and diseases, animal damage

or theft as well as economic risks such as price, supply and demand, regulatory and liquidity risks could affect the viability of investments in agroforestry. Although production costs are increased remarkably by distributing management costs between the timber and livestock components, marketing flexibility is enhanced. Comprehensive land utilization in silvopastoral systems provides a relatively constant income from livestock sale and selective sale of trees and timber products. Well-managed forage production provides improved nutrition for livestock growth and production. Research studies conducted in various settings worldwide have demonstrated that agroforestry systems are financially and economically viable and attractive land use options (e.g. in Qun, 1991; Willis *et al.*, 1993; Knowles and Middlemiss, 1999; Burgess *et al.*, 2003; Grado and Husak, 2004).

It is only recently that agroforestry systems, especially for temperate climates, began to receive much attention in recognition of the wide range of ecosystem services that trees can provide (Jose, 2009; Smith *et al.*, 2012a). Interest in agroforestry in the UK rekindled in the mid 80's. It was seen as a potential land-use system, which would reduce agricultural surplus in the European community, increase quality timber production, environmental diversity and protect rural employment (Sibbald & Sinclair, 1990). Burgess (2012) noted that the UK, but particularly England, Wales and Northern Ireland already comprises an agroforestry landscape. He went further to observe that currently the greatest opportunities for agroforestry in the UK relate to systems offering animal welfare and environmental benefits, and new methods of reinvigorating traditional hedge, orchard, parkland and wood pasture systems (Burgess, 2012). The reported success of silvopastoral systems in other temperate environments, e.g. New Zealand (Knowles, 1991) indicated that such systems could be applied in the UK.

The agroforestry research programme came under the stewardship of the UK Discussion Forum (UK Farm Woodland Forum); an informal group of scientists who set up a national network of silvopastoral experiments (UK's Silvopastoral National Network Experiment). Much of the research has involved detailed studies of ecological and physical processes. More recently, the Farm Woodland Forum has become affiliated with the European Agroforestry Federation (EURAF). The EURAF has about 280 members from 20 different European countries and aims to promote the use of trees on farms as well as any kind of Silvopastoralism throughout the different environmental regions of Europe (EURAF, 2012).

The rationale behind the setting up of the UK's Silvopastoral National Network Experiment (SNNE) was to provide knowledge, information and experience on the establishment of silvopastoral systems over a range of climatic and edaphic conditions in the UK using, wherever possible, common treatments and management protocols (Sibbald et al, 2001). The network of six sites (Figure 1.1) in the UK has common treatments and is run to agreed protocols. Sycamore (*Acer pseudoplatanus* L.) is the common tree species at all sites.



Figure 1-1 Sites of the UK's Silvopastoral National Network Experiment

The idea to establish a national network experiment was a step in the right direction to inform research and practice in this much-understudied area. However, no attempt has been made to date to examine the ecosystem services such a system could provide, and the financial and economic implications of transitioning from conventional pasture grazing system to silvopastoral system. This study will therefore evaluate some of the physical and bioeconomic potentials of a lowland silvopastoral agroforestry system in the United Kingdom.

1.1.1. General objective:

To investigate the ecosystem service potentials of the Silvopastoral National Network Experiment (SNNE) at Henfaes in North Wales with concentration on the red alder (*Alnus rubra* Bong) component. Red alder was chosen as the ‘optional’ species for the Henfaes site of the UK Silvopastoral National Network Experiment (sycamore being the ‘standard’ species found across all six sites) primarily because of its fast growth rate, ability to fix atmospheric nitrogen (Smith, 1968), tolerance of wet sites and potential to produce a range of quality wood (Mmolotsi and Teklehaimanot 2006) and maximum fibre yield (Gordon, 1978).

Specific objectives:

1. Review and synthesize research papers written on the Henfaes SNNE since inception.
 2. Evaluate the temporal and spatial changes in botanical composition of pasture species in alder plots over time – initial vs 20 years (pre-and post thinning).
 3. Develop biomass allometric equations for open-grown red alder.
 4. Study the effect of light on pasture productivity and quality in red alder blocks
- Conduct a bio-economic analysis to compared the economic viability of conventional livestock grazing, forestry, and silvopastoral agroforestry investment options

1.1.2. Thesis organisation

This thesis is organised in seven chapters. Chapter 1 introduces the background of the studies while Chapter 2 reviews and synthesises research papers written on the UK’s Silvopastoral National Network Experiment since inception. Chapter 3 evaluates the changes in pasture species composition and abundance in red alder blocks since

establishment. Chapter 4 develops allometric equations to estimate the aboveground biomass, carbon stock, and carbon dioxide emission potentials of two forms of open-grown red alder in a silvopastoral system. Chapter 5 evaluates the influence of solar radiation on understorey pasture productivity and quality in thinned red alder blocks. Information gathered from previous two chapters were used in Chapter 6 in the conduct of bioeconomic analysis to evaluate conventional pasture grazing system against preferred agroforestry system. The synthesis of the research findings is presented in Chapter 7.

1.1.3. Study area description

The study was conducted at the Silvopastoral National Network Experiment (SNNE) at Henfaes in North Wales, which is one of six National Network Experiments established across the country with trees planted at different arrangements and densities to investigate the potential of silvopastoral agroforestry on UK farms (Sibbald and Sinclair, 1990). The site (53°14'N 4°01'W) [Figure 1.2] is located in Abergwyngregyn, Gwynedd, approximately 12 kilometres east of Bangor City in North Wales, United Kingdom. The site was established in 1992 on 14.47 ha of agricultural land at Henfaes, owned by the Bangor University, Wales.



Figure 1-2 The location of the Henfaes study site.

Topography consists of a shallow slope of 1:20 on a deltaic fan and the aspect is northwesterly, at an altitude of 4-14 m above sea level. The site's climatic

characteristics are hyperoceanic with annual rainfall of 1000 mm. The climatic variables of the site for the period 2012 – 2014 are presented in Table 1.1 and Figure 1.3 below. Soil is a fine loamy brown earth over gravel (Rheidol series) classified as a Dystric Cambisol in the FAO system (Teklehaimanot and Mmolotsi, 2007). The parent material consists of postglacial alluvial deposits from the Aber River, comprising Snowdonian rhyolitic tuffs and lavas, microdiorites and dolerite in the stone fractions and Lower Paleozoic shale in the finer fractions. The ground water table at the site ranges between 1 and 6 m.

Table 1.1 Climatic variables in the study area (2012 – 2014)

Month/Annual	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual total
Relative humidity (%)	76	76	74	73	75	76	77	77	78	78	78	78	-
Rainfall (mm)	48	68	50	25	45	42	52	78	83	102	108	114	815
Light ($\mu\text{mole m}^{-2}\text{d}^{-1}$)	7.1	14.6	27.3	42.7	49.0	56.7	64.1	46.9	28.1	14.8	7.5	5.0	363.9
Min temperature (°C)	3.4	3.8	4.0	5.5	7.8	10.2	12.7	12.7	10.4	9.7	6.3	5.2	-
Max temperature (°C)	8.8	9.0	10.4	11.8	14.8	15.6	20.0	19.1	17.5	15.1	11.4	10.1	-
Mean temperature (°C)	6.1	6.4	7.2	8.7	11.3	12.9	16.4	15.9	14.0	12.4	8.8	7.7	-

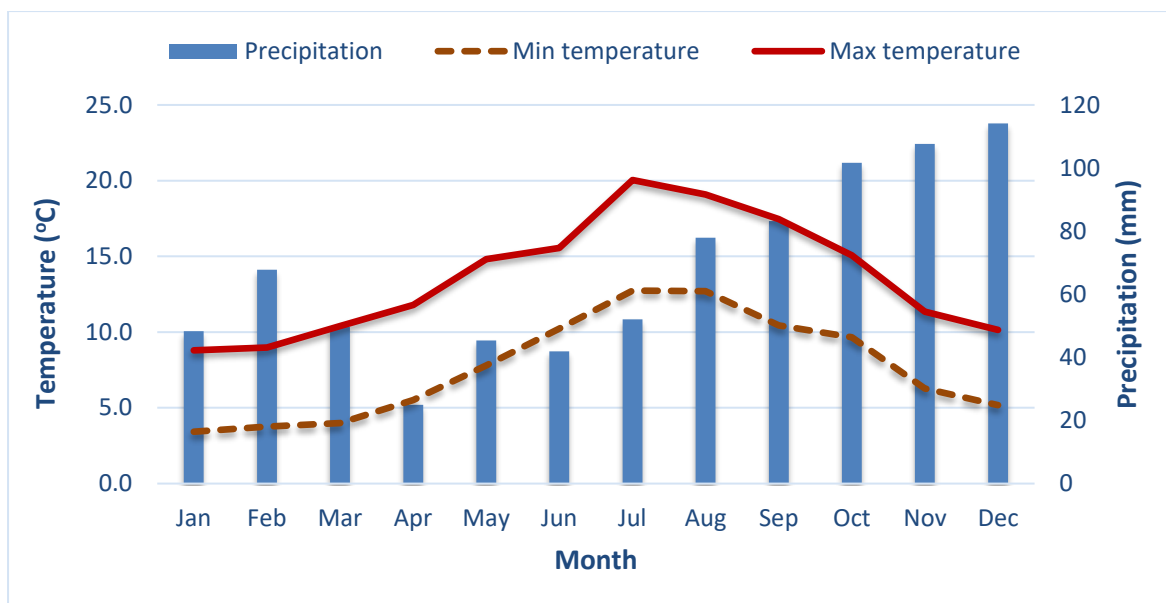


Figure 1-3 Climatic graph of the study area during the period 2012 - 2014

Sycamore (*Acer pseudoplatanus*) and red alder (*Alnus rubra*) were planted on the site at establishment in 1992 to investigate their use in agroforestry systems. Both species were chosen because they are fast growing broadleaf, medium strength with potential to grow well over the wide range of sites represented in the Network. The treatments include:

1. Sycamore agroforestry planted at 100 trees per hectare with sheep grazing;
2. Sycamore agroforestry planted at 400 trees per hectare with sheep grazing;
3. Sycamore agroforestry planted in clumps;
4. Sycamore farm woodland control planted at 2500 trees per hectare with no grazing;
5. Treeless agricultural control with sheep grazing;
6. Red alder agroforestry planted at 400 trees per hectare with sheep grazing;
7. Red alder farm woodland control planted at 2500 trees per hectare with no grazing.

Each treatment is 0.42 ha while the woodland control plot is 0.1 ha. Trees are individually protected only in agroforestry treatments but in Woodland Control treatments, fences exclude grazing and browsing animals. All treatments are replicated three times in a complete randomised block design.

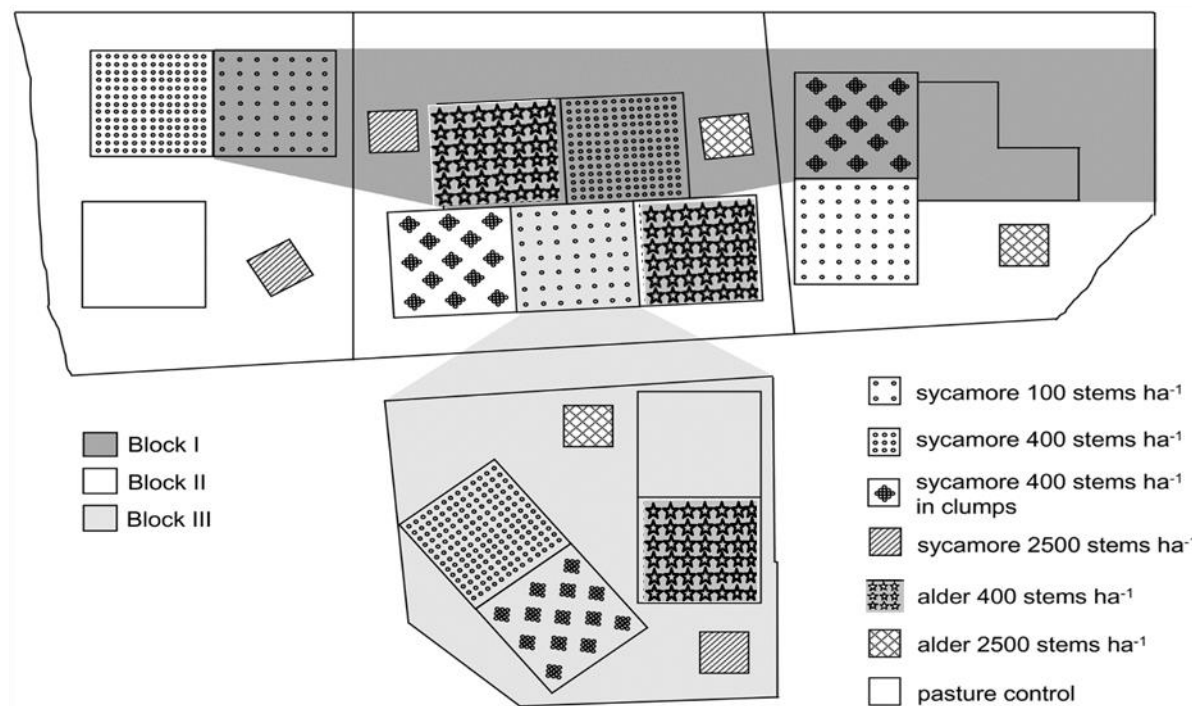


Figure 1-4 Diagram showing layout of silvopastoral experiment at Henfaes SNNE

At establishment in 1992, the entire site was sown with a mixture of perennial ryegrass (*Lolium perenne* L.), Talbot and Condessa varieties, and white clover (*Trifolium repens* L.), Gwenda and S184 varieties (Teklehaimanot et al., 2002). The species were sown at a range of 29 kg/ha (12.5 kg/ha of Talbot ryegrass, 12.5 kg/ha of Condessa ryegrass, 2 kg/ha of Gwenda clover and S184). The pasture has not been reseeded ever since. The grazing period lasts six months, from March to October; individual plots were fenced for the first eight years to closely control grazing – these fences were removed in 2000 and sheep can now move between treatments.

At establishment in 1992, all blocks had extra fertilization of N (160 kg/ha in four aliquots), except red alder blocks (which are N fixing trees) where no N was added. All blocks except forestry controls received treatments against weeds: Yorkshire fog (*Holcus lanatus*), spear thistle (*Cirsium arvense*) and common nettle (*Urtica dioica*) were treated with “Grazon 90” once at the end of March in 1993.

Red alder (*Alnus rubra* Bong) was introduced in 1992 to investigate the use of biological nitrogen fixation as an alternative to chemical fertilizer as well as for its rapid early growth rate, tolerance of wet sites and wide range of quality wood products. The red alder that was originally planted at 400 stems ha⁻¹ across three blocks (figure 1.4) were selectively thinned to 200 stems ha⁻¹ in 2000 and subsequently to 100 stems ha⁻¹ in the winter of 2012, respectively, primarily to improve the health and productivity of both the trees and the understorey pasture as well as to provide data for the construction of biomass allometric equations for open-grown red alder trees. All selected trees were cut at ground level by chainsaws, dragged from the plots, piled and chipped/processed for firewood.

In addition, crown-lifting operations have been routinely conducted as the need arose throughout the life of the crop. Stem diameter at breast height, total tree height, basal area and total volume variables were measured for each tree within a block. Measurements ranged from 15 cm to 43 cm in stem diameter and from 8 m to 14 m in tree height. The characteristics of the three alder blocks are summarised in Table 1.2.

1. 1: Thinning of the red alder blocks from 200 stems ha⁻¹ to 100 stems ha⁻¹ in 2012

Location	Category	No. of Trees	DBH (cm)	Height (m)	Basal Area (m ²)	Total Volume (m ³)
Block 1	Pre -thinning	86	22.0 - 43.0	8.5 - 13.5	0.04 - 0.15	0.11 - 0.58
	Thinned	43	22.0 - 43.0	8.5 - 13.0	0.04 - 0.15	0.11 - 0.58
	Retained	43	24.0 - 39.0	11.0 - 13.5	0.05 - 0.12	0.18 - 0.54
Block 2	Pre-thinning	88	20.0 - 37.0	8.6 - 13.5	0.03 - 0.11	0.09 - 0.48
	Thinned	45	20.0 - 37.0	8.6 - 13.5	0.03 - 0.11	0.09 - 0.46
	Retained	43	25.0 - 27.0	11.0 - 13.5	0.05 - 0.11	0.18 - 0.48
Block 3	Pre-thinning	85	15.0 - 38.0	8.3 - 14.0	0.02 - 0.11	0.05 - 0.53
	Thinned	43	15.0 - 37.0	8.3 - 13.5	0.02 - 0.11	0.05 - 0.48
	Retained	42	23.0 - 38.0	10.0 - 14.0	0.04 - 0.11	0.17 - 0.53
Blocks Pooled						
	Pre-thinning	259	15.0 - 43.0	8.3 - 14.0	0.02 - 0.15	0.05 - 0.58

The pasture is grazed by Welsh Mountain Ewes (Sheep) with single-cross bred lambs (Sibbald *et al.*, 2001). Sward height was maintained to between 3 and 6 cm governed by the UK National Network protocol, by adjustment of additional ewe and lamb numbers from a buffer flock.

Chapter 2 : REVIEW OF TWENTY YEARS OF ECOSYSTEM SERVICES RESEARCH AT HENFAES SILVOPASTORAL NATIONAL NETWORK EXPERIMENT

2.1. INTRODUCTION

The UK's Silvopastoral National Network Experiment (SNNE) was set up with a view to studying the potential of silvopastoral agroforestry on UK farms (Sibbald and Sinclair, 1990). Over the past two decades (1992-2012), much of the on-farm and on-station research efforts have involved detailed studies of ecological and physical processes, with a view to establishing a solid knowledge base on the functions and capabilities of silvopastoral agroforestry. However, no attempt has been made to date to synthesise and publicize this knowledge and this has led to a lack of appreciation of the environmental benefits of this land-use system.

This paper aimed to review and synthesise the state of current knowledge of ecosystem services of the UK's SNNE with specific focus on the Henfaes's Silvopastoral Systems Experimental Farm (SSEF) of Bangor University, Wales. The paper evaluates the status of the research in the farm's ecological and physical processes to establish what has been done to date, the gaps in our knowledge, and the priorities for future research. Overall, the following discussion uses the ecosystem services framework and relates the four major categories of ecosystem services (provisioning, regulating, cultural and supporting) identified by the UK National Ecosystem Assessment (2011) and the Millennium Ecosystem Assessment (2005) to the scientific domain of the research studies.

The review and synthesis of the ecosystem service issues addressed by the UK's SNNE, Henfaes SSEF and other studies in temperate Europe, along with the variables and nature of the studies, are aimed at bringing the knowledge to the fore that would undoubtedly lead to better understanding of the economic and environmental implications of silvopastoral systems.

2.1.1. Objectives

The objective was to conduct an in-depth review of research papers and articles written on the Henfaes SSEF during the period 1992 to 2012, to answer three questions:

- What has been done to date?
- What are the benefits and contributions to our knowledge base?
- What are the gaps and priorities for future research?

2.1.2. Information sources considered

This chapter presents a review of research and synthesis of the state of current knowledge of ecosystem services of the UK's SNNE with specific focus on the Henfaes's Silvopastoral Systems Experimental Farm (SEEF) of Bangor University, Wales. Other studies in the UK as well as in temperate Europe with similar environmental conditions to the UK are also included in this review. The review and synthesis of the ecosystem service issues addressed by the UK's SNNE, Henfaes SSEF and other studies in temperate Europe, along with the variables and nature of the studies, are aimed at bringing the knowledge to the fore that would undoubtedly lead to the appreciation of the economic and environmental benefits of silvopastoral systems, and hence more attention being paid to accelerating its adoption and institutionalization in national rural development policies.

In order to appraise the current status of research studies on silvopastoral systems with respect to the ecosystem services framework, all available papers, published and unpublished since the inception of the UK's SNNE in 1988 were reviewed. The screening and compilation of available, peer-reviewed, and non-peer-reviewed research papers were made primarily by accessing various electronic databases and existing library collections.

Specific sources include databases maintained by the Bangor University libraries; the School of Environment, Natural Resources and Geography, Bangor University; UK's Farm Woodland Forum; World Agroforestry Centre (formerly ICRAF); and the Food and Agriculture Organisation of the United Nation (FAO). Furthermore, there was scanning of the titles of the journals of *Agroforestry Systems*, *Agroforestry Abstracts*, *Agroforestry Today*, *Agroforestry Forum*, and conference proceedings.

In order to ensure that all research that has been carried out was reviewed, further investigation was undertaken to extract additional relevant publications and grey literatures. Since the Henfaes's SSEF serves as an outdoor laboratory for Bangor University research students, restricting the review to only peer-reviewed articles would have missed many important contributions made to the field by these students who have produced many theses on the experimental farm. Therefore, by using both peer-reviewed and non-peer reviewed research outputs this paper examines what has been accomplished, what major questions have been addressed so far.

To facilitate analysis and synthesis, the research papers were classified according to the following criteria:

1. Thematic groups: The research papers were split into two major thematic groupings: peer-reviewed (published) and non-peer-reviewed (unpublished student theses) papers.
2. Ecosystem service functions addressed: research papers were further categorised into ecosystem service function groups in relation to the following economic and environmental benefits of silvopastoral systems:
 - Carbon sequestration
 - Livestock management
 - Timber or fuel potential
 - Soil improvement
 - Water management, and
 - Biodiversity enhancement

2.1.3. What has been done to date

The trend line, Figure 2.1 below, shows that greater number of annual studies on ecosystem services of the UK's Silvopastoral National Network Experiment and temperate Europe were conducted in the mid and late 1990s than in any other time over the 20-year study period. Interest in the topic remained generally minimal in the early 1990s and from 2001 to the end of the decade. However, there is indication of rising trend in academic involvement thereafter.

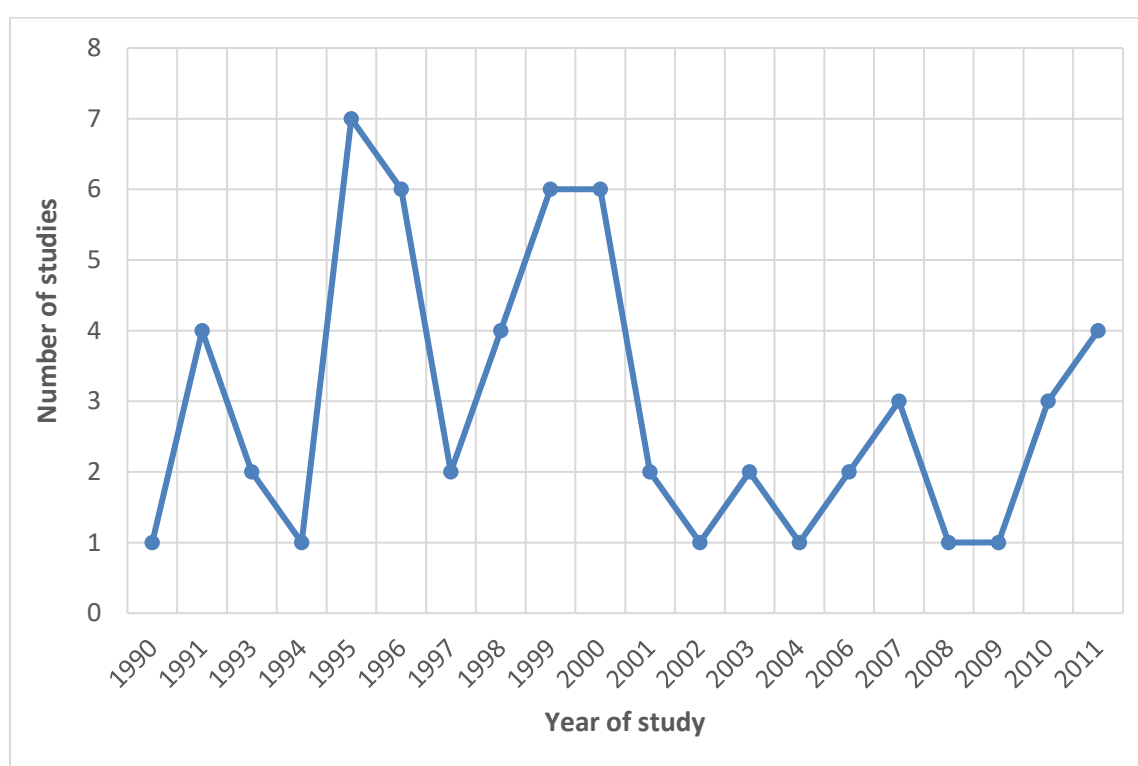


Figure 2-1: Number of identified scientific literature since 1988 on ecosystem services of the UK's Silvopastoral National Network Experiment and temperate Europe

Ecosystem services categories within different ecosystem systems services of silvopastoral systems trials are shown (Table 2.1, Figure 2.2 and Appendix 2.1), giving an overview on the major research question: What has been done to date?

Results of the categorisation of the research papers show that 66 research studies have been conducted since 1988 on ecosystem services of the UK's SNNE and temperate Europe (Table 2.1 and Appendix 2.1). Thirty (45%) of the 66 studies were produced based on studies at Henfaes SSEF, twenty-one (32%) at UK's SNNE, eight (12%) were from other silvopastoral systems trials in the UK, and seven (11%) were from European-wide silvopastoral systems studies. These 66 research studies are split into peer-reviewed (published) and non-peer-reviewed (unpublished) papers. 31 (47%) of these studies were classified as peer-reviewed, and 35 (53%) as non-peer-reviewed. The 35 non-peer-reviewed studies included 1 PhD thesis, 1 MPhil thesis, 20 MSc theses and 2 BSc theses at Bangor University and the rest 11 are information in various newsletters of the UK's SNNE. Only 2 BSc theses of Bangor University were included in this review because they were the only ones considered reliable and they were authenticated by the academic staff of the School of Environment, Natural Resources and Geography, Bangor University, Wales.

Table 2.1: Category of Research studies Reviewed

TYPE OF PAPER	Research site				TOTAL 1988-2012
	Henfaes SEEF	UK's SNNE	Other UK	Other Europe	
Peer- Reviewed	6	10	8	7	31
Non-Peer-Reviewed	24	11	0	0	35
TOTAL	30	21	8	7	66

Figure 2.2 shows the frequency of the different ecosystem services appearing in the 66 research studies and their share in ecosystem service categories. In all, 3 ecosystem service categories

and 6 different ecosystem service scientific domains have been studied. In general, 40 percent of the studied ecosystem service categories dealt with provisioning services, 13 percent with regulating services, and 47 percent with supporting services. However, the ecosystem service category of cultural service is yet to be studied. The most common ecosystem service scientific domains assessed in the sample are timber or wood-fuel potential (13 studies or 20%), pasture/livestock management (13 studies or 20%) and biodiversity enhancement (13 studies or 20%). Other ecosystem service domains studied include carbon sequestration (9 studies or 13%), soil improvement (8 studies or 12%), and water management (10 studies or 15%). It is not unusual to address more than one ecosystem service domain in a study (Figure 2.2 and Appendix 2.1).

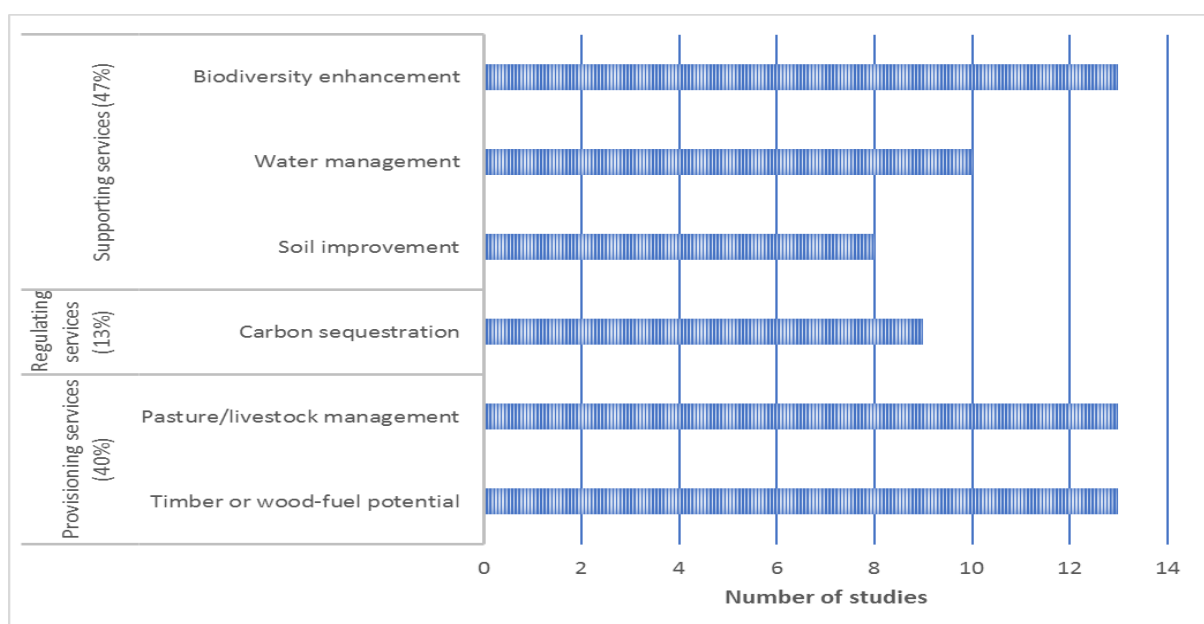
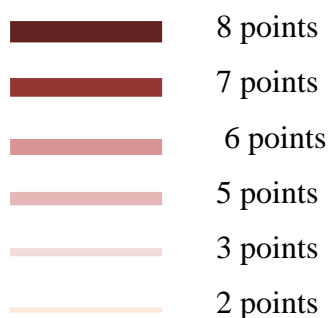


Figure 2-2: Frequency of the different ecosystem service domains appearing in the 66 publications and their share (%) in ecosystem service categories

2.1.4. Research domain and ecosystem services

The strength of linkages between categories of ecosystem service functions and components of scientific domain are illustrated in Figure 2.3. The scientific domain has multiple constituents including Tree, pasture and livestock productivity; Tree growth, form, phenology & wood properties; Carbon stock estimation; Water relation; Diversity of fauna; Nutrient composition and storage; Nitrogen-fixation & Nitrogenase activities; and Soil enrichment.

Arrow Width - intensity of linkages between scientific domain and key ecosystem service functions:



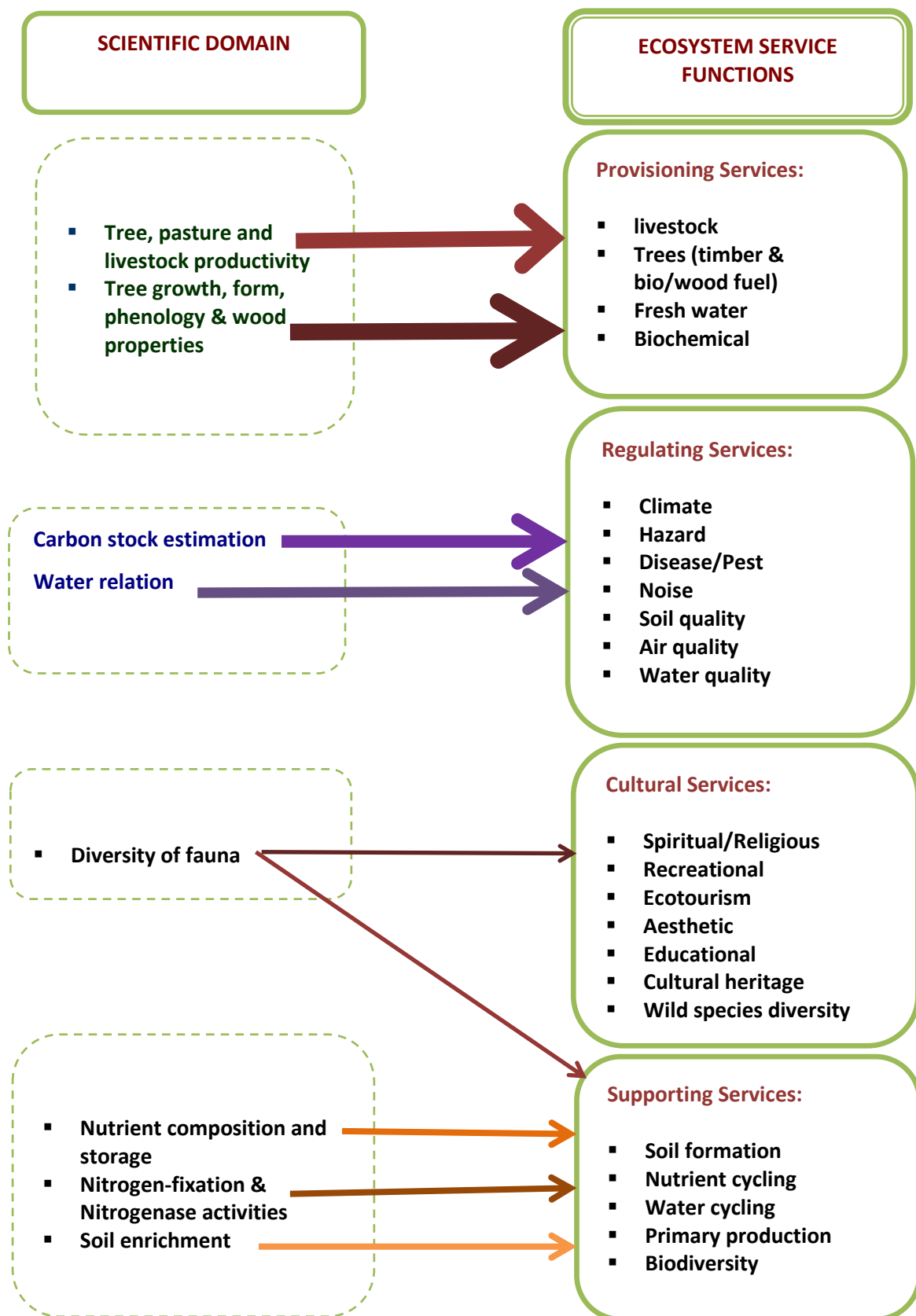


Figure 2-3: Linkages between Ecosystem Services and Research Scientific Domain

2.2. BENEFITS AND CONTRIBUTIONS TO KNOWLEDGE BASE

This section presents the results of the review on the benefits of silvopastoral agroforestry systems in relation to: timber or fuel potential, livestock management, carbon sequestration, water management, soil improvement, and biodiversity enhancement. In general, the discussion below cuts across the four major categories of ecosystem services (provisioning, regulating, cultural and supporting) identified by the Millennium Ecosystem Assessment (2005) and the UK National Ecosystem Assessment (2011).

2.2.1. Timber or fuel potential

Silvopastoral systems are designed to produce either timber or firewood, while providing intermediate cash flow from the livestock component.

The potential of growing timber or firewood species into pasture was investigated in the UK since 1988 as part of UK's SNNE. Sibbald *et al.* (2001) provided results of the performance of timber trees for the first six years (establishment phase) of the UK silvopastoral national network experiment. There were no significant differences in tree survival between the silvopastoral treatments and woodland control (mean 92.5% \pm 0.74). By year six, woodland control and trees at 100 stems ha⁻¹ were similar (180.7 \pm 17.31 cm) while trees at 400 stems ha⁻¹ were taller (219.0 \pm 22.80 cm: $p < 0.05$). It was concluded that tree shelters maintained silvopastoral tree survival at the level of conventional woodland. Tree height extension was, however, compromised on 100 stems ha⁻¹ plots where a higher animal: tree ratio resulted in greater animal activity and soil compaction around trees compared to 400 stems ha⁻¹ (Sibbald *et al.*, 2001).

Tree performance in relation to tree density and planting configuration in a silvopastoral system was also investigated at Henfaes SSEF by Roberts (1995), Englund (1995), Winslade (1996), Howe (1997), Ng'atigwa (1997), Zapater (1998), Gerety (1998), Islam (2000), Teklehaimanot et al. (2002), and Mmolotsi and Teklehaimanot (2006). Generally, the results of these studies indicated that tree performance within silvopastoral treatments was better at the higher planting density (400 stems ha⁻¹) and in silvopastoral plots with trees planted in clumped pattern. Stem diameter and tree height, which are indicators for tree growth, were generally better for all trees at 400 stems ha⁻¹ and for trees planted in clumped pattern, and that alder demonstrated better growth than sycamore. The authors attributed the poor performance of the wider spaced trees (100 stems ha⁻¹) to greater exposure to wind of widely spaced trees (Green et al. 1995), and to the effects of animals, either through browsing or soil compaction (Sibbald et al., 1995; Sibbald et al., 2001; Bezkorowajnyj et al., 1993).

The detailed results of the study by Islam (2000), who investigated the effect of spacing, planting pattern and sheep on tree growth, form and phenology of trees at Henfaes SSEF, showed that height and diameter did not vary significantly with treatments in sycamore or red alder seven years after planting. However, there were significant differences in tree form and phenology. A significantly higher height: diameter ratio was found in the woodland control than in the widely spaced 400 stems ha⁻¹ treatment in red alder. The number of shoot reiterations per tree (in total and the number of adaptive reiterations) were significantly lower in the woodland control than in two of the widely spaced silvopastoral treatments of both species (100 stem ha⁻¹ for sycamore and 400 stems ha⁻¹ for red alder) but in red alder the woodland control resulted in a significantly larger number of traumatic reiterations (The development of a shoot with a seedling growth form from the trunk, branch or root of a mature tree as a result of damage) per tree than in the 400 stems ha⁻¹ treatment. In red alder, the number

of dead branches and the rate of branch mortality were significantly higher in the 400 stems ha⁻¹ treatment than in the woodland control. Spacing also had significant effects on different phenological variables of both species. Shorter winter shoot dormancy periods were found in the woodland control than in the 100 stems ha⁻¹ treatment in sycamore, and a longer period of leaf production and a shorter winter dormancy period were found in the 400 stems ha⁻¹ than in the woodland control in red alder. In red alder, the 400 stems ha⁻¹ treatment resulted in a greater number of male catkin clusters and fruit clusters. The author concluded that tree forms were the best in the closely spaced woodland control and the least in the widely spaced silvopastoral treatment of 100 stems ha⁻¹ in sycamore. The author associated the poor performance in the 100 stems ha⁻¹ treatment with direct effects of animal activity that caused soil compaction and direct damage to the trees (Sibbald et al., 1995; Sibbald et al., 2001). The effect of livestock on soil compaction was also evaluated at Henfaes SSEF by Jarju (2000). Results showed that there was significant difference between treatments in soil compaction ($P < 0.001$). The highest mean penetrometer pressure weight was recorded at sycamore 100 stems ha⁻¹ (66.23 kg) followed by 56.70 kg and 57.15 kg in 400 stem ha⁻¹ alder and sycamore treatment plots, respectively.

Mmolotsi and Teklehaimanot (2006) assessed the timber and wood-fuel properties of red alder and sycamore at Henfaes SSEF. They found that tree-planting density had no significant effect on wood density and modulus of rupture in both tree species. However, tree-planting density had border line significant effects ($P < 0.05$) on the modulus of elasticity (MOE) in red alder and compression strength ($P < 0.01$) in sycamore. Wood samples taken from red alder in woodland control had a significantly higher MOE than those from trees in low-density plots of 400 stems ha⁻¹ (silvopastoral system). Sycamore wood from the woodland control had significantly higher compression strength than that from the 400 stems ha⁻¹ (silvopastoral) plots. In general, wood mechanical properties of red alder were found to be significantly

different from that of sycamore. Sycamore yielded higher wood density (0.64 g cm^{-3}), modulus of rupture (90.24 MPa) and compression strength (36.49 MPa) than red alder (0.49 g cm^{-3} , 73.48 MPa and 32.13 MPa, respectively). However, modulus of elasticity was higher in red alder (7614.64 MPa) than in sycamore (7430.05 MPa), although it was not significantly different. Based on the results of wood properties of red alder it was concluded that red alder is a medium strength tree species with potential for furniture manufacturing and for ordinary non-structural uses such as paneling and studs.

Planting density did not also have any effect on the wood-fuel higher heating value of either red alder or sycamore. However, red alder wood had a significantly higher fuel-value index (1638) than sycamore (1481), owing to higher ash content of sycamore (Mmolotsi and Teklehaimanot, 2006). Thus, it was concluded that red alder has a potential to provide a better bio-energy than sycamore for heating homes and generating electricity. From these results, it may be concluded that high quality timber and firewood can be produced from silvopastoral systems as most of the wood properties were not affected by planting trees at wide spacing.

As shown by the results of the studies at Henfaes SSEF by Roberts (1995), Englund (1995), Winslade (1996), Howe (1997), Ng'atigwa (1997), Zapater (1998), Gerety (1998), Islam (2000), and Teklehaimanot et al. (2002), planting sycamore trees in clumps rather than as individuals resulted in silvicultural advantages due to the proximity of adjacent trees within the clump at the same time as silvopastoral advantages of permitting grazing between the clumps. The clumps required less than half the cost of tree protection of individual trees in widely spaced treatments of 100 and 400 stems ha^{-1} , initial tree growth in clumps was not significantly different from the woodland control and livestock productivity in clumped treatment was not significantly different from the pasture control.

According to McAdam et al. (2007a), at year 15, hurley quality ash butts were sold for €1048 ha⁻¹ from North Ireland silvopastoral experiment site. Hence, the authors concluded that silvopastoral systems have the potential to support rural wood-based industry.

Because of long rotation period for trees, most estimates concerning the benefits of growing timber in silvopastoral systems are based on computer models. For example, comparing the financial viability of silvopasture system and pasture system, Sibbald (1996) found that the net present value for ash (*Fraxinus excelsior* L.) growing in silvopastoral system in lowland UK was greater than the net present value for treeless pastures by 15%. McAdam et al. (1999a) and Thomas and Willis (2000) also found that, under a range of commodity prices and agricultural subsidy support scenarios, silvopasture has a net benefit over open grassland ranging from 34% to 181%. Even with no farm subsidy support, silvopasture was more profitable (by €20 ha⁻¹) than open grassland as the result of the additional output of timber from silvopastoral systems.

There is ample evidence from the high survival and reasonable growth rates of trees in silvopastoral systems that high quality trees for the purposes of either timber or firewood can be established in grazed pasture in Britain without affecting livestock production for at least the first ten years (Sibbald et al., 2001; Teklehaimanot et al., 2002). This has important implications because it means, on the one hand, that farmers do not lose annual agricultural income from the land under silvopastoral systems during the establishment period, but on the other, that this type of agroforestry may not necessarily contribute to short-term reductions in surplus agricultural production in the UK as had once been thought (Sheldrick and Auclair, 2000).

2.2.2. Pasture/Livestock management in silvopastoral systems

Silvopastoral systems offer a variety of benefits for livestock management. Silvopastoral systems can affect livestock productivity through mitigating heat or cold stress and by altering understorey pasture growth. Such benefits of silvopastoral systems have been researched in the UK at the UK's SNNE since 1988.

The effect of trees in silvopastoral systems on pasture production, which consequently has effect on livestock production, was studied by Ng'atigwa (1997), Onyeka (1998) and Zapater (1998) at Henfaes SSEF. Results of these studies, in general, indicated that there was no significant difference in pasture production between the silvopastoral and pasture control treatments six years after tree establishment. Sibbald et al (1991), based on silvopastoral systems experiment of re-spaced Sitka spruce trees (*Picea sitchensis*) in Glentress forest, Scotland, also found that grass sward growing beneath widely spaced trees, above ground conditions, did not greatly limit rates of herbage production, under trees of up to 8 m in height and at spacing as close as 6 m (about 300 stems ha⁻¹). However, a higher amount of pasture production was obtained from the pasture control without trees than in silvopastoral treatments nine years after tree establishment at Henfaes SSEF (Nghitoolwa, 2001). Thus, the impact of trees on pasture growth and consequently on livestock production in a silvopasture depends on several factors including the forage and tree species used, the age and size of each component, and tree spacing and orientation (Hawke, 1991; Sibbald et al., 1991; Teklehaimanot et al., 2002).

Sibbald and Dalziel (2000) reported that, in the UK's SNNE, no significant differences in lamb growth were observed between silvopastoral treatments and the pasture control until up to ten years after establishment of the sites. Results of the research at the silvopastoral system

experiment at Henfaes have also shown that there was no significant difference in livestock production between silvopastoral treatments and the pasture control during the first six years of the tree establishment phase (Teklehaimanot et al., 2002). Also, a study conducted by Green et al. (1995), in a silvopastoral experiment established by re-spacing Sitka spruce (*Picea sitchensis* (Bong.) Carr) plantation in Glentress forest, Scotland to create a silvopastoral system and study the effects of widely spaced trees on the microclimate and consequently on herbage production from sown grass swards (Sibbald et al., 1991), showed that widely spaced trees can significantly reduce wind speeds that have impact on livestock production. The trees were re-spaced at intervals of 4, 6 and 8 m by thinning the Sitka spruce trees originally planted at 2 m spacing. Trees also buffer spring and autumn temperatures extending the growing season of pasture in silvopastoral systems (Sibbald et al., 1991). These can enhance livestock productivity.

The lack of significant difference in lamb growth rate and livestock carrying capacity between treatments found in UK's SNNE as well as Henfaes SSEF up to ten years after establishment may be explained by the fact that the negative effects of trees on pasture production may have been compensated by the positive shelter effects of trees on livestock (McArthur 1991; Sibbald et al., 1991; Ainsworth et al., 2012). Once the tree canopy closes, however, pasture production, and thus livestock production could decline. For instance, Hawke (1991) in New Zealand found that lamb live weight gains from perennial ryegrass and white clover (*Trifolium repens* L.) was reduced approximately 50% in 15-year-old radiata pine (*Pinus radiata* D. Don) plantations with 200 stems ha⁻¹ compared to pastures without trees. However, this depends, as mentioned above, on the forage and tree species used, the age and size of each component, and tree spacing and orientation. Thus, the positive shelter effects of trees on livestock production as reported

above may continue for many years even after tree canopy closes in the UK silvopastoral systems.

In two of the five UK's SNNE trials, one in Scotland and the second in Northern Ireland, it was found that sheep spent more time in the shade and shelter of trees on hot sunny days and cold windy days than they did in the open (Sibbald et al., 1995; Hislop and Claridge 2000). This amelioration of conditions could also be a positive welfare benefit to livestock.

2.2.3. Carbon sequestration in silvopastoral systems

Carbon sequestration is an important ecosystem service provided by silvopastoral systems. An interesting recent development in the UK is the increasing recognition of the value of such ecosystem service, in the context of increasing concerns about global climate change, provided by sustainable land management systems such as silvopasture. According to Nair (2012), silvopastoral systems are able to sequester more carbon in soil when compared with silvoarable practices due to accelerated decomposition of soil organic matter following soil tillage done as a soil management practice for crop production in silvoarable systems.

Silvopastoral systems are, therefore, believed to offer a low-cost method to sequester carbon because of their perceived ability for greater capture and utilization of growth resources (light, nutrients, and water) than single-species crop or pasture systems (Pandey, 2002; Montagnini and Nair, 2004). Carbon (C) sequestration is estimated by assessing the C stored both aboveground and in the soil. The estimates of C stored in agroforestry systems, in general, range from 0.29 to 15.21 Mg ha⁻¹ yr⁻¹ aboveground and 30 to 300 Mg C ha⁻¹ down to 1 metre depth in the soil (Pandey, 2002; Montagnini and Nair, 2004). Dixon et al. (1994) also evaluated

the C sequestration potential of agroforestry and alternative land use practices in 94 nations worldwide and found that the carbon storage values (including below-ground storage) for agroforestry ranged between 12 and 228 Mg C ha⁻¹ with a median value of 95 Mg C ha⁻¹ and concluded that the potential for C accretion via biomass production is greatest within tropical latitudes. There is, however, limited research investigating the C sequestration potential of silvopastoral systems in temperate Europe.

Studies were carried out at Henfaes SSEF to estimate the C sequestration potential of silvopastoral systems as described below.

Research carried out by Kasahun et al. (2011) quantified and compared the amount of C stored under different tree species in silvopastoral systems at Henfaes. The mean Soil Organic Carbon (SOC) content under 19-year-old red alder (*Alnus rubra* Bong.) and sycamore (*Acer pseudoplatanus* L.) were 4.30% and 4.51%, respectively. These values were almost two times higher than the SOC content of the soil under the pasture control (2.06%) that was not integrated with any tree species. The authors concluded that both red alder and sycamore have a positive impact in increasing the C pool potential in silvopastoral systems.

Ramdial (2010) quantified the ecosystem C stocks of sycamore at the age of 18 years planted at different densities at Henfaes SSEF. Ecosystem C significantly increased ($p < 0.05$) with an increase in tree density. Tree biomass C stocks ranged from 7.62 ± 4.28 in 100 stems ha⁻¹ to 80.43 ± 1.89 t C ha⁻¹ in 2500 stems ha⁻¹ (woodland control) while SOC, to a depth of 30 cm, ranged from 202.44 ± 11.78 in 100 stems ha⁻¹ to 244.98 ± 8.12 t C ha⁻¹ in the woodland control, indicating that a major portion of ecosystem C stocks was stored in the SOC pool. SOC was found to decline with soil depth. Similar values of tree biomass C stocks were also reported

by Khanal (2011) who found that tree biomass C stock in woodland control plots (2500 stems ha⁻¹) at Henfaes SSEF was 130.29 ± 6.39 t ha⁻¹ which was 10.7 and 6.7 times more than in 100 and 400 stems ha⁻¹, respectively. Model simulations run by Ramdial (2010) using the CO2FIX model showed that reducing thinning volume and extending rotation length increased C stored in tree biomass and soil over the long term but, managing stands for bio-energy provided additional C sequestration benefits.

In another study conducted by Agard (2011) at Henfaes, it was found that hedges have a positive influence on SOC content causing an increase of around 16% above the natural field content closer to the hedge. This result agrees with some of the findings reported by Follain *et al.* (2007) who provided a comprehensive review of authors that substantiate the increases SOC content with the presence of hedges. Also in another study conducted by Benjamin (2010) on twelve provenances of ash (*Fraxinus excelsior*) planted at Henfaes, the aboveground carbon stocks in the twelve provenances ranged from 188.69 to 208.26 t ha⁻¹. However, there was no statistical difference between provenances in SOC content, which varied from 188.04 to 199.68 t ha⁻¹, but differences were found with increases in soil depth.

Rodwell (2009) conducted a biophysical and economic appraisal of lowland silvopastoral systems in Wales to determine their suitability as carbon sequestration schemes for farmers. The author found that tree planting can increase aboveground biomass, and therefore total carbon sequestered, with increasing planting density. This implies that if carbon sequestration is the primary objective, woodland will sequester the most carbon and open pasture the least. The author suggested that changes in government policy and farm subsidies could be the most cost-effective way to encourage silvopasture as an agricultural land use in Wales.

The results of the above experiments have shown that silvopastoral systems have a higher C sequestration potential than pure pasture, but the C stocks in silvopastoral systems were less than pure woodlands. This is expected because as planting density increases aboveground biomass increases, and consequently the amount of C sequestered increases. Tree stands that have denser canopy cover continuously add organic matter to the soil resulting in higher soil organic matter content (Patenaude *et al.*, 2003). The fact that the amount of carbon sequestered in the soil decreased with depth is also an expected result of the downward movement of organic matter by leaching and eluviations. The results of the above experiments have also shown that soil organic carbon (SOC) was a major component of ecosystem carbon stocks. The findings of the above studies of 245 t C ha⁻¹ for the woodland control plots at Henfaes is comparable to the value of 228 t C ha⁻¹ reported by Broadmeadow and Matthews (2003) for woodlands in Wales. Thus, based on the above estimates made at Henfaes SSEF, it may be concluded that silvopastoral systems with higher density of trees (400 stems ha⁻¹) have a potential to sequester more C than open pasture and lower density silvopastoral systems (100 stems ha⁻¹).

2.2.4. Soil improvement and maintenance

One of the environmental benefits of incorporating trees onto pasture is soil amelioration by the trees. Trees are known to improve the productivity of the soil beneath them. Research results have shown that the main tree-mediated processes that determine the extent and rate of soil improvement by trees include increased nitrogen (N) input by N₂-fixing trees, enhanced availability of nutrients resulting from production and decomposition of tree biomass, and greater uptake and utilization of nutrients from deeper layers of soils by deep-rooted trees (Young, 1997).

The role of N₂-fixing trees in improving soil fertility in silvopastoral systems was investigated at Henfaes SSEF by Martin (1995), Teklehaimanot and Martin (1998), Mmolotsi (2004), Teklehaimanot and Mmolotsi (2007), and Mmolotsi and Teklehaimanot (2008).

Martin (1995) and Teklehaimanot and Martin (1998) assessed the nitrogen fixing capability of red alder (*Alnus rubra*) in silvopastoral systems at Henfaes SSEF by comparing it with the pasture component, white clover (*Trifolium repens*). The diurnal and seasonal patterns of nitrogenase activity of red alder and white clover was assessed using the acetylene reduction assay. No obvious diurnal patterns of nitrogenase activity were found in either red alder or white clover in summer and no significant variations in nitrogenase activity were observed between day and night. However, in autumn, pronounced diurnal patterns were observed in both species. Significantly higher rates of nitrogenase activity per unit dry weight (dwt) of nodules were detected at 1500 hours in red alder, whereas, in white clover, significantly higher rates were obtained at 2100 hours. Seasonal rates of nitrogenase activity showed significantly higher activity in summer, which subsequently decreased in autumn, to reach very low levels in the winter.

The rates of nitrogenase activity of white clover were consistently higher than those of red alder both diurnally and seasonally. In the three seasons sampled, the average nitrogenase activity for white clover was 66.42 $\mu\text{mol C}_2\text{H}_4 \text{ g nodule dwt}^{-1} \text{ h}^{-1}$, which was 3.5 times higher than the 18.67 $\mu\text{mol C}_2\text{H}_4 \text{ g nodule dwt}^{-1} \text{ h}^{-1}$ obtained for red alder. The low nitrogenase activity in red alder may be due to the young age of the trees. They were only three years old at the time of the investigation. Yet, the trees were actively fixing N and thus the results show that the trees were playing their soil amelioration potential role in silvopastoral systems as early as at three years of age.

Mmolotsi (2004) and Teklehaimanot and Mmolotsi (2007) also studied the nitrogen fixing capability of red alder at the age of 11 years in silvopastoral systems at Henfaes SSEF using nitrogen-15 natural abundance method. Results showed that depleted $\delta^{15}\text{N}$ values close to zero were recorded in red alder plant parts except in root nodules and the soil, indicating that a large proportion of nitrogen in red alder was fixed from the atmosphere. The depleted $\delta^{15}\text{N}$ values indicate a signature of ^{15}N that was similar to that of the N in the atmosphere, showing that the atmosphere was the main N source in red alder. This also indicates that the red alder was efficiently fixing atmospheric N as shown by the high fraction of N derived from the atmosphere (FNdfa) (average 85%).

The results of the study showed that $\delta^{15}\text{N}$ varied between seasons. $\delta^{15}\text{N}$ values for the summer and autumn seasons were negative for red alder indicating active N fixation during these periods. The FNdfa values were 90 and 99% for summer and autumn, respectively. The average values of $\delta^{15}\text{N}$ for the winter and spring seasons were positive but close to zero in the red alder that indicate reduced N fixation during these periods. FNdfa values were 85 and 64% for winter and spring, respectively.

Nitrogen fixation estimates by the acetylene reduction assay method also showed that the mean nitrogenase activity in red alder nodules was high in summer and autumn when temperature and moisture regimes were favourable, and N fixing activity was significantly reduced in the winter periods, probably due to low temperature (Teklehaimanot and Martin 1998; Tripp et al. 1979). Similar results were obtained by Binkley et al. (1985) in red alder.

The root nodules of the red alder and the soil of the site showed enriched values of ^{15}N . The positive $\delta^{15}\text{N}$ values in root nodules show that root nodules were enriched with ^{15}N indicating that the soil was the source of nitrogen for root nodules of red alder. The results of the present study are consistent with those reported by Tjepkema et al. (2000) who reported that the nodules of *Alnus glutinosa* were consistently enriched in ^{15}N relative to other plant parts. The enriched $\delta^{15}\text{N}$ values (5.95‰) of soil were also consistent with those reported between 5.0 and 5.8‰ in forest soils by Kreibich and Kern (2000).

Overall, planting density had significant effect on the rate of N fixation in red alder. It was estimated that 65.55 and 334.14 kg N ha⁻¹ was fixed by red alder in the silvopastoral treatment plots and woodland controls, respectively (Teklehaimanot and Mmolotsi, 2007). This included N fixed in leaves, wood, and roots. By considering only leaves, as done in most of the studies on N fixation (Coté and Camire, 1984), the amount of N derived from the atmosphere in leaves was 9.22 and 57.63 kg N ha⁻¹ in silvopasture and woodland control, respectively. The woodland control value agrees with that (53 kg N ha⁻¹ in both mixed and pure stands of alder grown in Canada) reported by Coté and Camire (1984).

A study was carried out by Mmolotsi and Teklehaimanot (2008) to estimate the contribution of red alder to soil organic matter and nitrogen content in a silvopastoral system at Henfaes SSEF. They quantified fine roots and roots nodules over the four seasons in silvopastoral and woodland control plots of red alder by collecting soil samples in each season at three sampling points (0.30 m, 0.50 m and 1.00 m distance from the base of each tree) from nine trees. Results showed that there were significant differences in the density of live fine root between seasons and treatments ($p < 0.001$). The mean weight density of live fine root over the four seasons in silvopastoral and woodland control was $0.27 \pm 0.01 \text{ kg} \cdot \text{m}^{-3}$ and $0.54 \pm 0.03 \text{ kg} \cdot \text{m}^{-3}$, respectively.

Weight density of dead root in each treatment plot remained constant throughout the year. The mean weight density of dead root was also significantly different ($p < 0.01$) between woodland control and silvopastoral treatments. Weight density of live and dead root nodule was both constant throughout the year and between the different sampling distances. Live and dead fine root densities of red alder were 2700 and 5400 kg ha⁻¹ and 360 and 790 kg ha⁻¹ in silvopastoral plots and woodland controls, respectively.

The mean weight densities of live and dead root nodule over the four seasons were 0.09±0.03 kg· m⁻³ and 0.05±0.03 kg· m⁻³ in silvopastoral and 0.08±0.02 kg· m⁻³ and 0.03±0.01 kg· m⁻³ in the woodland control plots, respectively. Live and dead root nodule weight densities of red alder yielded 880 and 520 kg ha⁻¹ in silvopastoral plots, and 800 and 310 kg ha⁻¹ in woodland controls, respectively. According to the results of the present study, the amount of organic matter potentially added to the soil due to senescent leaves and dead roots and root nodules was estimated at 4.0 and 9.1 t ha⁻¹ yr⁻¹, in silvopastoral plots and woodland controls, respectively. These results showed that red alder has a potential to improve and maintain soil fertility in silvopasture.

The study showed that significantly large quantities of dead fine roots and root nodules were found in soils within the silvopasture and woodland control treatments. These contribute significantly to soil organic matter and nitrogen content of the soil (Mmolotsi and Teklehaimanot, 2008). Consequently, the results of the above studies show that by planting N₂ fixing trees in silvopastoral systems, N is provided, and the level of nutrients in the soil is increased.

The high soil organic matter content at Henfaes SSEF has important and diverse implications for soil quality. Accumulation of more soil organic matter in the soil of silvopastoral systems than in open pasture improves and maintains soil physical, chemical and biological properties by reducing bulk density and increasing water-holding capacity and nutrient availability. Soil organic matter is a key attribute of soil quality vital to many of the soil functions (e.g. erosion control, nutrient cycling, and water infiltration and quality) (Young, 1997). Therefore, the high soil organic matter in silvopastoral systems as observed at Henfaes SSEF indicates a high ecosystem service function of Silvopastoral systems in terms of soil quality improvement and maintenance.

2.2.5. Water management

The amount of water present in a system is a useful measure of plant and soil water status. Climate, soil and vegetation influence water relations of forests (Whitehead, 1984) and agroforestry systems. Water balance has a great influence on tree growth and survival as tree growth depends on water availability. On the other hand, trees are also considered as important factors in the water balance in terms of their influence on interception, transpiration, and, hence, runoff and drainage (Landsberg and Gower, 1997).

Introducing trees on pasture or re-spacing of existing tree stands to create silvopastoral systems can change the water balance of the site. This can have direct effects on the productivity of trees and livestock at the site or on water resource management of a catchment or region.

An experiment was undertaken to study the effect of widely spaced trees on the water balance of a silvopastoral system in Glentress forest, Scotland by Teklehaimanot et al. (1991a, 1991b). The trees were re-spaced at intervals of 4, 6 and 8 m by thinning the Sitka spruce trees originally

planted at 2 m spacing to create a silvopastoral system and study the effects of widely spaced trees on the microclimate and consequently on herbage production from sown grass swards (Sibbald et al., 1991).

Teklehaimanot et al. (1991a) measured rainfall interception loss, which is an important component of the water balance of the system, at Glentress forest silvopastoral systems experimental site. Measurement of rainfall interception loss is an essential prerequisite for a quantitative prediction of the effects of widely spaced trees in silvopastoral system on the water management of a site. They measured throughfall, stemflow and interception loss using the traditional volume balance method in three widely spaced treatment plots of 4, 6 and 8 m and a forestry control plot of 2 m spacing.

The results showed that, on average, rainfall interception loss as a percentage of gross rainfall was 33, 24, 15 and 9% in the 2, 4, 6 and 8 m spacing treatments, respectively. The results also showed that rainfall interception loss was not directly proportional to the density of trees but it was related to the number of trees per hectare by a rectangular hyperbolic function.

The difference in interception loss between spacing treatments was, therefore, attributed to the difference in boundary layer conductance. Boundary layer conductance per tree increased with spacing from 0.82 mm s^{-1} in the 2-m spacing forestry control treatment to 5.92 mm s^{-1} in the widely spaced 8 m treatment (Teklehaimanot et al., 1991b). The high boundary layer conductance per tree in the 8-m spacing was caused by a high rate of evaporation per tree due to increased ventilation and air turbulence in widely spaced stands.

The effects of trees on the components of water balance of silvopastoral systems was also investigated at Henfaes SSEF by Rakkibu (1998), Temba (1999), Kuflu (2000), and Kondziela, (2011). With the exception of Kondziela (2011), all the others found significant effects of tree density and tree species on some of the components of the water balance of Henfaes SSEF and these are described in detail below.

Rakkibu (1998) assessed the effect of tree density on tree transpiration by measuring sapflow in red alder trees using Thermal Dissipation Probe. Planting density was found to modify individual tree sapflow. Mean sapflow per tree was found to be lower at woodland control being $0.29 \text{ dm}^3\text{h}^{-1}$ compared with $0.44 \text{ dm}^3 \text{ h}^{-1}$ at $400 \text{ stems ha}^{-1}$ silvopastoral treatment and the difference was highly significant ($P < 0.001$). The higher sapflow in widely spaced silvopastoral plot was related to a high rate of crown transpiration caused by greater canopy exposure to wind and sunlight. Sapflow was found positively correlated with solar radiation and air temperature and negatively correlated with relative humidity. Rakkibu also found high significant difference ($P < 0.001$) in sapflow between the three treatment blocks of red alder at the site. This could be attributed to the difference in water table between the three blocks. Water table was found to be very shallow in block 1 (1 m) and fairly deep in block 3. Trees in block 1 are more likely to transpire more water because of their greater proximity to soil water than in block 3. Similar results were obtained by Lu et al. (1995) who observed significant differences in transpiration between dry and wet plots. Again, due to abundance of soil water, trees in block 1 and block 2 grew faster than trees in block 3 as alder is known to prefer wet sites. Thus, trees in block 1 transpired significantly more water than those in block 3 due to their larger diameter and bigger crown area. The greater mean sapflow in alder silvopastoral plots than in alder woodland control plot was probably because trees at wide spacing are more exposed to sunlight and wind than trees in closely spaced trees. Similar results were also

reported by Morikawa et al. (1986) who observed that the rate of sapflow per tree was higher in a thinned plot at a given level of solar radiation, and that the difference between sapflow before and after thinning increased with solar radiation in *Chamaecyparis obtusa* stands. Lower sapflow per tree in alder woodland control may be due to mutual shading of trees as reported by Granier et al. (1996). Similar relationship of sapflow with tree densities have been shown by Eastham et al. (1990) who reported that trees planted at lower densities were able to maintain higher sapflow rates per tree than trees at higher densities despite higher evaporation from pasture in low densities. This indicates that trees may successfully compete with pasture for soil water, possibly because rooting patterns lead to withdrawal of water predominantly from different soil horizons (Eastham et al., 1990).

Temba (1999) measured soil moisture content in 400 stems ha⁻¹ silvopastoral plots and 2500 stems ha⁻¹ woodland control plots of both red alder and sycamore as well as in the pasture control plots without trees at Henfaes SSEF. Measurements in plots with trees were made at 0.5 meter and 2.5 meter distance from the tree base, and in four directions (North, South, West and East). Soil moisture was found to be consistently higher in the pasture control (0.35 m³m⁻³) compared to silvopastoral and woodland control plots. Also, soil moisture was found to be higher in silvopastoral than in woodland control plots. Soil moisture at 0.5 m distance from tree base was found to be higher in sycamore (0.28 m³m⁻³) compared to red alder (0.22 m³m⁻³). At 2.5 m distance, however, soil moisture was found to be similar to that of pasture control plots but significantly higher (P<0.001) than at 0.5 m distance. The significant variation in soil moisture as distance increased from the tree, that is, the increase in soil moisture with increasing distance from the tree trunk, can be explained in various ways. The lower soil moisture at the base of tree canopy may be due to more soil compaction at 0.5 m compared to 2.5 m distance from the tree trunk as a result of trampling by sheep that tend to concentrate

around trees for shade (Penn et al. 1994; Sibbald et al., 1995; Sibbald et al., 2001; Bezkorowajnyj et al., 1993). Compaction impedes infiltration which results in low soil moisture content. Penn et al. (1994) found that the soil at a depth of 10 cm on the agroforestry plot was drier closer (0.5 m) to tree throughout the year compared to the soil at 2.5 m away from the trunk.

Kuflu (2000) investigated the effect of tree density and species on throughfall and soil moisture content at Henfaes SSEF. Results showed that throughfall was higher under sycamore (93 and 89% of gross rainfall) than under alder (74 and 49%) at silvopastoral and woodland control plots, respectively. The author attributed this difference to the smaller canopy cover and shorter height of sycamore. The 49% throughfall value of alder in the woodland control plot is in close agreement with the 50% reported by Teklehaimanot et al. (1991a) for 19-year-old Sitka spruce plantation with a density of 2500 trees ha⁻¹. There was more throughfall in silvopastoral plots than in woodland control plots at Henfaes SSEF because the lower tree density in the silvopastoral plots (400 stems ha⁻¹) intercepted less rainfall than the higher tree density in woodland control plots (2500 stems ha⁻¹). Teklehaimanot et al. (1991a), also observed that there was more throughfall at wider spacing. The higher throughfall in sycamore than in alder may be attributed to the variation between the species (sycamore and alder) in the height and diameter and shape of trees as they grow, and on the degree of canopy cover. Soil moisture was also observed by Kuflu (2000) to be higher in silvopastoral plots (0.304 and 0.273 m³m⁻³) than in woodland control plots (0.265 and 0.189 m³m⁻³) under sycamore and alder, respectively, but there were no significant differences in soil moisture content between the sycamore silvopastoral plots and the pasture control plots (0.293 m³m⁻³). The author attributed this result to the lower tree density in the silvopastoral treatments and the absence of trees in pure pasture.

Based on the results of the above studies, it may be concluded that silvopastoral systems are better than pure woodlands in improving and maintaining the water balance of the soil by modifying one or two of its components.

2.2.6. Biodiversity enhancement

Silvopastoral system enhances biodiversity due to the diverse environmental conditions that are created within (vegetation structure, shading and moisture). It increases connectivity within landscape components which benefit the mobility of animals thus reducing habitat fragmentation (Rois-Díaz et al., 2006).

Various authors have examined how silvopastoral systems enhance biodiversity (e.g. McEvoy 2005; Burgess, 1999; McAdam et al. 1999, 2007a; Agnew and Sibbald 1996; Wang, 1999; Cuthbertson and McAdam, 1996; Toal and McAdam 1995; Crowe and McAdam, 1993).

Burgess, (1999) notes that experience in a number of trials across the UK has shown that biodiversity increases as a result of silvopastoral agroforestry. In his review of the potential impact of agroforestry systems on the diversity of plants and animals on British farms, he suggests that the introduction of silvopastoral systems can lead to an increase in the diversity of invertebrates and perhaps birds on grassland farms.

The impact of silvopastoral systems on aspects of biodiversity (carabid beetles, spiders, birds and flora) was investigated when trees had been established for up to 8 years at the North Ireland's UK National Network Experiment. McAdam et al. (1999b) found greater biodiversity levels in silvopastoral systems than in both open grassland and pure woodland systems. More spiders were collected from silvopasture than either pasture or woodland treatments. Carabid

beetles were more numerous and from a wider range of species in the silvopasture than open pasture (Cuthbertson and McAdam 1996; Whiteside et al. 1998). Numbers of juvenile earthworms were higher in silvopasture than grassland (Whiteside et al., 1998). Plant diversity was also greater (but not significantly so) near trees than in open pasture (McAdam 1996; McAdam and Hoppé 1996). Toal and McAdam (1995) found that, generally, significantly more birds were recorded on silvopasture in summer and winter than either open pasture or woodland. Similar findings were reported by Bergmeier et al (2010), in their survey of wood-pasture habitats in Europe, that silvopastures are a “habitat of importance” for at least 37 European bird species, while for another 18 species, a high proportion of their European populations uses this habitat too.

McAdam et al. (2007a) also reported the results of the studies carried out on the effect of silvopastoral systems on biodiversity in all the sites of the UK’s SNNE. The results showed that silvopastoral systems attracted invertebrates of epigeal groups which may have provided an enhanced food supply which attracted birds. It was concluded that, even at this early stage, silvopastoral systems have an impact on birds: birds normally associated with woodland are being attracted to silvopasture along with birds normally found in open fields.

Heron (1999) assessed ground fauna six years after the establishment of the silvopastoral experiment at Henfaes. All the silvopastoral treatments had bigger populations of ground beetle and woodlice than the woodland control treatments and were found in higher numbers near the tree bases. Centipedes/millipedes found to occur mainly in the woodland control plots decreased as tree density decreased. Springtails were found to occur less in the silvopastoral treatments than in the woodland control treatments suggesting that a competitive interaction

between ground beetles and springtails may exist. Based on these findings the author concluded that silvopasture treatments had greater habitat diversity.

Overall, the results of the above studies show that, even at an early stage, silvopastoral systems can significantly enhance biodiversity, confirming a general trend noticed across Europe by Benton et al. (2003) that structural heterogeneity created by agroforestry can increase biodiversity in previously intensively farmed grassland. Although the plot size in the above experiments (overall mean 0.56 ha), spatial arrangement of the plot and influence of adjacent habitats cast concern on the actual quantifiable value of the results, for comparative purposes they indicate trends which will have important long-term wildlife habitat implications. The creation of land-use mosaics involving both silvopasture and open grassland will offer further opportunity for landscape heterogeneity which will benefit birds and other fauna.

2.3. GAPS AND AREAS FOR FUTURE RESEARCH

The foregoing review and discussion of the ecosystem service topics clearly suggest that the UK's SNNE, though at its infancy, is contributing considerably to the biophysical and economic understanding of tree-pasture-livestock interactions. Substantial and impressive research studies have been conducted at the experiment so far, but the link between scientific knowledge and effective field application for the benefit of the sheep farmers and woodland owners is still lacking. However, some areas remained either under-studied or completely neglected. The following research gaps and areas deserve attention:

- There is the need to conduct a more exhaustive, holistic and updated review of research papers and articles written on the UK's SNNE to include more information from other institutions and experimental sites.
- As management interventions and invasion of pasture by unwanted weeds may have taken place over the years, there is the need to study the pattern of temporal and spatial changes in the understorey pasture species composition and abundance in silvopasture.
- Applying biomass allometric equations developed for trees grown in pure forestry systems to trees on agroforestry scale analysis can be challenging because of the disparity in their growth forms. This underscores the imperative need for the development of species specific allometric equations for trees grown in agroforestry systems to evade this constraint.
- Forage production and nutrient content are known to determine the productivity of the grazing animal, and thus the productivity of the grazing system. Since there is a strong relationship between understorey forage production and canopy closure/light intensity, there is the need to evaluate forages grown in silvopastoral practices for the effect of canopy/light on forage production and nutrient quality.

- While the environmental benefits from agroforestry systems are relatively well understood, considerable uncertainty remains about the potential economic profitability of silvopastoral systems compared to monoculture, and given the current difficulty in securing incentives to engage in agroforestry, there is the need to explore the financial and economic viability of silvopastoral system vis-à-vis non-agroforestry farm management systems.

Chapter 3 : CHANGES OVER TIME IN PASTURE SPECIES COMPOSITION IN A SILVOPASTORAL SYSTEM

3.1. INTRODUCTION

Widely spaced red alder (*Alnus rubra* Bong) trees were originally planted into blocks of weed free, mixed perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*), pastures in the UK's silvopastoral agroforestry network experiment at Henfaes in 1992. The choice of planting widely spaced trees onto pastoral agricultural land has the advantage of sustaining understorey pasture with associated livestock production for longer period than for closely spaced trees (Hawke and Gillingham, 1997). Monitoring change in these trials over time will involve not only looking at tree growth dynamics but also estimating botanical composition and diversity of the understorey pasture. The productivity and stability of ecosystems is thought to depend on the presence of key species and functional types rather than on the number of species (Grime, 1997; Huston, 1997).

Long-term routine monitoring of changes in pasture species composition will clarify ecological processes that bring about changes. However, lack of sustained routine measurements at intervals over long period of time will militate against the understanding and interpretation of these changes (Clark *et al.*, 2005).

Modifications to pasture species composition may be caused by a number of factors including changes in the microclimate and soil properties, tree density, tree spacing, crown size and magnitude of foliation, livestock grazing and trampling including frequency and intensity

(Benavides et al., 2009), These changes are more perceptible below widely spaced evergreen trees than under deciduous trees. Changes in perennial pasture species composition in the temperate region is a continuous process, though stable conditions may appear to be present sometimes. Snaydon (1987) has reported that, for temperate pasture, sown species in pastures is dominant only for a limited time before the sward is overtaken by local species usually from neighboring pastures.

The main bulk of the productive biomass of improved pasture in the United Kingdom is composed of traditional species such as rye grass (*Lolium perenne*) and clover (*Trifolium repens*) which occur together with a variety of other species. While some species are rather uncommon and unimportant to pasture production, others are found in abundance and are sufficiently palatable to grazing animals. But, some species are obviously unpleasant to grazing livestock, diminish farm output, and may be categorized as weeds.

Tree and pasture productivity is usually enhanced by the method of pruning and thinning forests. The effect of reducing or eliminating competition from overstory trees have been studied by other authors (e.g., Ducherer et al., 2013; Thomas et al. 1999; Streigl and Wickland 1998; Smit and Rethman, 2000).

This chapter focuses on what remains in the pasture in later years when a simple mixture is sown, whether the plant community will be representative of what was sown 20 years after sowing a pasture. An objective of any diversity study in a Silvopastoral system will be to determine changes over time in the composition of pasture mixtures. There is a paucity of information about the patterns of changes in pasture species composition, richness and diversity in silvopastoral systems. This study will examine the short- and medium-term changes over

time in botanical composition and variety of pasture species under an alder based silvopastoral system. Knowledge of the changes may help to inform management policies and techniques in order to manage species diversity with a view to increasing forage production in silvopasture.

3.1.1. Objectives

The objective of this study was to determine the temporal and spatial changes over time in botanical composition of pasture species under red alder (*Alnus rubra* Bong) in a silvopastoral agroforestry system.

Specifically, the study did the following:

- Measured the short-term effects of thinning on understory pasture species composition on the same plot before and after thinning; (up to 1 year), and no tree control was considered.
- Determined the medium-term changes (up to 20 years) in pasture species composition and variability in red alder plots and open pasture control from establishment to date.

3.1.2. Hypothesis

Thinning will not change the understory pasture species composition, abundance and diversity in the short-term (up to 1 year).

There will be no change in understory pasture species composition, abundance and diversity in the medium-term (up to 20 years).

3.2. LITERATURE REVIEW

3.2.1. The ecological influences of thinning

Forest management practices such as pruning and thinning reduce stand density and leaf area, increase light transmission to the understorey, affect forest soil, improve tree and understorey pasture productivity, increase the amount of organic matter stored in soils, reduce wildfire risk and maintain general health of forests (Janssens et al., 2001). Ducherer *et al*, (2013) noted that pruning of trees will increase the amount of light reaching understory vegetation, and consequently, increase understory plant cover and species richness, increase soil temperatures and decrease soil respiration. In addition, they noted that reducing or eliminating competition from overstory trees can increase soil water and mineral nutrients for understory plants, and allow colonization of bare soil by herbaceous plants. Since understory vegetation responds to increased light, the greatest responses to the increases in light following thinning are expected within the first several years (Ducherer *et al*, 2013). Furthermore, thinning disturbances can reduce understory vegetation cover, in particular, the process of felling trees and extracting leads to trampling, smothering and compaction of vegetation (Thomas et al. 1999).

Disturbance brought on by thinning can increase the availability of new microsites for plant establishment and growth, possibly leading to increased species richness (Brockway *et al*. 2002). The disturbances associated with thinning can reduce understorey vegetation cover, in particular, trampling and smothering of vegetation by cut trees (Thomas *et al*. 1999). Other research has demonstrated that timber harvesting and extensive site preparation can reduce the amount of surface organic matter (Jurgensen *et al*. 1997). Debris generated by tree pruning and thinning can decrease pasture production despite the increasing light transmission to the understorey after these operations, because of shading effect of the debris on the available

pasture area (Percival and Hawke 1985; Hawke 1991; Kellas et al., 1995; Hurst et al. 2000; Benavides et al., 2009).

Soil respiration can be partitioned into autotrophic respiration (metabolic activity of plant roots) and heterotrophic respiration (decomposition of dead organic material) (Ryan and Law, 2005; Jonsson and Sigurdsson, 2010). These two processes of soil respiration can respond independently to forest management practices (such as pruning, prescribed burning, thinning and clear cutting), weather conditions, and forest and soil types (Jonsson and Sigurdsson, 2010).

The effects of forest thinning on soil respiration are determined by a complex of many interactive factors including changes in soil temperature, soil water, microbial respiration, root respiration rate, organic carbon content and decomposition of soil microbes, dead roots, leaf and branch litter (Jonsson and Sigurdsson, 2010; Olajuyigbe et al., 2012). These effects of thinning on soil respiration increase with thinning intensity. Thinning reduces tree density and basal area, leading to similar relative reduction in leaf area index (LAI), which may therefore reduce both photosynthesis per area and the amount of respiring roots per area in the post-thinning period. The decrease in total soil respiration after thinning may be attributed to reduction in root respiration per ground area (Olajuyigbe et al., 2012). On the other hand, a short-term increase in soil respiration may also result from the release of carbon from the decay of litter, wood and root debris (Sullivan et al., 2008; Olajuyigbe et al., 2012).

The diverse and complex nature of these interactions have given rise to confounding results from research studies on the impact of thinning on soil respiration. For instance, soil respiration has increased (Kaye and Hart, 1998; Ohashi et al., 1999; Selig and Seiler, 2004; Selmants et

al., 2008; Olajuyigbe et al., 2012), decreased (Nakane et al., 1986; Kaye and Hart, 1998; Tang et al., 2005; Sullivan et al., 2008; Jonsson and Sigurdsson, 2010) or stayed unchanged (Toland and Zak, 1994) following thinning.

The detrimental effects of tree ingrowth must be weighed against the disturbances associated with thinning, because both can reduce the cover of many understorey species (Ducherer *et al*, 2013), depending on whether the species is a pioneer, a light demander, or a shade tolerant species. In particular, cover of herbs, pinegrass and bryophytes respond negatively to thinning, however, there is no indication that the long-term benefits of thinning are outweighed by initial decreases in cover and biomass of understorey vegetation (Ducherer *et al*, 2013).

3.2.2. Diversity indices

Diversity in ecological studies relates to the different forms of life, which are present in a particular site; precisely, it concerns the different species of a particular genus that are present in an ecological community (Frosini, 2006). Magurran (2004), in her comprehensive account of methods for measuring biodiversity, defines biodiversity as ‘*the variety and abundance of species in a defined unit of study*’. Abundance is easily defined as total numbers of individuals or the density of individuals, though biomass or percentage ground cover (for terrestrial plants) may also be appropriate measures. Biodiversity is commonly expressed through indices, which are mathematical functions that combine the number of species (species richness, number per unit area) and their relative abundance (evenness, an estimate of species distribution within a community) in a single measure (Whittaker 1972; Lande 1996; Purvis and Hector 2000; Magurran 2004). Species diversity is therefore the product of species richness and evenness. A diversity index, must be sensitive to both factors, thus must also be sensitive to the different number of species in two or more communities (Mueller-Dombois and Ellenberg, 1974;

Frosini, 2006). Other things being equal, there is greater diversity when the number of species grows, and when all the species are fairly represented.

The diversity index provides information about species endemism, rarity and commonness as well as information about community composition than simply species richness and relative abundances of different species (Frosini, 2006). The ability to quantify diversity in this way is an important tool for biologists trying to understand community structure. Also, measuring diversity has been of historical significance due to the evident declines in habitat diversity (Frosini, 2006). Although there are many others, the most commonly used diversity indices in ecology are Shannon index (H') and Simpson's diversity index (Mueller-Dombois and Ellenberg, 1974). Comprehensive reviews of various diversity indices have been presented by Peet (1974) and Magurran (1988).

The simplest measure of species diversity is species richness. It is a measure of the number of different species in a given site. The more species present in a sample, the 'richer' the sample. Species richness as a measure on its own does not consider the number of individuals of each species present but rather gives as much weight to those species which have very few individuals as to those which have many individuals. A richness index may simply coincide with the number of species present in a community, but may also be a function of the number of all the individuals in the community. The species richness of each community is simply the number of species present with at least one individual in a given area. The index is essential in assessing taxonomic and ecological values of a habitat (Mueller-Dombois and Ellenberg, 1974; Frosini, 2006). As a simple measure, richness has been a popular diversity index in ecology, where abundance data are often not available for the datasets of interest. Because richness does

not take the abundances of the types into account, it is not the same thing as diversity, which does take abundances into account.

The second factor, evenness, is a measure of the relative abundance of different species making up the richness of the area (Mueller-Dombois and Ellenberg, 1974). According to Frosini (2006), an evenness index is a function of the frequencies or proportions pertaining to the species; such an index increases when the proportions tend to be equal or perfect homogeneity and decreases when one species tend to dominate all the others. The interpretation of evenness is strictly dependent on the richness. Evenness is not calculated independently, but rather is derived from other compound species diversity measures that inherently contain richness and evenness components.

3.2.3. Shannon index

The Shannon index (H'), also known as the Shannon's diversity index, the Shannon–Wiener index, the Shannon–Weaver index and the Shannon entropy (Spellerberg and Fedor 2003), is the most commonly used index to characterize species diversity in a community, and can be used to compare different populations as it takes both species abundance and species richness into account and is sensitive to changes in the importance of the rarest classes (Kent and Coker, 1992; Nolan and Callahan, 2005). The idea behind this index is that the diversity of a community is similar to the amount of information in a code or message. The Shannon Index is calculated using Equation 3.1, as shown below:

$$H' = - \sum_{i=1}^s p_i \ln p_i$$

Equation 3.1

Where H' = Shannon index; S = total number of species; p_i = is the proportion of each plant species in the sample; and $\ln p_i$ = natural logarithm of the proportion of each species in the sample.

The Shannon index increases as both the richness and the evenness of the community increase. Typical value of the Shannon-Wiener Index generally lies between 1.5 and 3.5 for ecological data and rarely exceeds 4.0 (Magurran 2004). High values of Shannon-Wiener diversity index is a representative of more diverse communities (Kent and Coker, 1992; Frosini, 2006).

Shannon's equitability or Evenness index measures the evenness of a community and can be easily calculated by dividing the value of Shannon diversity index by the maximum diversity.

The Shannon's Equitability is given by Equation 3.2 see, e.g., (Kent and Coker, 1992)

$$E_H = H'/H_{\max} = H/\ln S \quad \text{Equation 3.2}$$

where E_H = Shannon equitability index; H' = Shannon index; H_{\max} = maximum diversity; S = total number of species; and \ln = natural log

The value of E_H is between 0 and 1 with 1 being complete evenness. If the species are evenly distributed, then the H' value would be high. So, the H' value allows us to know not only the number of species but how the abundance of the species is distributed among all the species in the community (Frosini, 2006).

3.2.4. Simpson's diversity index

Simpson's index (D') is a measure of diversity, which takes into account both species richness, and the relative abundance of each species. In essence, it measures the probability that two individuals randomly selected from an area will belong to the same species. The general formula (Simpson 1949) for calculating D' is presented in equation 3.3 below:

$$D' = \frac{1}{\sum_{i=1}^S p_i^2} \quad \text{Equation 3.3}$$

Where D' = Simpson's diversity index; S = total number of species; P_i = is the proportion of total sample belonging to the i th species;

The value of D' ranges from 0 to 1. With this index, biodiversity increases as the Simpson index decreases. That is, the bigger the index value the lower the diversity. To get rid of this evident ambiguity, some texts use derivations of the index, such as the inverse ($D' = 1/D'$) or the difference from 1 ($D' = 1 - D'$).

Both Shannon and Simpson diversities increase as richness increases, for a given pattern of evenness, and increase as evenness increases, for a given richness, but they do not always rank communities in the same order (Colwell 2009). Simpson diversity is less sensitive to richness and more sensitive to evenness than Shannon diversity, which, in turn, is more sensitive to evenness than is a simple count of species.

3.2.5. Dry-weight rank method of measuring botanical composition of pastures

The Dry-weight Rank method (DWR) is specifically designed to estimate quickly and accurately the composition of pastures on a dry weight basis by providing a measure of the relative contribution of various species to the total biomass for a site. It involves the visual observation of various quadrats and the ranking of the three species which contribute the most weight in the quadrat (Mannetje and Haydock, 1963). It is a fast method because it eliminates the need for the labour intensive clipping and hand-sorting of samples. Results are expressed only as percentage values, and do not quantify the actual biomass for each species.

This method entails the selection of the first, second and third heaviest species within each quadrat, each of which is then assigned a weighting based on standard multipliers, which have been shown to be applicable over a range of pasture types in Australia, the United States and Zimbabwe (Jones and Tothill, 1985). The observer decides which three species in each quadrat have the greatest yield of current year's growth on a dry matter basis. The species with the highest yield is given a rank of 1, the next 2, and the third highest a 3. All other species present are ignored. If there are not three species present in the quadrat, a multiple rank is assigned.

The Dry-weight Rank method assumes that a rank of 1 corresponds to 70% composition, rank 2 to 20%, and rank 3 to 10%. If only one species is found in a quadrat, it would be ranked 1, 2 and 3 (100%). If two species are found, one may be given ranks of 1 and 2 (90%), ranks 1 and 3 (80%), or ranks 2 and 3 (30%), depending on the relative weight for the two species. The values for each quadrat are then summed for each species and expressed as percentages of the total score. This approximates the percentage contribution by weight of each species, from which the overall composition of the sample area is derived. Quadrat size has no effect on the

results of DWR and the method is universally applicable regardless of the type of vegetation because it is based on dry weights (Mannetje & Haydock 1963).

The Dry-weight Rank method is suitable for grassland and small shrubs types or understory communities of large shrub or tree communities, however it does not work well on large shrubs and trees. An advantage of this method is that a large number of samples can be collected quickly. It is useful because it deals with estimates of production, which allows for better interpretation of the data to make management decisions. It is also easier to rank the top three species in a quadrat and easier to apply them similarly by individual observers, resulting in less observer bias.

However, the disadvantage with this technique is that, by itself, it will not give a reliable estimate of plant standing crop, and it assumes there are few empty quadrats. In many large shrub or sparse desert communities, a high percentage of quadrats are empty or have only one species present. The quadrat size required to address these concerns is often impractical. Sufficient training for evaluators performing this method is required.

3.3. MATERIALS AND METHODS

3.3.1. Study area

The experiment was conducted in the years 2012 - 2014 on three red alder blocks at the United Kingdom's Silvopastoral National Network Experiment (SNNE) located in the Bangor University's Henfaes research farm (53°14'N 4°01'W), Abergwyngregyn, Gwynedd, Wales. The local climate in Henfaes is cool and temperate. Mean monthly temperature over the course of this study (2012 – 2014) was 10.6 °C, and temperatures of the warmest and coldest month was 20.0 °C in July and 3.4 °C in January, respectively. Average monthly precipitation ranged from a minimum of 25 mm in April to a maximum of 114 mm in December. The red alder blocks were 4,225 m² (0.42 ha) each and sown to a mixture of perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) at establishment in 1992. The experimental area was rotationally grazed by sheep through the period of the study at average stocking rate of 0.5 to 1.0 AU per ha (Teklehaimanot et al., 2002). A detailed description of the study area including weather data, vegetation and earlier treatments to pasture establishment for the period of the study has been previously reported in chapter 1.

3.3.2. Experimental design and data collection

The study consisted of two similar experiments structured to systematically measure pasture species composition and diversity in red alder blocks and in adjacent open pastures (agricultural control blocks). The treatments were replicated three times in a randomized complete block design. A sample area of 60 m x 60 m in size was demarcated in each of the three red alder blocks as well as in the open pastures adjacent to each alder block to allow greater area within the study area to have an equal chance to be sampled. Experiment 1 assessed the short-term (1

year) changes in pasture species composition and diversity using the Dry-Weight Rank method, which involved rapid estimation of the percentage dry weight of the three most dominant pasture species that are present in a quadrat (Mannetje and Haydock, 1963), while experiment 2 evaluated the medium-term (20 years) changes in pasture diversity utilising the more intensive Point-Intercept method to estimate the percentage cover of pasture species (Cook and Stubbendieck, 1986) along transverse transects.

Earlier surveys of the botanical composition of pasture in all treatment blocks at Henfaes experimental site were conducted in 1993 and planned to be repeated every three years thereafter but routine measurements were stopped abruptly after 1993 with the hope of continuation at a future date. The site has been retained on a care and maintenance basis ever since. However, botanical surveys were conducted later in 1997, 1998, and 2001 by postgraduate students of Bangor University, School of Environment, Natural Resources and Geography, and in 2013 and 2014 by a PhD student and author of this research paper. A higher degree of confidence is placed on the survey data for 1993, 2013 and 2014 as they were collected by experienced surveyors whereas some caution is needed for the cover data for 1997, 1998 and 2001 which were collected by less experienced M.Sc. Students.

3.3.3. Experiment 1: Assessing short-term changes in pasture species composition and diversity (1 year)

This section made a rapid evaluation of the composition of pasture species and the percentage contributions of their dry weight to the pasture under alder trees and in the open before and after thinning. Botanical surveys were conducted in June 2013, before the three 200 stems ha⁻¹ alder blocks were thinned to 100 stems ha⁻¹, and in June 2014 after the thinning.

Three parallel transect lines, 60 metres long and 20 metres apart, were established within each red alder block, and pasture sampling was conducted by systematically placing 12 quadrats (0.25 m^2) along each transect line at 5-metre intervals and estimating pasture species composition within the quadrats using the dry-weight rank method of Mannetje and Haydock (1963) as modified by Jones and Hargraves (1979). A total of 108 quadrats were sampled along 9 transects in the three blocks.

Each quadrat was visually observed and the three most dominant species were ranked depending on their relative weight (Mannetje and Haydock, 1963). The first, second, and third most abundant species (on a dry weight basis) were identified to which the ranks of 1, 2, and 3, were respectively assigned. All other species present were ignored. At the end of the sampling, ranks were tallied for each species, and weighted by a set of multipliers provided by Jones and Hargreaves (1979). The result of the weighted values of the three ranks were then added together for each species to represent species percent composition. Recommended procedures for treating equal-ranked species, less than 3 species, and species which consistently formed a high proportion of the biomass were followed. The details of the method are available in Mannetje and Haydock (1963), Tothill et al. (1978) and Jones and Hargreaves (1979).

The distance between a quadrat and the closest tree was no less than 2.5 metres (Figure 3.1). For effective measurement of changes in the pasture species composition in both understorey and open pasture, the same transect lines and quadrat points were used for measurements before and after thinning. Each sampling area was set in a grid pattern and transect lines and midpoints

of quadrat were marked with labelled pegs the first year for easy identification. Transects were oriented north to south, and sampling positions were geo-referenced using a GPS to an accuracy of ± 5 cm.

There was no estimation of the level of forage utilisation by livestock.

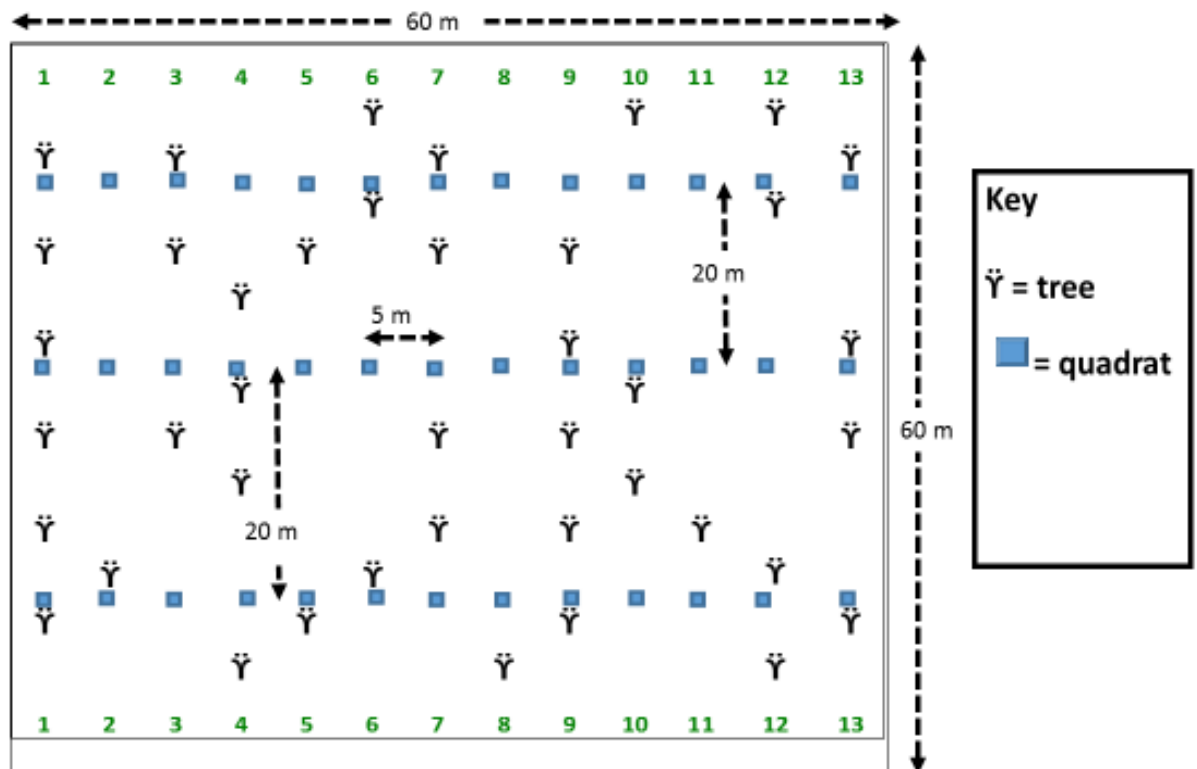


Figure 3-1 Layout of transects and quadrats in red alder blocks at Henfaes SNNE

3.3.4. Experiment 2: Assessing changes (after 20 years) in pasture species composition and diversity

This study used the pasture species cover data for the years 1993, 1997, 1998, 2001 and 2014 to assess the medium-term changes in the pasture species composition and diversity. The 2014 pasture sampling was conducted in the month of June, at the peak of the flowering/growing season, when individual species were most readily identified, using the Point Intercept method (Cook and Stubbendieck 1986) as had been used in the earlier pasture studies at the Henfaes SNNE. This involved taking 1000 hits per red alder block along two diagonal transects (500 hits per transect) in the NE-SW and the NW-SE directions with a reading taken every 10-cm interval. This method was repeated in the red alder block and the agricultural control block. Hits were recorded on an individual species basis. Percentage cover was calculated by dividing the number of hits for each plant species or ground cover class by the total number of points along a transect.

All species within each quadrat were sampled, initially recorded by species but later grouped into functional types: sown species, grass weeds and forb (broadleaf) weeds (Table 3.1). Assessments were made on a percentage of cover basis.

3.3.5. Analysis of data

For short-term changes, species percent composition data from quadrats along the three parallel 60-metre transects were pooled for each block. A Chi Square analysis was used to determine if the frequency of each species in each rank tally group (1,2, or 3) has changed from one sampling period to another (Coulloudon et al., 1999). Each species was analysed separately.

For medium-term changes, species diversity was determined for each block sample using three measures: species richness (R'), the Shannon-Wiener diversity index (H') and the Shannon-Wiener evenness index (E_H). Species richness was calculated by summing the number of different species for the sown, grass weeds and forb weeds. The Shannon-Wiener diversity index was calculated with the formula $H' = - \sum p_i \ln p_i$ (equation 3.1) while the Evenness, expressed by the Shannon equitability index, was calculated with the formula $E_H = H/H_{MAX}$ (equation 3.2). Species richness, diversity and evenness for the three alder blocks were calculated for the years 1993, 1997, 1998, 2001 and 2014. The statistical significance for all analyses was determined by Analysis of Variance using the General Linear Model procedures. Significant level was set at $\alpha = 0.05$. A correlation analysis was also conducted across blocks and years to examine how overall species diversity (R , H' and E) and diversity within functional types related to the percentage cover of each functional type.

3.4. RESULTS

3.4.1. Experiment 1. Assessing short-term effects of thinning on understorey pasture species composition; (1 year)

Results of the dry-weight rank method of determining short-term changes in pasture species composition are presented in Table 3.1 and Figure 3.2. Generally, species richness was the same for the treatments. This approach generated eight most dominant pasture species in each of the three blocks and their adjacent control (open pasture) blocks across the treatments comprising of two sown species, four grass weeds, and two forbs weeds, respectively. However, there was a decline in the percentage composition by weight of the sown species (*L. perenne* and *T. repens*) and a slight increase in both grass weeds (*A. capillaris*, *H. lanatus*, *P. pratensis* and *F. pratensis*) and forb weeds (*U. dioica* and *C. arvensis*) after thinning compared to the adjacent open pastures. *A. capillaris* dominated the pastures in all treatments. However, in the open pasture, *L. perenne* is as dominant as *A. capillaris* (Table 3.1 and Figure 3.2).

Specifically, the under-tree *L. perenne* (a sown species) declined from 17% by weight before thinning to 8% by weight one year after thinning compared to the same species in the open pasture (Table 3.1 and Figure 3.2). Similarly, the under-tree *T. repens* (a sown species) decreased from 8% by weight before thinning to 3% by weight one year after thinning compared to the same species in the open pasture. On the other hand, the under-tree *A. capillaris* (a grass weed) increased from 27% by weight before thinning to 30% by weight one year after thinning compared to the same species in the open pasture. Again, the under-tree *U. dioica* (a forb weed) increased from 13% by weight before thinning to 20% by weight one year after thinning compared to the same species in the open pasture.

These results indicate that there were changes in the percent composition by weight of each pasture species after thinning compared to the treeless open pasture, suggesting that thinning does have some effect on the composition of pasture species in the short-term. However, results of the Pearson Chi-Square analyses show that the observed changes in both undertree and open pasture species composition by weight is not statistically significant $\chi^2(1) = 2.0, p = 0.157$.

Table 3.1: Mean percent composition by weight of pasture species in pre- and post-thinned treatments at Henfaes SNNE using the DWR method of pasture assessment.

SPECIES	Under tree				Open pasture			
	Mean % composition		Mean difference	Standard Error of mean	Mean % composition		Mean difference	Standard Error of mean
	Pre-thinning	Post-thinning			Pre-thinning	Post-thinning		
SOWN SPECIES								
<i>Lolium perenne</i>	17	8	-9	4.5	25	26	1	0.5
<i>Trifolium repens</i>	8	3	-5	2.5	10	11	1	0.5
GRASS WEEDS								
<i>Agrostis capillaris</i>	27	30	3	1.5	25	26	1	0.5
<i>Holcus lanatus</i>	11	12	1	0.5	14	15	1	0.5
<i>Poa pratensis</i>	14	15	1	0.5	12	11	-1	0.5
<i>Festuca pratensis</i>	8	9	1	0.5	9	8	-1	0.5
FORB WEEDS								
<i>Urtica dioica</i>	13	20	7	3.5	4	2	-2	1.0
<i>Cirsium arvense</i>	2	3	1	0.5	1	1	0	0.0

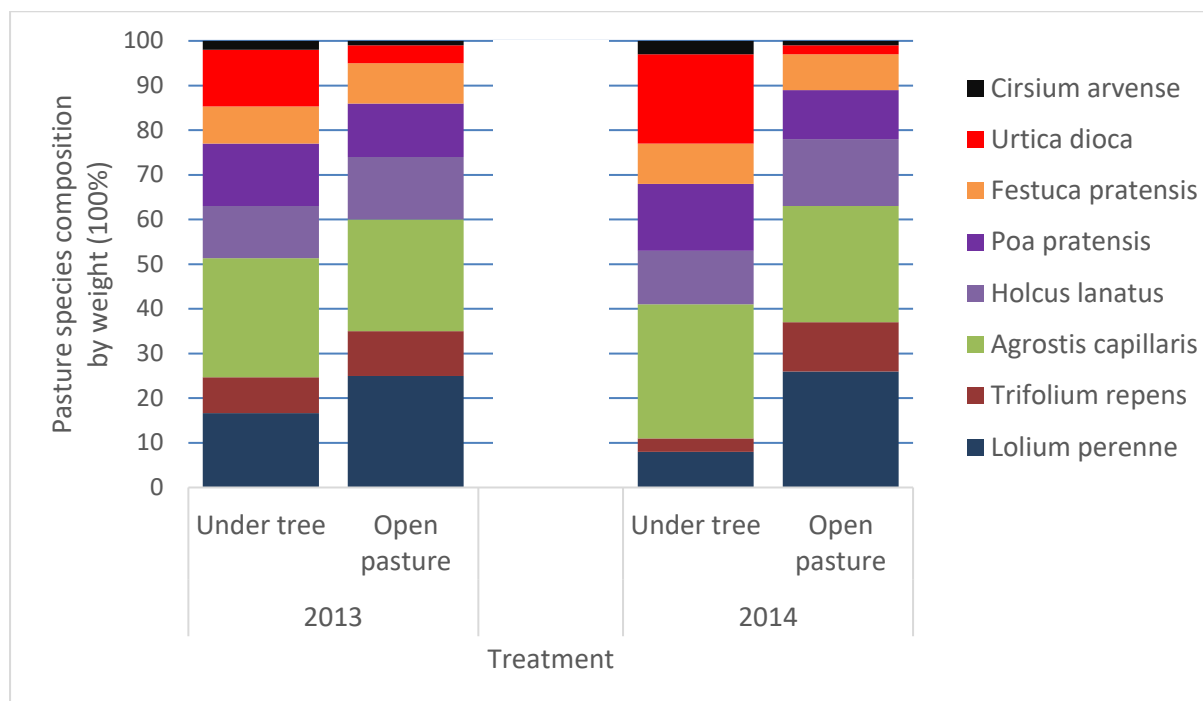


Figure 3-2: Relative percent composition by weight of pasture species from quadrat data before and after thinning (June 2013 – June 2014) at the Henfaes SNNE in North Wales, UK.

3.4.2. Results: Experiment 2: Assessing changes (after 20 years) in pasture species composition and variability in alder plots over time

The use of the point-intercept method of determining medium-term changes in pasture species composition, diversity and abundance generated a total of 19 pasture species in this experiment, out of which two were sown species, five were grass weeds, eleven were forb (broadleaved) weeds and the rest were unidentifiable graminoids and bryophytes (Table 3.2). Accordingly, pasture on the three alder blocks, 20 years after the establishment of the Silvopastoral National Network Experiment at Henfaes, was predominantly grass weeds (46-48%) followed by forbs or broadleaf weeds.

Table 3.2: The relative mean percentage cover of under canopy pasture and open pasture (1993 – 2014).

SPECIES		1993		1997		1998		2001		2014	
Botanical name	Common name	Under tree	Open pasture	Under tree	Open pasture	Under tree	Open pasture	Under tree	Open pasture	Under tree	Open pasture
SOWN SPECIES											
<i>Lolium perenne</i>	Perennial ryegrass	55.7	60	67.7	66.33	77.7	72.9	80.3	75	10	23
<i>Trifolium repens</i>	White clover	29.7	27	26.7	27.33	19	23.2	15.5	22.7	6.7	18.9
GRASS WEEDS											
<i>Agrostis capillaris</i>	Common bent	-	-	-	-	-	-	-	-	27.9	22.3
<i>Holcus lanatus</i>	Yorkshire fog	-	-	-	-	-	-	-	-	10.3	6.4
<i>Poa pratensis</i>	Smooth meadow grass	4.7	8	0.5	0.7	-	-	-	-	13.6	7.4
<i>Festuca pratensis</i>	Meadow fescue	-	-	-	-	-	-	-	-	5.7	7
<i>Bromus hordeaceus</i>	Common soft brome	-	-	-	-	-	-	-	-	1.1	5
FORB WEEDS											
<i>Urtica dioca</i>	Common nettle	-	-	-	-	-	-	0.7	-	16	2.4
<i>Cirsium arvense</i>	Spear thistle	-	-	-	-	-	-	-	-	3.6	2.2
<i>Stellaria media</i>	Common chickweed	3	2	0.4	-	-	-	-	-	0.7	1
<i>Ranunculus acris</i>	Meadow buttercup	-	-	-	-	-	-	-	-	0.7	0.9
<i>Rumex obtusifolius</i>	Broad-leaved dock	6.5	1	-	0.3	1	-	0.2	-	0.6	0.5
<i>Capsella bursa-pastoris</i>	Shepherd's purse	-	-	-	-	-	-	-	-	0.5	0.5
<i>Fumaria officinalis</i>	Earth smoke	-	-	-	-	-	-	-	-	0.4	0.3
<i>Taraxacum officinale</i>	Dandelion	-	-	-	-	-	-	-	-	0.8	0.8
<i>Plantago spp</i>	Common plantain	-	-	-	-	-	-	-	-	0.5	0.5
<i>Cerastium fontanum</i>	Common mouse ear	-	-	-	-	-	-	-	-	0.9	0.9
<i>Eryngium campestre</i>	Field eryngo	-	-	-	-	-	-	0.6	-	-	-
Others		0.4	2	4.7	5.34	2.3	3.9	2.7	2.3	-	-

3.4.3. Species composition

The mean percentage cover of pasture species in red alder silvopasture and in the open is shown in Table 3.3 while the graphical representation of the trends over the years are presented in Figure 3.3. Percentage cover for the sown species remained relatively stable from 1993 until 2001 when it dropped significantly ($p < 0.05$). *Lolium perenne* was observed to be dominant over *Trifolium repens* in both the under tree and the open pasture. The decline in *Lolium perenne* and *Trifolium repens* percentage cover over the sampling period was more rapid than for other functional groups.

Percentage cover for the grass weeds remained relatively low from 1993 until 2001 but rose to significant ($p < 0.05$) level in 2014. *Agrostis capillaris* dominated the grass weeds category under canopy with a mean cover value of 27.9% compared to 22.3% in the control, while *Holcus lanatus*, *Poa pratensis*, *Festuca pratensis* and *Bromus hordeaceus* occupied lower and similar levels of abundance.

The composition of the forb weeds showed similar trends to the grass weeds, dropping from 9.67 percent in 1993 to only 1.67 percent in 2001 on the under-canopy pasture but increased significantly ($p < 0.05$) to 24.67 percent thereafter. The pasture composition on the open pastures showed similar trend. *Urtica dioica* was dominant while the rest of the species in the group indicated lower abundance levels. The percentage cover for other unidentifiable species remained relatively the same over the study period.

Table 3.3: Relative percentage cover of pasture species in relation to functional grouping, year and treatment.

	Sown spp		Grass weeds		Forb weeds		Others	
Year	Under tree	Open pasture	Under tree	Open pasture	Under tree	Open pasture	Under tree	Open pasture
1993	85.33 ^b	87.00 ^b	4.67 ^a	8.00 ^a	9.67 ^b	3.00 ^a	0.33 ^a	2.00 ^a
1997	94.33 ^b	93.67 ^b	0.67 ^a	0.67 ^a	0.67 ^a	0.33 ^a	4.67 ^a	5.33 ^a
1998	96.67 ^b	96.33 ^b	0.00 ^a	0.00 ^a	1.00 ^a	0.00 ^a	2.33 ^a	4.33 ^a
2001	95.67 ^b	97.67 ^b	0.00 ^a	0.00 ^a	1.67 ^a	0.00 ^a	2.67 ^a	2.33 ^a
2014	16.67 ^a	42.00 ^a	58.67 ^b	48.33 ^b	24.67 ^c	10.00 ^b	0.00 ^a	0.00 ^a

^{abc} Means within a column followed by different letters vary significantly ($p < 0.05$).

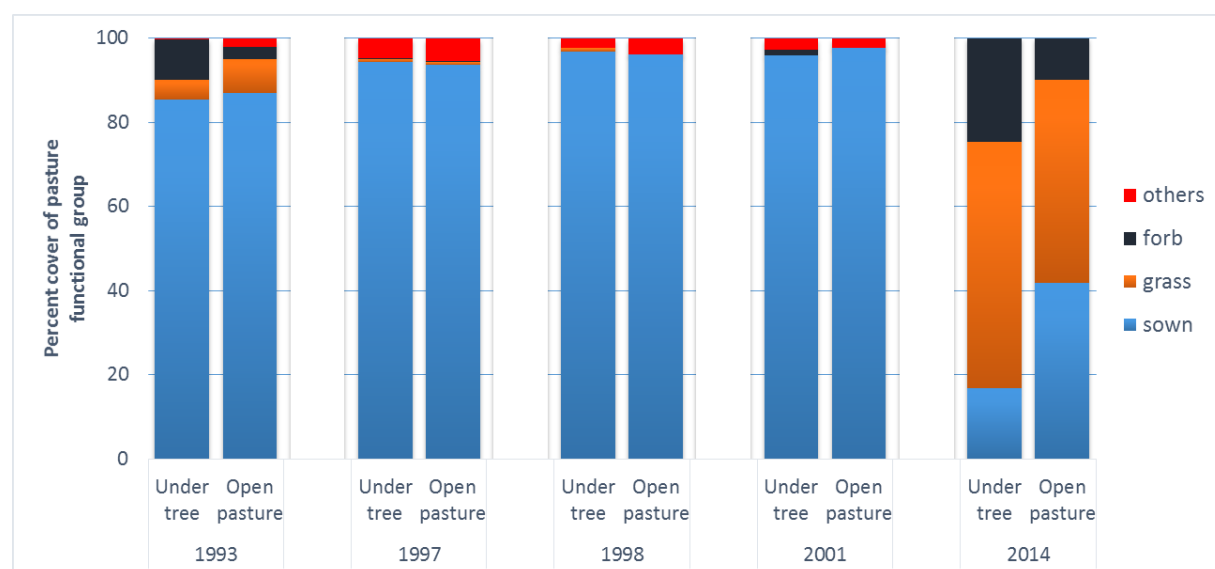


Figure 3-3: Graphical representation of relative percentage cover of pasture species in relation to functional grouping, year and treatment at Henfaes SNNE for the period 1993 – 2014.

3.4.4. Species diversity

Species richness (R'), Shannon-Weiner diversity (H') and Evenness (E') indices under canopy and open pastures are presented in Table 3.4. Generally, species richness on both under tree and open pasture demonstrated similar and parallel trends. Species richness differed significantly ($p < 0.05$) across years and between treatments but not between blocks ($p = 0.474$) nor between treatments ($p = 0.150$). There was no significant year-treatment interaction ($p = 0.692$).

Shannon-Weiner diversity index was significantly different across years ($p < 0.001$) but not across blocks ($p = 0.156$) nor between treatments ($p = 0.891$), and no significant interaction was observed between year and treatment ($p = 0.734$). Species diversity declined steadily from 1993 to 2001 but rose to high level in 2014. The open pasture diversity indicated parallel trend as the under-canopy diversity.

The effect of year, block and treatment was highly significant ($p < 0.05$) for species evenness values. Generally, evenness differed significantly across years ($p < 0.001$), across blocks ($p = 0.024$) and between treatments ($p = 0.035$). The year-treatment interaction was also significant ($p = 0.013$). There was a decline in evenness from 1993 to 1998 but this increased again in 2014. Evenness of species on the open pasture followed a similar pattern but converged in 2014.

Table 3.4: Mean species richness, diversity and evenness as affected by year and treatment

Year	Richness (R')		Diversity (H')		Evenness (E')	
	Under tree \bar{x} (σ)	Open pasture \bar{x} (σ)	Under tree \bar{x} (σ)	Open pasture \bar{x} (σ)	Under tree \bar{x} (σ)	Open pasture \bar{x} (σ)
1993	5.67 ^{bc} (0.57)	5.33 ^{bc} (0.57)	1.11 ^b (0.28)	0.99 ^b (0.21)	0.64 ^a (0.13)	0.59 ^a (0.08)
1997	4.00 ^c (1.00)	3.67 ^c (0.57)	0.77 ^{bc} (0.05)	0.77 ^{bc} (0.08)	0.58 ^{ab} (0.08)	0.61 ^a (0.15)
1998	7.67 ^b (2.08)	7.33 ^b (2.51)	0.65 ^c (0.07)	0.75 ^{bc} (0.14)	0.32 ^c (0.02)	0.38 ^b (0.01)
2001	4.67 ^c (0.57)	2.67 ^c (0.57)	0.59 ^c (0.09)	0.62 ^c (0.06)	0.39 ^{bc} (0.03)	0.67 ^a (0.13)
2014	16.67 ^a (0.57)	16.33 ^a (0.57)	2.10 ^a (0.11)	2.08 ^a (0.15)	0.74 ^a (0.04)	0.79 ^a (0.13)

^{abc} Means within a column followed by different letters vary significantly ($P < 0.05$)

3.5. DISCUSSION

3.5.1. Short-term changes

The results suggest that thinning had some effect on flora species composition by weight between 2013 and 2014, though the change was not statistically significant. The hypothesis that thinning will not change the understory pasture species composition, abundance and diversity in the short-term (up to 1 year) was therefore not corroborated in this study. As expected, the percent pasture species composition by weight of the open pastures were similar before and after the thinning treatments compared to the under-tree pastures. The most consistent difference in species composition was a greater content of *Agrostis capillaris* and *Urtica dioica* in the under-tree pasture. The change may be short-term disturbance impacts due to removal of trees, which agrees with previous research studies by Ducherer *et al.* (2013) who reported that thinning had little effect on the understorey species richness and diversity of ponderosa pine and Douglas-fir tree compared to the pasture control. Thomas *et al.* (1999) observed that thinning could crush and smother understory flora, while other research has upheld that extensive site preparation tree cutting can have detrimental effect on the amount of surface, organic matter (Jurgensen *et al.*, 1997). Tree in-growth have also been reported to have damaging effects on many understory species (Page, 2002).

On the other hand, disturbance occasioned by thinning can increase the availability of new microsites for plant establishment and growth, possibly leading to increased species richness (Brockway *et al.*, 2002). It is common for understorey pasture species to respond to increases in light following thinning. Since understorey vegetation responds to increased light, the

greatest response to the increases in light following thinning are expected within the first several years (Thomas *et al.*, 1999).

Citing previous studies by Bailey *et al.* (1998) and Thomas *et al.* (1999), Ducherer *et al.* (2013) reported that thinning trees enhances understorey species diversity, but that the increase usually occurs more than 10 years after thinning. Thinning had little effect on the understorey species richness and diversity of ponderosa pine and Douglas-fir tree the first 4 years after treatment compared to the pasture control (Ducherer *et al.*, 2013). Species richness improved within three years in a repeatedly thinned Douglas-fir forests in the Pacific Northwest of the U.S after early decline one year after thinning (Thysell and Carey 2001; Ducherer *et al.* 2013). Species richness was also reported to have improved two years after thinning in pinyon–juniper forests in central Mexico (Brockway *et al.* 2002; Ducherer *et al.* 2013). Bailey and Tappeiner (1998) reported similar improvement in shrub cover 10 to 25 years following thinning in Douglas-fir forests. Short term increase in species richness has been linked with exotic species and high treatment intensity (Griffis *et al.* 2001). Metlen *et al.* (2004) suggested that the short-term effect of thinning could be as a result of adaptation of the environments to minor disturbances.

Winter season, minimum disturbance to the soil during thinning using chainsaw, and relatively low tree density before thinning in this study at the Silvopastoral Network Experiment may have contributed to the observed changes in understorey species composition by weight one year after thinning treatment.

3.5.2. Medium-term changes

There was variation in pasture species richness, diversity and evenness across the three alder blocks, over the 20 years since establishment, which may be attributed to interactions between various factors such as amount of disturbance that has occurred, environmental variations, climate, soil nutrient levels as determined by edaphic factors, fertilizer application and recycling of nutrients by animals, habitat management, and the pattern, amount and intensity of grazing by livestock and wild animals such as deer and rabbit.

Benavides et al., (2009) have observed that the botanical composition of understorey pasture generally deteriorates over time because there is a decline in legume and *L. perenne* contents, and an increase in the contents of litter and dead matter, and that the overall content of grasses increases under trees. These authors have further observed that changes in botanical composition are brought about by variations in the microclimate and soil properties, tree spacing, crown size and extent of foliation, and livestock grazing pattern, and that greater changes are more noticeable below widely spaced evergreen trees than under deciduous trees (Benavides *et al.*, 2009).

Lack of light and low temperatures reduces the productivity of pasture species, and consequently may delay their life cycle (Benavides *et al.*, 2009; Balocchi and Phillips 1997; Peri *et al.*, 2001). The shade tolerance of a species will determine its capacity to survive and complete its life cycle beneath trees. The most shade-tolerant species are those that maintain a comparatively high absolute and relative shoot yield under low PAR. Shade tolerance of pasture species is important, particularly as clovers (*Trifolium* spp) are light demanding. There is a 50% loss of productivity of legumes with about 35% shade (Dodd *et al.*, 2005). Usually

light demanding grasses such as ryegrass (*Lolium perenne*) decrease and are replaced with less valuable species (Hawke and Gillingham 1997; Dodd *et al.*, 2005; Benavides *et al.*, 2009).

In the present study, the grass weeds (*A. capillaris*, *H. lanatus*, *P. pratensis* and *F. pratensis*) and the forb weeds (*U. dioca* and *C. arvense*) increased over the years, suggesting novel life form and could proliferate in silvopastoral farms in response to frequent managerial disturbances. This result agrees with other trials reported by Benavides *et al.*, (2009) (Appendix 3.2), which show that the content of *H. lanatus* L., *D. glomerata* L., *Agrostis stolonifera* L. and *P. annua* L. increases beneath trees, and that it is likely that these changes are related to their shade tolerance, tillering ability, phenological development and growth in winter.

The turf-forming grass weeds easily invade managed silvopasture, while the forb weeds easily colonize new sites by seed which establish best in bare or disturbed ground. Silvicultural practices such as clear-cutting, burning, and thinning create opportunities for these weeds to become established and may pose a serious threat to the survival of the desired sown species (Thysell and Carey 2000; Peltzer *et al.*, 2000).

This study has shown that the decline in the sown species rye grass (*Lolium perenne*) and white clover (*Trifolium repens*) percentage cover over the sampling period was more rapid than for other functional groups. Most pasture species are prone to decline due to environmental conditions that do not favour their long-term persistence (Benavides *et al.*, 2009). A decline in the sown species over time or a fluctuation between years has been reported in many trials to be associated with seasonal effects (Cameron and Cannon 1970); it is therefore important to consider these alongside the effect of tree growth and thinning in a silvopasture system.

Previous pasture surveys have shown that most improved pastures have a low prevalence of introduced grasses and clover (Benavides *et al.*, 2009). A reduction in perennial grass and clover is frequently linked with dry summers and autumns (Hutchinson 1990). In these situations, plants are stressed not only by lack of moisture, but also by overgrazing. This can lead to the development of a large percentage of bare ground (Cameron and Cannon 1970) which in turn provides the opportunity for weeds to invade (Medd *et al.*, 1987). Within species comprising of pastures, however, there are varying persistence rates and weed levels that cannot be accounted for by the association between over-grazing, pests and climatic stress.

The decline in the composition of white clover and the proliferation of the grasses and forbs over the years in the present study may have been caused by the continued fixation of N by red alder. Other studies have shown that fertilization with nitrogen affects the botanical composition of pastures negatively by stimulating grass growth and at the same time depressing legume growth (Lee and Lee, 2000). Legumes such as *Trifolium repens* are an integral part of most productive temperate perennial pastures. The decline of legume in a pasture will not only reduce the pasture quality but may lead to the decline of sown grasses and subsequent weed invasion. Under moderate to intense grazing pressure more productive grasses rely on the nitrogen-fixing ability of the legumes to remain competitive.

Intermittent management intervention practices over the years may have been responsible for the erratic fluctuations in the species abundance and diversity observed in this study. It is important to note that since the establishment of the site in 1992 with a mixture of perennial ryegrass and white clover, the pasture has not been reseeded. Again, the red alder blocks have

not received any fertilizer treatment since establishment primarily because of the capacity of red alder tree and white clover to fix Nitrogen. Furthermore, the weeds, *Holcus lanatus*, *Urtica dioica* and *Cirsium arvense*, were treated with “Grazon 90” once in early 1993.

For close control, grazing period of six months, from March to October, was in place for the first eight years of the Henfaes SNNE but was relaxed in 2000 to allow free access to all treatments (Teklehaimanot *et al.*, 2002). Differential grazing may have resulted in the selection of different pasture species in the diets of sheep during particular parts of the year. Over the years, the diets of the sheep could have been influenced by the relative proportions of species available, and the intensity of grazing on the blocks would have varied considerably culminating in interaction with phenological development of different pasture species to influence medium-term changes in botanical composition.

3.6. CONCLUSION

The responses of pasture species to thinning in this study was variable. The percentage composition by weight of the sown species declined, while that of the grass weeds and the forb weeds increased slightly one year after thinning compared to the adjacent open pastures. Again, the greatest composition of the weeds was observed in the open pasture. The hypothesis that thinning will not influence understory pasture species composition, abundance and diversity in the short-term was therefore rejected in this study. It is evident from the results that there were little changes in pasture species composition one year after thinning compared to the open pastures, suggesting that minimum disturbance to the soil during thinning using chainsaw, together with cold winter condition and relatively low tree density before thinning in the study area, may have contributed to the minor change. Consequently, it is expected that the

availability of new microsites essential for plant development as well as the response of the understory to increased light, as a result of more open canopy (less shade), would eventually lead to increased species diversity and abundance within the first few years. These results indicate that there were changes in the composition by weight of the pasture species after thinning compared to the treeless open pasture, suggesting that thinning does have some effect on the composition of pasture species in the short-term.

Compared to the under-tree pastures, the percent pasture species composition by weight of the open pastures remained relatively unchanged over the years. The second hypothesis that there will be no change in understory pasture species composition, abundance and diversity in the medium-term was also rejected. Variations were detected in pasture species composition, diversity and abundance across the three red alder blocks studied over the twenty years since establishment.

Percentage cover of the sown species, rye grass (*Lolium perenne*) and white clover (*Trifolium repens*), in both the under-tree and open pasture remained relatively stable from establishment in 1993 until 2001 but dropped significantly ($p < 0.05$) in 2014 to lower level. In contrast, the percentage cover of the grass and forb weeds in both under-tree and open pastures were also relatively low until 2001 but rose significantly ($p < 0.05$) in 2014. Again, the under-tree percentage cover of the sown species, declined more rapidly than for other functional groups compared to the open pasture. Furthermore, both under-tree and open pasture species richness, diversity, and evenness indices followed similar trend as the species percentage cover, declining progressively from 1993 to 2001 but rising to high levels in 2014. Generally, the grass weeds dominated both under-tree and open pastures 21 years after the establishment of

the Silvopastoral National Network Experiment at Henfaes, accounting for about 46% to 48% of pastures on the three red alder blocks.

The observed temporal changes in both under-tree and open pastures species richness, diversity and evenness in this study may be attributed to a number of factors including the continued fixation of N by red alder, sporadic management intervention practices over the years, microclimate and soil properties, tree density and spacing, differential grazing by livestock and wild animals such as rabbits and deer, and invasion and proliferation of other species.

Chapter 4 : ALLOMETRIC EQUATIONS FOR ESTIMATING BIOMASS AND CARBON STOCK OF OPEN-GROWN RED ALDER (*Alnus rubra* Bong) IN A SILVOPASTORAL AGROFORESTRY SYSTEM

4.1. INTRODUCTION

The mitigation of carbon dioxide emissions is a global concern and has underscored the need to develop the skills for accurate measurement of carbon stored and sequestered in forests (Brown *et al.*, 1996; Kauppi and Sedjo, 2001). It is generally recognised that the emission of carbon dioxide, the main greenhouse gas (GHG), to the atmosphere is mainly as a result of worldwide burning of fossil fuels. Disturbances such as indiscriminate exploitation of forest resources, wildfire, pest and disease occurrences, and conversion to non-forest use, particularly infrastructure, agriculture and pastures, are the sources of carbon dioxide because total respiration or oxidation of plants, soil, and dead organic matter have surpassed net primary productivity (Houghton *et al.*, 2001).

The United Nations Framework Convention on Climate Change and in particular the Kyoto Protocol (Breidenich *et al.*, 1998) highlights the need to monitor, regulate and maintain forest carbon stock. The steady rise in atmospheric carbon dioxide (CO₂) in recent times is causing a serious global concern. The process of photosynthesis in plants sequesters and stores Carbon (C) as biomass in different components of the tree. The absorption of CO₂ from the atmosphere and the sequestration and storage of C in different plant tissues as biomass result in the development of different tree components (Ali *et al.*, 2014). Tree increases in growth as more

CO₂ is absorbed and excess C is stored in different plant organs. Trees therefore act as a sink for CO₂ by fixing C and sequestering excess C as biomass in different tree sections.

The rate of C sequestration in trees is species dependent. The higher the quantity of tree biomass the higher the C sequestered in the whole tree as well as in tree components (Ali *et al.*, 2014). In a forest setting, the level of CO₂ sequestration is a function of forest type, dominant species, density and age (Huy and Anh 2008; Ali *et al.*, 2014).

The importance of forest trees in the global C cycling is well recognised as the amount of C stored in plant biomass globally surpasses that of atmospheric CO₂. It has been reported that almost 90% of the plant biomass C is stockpiled in tree biomass (Ali *et al.*, 2014), and that forest biomass represents about 44% of the globe forest C pool (Pan *et al.*, 2011). Hence, trees play significant role in climate change mitigation. This stresses the imperative need to accurately regulate the amount of C stored and CO₂ sequestered in specific forest ecosystems of which silvopastoral agroforestry system is a part.

The integration of trees on farms or pastures is known to increase the amount of C sequestered compared to a monoculture field of crop plants or pasture (Sharrow and Ismail 2004; Kirby and Potvin 2007). Agroforestry systems can also sequester significant amount of C stored in both aboveground and belowground biomass.

In an analysis report by the International Panel on Climate Change (IPCC) on Land-Use Change and Forestry, re-/afforestation, the conversion of agricultural land into agroforestry systems, has been recommended as one of the measures to mitigate increasing CO₂ emissions

(Godal, 2003; Jose and Bardhan, 2012). Agroforestry systems (including silvopasture) have therefore emerged as land use with the greatest potential for carbon sink because the integration of trees results in greater CO₂ sequestration from the atmosphere and thus enhance carbon storage in permanent tree components (Dixon 1995; Montagnini and Nair 2004).

In the light of the argument for the imperative need to monitor, regulate and maintain forest carbon stock, the Kyoto Protocol has led to greater worldwide attention being given to agroforestry as a strategy to sequester carbon. Amount of carbon sequestered in agroforestry has been estimated to range from 0.29 to 15.21 Mg C ha⁻¹ yr⁻¹ above ground, and 30–300 Mg C ha⁻¹ yr⁻¹ up to 1 m depth in the soil (Nair et al. 2010; Jose and Bardhan, 2012). Trees growing in agroforestry systems can be crucial in national carbon budgets and the climatic regulation system.

Carbon sequestration potential in agroforestry systems in the tropical zone has been estimated to range between 21 and 240 t C ha⁻¹, achieved within a cutting cycle of ten or twenty years (Dixon 1995; Adesina *et al.*, 1999; Montagnini and Nair 2004). while the estimates for Carbon sequestration potential in temperate climate agroforestry has been placed between 10 and 208 t C ha⁻¹ over a longer cutting cycle of twenty to fifty years (Schroeder 1994; Dixon 1995; Turnock 2001; Montagnini and Nair 2004).

Despite the recognition of these potentials and the paucity of data on biomass and carbon stored in trees on farms, there is a limited understanding of biomass and carbon sequestration in specific agroforestry practices from around the world. Moreover, the carbon stock of these landscapes has not been adequately quantified, and attempts to measure these accurately have

been impeded by the absence of effective allometric equations that would enable the conversion of tree measurements into biomass and finally into carbon (Jose and Bardhan, 2012; Kuyah *et al.*, 2012). This may in turn lead to a lack of appreciation of the significant quantities of carbon that are stored in agroforestry systems, thereby reducing the attention and value assigned to this form of land use (Kuyah *et al.*, 2012). However, with increasing interest on farm forestry, agroforestry and extension of agricultural boundary in many countries, the interest in developing the potentials of such open-grown trees for carbon sequestration is on the rise (Kuyah *et al.*, 2015).

Open-grown trees planted in agroforestry landscapes are potentially useful in storing additional carbon and can be harvested and used as a fuel in place of non-renewable energy sources (e.g. coal, oil, gas...) in addition to their primary function as wind breaks, microclimate amelioration, conservation of soil and water, and wildlife habitat (Zhou *et al.*, 2011). Variations in the carbon stock from these systems modifies the amount of CO₂ in the atmosphere.

The principal element for the reliable estimation of agroforestry carbon stocks is the estimation of tree biomass. Tree biomass estimation provides vital biological information and has been used for various purposes, such as in assessing forest structure and condition, forest productivity and sustainability, timber extraction, estimating the carbon stocks of forest and CO₂ dynamics and their greenhouse effect, and studying biogeochemical cycles (e.g., Zianis and Mencuccini 2003; Cole and Ewel 2006; Vashum and Jayakumar 2012). For proper accounting of carbon stock, reliable volume estimation of the component parts of trees are essential for total biomass to be derived. Carbon stock quantification efforts are typically forest-based, to the exclusion of trees in agroforestry landscape. Consequently, allometric

relationships to predict biomass and carbon have typically only been developed for tree species grown in forest environments to the virtual exclusion of agricultural lands.

It is important to note that allometric relationships have been developed for trees growing in forests because these trees are typically grown with a final end-use in mind (e.g. construction), so it's useful to be able to predict shape, volume, yield etc. On the other hand, objectives for growing trees on agricultural farms are typically much broader and therefore there has been much less need for accurate allometric relationships to predict volume and carbon.

The development of allometric equations that are applicable to varied forms of open-grown trees is important not only for proper accounting of carbon stock in agroforestry landscapes but also for policy makers as tool for formulating the best suitable environmental policy decisions. Open-grown trees have larger crowns and sharper trunk tapers than more closed-canopy counterparts (Cole and Jensen, 1982; Sharma and Parton, 2009). The larger crown of open-grown trees is empirically expected to contribute more branch biomass. Trees growing in plantations respond to crowding by growing taller while allocating relatively less to stem diameter increments. On the other hand, Holbrook and Putz, 1989; Zhou *et al.*, 2011 have observed that open-grown trees allocate relatively more biomass to leaves and branches than do trees crowded in dense plantations, that relative to forest-grown trees, open-grown trees are established in shelterbelts mainly as wind shields because of the larger relative crown that is then exposed to heavier wind force capacity from open fields. They postulated that open-grown trees support more branch weight on the same trunk biomass base, a strength which they attributed to stronger mechanical stem structure in consolidation with sharper trunk taper and increase in specific gravity.

Tree allometry relates easily measurable variable such as tree stem diameter and height to other structural and functional tree characteristics (Wang, 2006). Allometric equations used in estimating biomass of trees in agroforestry systems are most often derived from trees grown in forest systems that are totally different in their growth form from open-grown trees typically found in agroforestry systems. Therefore, applying such models to trees on agroforestry scale analysis can be challenging as there can be errors in estimating both biomass and carbon stock potentials. This highlights the imperative need for the development of species specific allometric equations for trees grown in agroforestry systems to circumvent this limitation.

Although several allometric equations have been developed for temperate forests (e.g. Wang, 2006) and specifically for red alder, little is known about red alder production in agroforestry plantations. Despite more than 20 years of agroforestry research at the SNNE site at Henfaes, no allometric equations have yet been developed for red alder grown in agroforestry configurations

Red alder is known for its fast growth rate and potential to produce a range of quality wood (Mmolotsi and Teklehaimanot 2006). In 8 to 10-year-old naturally established red alder plots, annual biomass production can reach 29 Mg ha⁻¹ (Smith, 1977). The aboveground biomass of red alder growing in a natural stand increased very rapidly during the first 15-20 years and reached about 240 metric tons/ha by the age of 33 years under ideal soil-moisture conditions (Zavitkovski and Stevens, 1972).

A wealth of allometric equations that relate stem volume and biomass of the various tree components to diameter at breast height and/or to tree height has been compiled for European tree species (Zianis *et al.*, 2005). However, allometric equation for red alder (*Alnus rubra* Bong.) is conspicuous in its absence from the compiled equations. Since there is a dearth of information on allometric equations for red alder in the UK, this study partially fills that deficiency and falls into site-specific studies as it is focused on the determination of allometric relationships in a silvopastoral national network experiment setting. Allometric equations developed in this study can therefore be useful not only in the estimation of tree biomass and carbon sequestration in agroforestry landscapes but also serve as tool for policy makers for the formulation of appropriate environmental policy decisions. Obtained allometric relationships could also form the inputs into eco-physiological studies and growth or carbon uptake models.

4.1.1. Objectives:

This study aims to assess the aboveground biomass and carbon stock of two forms of open-grown red alder (*Alnus rubra* Bong) trees in a lowland silvopastoral system in North Wales, UK.

Specifically, the study will seek to:

1. Determine the total aboveground biomass of samples of ‘good’ and ‘poor’ form open-grown red alder trees through destructive sampling.
2. Estimate the carbon and carbon dioxide sequestration potentials of the two forms of open-grown red alder trees.
3. Develop yield prediction models for the two forms of open-grown red alder trees.

4.1.2. Hypotheses

The estimates of dendrometric measurements, aboveground biomass, carbon and carbon dioxide contents of the good form red alder samples do not differ significantly from that of the poor form red alder samples.

4.2. LITERATURE REVIEW

4.2.1. Forest tree biomass and carbon stock estimation

Biomass, in relation to the forest biomass issue, is a vital indicator of ecosystem energy potential and productivity. Typically expressed in terms of dry weight of organic matter, biomass refers to the weight or mass of its living plant tissue and is generally expressed in units of metric tonnes (t) or oven-dry tonnes of matter per unit area (individual plant, hectare, region or country). Biomass, in general, includes the above ground biomass (leaves, branches and stems) and below ground biomass (roots) components. Estimates may be restricted to the aboveground section of trees only, or to tree components (such as leaves, wood, etc.), or to belowground portions. Most past research studies on biomass estimation centred on aboveground biomass because of the difficulty in collecting field data of below ground biomass (Lu, 2006). It is therefore most common to estimate the aboveground live dry biomass of a tree, which is the weight of the living aboveground plant tissue after all the water has been removed, i.e., after the leaves, branches, and stems have been dried thoroughly, often using a special laboratory oven. In general, water accounts for approximately 50% or ½ of the weight (or wet biomass) of a live tree.

In addition to widespread use in estimating the carbon stocks of forest and CO₂ dynamics and their greenhouse effect (Rokityanskiy *et al.*, 2007; Wulder *et al.*, 2008), biomass estimates are important for a broad range of applications, including: characterizing forest structure, conditions and processes (Wulder *et al.*, 2008); assessing forest productivity, timber extraction and sustainability; modelling impacts of fire and other disturbances; modelling the environmental and economic consequences of energy production from biomass; Monitoring changes in biomass over time; and for studying biogeochemical cycles (e.g., Zianis and

Mencuccini 2002; Cole and Ewel 2006; Vashum and Jayakumar 2012). The usual approaches to estimating aboveground biomass (AGB) are through traditional field-based measurement and remote sensing and geospatial information system (GIS) methods (Brown and Gaston 1995; Schroeder *et al.*, 1997; Houghton *et al.*, 2001; Santos *et al.*, 2003; Zhang *et al.*, 2004; Lu 2006; Vashum and Jayakumar 2012).

There are two field-based methods of measuring forest biomass and carbon storage in the forest ecosystems: the destructive and the non-destructive methods. The destructive, also known as the harvest method, is the most direct and accurate method of biomass estimate and consists of felling the trees in an area and measuring the weight of the different components like the roots, stem, branches, and foliage (Zianis and Mencuccini 2002; Segura and Kanninen 2005; Vashum and Jayakumar 2012) and measuring the weight of these components after they are oven dried. The biomass of an area can be accurately measured. However, this approach is destructive, strenuous, time and resource consuming, and expensive, and is limited to small area or small tree samples destructive, difficult to implement, especially in remote areas, and are only feasible for a small-scale analysis (Kuyah 2012; Vashum and Jayakumar 2012), and cannot be applied to degraded forests having rare or protected species (Montes *et al.*, 2000).

The aboveground forest biomass can also be estimated directly but non-destructively by climbing the tree to measure the various component parts or by simply measuring the diameter, height, volume, and wood density of the tree and applying biomass expansion factors, as well as by using available generalized or species-specific allometric equations (Brown *et al.*, 1989 Aboal 2005; Ravindranath and Ostwald 2008; Vashum and Jayakumar 2012). Again, this

method is limited to a small area or to small tree samples and are often labour and time intensive, and expensive, and climbing can be strenuous and risky.

To avoid the challenges associated with destructive sampling and climbing of trees, indirect approaches have been conceived. The indirect method estimates the biomass of a tree without felling (non-destructive) and is usually used when the tree has large dimensions and in environments where the harvesting of rare or protected tree species is not very practical or feasible (Vashum and Jayakumar 2012). Indirect methods include use of allometric relationships (Brown, 1997), functional branch analysis (van Noordwijk and Mulia, 2002), photographic techniques (Jonckheere *et al.*, 2004), remote sensing, and geospatial information system (Brown 2002; Suarez *et al.*, 2005; Gibbs *et al.*, 2007; Wulder *et al.*, 2008; Vashum and Jayakumar 2012). Although indirect methods have many advantages over direct methods and since there is no felling of tree species, it is not easy to validate the reliability of these methods.

4.2.2. The use of remote sensing and GIS techniques for biomass estimation

In recent years, geospatial based technologies such as remote sensing, global positioning system (GPS) along with geographic information system (GIS) have increasingly attracted scientific interest as a solution to the problems posed by the conventional field measurement method of estimating forest biomass and carbon stock (Patenaude *et al.*, 2005; Lu 2006; Szwagrzyk and Gazda 2007; Wulder *et al.*, 2008; Gonzalez *et al.*, 2010; Koch, 2010; Zhang *et al.*, 2014; Chen *et al.*, 2015; Dube and Mutanga 2015). Remote sensing is a useful and important source of data for developing land-use and forest cover maps and estimating aboveground biomass and forest carbon stock of different vegetation types including

agroforestry and for analysing change across wide areas that are otherwise difficult to access (Dube & Mutanga 2015; Chen *et al.*, 2015; Gilani *et al.*, 2015).

This technique involves the use of instruments to remotely acquire and analyse data about the properties of an object or area from a distance without physical contact between the collecting device the object or area under observation. Again, the technique allows for the acquisition of continuous and repetitive digital data with different spatial resolutions from the same area and at multiple scales over large and even difficult or inaccessible areas. The remotely sensed digital data may be processed by a computer and integrated with other datasets in a GIS, or processed automatically thereby interpreting the images with less bias. There are three main remote sensing systems with different features of spectral, spatial, and temporal resolutions that are either passive or active sensors currently used for biomass and carbon stock estimation (Kumar *et al.*, 2015). The precise suitability of each depends on the scale of study and the nature of the observed objects or processes (Suarez *et al.*, 2005):

Optical remote sensing (satellites and airborne sensors) are passive sensors that detect emitted or reflected natural radiation from the object or surrounding area being observed. Optical remote sensing techniques involve the use of reflected sunlight energy in the visible, near and middle infrared domains (400 - 2500 nm wavelengths) (Malthus *et al.*, 2002) and provides a two-dimensional view of topographic features on earth surface. Data for AGB estimation can be secured at different spatial resolutions (fine, medium and coarse), and are the most established remote sensing methods in forestry (Malthus *et al.*, 2002).

Optical sensor data include aerial photographs, IKONOS and QuickBird images, Landsat, SPOT, IRS, TM, AVHRR, MODIS and ASTER data. Their areas of greatest potential are the modelling of tree parameters or forest canopy structure, the classification of vegetation types and valuation of stocking levels. One of the advantages of optical remote sensing technique is that it provides consistent spatial data at both local and global scales (Timothy *et al.*, 2016).

RADAR (radio detection and ranging) and LiDAR (light detection and ranging) are active sensors that emit energy in order to scan objects and areas and detect and measure the radiation that is reflected or backscattered from the target (Lu 2006). Radar sensors send out electromagnetic radiation beams in the form of microwaves and measure the backscattered energy of the microwaves reflected from surfaces beneath. The high sensitivity of the microwaves to the moisture content of different forest structures enables them to penetrate into the forest canopy and scatters back from stems, branches, leaves, and soils and so generate complete third dimensional statistics about the forest canopy. The radar systems have unique advantage over optical sensors as it can penetrate the cloud and can collect topographic feature data irrespective of light or weather conditions. The radar data have been used extensively in the collection of various forms of forest information, including data for the estimation of forest biomass and carbon sequestration (Suarez *et al.*, 2005).

LiDAR (Light Detection and Ranging) is another active sensor used for forest parameter estimation (Lefsky *et al.*, 2002; Lu 2006). LiDAR sensor emits laser pulses of near infrared laser light towards a target, and measures the return time for the backscattered or reflected light using ultra accurate clocks (Suarez 2005; Dube and Mutanga, 2015), and has been described as very robust and most suitable for the generation of data for accurate characterisation of

vertically-distributed forest structural attributes such as canopy heights, stand volume, and the vertical structure of the forest canopy necessary for AGB estimation (Patenaude *et al.*, 2005b; Koch *et al.*, 2010; Dube and Mutanga, 2015).

Several studies have been conducted to demonstrate the use of remote sensing (with the data collected from the field) in the estimation of forest biomass:

Hudak *et al.* (2012) found that high resolution and spatially explicit biomass and carbon dynamics in conifer forests can be accurately estimated by combining repeated LiDAR surveys with field sampling and statistical modelling. Ene *et al.* (2012) used the airborne laser scanning (ALS) sampling approach to evaluate the accuracy of LiDAR-based biomass estimation. Their finding suggested the systematic ALS assisted survey was more efficient than the ground-based inventory. Attarchi and Gloaguen (2014) used data from both optical and L-band Synthetic Aperture Radar (SAR) to develop models for the estimation of above ground biomass in mountain forests. Their results established that topographically and atmospherically corrected data are indispensable for the estimation of mountain forest's physical properties. Using optical satellite image data, Tomppo (1990) estimated forest area and volume in Finland. Again, airborne LiDAR sensing technique is in use for mapping tree heights at individual tree level (Persson, *et al.*, 2002; Hyypä *et al.*, 2003) or at stand level (Naesset, 2003). High resolution satellite imagery has been used in Canada for mapping individual tree canopies that subsequently are classified into a species distribution map for logging operations (Gougeon and Leckie, 2003). Lefsky *et al.* (2002) used LiDAR remote sensing to estimate the above-ground biomass in three biomes-temperate deciduous, temperate coniferous and boreal coniferous. Omasa *et al.* (2003) used a high resolution, helicopter-borne 3-dimensional (3-D)

scanning LiDAR system to develop a methodology for estimating carbon stocks in a Japanese cedar forest. Their LiDAR system measures the 3-D canopy structure of every tree in the forest.

Popescu (2007) discovered that LiDAR data can be used to measure precisely the diameter at breast height of individual trees, which is one of the commonly used variables for biomass estimation of forest. Calders et al. (2014) compared estimates of AGB from Terrestrial laser scanning (TLS) against estimates of AGB derived from destructive harvesting and from allometric equations. They also evaluated tree parameters, diameter at breast height and tree height, estimated from traditional field inventory and TLS data. Their results demonstrated not only that diameter at breast height can be extracted accurately from LiDAR data, but also that AGB estimates derived from TLS show better agreement with the reference values from destructive sampling than from allometric equation.

4.2.3. Modelling of total aboveground biomass

The most common tools for estimating biomass of a given forest stand is using tree allometric equations combined with forest inventories (Henry *et al.*, 2013). Tree allometric equation relates aboveground biomass (AGB), wood volume or that of several tree components to stem diameter at breast height DBH and/or to tree height (HT) and/or other dendrometric variables. An area or a few trees are destructively sampled and the weight of each component determined and related by regression to some dimensions of the standing tree. DBH is commonly used to estimate AGB because it is easy to measure accurately, repeatedly and conventionally (Kuyah 2012). Consequently, in specific forest ecosystem studies, allometric equations based on DBH can be refined by including other variables such as HT, crown area (CA) or wood density (WD)

to improve the tree AGB accuracy (Ketterings et al., 2001; Chave *et al.*, 2005). Sometimes, regression may be calculated using combinations of some of the variables (usually DBH^2HT) to obtain a linear relation in arithmetic units. Logarithm transformation and back transformation to arithmetic units are usually employed when the necessary assumptions of regression analysis are violated (Baskerville 1972).

Parresol (1999) emphasized that regression analysis (both linear and nonlinear) is the most commonly used mathematical models for calculating tree biomass, but most equations take the simple nonlinear form (Zianis et al., 2005; Niklas and Spatz, 2004): $Y = aX^b + \varepsilon$

Where Y is the tree biomass/volume, X is usually the independent variable (such as DBH, HT, ...) a is a constant (the allometric coefficient) and b is the allometric exponent (which defines the relationship between X and Y), and ε is the error term. The variability of Y is often largely explained by the variability of X .

The data for X and Y measured from destructively sampled trees that represent the diameter range within the stands under study are usually transformed using natural logarithms, and then regression equations are fitted using ordinary least-squares techniques to obtain estimates for the coefficients a and b . The transformation improves parameter estimation by reducing variability and heteroscedasticity. These equations take the form of $\ln(Y) = a + b \times \ln(X)$ and can be rewritten as $Y = a e^{bx}$ where e is the natural logarithm base. The linear equation form is $Y = a + bX$.

Transforming of nonlinear model to linear logarithmic form and fitting by the ordinary least square method generally produce an inherent negative bias estimates of biomass following

back-transformation from logarithm. Several correction factors have been studied and presented by many researchers (Finney 1941; Baskerville 1972; Beauchamp and Olson 1973; Snowdon 1991; Zeng and Tang 2011) of which the correction factor $\text{EXP}(S^2/2)$ presented by Baskerville has been most commonly used in practice (Zeng and Tang 2011).

The predictors for AGB estimates used by researchers vary depending on the objective, forest type, tree size, accessibility of the tree, forestry law, technical, financial and human capacities (Henry et al., 2013). Hence, the quality of the biomass estimates differs among allometric equations and depends on the method used in the development of the allometry. Values for aboveground tree biomass are usually influenced by species, stand age, site quality, climate, and stocking density of stands (Zianis and Mencuccini 2002). Allometric equations are site and size specific and would be viewed with caution when implemented outside the specific location for which they were developed. To avoid this difficulty, generalised allometric equations for several North-American species have been developed of which some good estimates were derived (e.g. Pastor *et al.*, 1984). This view was reinforced by Ketterings *et al.* (2001) who upheld that biomass could be estimated non-destructively and suggested that parameters of allometric equations in biomass studies should be contingent on the average wood density, and on the exponent of the tree height-diameter relationship (Ketterings *et al.*, 2001). Brown (2001) and Brown and Iverson (1992) used mainly DBH and HT in their research while some researchers used WD (e.g. Chave *et al.*, 2005; Basuki *et al.*, 2009). Again, some authors recommended the use of CA (e.g. Kuyah *et al.*, 2012; Henry *et al.*, 2010). Furthermore, inclusion of tree height, specific wood density, and/or crown area variables have been reported to improve the accuracy of DBH based biomass equations (Chave *et al.*, 2005; Ketterings *et al.*, 2001; Kuyah *et al.*, 2014). Remotely determined crown area measurements for certain plant

types have been used to develop estimates of aboveground biomass using allometric equations (Drake *et al.*, 2003). Other researchers used parabolic equations of higher order to estimate AGB (e.g. Chave 2005; Basuki *et al.*, 2009).

4.2.4. Biomass equations for open-grown trees

There is reason to believe that the tree allometry of traditional forests does not accurately represent open-grown trees. An important characteristic associated with open environments is low tree density that reduces potential competition for light and other resources with surrounding trees. In general, trees in traditional forests experience a change in growth and allocation with reduced competition. After thinning, there is a tendency for trees in traditional forests to increase cambial activity and radial growth toward the base of a tree rather than the crown, producing a more tapered trunk (McHale *et al.*, 2009).

Though the scientific literature is replete with allometric equations developed for temperate, tropical, sub-tropical and semi-arid tree species and forests (e.g. Zianis and Mencuccini 2002; Jenkins *et al.*, 2004; Navar 2010), there is scarcity of literature presenting biomass equations for trees grown in open conditions. Current methods used to model carbon sequestration and uptake in agroforestry landscapes consist of applying allometric equations derived for natural forest-grown trees to agroforestry trees that grow in the open. These equations are most often derived from forest-grown trees that are different in their growth form from those open-grown trees in agroforestry configurations. The use of those broad scale allometric models on open-grown trees can be challenging as they generally lack accuracy because they are either too location-specific or much generalised (Nair *et al.*, 2009). This can introduce errors in

estimating not only biomass production potential, but carbon sequestration as well (Nair *et al.*, 2009). Tree canopy architecture differs between open-grown and restricted canopy conditions. When grown in the open, a tree's canopy can reach its full size and not be restricted; however, in a forested situation, tree canopies compete for limited growing space and, therefore, may not reach maximum expansion potential. However, other factors, such as soil, light, moisture, and crown loss due to storms or pruning, can also be limiting factors for crown width (Martin *et al.*, 2013). With the greater light exposure and less competition for water and nutrients in these settings, trees tend to allocate a larger portion of growth into canopy biomass than would generally occur in a forest understory (Zhou *et al.*, 2007).

Open-grown trees have larger crowns (Cole and Jensen 1982) and sharper trunk tapers (Sharma and Parton 2009) than more closed-canopy counterparts. The larger crown of open-grown trees is empirically expected to contribute more branch biomass. This tree architectural characteristic causes redistribution of biomass among tree parts and changes in tree specific gravity (Enquist and Niklas 2001). Zhou *et al.* (2011) demonstrated that “trunk-specific gravity” values for open-grown trees were greater than those published for forest-grown counterparts within the same geographic region (Zhou *et al.*, 2011). These findings infer that using a forest-derived equation could potentially either underestimate (due to greater specific gravity) or overestimate (due to sharper trunk taper) trunk biomass.

This underscores the need for the development of species specific allometric equations for different agroforestry practices to overcome this serious weakness in agroforestry research (Zhou *et al.*, 2007 and 2011). The development of allometric equations that are applicable to varied forms of open-grown trees is important not only for proper accounting of carbon stock

in agroforestry landscapes but also for policy makers as tool for formulating the best suitable environmental policy decisions. Trees growing in plantations respond to crowding by growing taller while allocating relatively less to stem diameter increments. Open-grown trees, on the other hand, allocate relatively more biomass to leaves and branches than do trees crowded in dense plantations (Holbrook and Putz 1989). Nowak (1994) observed that allometric equations developed for trees grown in natural forests are used at present to estimate carbon sequestration in open urban forests, and that there are limited research studies of estimation methods in urban settings. Furthermore, he discovered that allometric equations for forest-grown trees overestimated urban tree biomass, and he multiplied estimates by a factor of 0.8 to adjust for the over estimation. (Nowak, 1994).

4.3. MATERIALS AND METHODS

4.3.1. Study area description

The experiment was conducted in the year 2012 - 2014 on three red alder blocks at the United Kingdom's Silvopastoral National Network Experiment (SNNE) located in the Bangor University's Henfaes research farm (53°14'N 4°01'W), Abergwyngregyn, Gwynedd, Wales. A detailed description of the study area including weather data, vegetation and earlier treatments to pasture establishment for the period of the study has been previously reported in chapter 1.

4.3.2. Data collection

4.3.3. Selection and measurement of 'good' and 'poor' forms of trees

The three blocks of red alder 200 stem ha⁻¹ (400 stems ha⁻¹ at the start of the experiment in 1992) at Henfaes research farm were inventoried during summer (full leaf) season in 2012. In the winter of 2012, ten good form and ten poor form red alder trees were randomly selected during the thinning of this treatment to 100 stem ha⁻¹. The criteria for the selection of the thinned trees were:

- Good form trees generally free of defects, showing good health and expected to reach their full age and size potential, and chosen arbitrarily on the basis of their location.
- Poor form trees with moderate to high risk of failure: trees in obvious decline, or with significant health and/or irremediable structural defects, including advanced decay and crack, root problems, weak branch union, canker, poor architecture, stunted growth, and dead wood.
- Out-of-line trees irrespective of form.

The 20 individual alder trees were measured before harvest for diameter at breast height: DBH, total height: HT, and crown area: CA. Prior to harvesting, GPS was used to geo-reference the locations of the selected trees. The DBH was directly measured overbark in cm using a diameter tape at 1.3 m above ground level. A clinometer was used to measure HT of each tree in metres. The crown area was estimated in square metres by taking the average of the maximum and minimum crown diameters. Assuming an elliptic shape, the following formula (Dietz and Kuyah, 2011) was applied:

$$CA = [(d_1 \times d_2)/4] \times \pi$$

Where

CA = crown area (m²), d₁ = larger crown diameter (m), d₂ = smaller crown diameter (m), and $\pi = 3.142$



Figure 4-1: Good form (left) and poor form (right) red alder trees in 2012 at Henfaes SNNE

4.3.4. Destructive measurement of fresh biomass of trees

Each sampled tree was cut off as close to ground level as practicable with a chain saw. Felled trees were measured for total tree height (from the stump to the top of the crown) using a 50-m measuring tape while length of tree bole was measured from the stump to the first main branch.

For determination of whole tree weight and volume, the trees were segmented into trunk, branches and twigs, and the fresh weight of each component determined on-site by weighing on a balance (300 kg) to the nearest 0.1 kg sections. The trunk and branch sections were then sawn into logs of varying sizes (Figure 4.2) which could be lifted by hand and weighed using tractor based weighing scales (Figure 4.2). The twigs were chipped (Figure 4.2) straight into a 1 m³ bag and then weighed using the tractor based weighing scales. The tractor based weighing scales were accurate to 0.1 kg (Dietz and Kuyah, 2011).



Figure 4-2: Trunk and branch sections of red alder sawn into logs of varying sizes (a), weighing of logs using tractor based weighing scales (b), and chipping of twigs (c).

4.3.5. Measurements for timber/log size classification

The weighable log sections of the trunks and the branches were then measured to determine the volume of timber in the trunk and branch sections of each tree. The measured log segments (stem and branches) were sorted and classified into six diameter size classes of: 5 – 10 cm; 10.1 – 20 cm; 20.1 – 30 cm; 30.1 – 40 cm; 40.1 – 50 cm; > 50 cm. The volume of each segment was calculated with the use of Smalian's formula (Avery and Burkhardt, 2002) as follows:

$$V = \frac{A_1 + A_2}{2} \times L$$

Where:

V = Smalian's volume (m³), A₁ = cross section of large end (m²), A₂ = cross section of small end (m²), L = log length (m).

All volume measurements were made over-bark.

A paired-sample *t*-test was conducted to test the null hypothesis that the distribution of the timber/logs by size class did not differ significantly between the good form and the poor form red alder trees.

4.3.6. Comparison of branching process

Prior to cutting the trunks and the branches into weighable logs of varying sizes, the bifurcation frequencies of each alder tree were examined. This was accomplished by numbering and measuring and grouping the branches by order. Following Strahler's (1957) centripetal ordering systems, the ultimate branch was labelled as first-order (e.g. Borchert and Slade, 1981). The second-order was the segment next to where the two first-order branches meet. Where two branches of unequal order meet, the resulting branch maintained the higher order.

To calculate the bifurcation ratio, the number of first and second order branches were counted. The ratio of the number of branches of one order to the number of branches of the next order constitutes the bifurcation ratio (BR):

$$BR = N_n / (N_{n+1})^{-1} \quad (1)$$

Where N is the total number of branches of an order and n is the order number.

The bifurcation ratio for the entire tree was estimated by the formula (Steingraeber et al., 1979):

$$BR = (N - N_{\max}) / (N - N_1)^{-1} \quad (2)$$

Where N is the total number of branches of all orders,

N_{\max} is the number of branches of the highest order,

N_1 is the number of branches of the first order.

The BR of the two forms of alder were compared using the independent sample t-test to test the null hypothesis that there was no significant difference between the means with α set at 0.05.

4.3.7. Sampling for oven dry mass and wood density analysis

Representative subsamples of twig and wood were randomly collected for fresh weight determination. Discs of about 2 – 3 cm thick were cut from the lower, middle and upper ends of the trunks, and from different parts of the branches. The discs were taken from different branches to be representative of the architecture of a standard branch. About 500 g of twig subsample was collected from each tree in tared sample bags (Kuyah et al, 2012). The sub-samples from the segmented trees were collected in duplicate and stored in sealed polythene bags, and then taken immediately to the Henfaes research laboratory to determine their moisture content,

wood density and biomass. In the laboratory, the subsamples were weighed in situ using a 0.1 g precision scale as the trees were being processed to reduce the loss of moisture (Figure 4.3).



Figure 4-3: Weighing of disc and twig subsamples in the laboratory using a 0.1 g precision scale.

The volume of a disc was calculated as the cross-sectional area of the disc times the thickness (measured at four points, 90° to each other). Dry weights were obtained by drying the subsamples at a temperature of 105 °C for 48 hours and re-weighed daily until constant weight was achieved for each sample (Ketterings *et al.*, 2001). Dry weight (biomass) of the subsamples was determined after the drying process, and the values obtained for each subsample were used to convert fresh weight into dry weight for each tree. Separate biomass estimates were

computed for the various tree components. The total above ground biomass per tree was derived by summing the biomass of the trunk, branches and twigs. The value obtained was then multiplied by the number of trees per block and extrapolated to a hectare. The mean annual increment in biomass was computed by multiplying the mean aboveground biomass per tree (expressed in Mg) by the tree density of 200 trees ha⁻¹ and dividing the product by 20 years, the age of the trees at the time of study.

4.3.8. Carbon and carbon dioxide stocks estimation

Carbon (C) and carbon dioxide (CO₂) stocks of the sampled trees and their component parts were estimated. The amount of carbon sequestered by the trees were determined by multiplying the total tree biomass by an international standard conversion coefficient of 0.5. This conversion factor indicates that 50% of the total tree biomass consists of elemental C (Dixon *et al.*, 1994; FAO 2004). The amount of carbon dioxide that would be emitted to the atmosphere if the 20 sampled trees were cut down and burned completely was calculated by taking the estimated weight of Carbon in the red alder tree samples and multiplying by 3.6663 as follows:

$$\text{Weight of CO}_2 \text{ sequestered in tree (in kg)} = \text{Weight of carbon in tree} \times 3.6663$$

where 3.6663 is the universal conversion factor of C content values to CO₂ values. The factor 3.6663 is derived from a calculation measuring the atomic weights of Carbon, Oxygen and Carbon Dioxide and then calculating the ratio of the atomic weight of CO₂ (44) to the atomic weight of C (12) (Walker *et al.*, 2011; Ali *et al.*, 2014).

Average amount of carbon content and carbon dioxide sequestered per year over the life of the tree were calculated by multiplying the mean aboveground carbon/carbon dioxide (expressed in Mg C per tree) by the tree density of 200 trees ha⁻¹ and dividing the product by 20 years, the age of the trees at the time of study. The descriptive statistics of the computations are presented in Appendix 4.1.

4.3.9. Development of above-ground biomass prediction models

Preliminary evaluations using transformed and untransformed data were performed for the development of models for the two different forms of red alder and their components. Scatter diagrams were plotted using raw data to assess the relationships between the dependent variable and the independent variables.

The dependent variable, aboveground biomass (AGB), was regressed against the independent variables, diameter at breast height (DBH), tree height (HT), crown area (CA), bifurcation ratio (BR) and wood density (WD) using the ordinary least squares regression analysis. The influence of heteroscedasticity was eliminated by logarithmically transforming the power function into the linear form $\ln(Y) = a + b \times \ln(X) + e$, where Y is the dependent variable, X is the independent variable, a is the intercept, b is the scaling allometric exponent, and e is the error term. This log-transformation tends to introduce systematic bias which needs to be corrected with a bias correction factor (CF) before back-transformation (Baskerville 1972; Parresol 1999; Chave *et al.*, 2005). Correction factor was computed using the formula (Sprugel 1983): $CF = \exp(SEE^2/2)$ where SEE is the standard error of estimate from the regression model.

The accuracy of the estimates was determined by evaluating the difference between the predicted and actual value expressed as a percentage of relative error as outlined by Chave *et al.* (2005): $\% \text{ error} = (\text{AGB}_{\text{Predicted}} - \text{AGB}_{\text{Actual}}) / (\text{AGB}_{\text{Actual}}) \times 100$, where % error is the percent relative error of the estimate, $\text{AGB}_{\text{Predicted}}$ is the predicted aboveground biomass, and $\text{AGB}_{\text{Actual}}$ is the actual aboveground biomass.

To measure the extent of information lost in specific models, the Akaike Information Criterion (AIC) was computed using the residual sum of squares model: $\text{AIC} = n \ln (RSS/n) + 2k$, where n is the number of samples, k is the number of independent variables in the model plus 1, and RSS is the residual sums of the squares of the regression model. Since bias adjustment is required for small sample size ($n/k < 40$) such as found in this study, a recommended modified version of AIC (AIC_C) was used: $\text{AIC}_C = n \ln (RSS/n) + 2k (n/n-k-1)$.

The Stepwise multiple regressions method was used where one independent variable is included or excluded at each step, based (by default) on the probability of F (p-value). The focus of interest in the use of the stepwise method was to determine the best combination of the independent variables (DBH, HT, CA, BR, WD) that would be required to predict aboveground biomass (AGB) for the two forms of red alder. Models for the tree components (trunk, branch and twig) were developed by independently regressing the biomass (dry weights) of the components against the DBH.

4.3.10. Model selection

The goodness of fit of all the generated regression models was determined by examining the coefficient of determination (R^2) and adjusted R^2 ($adj. R^2$) in the case of multiple independent variables, the standard error of estimate (SEE), the bias correction factor (CF), the percent relative error (% error) and the corrected Akaike Information Criterion (AIC_C). The best-fit models were considered as having the highest adjusted R^2 and the lowest SEE , CF , % error and AIC_C . (Chave *et al.*, 2001; Burnham & Anderson 2002; Litton and Kauffman, 2008; Basuki *et al.*, 2009)

The normality and collinearity of the linear regression models were verified by examining the residual statistics of the regression models and interpreting them in plots and diagrams. The standardised coefficient Beta was used to measure the strength of the effect of each predictor variable while the variance inflation factor (VIF) was used to test the severity of multicollinearity associated with the log-transformed regression model. Larger Beta indicates stronger effect of the predictor variable while smaller VIF ($VIF < 10$) establishes the validity, applicability and comparability of the predictive model.

4.4. RESULTS

4.4.1. Descriptive statistics of tree variables

Variables, diameter at breast height (DBH), tree height (HT), basal area (BA), Volume (V), crown area (CA), wood density (WD) and bifurcation ratio (BR), were measured for each of the good form and poor form red alder trees, respectively. The general properties of the measured red alder trees are summarised in Table 4.1.

Table 4.1: Summary of descriptive statistics for good form and poor form alder tree variables (n=20)

Variables	Form	No.	Minimum	Maximum	Mean	Stdev	CV (%)
DBH (cm)	Good	10	21.00	38.00	29.66	5.25	17.70
	Poor	10	13.00	29.00	23.00	4.24	18.44
HT (m)	Good	10	11.00	13.50	12.20	0.78	6.39
	Poor	10	9.00	13.00	11.13	1.26	11.32
BA (m ²)	Good	10	0.03	0.11	0.07	0.02	28.57
	Poor	10	0.01	0.07	0.04	0.01	25.00
V (m ³)	Good	10	0.13	0.47	0.29	0.11	37.93
	Poor	10	0.03	0.27	0.16	0.06	37.50
CA (m ²)	Good	10	21.21	62.84	53.45	12.23	22.88
	Poor	10	12.19	40.14	31.97	8.56	26.77
WD (kg/m ³)	Good	10	0.26	0.75	0.44	0.18	40.90
	Poor	10	0.15	0.62	0.39	0.15	38.46
BR	Good	10	2.00	5.00	3.22	0.94	29.19
	Poor	10	1.82	3.20	2.35	0.40	17.02

It can be observed that stem diameter at breast height measurements ranged from 21 cm to 38 cm for the good form trees and from 13 cm to 29 cm for the poor form trees with coefficient of variation of 17.70% and 18.44%, respectively, whereas height varied from 11 m to 13.50 m for good form trees and from 9 m to 13 m for poor form trees with coefficient of variation (CV)

of 6.39% and 11.32%, respectively. Tree basal area in the red alder blocks varied from 0.01 m² in poor form trees to 0.11 m² in good form trees, while tree volume ranged from 0.03 m³ in poor form trees to 0.47 m³ in good form trees. The mean value of the measured variables is greater in the good form trees than in the poor form trees. Again, among all the variables measured, tree height had the smallest CV (6.39% for good form trees and 11.32% for poor form trees) compared to other variables (Table 4.1).

4.4.2. Timber/log size classification

The amount of timber/log of different size classes that can be found in the two forms of red alder trees and their relationships with total volume per cubic metre (m³) is presented in Figure 4.4. About 33.22 % by volume per m³ of the good form trees and 37.69 % by volume per m³ of the poor form trees fall within the 20 – 30 cm size class, respectively. Again, about 74 % by volume per m³ of the good form trees and about 77 % by volume per m³ of the poor form trees are of merchantable size. The poor form trees showed greater percent volume per m³ than the good form trees only in the 10 – 30 cm size classes (Figure 4.4).

The statistical ‘paired-sample *t*-test’ agreed with the null hypothesis that the distribution of the timber/logs by size class did not differ significantly between the good form ($\bar{x} = 14.28$, $\sigma = 116.76$) and the poor form trees ($\bar{x} = 14.28$, $\sigma = 253.08$), $t(6) = 2.44$, $p > 0.05$. This suggests that the assumption of the null hypothesis is valid.

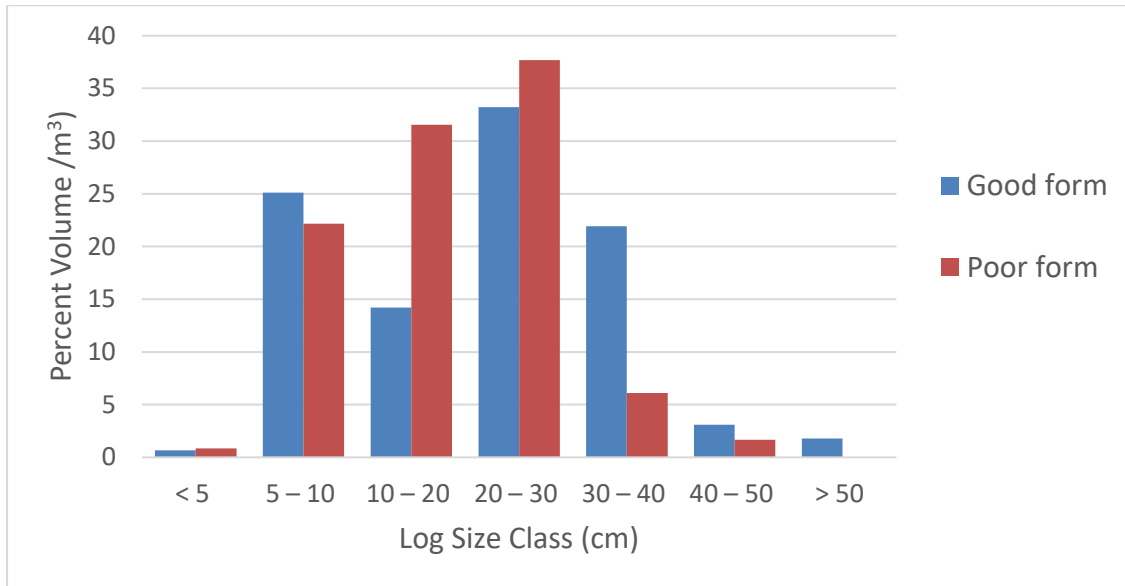


Figure 4-4: Relationships between log size class and total volume according to form.

4.4.3. Comparison of branching processes

The descriptive statistics for the branching process of the two forms of alder is shown in Table 4.2. The result of the comparison of the bifurcation ratio (BR) of the two forms using the independent-sample *t*-test for the comparison of the means show that bifurcation ratio is significantly higher for good form alder ($\bar{x} = 3.22$, $\sigma = 0.94$) than for poor form alder ($\bar{x} = 2.35$, $\sigma = 0.40$), $t(18) = 2.704$, $p = 0.015$. This disagrees with the null hypothesis of no difference and implies that the branching process of the two forms of red alder actually differs.

Table 4.2: Bifurcation ratios (BR) of good and poor forms of alder in a Silvopastoral system.

Form	No. of trees	BR		
		\bar{x} (σ)	Minimum	Maximum
Good form	10	3.22 ^a (0.94)	2.00	5.00
Poor form	10	2.35 ^b (0.40)	1.82	3.20
Overall	20	2.79 (0.83)	1.82	5.00

^{ab}Mean BR values are significantly different with $P < 0.05$.

4.4.4. Aboveground biomass, carbon and carbon dioxide content

The results of the estimated aboveground biomass (AGB), carbon stock (C) and carbon dioxide (CO₂) mitigation potential of the good and poor forms of red alder are presented in Table 4.3 to Table 4.8 while the percentage distribution of the biomass of the tree components are presented in Figure 4.5. The estimated mean aboveground biomass per tree for the good form trees was 245.43 kg/tree (49.09 Mg ha⁻¹) with a standard deviation of 51.25 kg/tree and mean annual biomass increment of 2.46 Mg ha⁻¹ yr⁻¹, whereas the mean aboveground biomass per tree for the poor form trees was 129.52 kg/tree (25.91 Mg ha⁻¹) with a standard deviation of 42.19 kg/tree and a mean annual biomass increment of 1.30 Mg ha⁻¹ yr⁻¹ (Tables 4.3 and Table 4.4).

The biomass of tree component parts of both forms showed similarity in their percentage compositions. For the good form trees, about 69.97% of the aboveground biomass is contained in the stem, 23.02% in branches, and 7.00% in twigs (Figure 4.5). For the poor form trees, estimated contributions of stem, branches and twigs to aboveground biomass were 69.49%,

23.96%, and 6.56%, respectively. Specifically, the respective mean stem, branch and twig biomass per tree were 171.73 kg/tree, 56.50 kg/tree and 17.19 kg/tree for good form trees, and 90.00 kg/tree, 31.02 kg/tree and 8.49 kg/tree for poor form trees.

Table 4.3: Descriptive statistics of estimates of aboveground biomass according to the components of the two forms of red alder trees at the Henfaes SNNE (n = 20)

Component Variables	Form	Min	Max	Sum	Mean	Stdev	CV (%)
Stem biomass (kg)	Good	102.81	225.52	1717.30	171.73 ^a	35.66	20.77
	Poor	24.58	133.00	900.02	90.00 ^b	20.36	22.62
Branch biomass (kg)	Good	33.70	75.00	565.07	56.50 ^a	11.89	21.04
	Poor	8.00	45.89	310.27	31.02 ^b	10.25	33.04
Twig biomass (kg)	Good	10.00	23.00	171.90	17.19 ^a	3.70	21.52
	Poor	3.00	12.50	84.94	8.49 ^b	2.59	30.51
Aboveground biomass (kg)	Good	146.51	323.52	2454.26	245.43 ^a	51.25	20.88
	Poor	35.58	191.39	1295.22	129.52 ^b	42.19	32.57

^{ab} Means within a column followed by different letters vary significantly (P < 0.05)

Table 4.4: Mean biomass (Mg ha⁻¹) of the aboveground components of the two forms of red alder trees at the Henfaes SNNE

Form	No of Tree/ha	Stems (Mg ha ⁻¹)	Branch (Mg ha ⁻¹)	Twig (Mg ha ⁻¹)	AGB (Mg ha ⁻¹)	Annual increase in AGB (Mg ha ⁻¹ yr ⁻¹)
Good	200	34.35	11.30	3.44	49.09	2.46
Poor	200	18.00	6.21	1.70	25.91	1.30

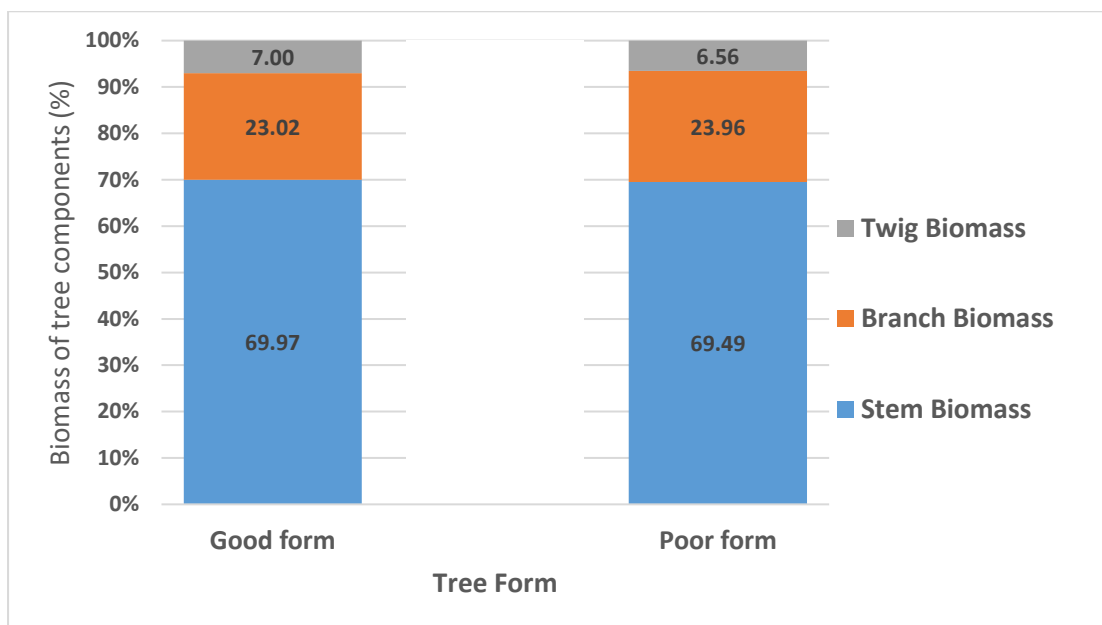


Figure 4-5: Percentage distribution of biomass of stem, branch, and twig.

Similarly, the annual aboveground carbon stock and sequestered carbon dioxide in the system was estimated by multiplying the mean carbon/carbon dioxide (expressed in Mg C per tree) by the tree stocking density of 200 trees ha⁻¹ and dividing the product by 20 years, the age of the trees at the time of study. Using the default coefficient of 0.50 for the conversion of biomass to carbon (FAO 2004), the estimated mean aboveground carbon stocks per tree were 122.73 kg/tree (24.55 Mg C ha⁻¹) with a standard deviation of 25.63 kg/tree and a mean annual carbon increment of 1.23 Mg C ha⁻¹ yr⁻¹ for good form trees, and 64.76 kg/tree (12.95 Mg C ha⁻¹) with a standard deviation of 21.10 kg/tree and a mean annual carbon increment of 0.65 Mg C ha⁻¹ yr⁻¹ for poor form trees, (Table 4.5 and Table 4.6). Mean stem, branch and twig carbon stock per tree were 85.86 kg/tree, 28.26 kg/tree and 8.59 kg/tree, respectively, for good form trees, and 45 kg/tree, 15.51 kg/tree and 4.24 kg/tree, respectively, for poor form trees.

Table 4.5: Descriptive statistics of estimates of carbon stock according to the components of the two forms of red alder trees at the Henfaes SNNE (n = 20)

Component Variables	Form	Min	Max	Sum	Mean	Stdev	CV (%)
Stem carbon stock (kg C)	Good	51.41	112.76	858.64	85.86 ^a	17.83	20.77
	Poor	12.29	66.50	450.01	45.00 ^b	14.68	32.62
Branch carbon stock (kg C)	Good	16.84	37.50	282.54	28.26 ^a	5.94	21.02
	Poor	4.00	22.95	155.13	15.51 ^b	5.12	33.01
Twig carbon stock (kg C)	Good	5.00	11.50	85.95	8.59 ^a	1.85	21.54
	Poor	1.50	6.25	42.47	4.24 ^b	1.29	30.42
Aboveground carbon stock (kg C)	Good	73.25	161.76	1227.13	122.71 ^a	25.63	20.89
	Poor	17.79	95.70	647.61	64.76 ^b	21.10	32.58

^{ab} Means within a column followed by different letters vary significantly ($P < 0.05$)

Table 4.6: Mean carbon stock (Mg C ha⁻¹) of the aboveground components of the two forms of red alder trees at the Henfaes SNNE.

Form	No of Tree/ha	Stems (Mg C ha ⁻¹)	Branch (Mg C ha ⁻¹)	Twig (Mg C ha ⁻¹)	AGB (Mg C ha ⁻¹)	Annual increase in AGB (Mg C ha ⁻¹ yr ⁻¹)
Good	200	17.17 ^a	5.65 ^a	1.72 ^a	24.55 ^a	1.23
Poor	200	9.00 ^b	3.10 ^b	0.85 ^b	12.95 ^b	0.65

^{ab} Means within a column followed by different letters vary significantly ($P < 0.05$)

Using the universal conversion factor of 3.6663 for the conversion of carbon stock values to carbon dioxide values, the estimated mean aboveground carbon dioxide sequestration potential was 449.90 kg/tree (89.98 Mg C ha⁻¹) with a standard deviation of 93.95 kg/tree and a mean annual carbon dioxide increment of 4.50 Mg C ha⁻¹ yr⁻¹ for good form trees, and 237.43 kg/tree (47.50 Mg C ha⁻¹) with a standard deviation of 77.35 kg/tree and a mean annual carbon dioxide

increment of 2.38 Mg C ha⁻¹ yr⁻¹ for poor form trees (Table 4.7 and Table 4.8). Correspondingly, mean stem, branch and twig sequestered CO₂ per tree were 314.81 kg/tree, 103.59 kg/tree and 31.51 kg/tree, respectively, for good form trees, and 164.99 kg/tree, 56.88 kg and 15.57 kg/tree, respectively, for poor form trees.

Table 4.7: Descriptive statistics of estimates of carbon dioxide (CO₂) sequestration according to the components of the two forms of red alder trees at the Henfaes SNNE (n = 20)

Component Variables	Form	Min	Max	Sum	Mean	Stdev	CV (%)
Stem CO ₂ (kg CO ₂)	Good	188.47	413.41	3148.06	314.81 ^a	65.38	20.77
	Poor	45.06	243.81	1649.87	164.99 ^b	53.82	32.62
Branch CO ₂ (kg CO ₂)	Good	61.77	137.49	1035.86	103.59 ^a	21.79	21.03
	Poor	14.67	84.12	568.78	56.88 ^b	18.80	33.05
Twig CO ₂ (kg CO ₂)	Good	18.33	42.16	315.11	31.51 ^a	6.79	21.55
	Poor	5.50	22.91	155.70	15.57 ^b	4.74	30.44
AGB CO ₂ (kg CO ₂)	Good	268.57	593.06	4,499.03	449.90 ^a	93.95	20.88
	Poor	65.22	350.85	2,374.35	237.43 ^b	77.35	32.58

^{ab} Means within a column followed by different letters vary significantly (P < 0.05)

Table 4.8: Mean carbon dioxide sequestration (Mg CO₂ ha⁻¹) of the aboveground components of the two forms of red alder trees at the Henfaes SNNE.

Form	No of Tree/ha	Stems (Mg CO ₂ ha ⁻¹)	Branch (Mg CO ₂ ha ⁻¹)	Twig (Mg CO ₂ ha ⁻¹)	AGB (Mg CO ₂ ha ⁻¹)	Annual increase in AGB (Mg CO ₂ ha ⁻¹ yr ⁻¹)
Good	200	62.96 ^a	20.72 ^a	6.30 ^a	89.98 ^a	4.50
Poor	200	33.00 ^b	11.38 ^b	3.12 ^b	47.50 ^b	2.38

^{ab} Means within a column followed by different letters vary significantly (P < 0.05)

Generally, biomass, carbon stock and sequestered carbon dioxide differ between the two forms of red alder trees as well as between their components. Again, standard deviations were comparatively largest in stems, and least in twigs. Result of the independent-sample t -test for the comparison of the biomass (and C or CO₂) of the two forms of red alder tree show that mean biomass (and C or CO₂) differs between good form trees ($\bar{x} = 245.43$, $\sigma = 51.23$, $n = 10$) and poor form trees ($\bar{x} = 129.52$, $\sigma = 42.19$, $n = 10$), $t(18) = 5.52$, $p < 0.05$ (Tables 4.5 to 4.8). The above results suggest that the silvopastoral agroforestry ecosystem in the United Kingdom represents a significant carbon sink.

4.4.5. Development of allometric model

Untransformed and log-transformed data were subjected to simple linear regression in order to estimate the parameters. Preliminary evaluation of model development justified the use of log-transformed data as transformed regression equation showed better goodness of fit for the models. Allometric models were developed for stem, branch and twig biomass using only DBH as the independent variable in a simple linear regression while a stepwise multiple linear regression analysis was performed to determine which of the variables, DBH, HT, CA, WD and BR, were most significant (0.05 probability of F-to-enter, and 0.1 probability of F-to-remove) to predict aboveground biomass of the good and poor forms of open-grown red alder.

4.4.6. Correlation of modelling parameters

A Pearson product-moment correlation results for the pooled good form and poor form tree data are summarized in Table 4.9. It can be seen that all the variables except wood density are

positively and significantly correlated ($p < 0.01$) with AGB, indicating their potential reliability in the estimation of tree biomass, while the correlation of wood density with AGB is low, negative ($r = -0.305$) and not significant ($p > 0.05$), indicating that lower wood density reduces the biomass. AGB showed highly significant correlation ($p < 0.01$) with diameter at breast height ($r = 0.948$) followed by crown area ($r = 0.904$) and height ($r = 0.789$), indicating that these variables are strong indicators of biomass. The result also showed that diameter at breast height correlated significantly ($p < 0.01$) with height ($r = 0.775$) and crown area ($r = 0.841$) while bifurcation ratio correlated moderately with AGB ($r = 0.504$) and diameter at breast height ($r = 0.516$) and height ($r = 0.446$) and crown area ($r = 0.512$)

Table 4.9: Pearson's correlation coefficients between diameter at breast height (DBH), height (HT), crown area (CA), wood density (WD), bifurcation ratio (BR) and total aboveground biomass (AGB)

	DBH (cm)	HT (m)	CA (m ²)	WD (g/m ³)	BR	AGB (kg)
DBH (cm)	1					
HT (m)	0.775**	1				
CA (m ²)	0.841**	0.668**	1			
WD (g/cm ³)	-0.256	-0.082	-0.340	1		
BR	0.516**	0.446*	0.512*	-0.214	1	
AGB (kg)	0.948**	0.789**	0.904**	-0.305	0.504	1

* Correlation is significant at the 0.05 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

4.4.7. Tree component model

Allometric models were developed for stem, branch and twig biomass using only DBH as the independent/predictor variable in a simple linear regression. Developed models along with

their coefficients and estimates of error for the components of good and poor forms of red alder trees are presented in Tables 4.10. Results of the use of DBH only as the predictor variable show that, generally, for both good form and poor form trees, more than 93% of the observed variation in component and aboveground biomass was explained by DBH alone, indicating a very strong positive correlation between diameter and biomass of aboveground tree components (Table 4.10). Scatter plots of biomass components and aboveground biomass plotted against their significant variables are summarised in Figure 4.6. Specifically, for good form trees (models 1, 3, 5, and 7), allometric equations showed that using only diameter in the basic functional model predicted the stem, branch, twig, and aboveground biomass with R^2 values of 0.939, 0.939, 0.931, and 0.939, respectively, % *error* values of 0.77%, 1.68%, 2.49%, and 0.63%, AIC_c of -38.60, -37.87, -40.75, and -4.31, respectively. Similarly, for poor form trees (models 2, 4, 6, and 8), allometric equations showed that using only diameter in the basic functional model predicted the stem, branch, twig, and aboveground biomass with R^2 values of 0.939, 0.939, 0.931, and 0.939, respectively, % *error* values of 0.77%, 1.68%, 2.49%, and 0.63%, AIC_c of -38.60, -37.87, -40.75, and -4.31, respectively.

Generally, good form trees had higher R^2 and F -value, and lower SEE , CF , % *error*, and AIC_c compared to poor form trees (Table 4.10). Again, the scatterplot of standardised residual values showed that the data met the assumptions of homogeneity of variance and linearity. The points in the residual plots showed randomly dispersed pattern in all the biomass equations, indicating a good fit for their applicability (Figure 4.7).

Table 4.10: Description of diameter-based allometric models for the estimation of component parts of red alder.

Allometric model	Model No.	Form	coefficient		R^2	SEE	CF	F-value	P-value	% error	AIC _c
			a	b							
Stem biomass Ln (Stem) = a + b * Ln (DBH)	1	Good	1.048	1.208	0.943	0.057	1.006	131.936	0.000	0.11	-53.81
	2	Poor	-2.096	2.092	0.939	0.122	1.008	124.108	0.000	0.77	-38.60
Branch biomass Ln (Branch) = a + b * Ln (DBH)	3	Good	-0.109	1.221	0.944	0.057	1.001	134.906	0.000	0.20	-53.81
	4	Poor	-3.405	2.169	0.939	0.127	1.008	124.190	0.000	1.68	-37.87
Twig biomass Ln (Twig) = a + b * Ln (DBH)	5	Good	-1.418	1.256	0.938	0.062	1.001	121.166	0.000	0.54	-52.05
	6	Poor	-3.366	1.748	0.931	0.109	1.005	107.848	0.000	2.49	-40.75
Total biomass Ln (AGB) = a + b * Ln (DBH)	7	Good	1.383	1.214	0.943	0.057	1.002	131.962	0.000	0.10	-19.15
	8	Poor	-1.708	2.084	0.939	0.121	1.007	124.032	0.000	0.63	-4.312

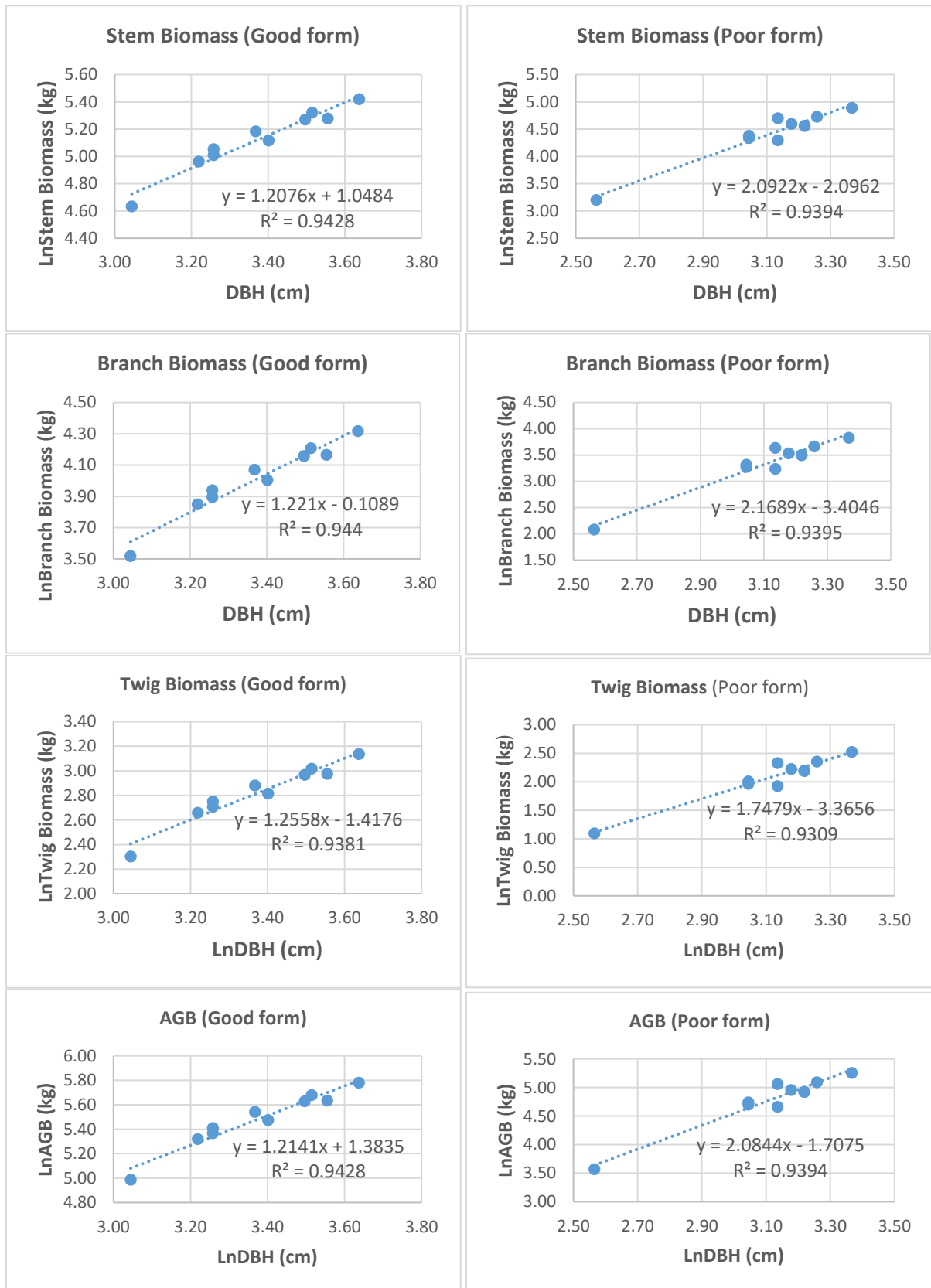


Figure 4-6: Regression between the natural logarithm of component biomass (kg) (stem, branches, and twigs) and the natural logarithm of diameter at breast height (cm) for the good form and poor form trees.

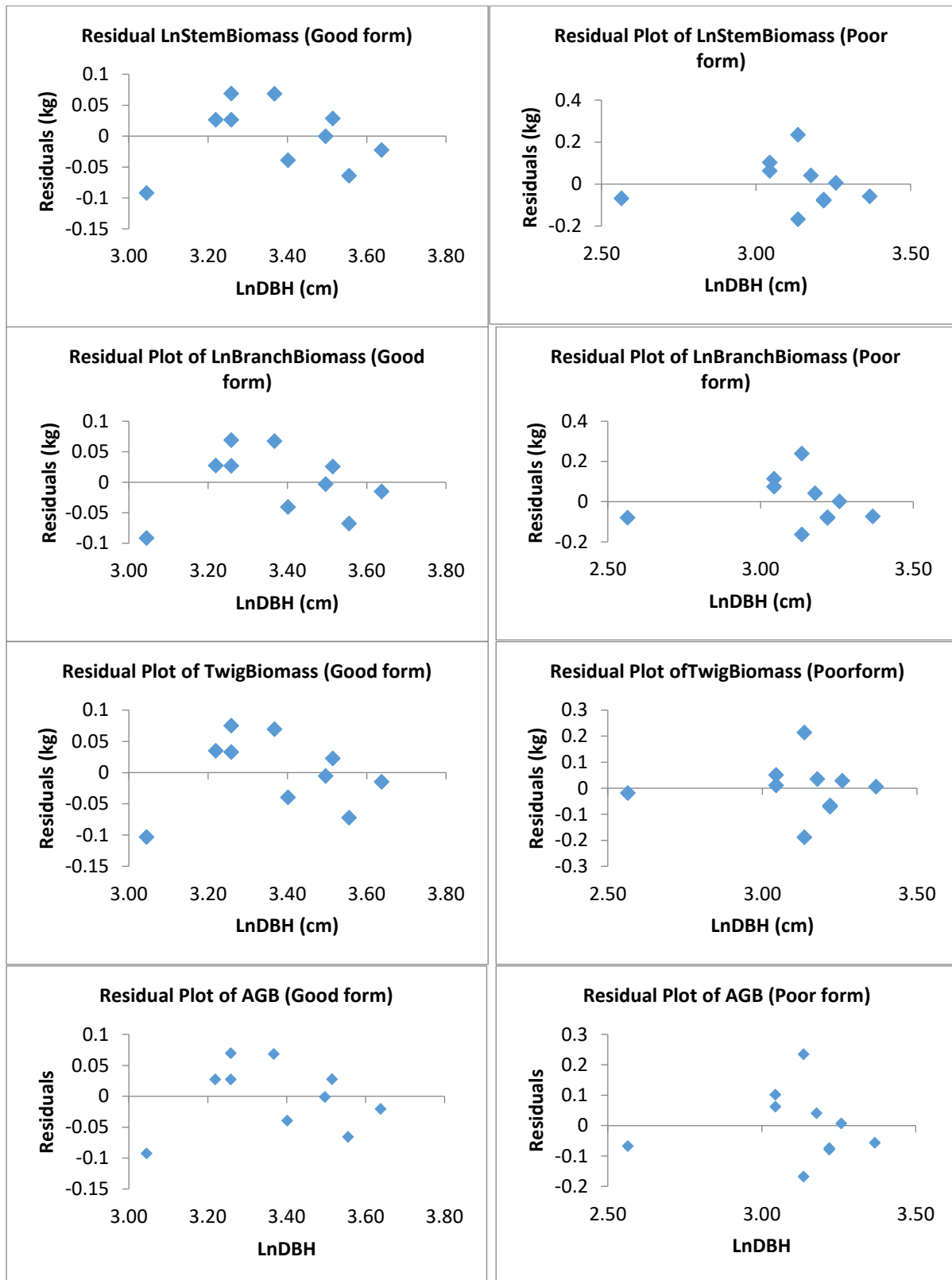


Figure 4-7: Residual scatter plot of regression between the natural logarithm of component biomass (kg) (stem, branches, and twigs) and the natural logarithm of diameter at breast height (cm) for the good form and poor form trees.

4.4.8. Aboveground biomass model

A stepwise multiple regression analysis was performed to determine which of the variables, DBH, HT, CA, WD and BR were necessary to predict aboveground biomass of the good and poor forms of open-grown red alder trees. The regression models along with their coefficients of determination and estimates of error for the two forms of alder are detailed in Table 4.11.

4.4.9. Good form trees

The SPSS stepwise regression generated two models for the good form variables (Table 4.8), and the better of the two models was chosen. DBH entered the regression equation (model 9) at step 1 as the only predictive variable and was found to significantly predict AGB with adjusted $R^2 = 0.936$, $F(1, 8) = 131.96$, $p < 0.001$ and could explain about 94% of the variance of AGB. The variables, HT, CA, WD, and BR, were excluded from the equation at step 1. Model 9 was therefore found to have a relative error (% error) values of 0.10%, a corrected Akaike Information Criterion (AIC_c) values of -19.15, and a correction factor (CF) of 1.002.

At step 2 of the analysis both DBH and CA were entered into the regression equation (model 10) and found to have a statistically significant relationship with AGB [$F(2, 7) = 149.63$, $p < .001$]. The adjusted R^2 was 0.971 indicating that approximately 97% of the variance of AGB could be accounted for by a combination of DBH and CA. The predictive power of DBH was much higher ($beta = .808$, $t = 10.627$, $p < .001$) than that of CA ($beta = .247$, $t = 3.241$, $p < .05$). On the other hand, HT, BR and WD did not enter into the equation at step 2 of the analysis (HT: $t = -0.452$, $p > .05$; WD: $t = 0.736$, $p > .05$; and BR: $t = -1.761$, $p > .05$). Model 10 showed a % error of 0.04, an AIC_c of -28.12, and a CF of 1.001. The variance inflation factor (VIF)

of 1.772 (< than 10) and *Tolerance* of 0.564 (> 0.1) for mean DBH and CA are indications of absence of collinearity.

Table 4.11 indicates that models 9 and 10 for good form trees were statistically similar. However, the second model (10) was selected for having a higher coefficient of determination, and a lower percent relative error and Akaike information criterion. Thus the optimal regression model for predicting good form aboveground biomass was:

$$\text{Ln(AGB)} = [1.385 + 1.011 * \text{Ln(DBH)} + 0.174 * \text{Ln(CA)}] * 1.001 \quad \text{Model (10)}$$

4.4.10. Poor form trees

The stepwise regression also generated two models for poor form trees (models 11 and 12) (Table 4.11). Model 11 displayed DBH as the only statistically significant predictor of AGB [adjusted $R^2 = 0.932$, $F(1, 8) = 124.062$, $p < .001$] and was reached in step 1, accounting for about 93% of the variance of AGB. The other variables, HT, CA, WD, and BR, were excluded from the model. DBH predicted AGB of poor form red alder trees with a % error of 0.63, an AIC_C of -4.31, and a CF of 1.007.

DBH and CA were included in the second model (model 12) at step 2 of the regression analysis as the only predictive variables while HT, WD, and BR variables were excluded. Statistically, DBH and CA predicted AGB significantly with adjusted R^2 of 0.965 ($F(2, 7) = 126.200$, $p < 0.001$) and accounted for about 97% of the variability in aboveground biomass. DBH had better predictive weight ($\beta = 0.695$, $t = 6.223$, $p < .001$) than CA ($\beta = .330$, $t = 2.952$, $p < .05$).

Model 12 was generated with a % *error* of 0.21, an AIC_C of -12.40, and a CF of 1.003. The VIF of 3.237 and *Tolerance* of 0.309 for mean DBH and CA indicate noncollinearity.

Models 11 and 12 for poor form trees exhibited rather comparable statistical indicators. However, the second model (12) was selected for having a higher coefficient of determination, and a lower percent relative error and Akaike information criterion. Therefore, the optimal regression equation for predicting poor form aboveground biomass was:

$$\text{Ln(AGB)} = [-1.339 + 1.495 * \text{Ln(DBH)} + 0.430 * \text{Ln(CA)} * 1.003] \quad \text{Model (12)}$$

4.4.11. Pooled (good form and poor form trees)

When good and poor form data were combined, the SPSS stepwise regression generated two models (Table 4.11). DBH entered the regression equation (model 13) at step 1 as the only predictive variable with significantly high relationship with AGB with adjusted $R^2 = 0.894$ ($F(1, 18) = 161.008, p < 0.001$) and accounted for about 89% of the variance of AGB. HT, WD, and BR variables were excluded from the model. The model was generated with % *error* value of 2.38, AIC_C value of -55.97, and CF value of 1.014.

The second model contained two of the five predictors and was reached in two steps with three variables (HT, WD, and BR) removed. DBH and CA entered the regression equation at step 2 of the analysis and the model was statistically significant, ($F(2, 17) = 129.228, p < .001$), and with adjusted $R^2 = 0.931$ accounted for approximately 93% of the variance of biomass. The predictive strength of DBH ($\beta = .643, t = 5.776, p < 0.001$) was greater than that of CA (β

= .364, $t = 3.271$, $p < 0.05$). Model 14 was generated with % error of 1.50, AIC_c of -65.73, and CF of 1.009. The VIF of 3.409 and $Tolerance$ of 0.293 for DBH and CA are indications of absence of collinearity.

Both models 13 and 14 for pooled good form and poor trees, respectively, obtained rather similar statistical indicators. However, the second model (14) was selected for having a higher coefficient of determination, and a lower percent relative error and Akaike information criterion. Thus, the optimal model for predicting overall aboveground biomass was:

$$\text{Ln(AGB)} = [-0.903 + 1.373 * \text{Ln(DBH)} + 0.429 * \text{Ln(CA)} * 1.009] \quad \text{Model (14)}$$

Table 4.11: Stepwise regression equations for Ln (AGB) using the natural logarithm of DBH, HT, CA, WD and BR.

Form	Model no	Allometric model	Coefficient			Adj. R ²	SEE	CF	F-value	P-value	% error	AIC	VIF
			<i>a</i>	<i>b</i>	<i>c</i>								
Good	9	Ln (AGB) = a + b * Ln (DBH)	1.383	1.214		0.936	0.057	1.002	131.96	0.000	0.10	-19.15	1.00
	10	Ln (AGB) = a + b * Ln (DBH) + c * Ln (CA)	1.385	1.011	0.174	0.971	0.039	1.001	149.63	0.000	0.04	-28.12	1.77
Poor	11	Ln (AGB) = a + b * Ln (DBH)	-1.708	2.084		0.932	0.121	1.007	124.05	0.000	0.63	-4.31	1.00
	12	Ln (AGB) = a + b * Ln (DBH) + c * Ln (CA)	-1.339	1.495	0.430	0.965	0.086	1.003	126.20	0.000	0.21	-12.40	3.23
Good + Poor	13	Ln (AGB) = a + b * Ln (DBH)	-1.445	2.027		0.894	0.164	1.014	161.08	0.000	2.38	-55.97	1.00
	14	Ln (AGB) = a + b * Ln (DBH) + c * Ln (CA)	-0.903	1.373	0.429	0.931	0.132	1.009	129.23	0.000	1.50	-65.73	3.40

4.5. DISCUSSION

The present study of aboveground biomass (AGB), carbon stock (C) and carbon dioxide (CO₂) mitigation potential revealed significant ($p < 0.05$) variation between the sampled good and poor forms of red alder trees in the silvopastoral system. On the average, aboveground biomass varied from 25.91 Mg ha⁻¹ to 49.09 Mg ha⁻¹ between the two forms of trees, resulting in carbon content of 12.95 Mg C ha⁻¹ to 24.55 Mg C ha⁻¹ and carbon dioxide mitigation potential of 47.50 Mg C ha⁻¹ to 89.98 Mg C ha⁻¹ (Table 4.3, to Table 4.8). Generally, biomass, carbon stock and sequestered carbon dioxide were found to be greater in good form trees than in poor form trees. Thus, the hypothesis that the estimates of dendrometric measurements, aboveground biomass, carbon and carbon dioxide contents of the good form red alder samples do not differ significantly from that of the poor form red alder samples was rejected by the result of this study. This disparity could be explained by the morphological differences between the two forms of trees. It is pertinent to recall that the tree forms were chosen on the basis of their conditions: Good form trees are generally free of defects, showing good health and expected to reach their full age and size potential while poor form trees are trees in obvious decline, or with significant health and/or structural impairments, and showing very little signs of life or remaining vitality, or with severe, irremediable structural defects, including advanced decay and crack, root problems, weak branch union, canker, poor architecture, stunted growth, and dead wood. It is evident that both forms of trees do not share comparable trunk shape and crown architecture (Figure 4.1). Consequently, good form trees are expected to have greater biomass, and hence carbon stock and carbon dioxide mitigation potential, than poor form trees.

The amount of biomass among the tree components for the two forms of red alder was largest in stems, intermediate in branches, and lowest in twigs (Figure 4.5). As expected, this result indicated that the component which constituted a maximum portion of biomass stored the maximum amount of carbon. Since the stem contributed more biomass as compared to other components hence it stored and sequestered more carbon in its biomass compared to the branch and twig. Factors that could influence the biomass of these aboveground tree components include tree density, growth habit of red alder, crown size, average number of branches on the trees, tree age, site condition, soil, moisture conservation, and management interventions, among others.

Again, the difference between carbon stocks of tree components in this study may be related to the fibre composition of plant tissues. Research studies have reported higher content of cellulose, hemicellulose and lignin in woody materials compared to herbaceous components. Ververis *et al.* (2004) have observed that lignin and cellulose content is contingent upon tissue maturity, but does not change significantly within each species. Lamtom and Savige (2006) reported cellulose has a mean carbon percentage of 42.1%, and that the carbon content varies between 40-44% and 63-72% in hemicelluloses and lignin, respectively.

The figures of biomass and carbon sequestration in this study were within the range reported by other researchers. Estimated carbon sequestration potential in temperate latitudes agroforestry systems has been placed between 10 Mg C ha⁻¹ and 208 Mg C ha⁻¹, which may

have been attained over a cutting rotation of 20 to 50 years (Dixon 1995; Kort and Turnock 1999; Peichl *et al.*, 2006; Turnock 2001; Montagnini and Nair 2004). Research studies in tropical climates have estimated the carbon sequestration potential in agroforestry systems to be between 21 Mg C ha⁻¹ and 240 Mg C ha⁻¹ within a rotation cycle of ten or 20 years (Dixon 1995; Peichl *et al.*, 2006; Adesina *et al.*, 1999; Montagnini and Nair 2004).

The underlying assumptions of regression such as homoscedasticity and no autocorrelation were completely met after the independent variables, DBH, HT, CA, BR and WD, were log-transformed. The choice of applicable biomass model is principally a function of the intended use and by available independent variables. Many independent variables may be applied to reduce a bias in the prediction, but it is always appropriate to minimise the number of predictors to reduce the variance of estimates (Wirth *et al.*, 2004; Cienciala *et al.*, 2006). Based on this view point, the stepwise method was adopted in this study to determine which of the five independent variables are very essential in the prediction of AGB. The method is easy to compute, use and explain. Moreover, it is widely used. On the other hand, it is possible to miss the ‘optimal model’ because the variables are added and dropped one at a time, the method may overstate the significance of results. However, both standard and stepwise regression methods gave reliable models based on the evaluation of their goodness of fit (Tables 4.10 and 4.11). The development of allometric models for stem, branch and twig biomass in a simple linear regression with DBH as the only explanatory variable provided better estimation of component biomass in good form trees with higher R^2 and F-value, and lower *SEE*, *CF*, % *error*, and AIC_c compared to poor form trees (Table 4.10). For the stepwise regression analysis,

models with DBH and CA as explanatory variables provided a better estimation of biomass for both forms of trees since the total variation explained by the relationship is high ($R^2 > 0.93$) and the associated relative error was small (Table 4.11). The results indicate that CA when added to DBH is a strong indicator of aboveground biomass, which implies that variability of aboveground biomass of open-grown trees in agroforestry landscape is largely a function of DBH and CA. This is in line with the other independent studies, which proved the importance of crown variables for the prediction of biomass of tree species (e.g. Wirth *et al.*, 2004; Ledermann, Neumann 2006; Gschwantner, Schadauer 2006; Kuyah *et al.*, 2012).

The disparity in the explained variances (R^2) of the two forms may be due to the large variation of physical shapes characterising this category of trees growing in the open. Poor form trees are irregularly shaped and multiple-stemmed and seem to allocate more biomass to branches than good form trees over the entire range of DBH measured. It may also be due to the fact that poor form trees were those for which the crowns had suffered from either a) Storm damage and loss of branches and b) Squirrel damage and disease and loss of branches. The strong correlation of DBH with biomass in this study is a confirmation of the prominent effect of diameter on biomass prediction, which suggests that the variability of biomass of open grown trees is largely explained by the variability of DBH. This is in agreement with previous findings (Brown *et al.*, 1989; Zianis and Mencuccini, 2004; Basuki *et al.*, 2009; Vahedi *et al.*, 2014) that DBH alone is a good predictor of biomass especially in relations to its accuracy, ease of measurement in the field and ready availability in forest inventories data. Generally, the

inclusion of additional variables in the models was expected to improve the accuracy of the models.

Based on the above regression results, estimating biomass and carbon sequestration potentials in these open-grown trees within agroforestry settings will require the development of more suitable relationships of biomass to measured characteristics; hence, incorporation of crown area could improve the accuracy of equations for trees in agroforestry landscapes. However, challenges associated with the measurement of CA are frequently given as the reason for not including it as a parameter in allometric equations. There are many challenges with using crown area as a predictor variable, ranging from the difficulty to measure crown area accurately to lack of consistent allometric equations (Gibbs *et al.*, 2007). Moreover, crown geometry in agroforestry landscapes is highly heterogeneous due to interplant competition and management. Due to their branchiness, open-grown trees at given diameter and/or height have considerable variability in their biomass.

In this study, wood density had no significant ($p > 0.05$) effect on biomass because of its weak correlation with other variables (Table 4.9), an indication of reduction in biomass. Chave *et al.* (2005) have observed that wood density is the second most important parameter after tree diameter in predicting the mass of a tree through destructive sampling, while Baker *et al.* (2004) have warned that ignoring variations in wood density would lead to poor prediction of the aboveground biomass and consequently in carbon calculation.

However, variation in wood density is so considerable that it occurs at multiple scales in forest ecosystems, ranging from within and between trees to regional changes in mean wood density among forest communities, which makes difficult the estimation of accurate average values by species (Muller-Landau, 2004; Chave *et al.*, 2006; Grabner and Wimmer, 2006; Swenson and Enquist, 2007).

Wood density is a function of the three major structural components of wood, α -cellulose, hemicellulose and lignin. The density of cell wall and wood tissue is determined by its structure such as solid material (cell walls) and air space (cell lumens) or void volume. Basic wood density is defined as oven-dry (0% MC) mass per green volume (kg m^{-3}), i.e. it is a reflection of the amount of dry cell wall material that is contained in a cubic metre of fresh wood, and this in turn is a function of the type and size of cells and the inherent differences between the amounts of earlywood versus latewood formation in the piece under consideration, among other factors (Jeong *et al.*, 2009; Saranpaa 2003). Additionally, relative percentages of latewood and earlywood are functions of location, climate, tree age, height, diameter class, growth conditions, radial growth, tree genetics, sensitivity to seasonal changes in both temperature and precipitation, and possibly management (Kettering *et al.*, 2001; Chave *et al.*, 2004; Auty *et al.*, 2014).

The equations in this study were developed using data from 10 good form and 10 poor form destructively sampled open-grown trees. This sample size of only 20 trees was adjudged inadequate for the development of equations that can be applied widely. However, these models

should be useful for providing reliable estimates of biomass for open-grown trees with similar species characteristics, tree forms and site conditions (soil and climate). Modelling the biomass of open-grown trees turned out to be challenging, possibly due to the branching habit and shape of stem and crown characterising this group.

For wider application and greater accuracy, these equations will need to be calibrated using data from a greater number of destructively sampled open-grown trees in other agroforestry landscapes. Further research on measures that could be used to improve biomass estimates for varied forms of open-grown trees would therefore be useful.

4.6. CONCLUSION

In conclusion, the species-specific allometric models presented in this study for determining aboveground biomass in two forms of open-grown red alder have some use, but their application is likely to be limited. The use of DBH as a sole predictor variable or in combination with crown area will facilitate the collection of more reliable inventory data to examine temporal and spatial variability in silvopastoral ecosystem structure and function. However, care should be taken in applying the allometric models developed in this study to other silvopastoral sites without knowledge of tree form. To determine how appropriate, the allometric models are for a given site, it is recommended that DBH versus tree CA curves be constructed for the area of interest and compared to that presented in this study.

Chapter 5 : EFFECT OF LIGHT ON PASTURE PRODUCTIVITY AND QUALITY IN A SILVOPASTORAL SYSTEM

5.1. INTRODUCTION

In a general sense, agroforestry is a term covering all farm practices that deliberately combine the production of trees and/or shrubs with other crops and/or livestock in a manner that may be collectively beneficial. Through the intentional integration of trees with livestock, silvopastoral practices strive to simultaneously optimize economic, environmental and social benefits, thereby ensuring the attainment of multiple objectives. There are four main components of silvopastoral practices that can be readily manipulated at any given site - trees, pastures, animals and the soil (Mead (2009). The interactions between these major components are dynamic and their understanding is important in the development of comprehensive management practices. Light usually becomes the dominant competitive factor with time as the trees shade the pasture. Trees, because of their growth habit, will shade pastures, the degree of shade being related to the density of the tree canopy. How quickly this happens depends on tree species, their temporal and spatial arrangement, age, and factors that influence tree vigour. The tree–pasture-animal interaction affects not only pasture production but also pasture quality and through that animal productivity and plant nutrient status (Mead, 2009). However, other interactions related to shelter and animal health can occur, of which reducing stresses on animals can often be very important. Thus, agroforestry is much more complex than either

pastures or forests on their own and the mixing of these two major components can result in some unexpected interactions, both positive and negative.

The influence of trees on the understory pasture is contingent upon the degree to which they modify the microclimate and soil properties (Benavides *et al* 2009). The quantity and quality of pastures beneath trees are useful indicators of the sustainability of farms, because they significantly influence both economic performance (farm output) and resource status (Lambert *et al.*, 1996).

Benefits of silvopastoral systems are numerous and have been summarized by Mosquera-Losada *et al.* (2005). Extension of the growing season of herbage via protection of swards from environmental extremes and overall increases in forage production have been shown by Sibbald (1999). Kephart and Buxton (1993) found that shade tended to decrease secondary cell-wall development and proposed that morphological changes in herbage grown under reduced light, e.g. under a tree canopy or in areas with prolonged cloudiness, would very likely increase the nutritive value of herbage, estimated in terms of its CP concentration. Peri *et al.* (2007) also found increased CP concentrations with increasing shade for herbage of *Dactylis glomerata*.

Conversely, low radiation levels have been shown to reduce forage production and nutritive value. Research in Scotland, UK (Sibbald *et al.*, 1994) showed that herbage production decreased with increased shading (or attenuation of full sunlight) when precipitation and

temperature favour herbage growth. Belesky (2005) and Peri *et al.* (2007) found that herbage plants grown in areas with lower light levels were smaller, had fewer numbers of tillers and produced less dry matter (DM) compared with treatments with higher levels of radiation. Shade-grown grasses of cool-temperate origin increase allocation of N to leaves to maximize light acquisition. Lin *et al.* (2001) found that, in general, acid–detergent fibre (ADF) concentration was either unaffected or increased because of shading. The high nitrate concentrations, along with depressed levels of total non-structural carbohydrates (TNC), found in shade-grown herbage (Deinum *et al.*, 1968; Chiavarella *et al.*, 2000) could compromise nutritive value. Concentration of TNC in herbage has been positively associated with improved dietary protein utilization in the rumen, and increased selection and intake by grazers (Chiavarella *et al.*, 2000; Mayland *et al.*, 2000). High levels of N in herbage have also been associated with off-flavours in meat from pasture-raised beef cattle (Lane and Fraser, 1999).

The measurement of pasture productivity in grazing systems is a complex issue that relates the variations in climate to seasons and the soil – plant-animal interaction, dynamics of water and nutrients from soil, changes in botanical composition, and seasonal variations in the stocking rate, grazing intensity and frequency. The performance of a pasture is mainly influenced by humidity, temperature and radiation (Mc Calla and Bishop-Hurley, 2003) and, in temperate regions, the best relationship between these factors are generated in the period of spring (Skinner *et al.*, 2009). Pasture biomass productivity and quality values are crucial in management of grazing lands and livestock. More accurate and timely estimation of pasture biomass production and forage quality during the grazing season can help livestock managers

make appropriate decisions of pasture fertilization and stocking rate. Laboratory analyses of the composition of feed or forage are used to assess their nutritive value. A typical feed analysis includes measurements of some important quality attributes or parameters (e.g., crude protein, fibre, digestibility, etc.) used to define nutritive value. Conventional laboratory chemical “wet chemistry” methods of laboratory chemical analysis have long been used for assessment of forage quality (Kellems and Church 1998). Neutral detergent fiber (NDF), acid detergent fiber (ADF) and crude protein (CP) concentrations are commonly used forage quality variables (Ball *et al.* 2001). These three quality variables are closely associated with intake potential, digestibility, and nutritive values of forage (Ball *et al.*, 2001). Conventional “wet chemistry” methods used to determine these quality variables are time consuming and costly, and also require personnel with special skills. Additionally, the hazardous waste generated from laboratory processes must be disposed of in order to reduce the risk of environmental pollution.

Another technique for the assessment of forage is the use of the near-infrared reflectance spectroscopy (NIRS) method. This is a rapid and inexpensive computerized approach to quantify the nutritive values of forage and grain crops (Marten *et al.*, 1989; Shenk and Westerhaus 1994). Studies indicate that strong correlations exist ($r > 0.95$ between NIRS and the various components of forage nutritive value (Norris *et al.*, 1976; Shenk *et al.*, 1979; Counts and Radloff, 1979; Ward, 1980). NIRS uses near-infrared light instead of chemicals as in conventional methods, to determine protein, fibre, mineral, energy and other variables of interest. In NIRS method, air dried and finely ground samples are exposed to infrared light in a spectrophotometer. The reflected infrared radiation is converted to electrical energy and fed

to a computer for the determination of the quantity of these components in the feed. It is based on the fact that each major organic component of forage and grain will absorb and reflect near-infrared light in a different way (Norris *et al.*, 1976; Stermer *et al.*, 1977; Shenk *et al.*, 1979). Though NIRS analysis is fast and very precise but its accuracy depends on appropriate calibration with adequate number of “wet chemistry” samples similar to those being analysed, and therefore requires a period of time for the preparation of the samples.

Understorey pasture productivity is mainly affected by the intensity of solar radiation reaching the forest floor, which in turn depends upon the degree of canopy closure and characteristics of the tree canopy. Citing Knowles *et al.*, (1997), Devkota *et al.* (2001) observed that canopy closure and available light provide measures of the potential shading by trees that are independent of tree stems per hectare and the height to which trees have been pruned. The authors concluded that canopy closure is the critical factor to manage if pasture production is to be maintained at an economic level and suggested a critical threshold range of 40-50% as the percentage of tree canopy closure that would be required to maintain pastoral enterprises under deciduous tree based silvopastoral systems.

In winter of 2012, the three alder 200-stem ha⁻¹ blocks at Henfaes SNNE were thinned down to 100-stem ha⁻¹. Understorey pasture DM production has been strongly linked to overstorey canopy closure (Knowles *et al.* 1999) and, unlike tree spacing, this index also accounts for both the arrangement and size of the trees. Hence, there is the need to develop a relationship between alder canopy closure and understorey pasture DM production, which can be used by farmers to

optimise the spacing of alder-stands for multiple objectives, such as soil conservation, pasture grazing and wood fuel. This study will therefore look at how the current tree density treatments (100-stem ha⁻¹) affects the light at ground-level and how this in turn affects the pasture productivity and quality.

5.1.1. Objectives

The general objective of this study was to evaluate the influence of solar radiation on pasture productivity and quality in thinned red alder blocks in a silvopastoral system.

Specifically, the study will do the following:

- Estimate seasonal pasture production under varying solar radiation intensity;
- Determine the effect of solar radiation intensities on pasture productivity and quality under varying canopy gap levels.

5.2. MATERIAL AND METHODS

5.2.1. Study area

The experiment was conducted in the year 2012 - 2014 on 0.42 ha red alder blocks at the United Kingdom's Silvopastoral National Network Experiment (SNNE) located in the Bangor University's Henfaes research centre (53°14'N 4°01'W), Abergwyngregyn, Gwynedd, Wales. The climate in Henfaes is cool and temperate. Mean temperature over the course of this study was 10.6 °C, and temperatures of the warmest and coldest month was 20.0 °C in July and 3.4 °C in January, respectively. Average annual precipitation ranged from a minimum of 25 mm in April to a maximum of 114 mm in December. Weather data for the period of the study are presented in Table 1.1 and Figure 1.3. A detailed description of the study area including vegetation and previous treatments to pasture establishment has been previously reported in chapter 1.

5.2.2. Determination of the seasons

Data on the climatic condition of the study area were collected from the on-site automatic weather station. Data for three years (2012-2014) were averaged for precipitation, solar radiation, temperature and relative humidity at the study site.

5.2.3. Assessing pasture biomass

All trees in the three red alder blocks were inventoried prior to and following random thinning operations in winter of 2012. In each of the three thinned alder blocks, seven 0.25 m² grazing exclusion cages were systematically placed in varying canopy gap levels created as result of

the thinning (Figure 5.1). The perimeters of these 7 canopy gaps were not delimited, however, the distance of each cage from the nearest tree was measured in metres with a 50 m measuring tape. For consistency in data collection, pastures in all exclusion cages were clipped to a residual level of 2.5 cm at the start of the setup on June 30, 2003. Thereafter, forages in the grazing exclusion cages were harvested to a residual sward height of 2.5 cm on the last day of every month from July 2013 to June 2014 (Appendix 5.1). The grazing exclusion cages were permanently positioned and maintained within the predetermined locations all through the study period.



Figure 5-1: 0.25 m² grazing exclusion cages in varying red alder shade level.

To determine their dry matter content, the harvested forages were taken immediately to the Henfaes Research Centre laboratory, weighed with a 0.005g precision balance, and subsamples were extracted for the evaluation of the dry matter content, weighed and dried in a forced air oven at 60°C (140°F) for 24 hours or until constant weight was achieved. This temperature is sufficiently high to decrease the water considerably and low enough not to significantly modify sample chemistry. The monthly production of dry matter per hectare was obtained by multiplying the green matter production value by the dry matter content and dividing by 100. The daily growth rate per hectare was obtained by dividing the monthly production by the number of days in each month. To express production in kg DM ha⁻¹, the daily production rate per hectare was multiplied by a conversion factor of 0.1.

5.2.4. Assessing forage quality

To assess pasture quality, the monthly oven-dried forage was then milled to pass a 1-mm sieve. The 12 months (July 2013 to June 2014) collection of dried and milled forages were pooled based on the seasonal differences in the condition of the tree canopy, that is, *with-leaves* condition (months of March to September when the trees are with foliage) and *without-leaves* condition (months of October to February when the trees are without foliage). The pooled samples were stored in sealed plastic bags and taken to the feed and forage analysis laboratory of Bioparametrics Ltd, Peter Wilson Building, West Mains Road, Edinburgh, Scotland, EH9 3JG, United Kingdom in early August 2014 for quality analyses. Forage quality was quantified in terms of protein content (Crude Protein), fibre content (Acid Detergent Fibre and Neutral Detergent Fibre), and energy potential (Metabolisable Energy). The laboratory applied the Near

Infrared Reflectance Spectroscopy (NIRS) (Model FOSS 6500, NIRSystems Inc., Silver spring, MD, USA) standard procedures of forage quality analysis as outlined by Stuth et al (2003) to determine the concentrations of crude protein (CP), acid detergent fibre (ADF), neutral detergent fibre (NDF), and metabolisable energy (ME) in the dried and milled forages. The amount of pasture CP, ADF, NDF, and ME availability in $\text{g m}^{-2} \text{d}^{-1}$ were estimated by multiplying pasture biomass by the CP, ADF, NDF and ME concentrations, respectively.

5.2.5. Light measurement

A total of 42 hemispherical photographs taken looking upward were used to estimate solar radiation penetration to the canopy gaps of varying sizes created by the thinning of the three red alder blocks. These comprised of 21 photos taken above the 21 grazing exclusion cages in August 2013, when the leaves were fully expanded and repeated in December 2013, after the trees had completely shed their leaves (Figure 5.3). Light transmission through the canopy gaps was measured with the use of a Nikon Coolpix 990 digital camera (Nikon Corporation, Tokyo, Japan) fitted with a Nikon FC-E8 fisheye converter (lens) pointed upward (Nikon Corporation, Tokyo, Japan). The digital camera was mounted on a tripod vertically above each exclusion cage in the gaps at a height of approximately 1.5 m above the ground and hemispherical view (HemiView) colour images of the overstory canopy gaps were taken (Figure 5.2). The top of the camera was oriented relative to magnetic north and positioned horizontally with the aid of a spirit level, and adjustment for magnetic declination was made during the photo analysis (Rich, 1990). Automatic settings were selected for aperture width and shutter speed (Inoue *et al.*, 2004). The digital images were downloaded directly to a personal computer and

analysed with image-processing software, HemiView Version 2.1 (Delta-T Devices, Cambridge, UK). The images were processed following the approach of Brunner (2002). This comprised the manual setting of a threshold value to separate canopy and sky elements into a binary black and white image. The lens distortion was corrected using the Coolpix 900 option (Hale and Edwards 2002).

The hemiView photos were used to calculate absolute amount of radiation beneath the tree canopies. Hemispherical photographs were analysed to derive a variety of solar radiation indices such as gap fraction (GF), direct site factor (DSF), indirect site factor (ISF), and global site factor (GSF).



Figure 5-2: Fisheye lens camera on tripod over grazing exclusion cages.



With leaves hemi-view



Without leaves hemi-view



With leaves hemi-view



Without leaves hemi-view

Figure 5-3: Hemispherical view (Hemi-View) images of red alder overstorey canopy gaps of trees with leaves and trees without leaves.

The primary indicator of canopy gaps was canopy gap fraction (GF). Gap fraction is the proportion of open area within a canopy, the fraction of view looking up from beneath the canopy that is not blocked by wood and foliage. Gap fraction measurements hinge on dividing the sky into several sectors and calculating the gap fraction for each sky sector (Awal 2008). A gap fraction of zero (0) means the sky is completely blocked (obscured) in the particular sky sector, whereas a gap fraction of one (1) means the sky is completely visible (not obscured). The gap fraction algorithm used in the instrument assumes a diffusely lit sky, so wherever possible, images were captured under calm and completely overcast sky conditions to maximize image contrast and minimise interference by direct sunlight (e.g. sunflecks).

Global (or total) solar radiation (also called global site factor) is the amount of solar radiation actually reaching a particular location and is influenced mainly by cloudiness, time of year, latitude, and surface geometry (Iqbal, 1983). Global solar radiation (global site factor) is the sum of the direct irradiance (direct site factor) in the sunlight and the diffuse solar radiation (indirect site factor) scattered from sunlight as it passes through the atmosphere (Rich 1990).

Data for photosynthetically active radiation (PAR), taken hourly over the study period (2012-2014), were procured from the automatic weather recording station in the study area (Henfaes research centre). Analyses of radiation flux was applied for photosynthetically active radiation (400-700 nm) photon flux density in order to determine radiation available for photosynthesis.

The relative contribution of direct sunlight and diffuse skylight to global radiation flux was expressed as direct and indirect site factors, respectively. Direct site factor is the proportion of direct sunlight and indirect site factor is the proportion of diffuse skylight under the canopy relative to that outside the canopy (Rich 1990).

Radiation flux density was calculated as the sum of direct sunlight and diffuse skylight that passed unimpeded through canopy openings (gaps) and expressed in absolute units as mole per square metre per day ($\text{mol m}^{-2} \text{d}^{-1}$). Calculation of percent solar radiation values was therefore based on actual measurement of global site factor that passed through the canopy gaps.

5.2.6. Data analysis

Statistical analysis was carried out by checking assumptions of normality, homogeneity of variance and multicollinearity and transforming the data to logarithms as appropriate. For log-transformed variables, the mean of the untransformed data was used to express central tendency and the standard error derived from log-transformed data was used to express precision.

An ANOVA was used to determine the impact of thinning on pasture productivity and quality parameters. The productivity and quality data with corresponding radiation flux densities were pooled over pasture blocks, gap locations, months and seasonal conditions ($n = 252$).

Where appropriate, a linear mixed model was fitted using the method of restricted maximum likelihood (REML) to take account of the variance components of the random effects of month, season, and canopy gap level (location) and their interactions on pasture productivity.

A subset of data comprising the condition of alder trees in two seasons (with- and without-leaves) were used for comparison. Herbage harvested from March to September corresponded to 7 months of production under the with-leaves condition, referred to here as with-leaves condition and harvests from October to February corresponded to 5 months, referred to as without-leaves condition.

Linear regression analysis was used to explore the functional relationships between solar radiation transmission and pasture productivity and quality. Differences were assessed at the significance level of $p < 0.05$. All analysis was conducted using the SPSS version 22 software.

5.3. RESULTS

5.3.1. Pasture production

Results of the monthly, seasonal and locational pasture productions parameters obtained from 21 grazing exclusion cages in the canopy gaps in three alder blocks in the 2013/2014 growing seasons are presented in Figure 5.4, to Figure 5.7. Significant ($p < 0.05$) differences were observed between the monthly, seasonal and locational pasture dry matter (DM) yield for the amount of solar radiation transmitted through the tree canopy of red alder.

Linear fitted curves were attempted to predict the trend of monthly, seasonal and locational averages of pasture dry matter yield. The results of this study show that all coefficients were statistically significant ($p \leq 0.0001$). Analysis of the mixed model determined by the variance of components, indicated that about 45% of the total variation in pasture dry matter yield can be attributed to seasonality, 40% to month, and 15% to location (grazing exclusion cage locations). Only 0.6% of the variation was attributable to the blocks, which did not show significant values ($p = 0.05$).

Strong significant relationship was obtained between solar transmission and monthly pasture production ($R^2 = 0.90$, $p < 0.05$). Pasture productivity increased with increasing solar radiation intensity (Figure 5.4). Solar radiation reaching the various grazing exclusion cages ranged from a minimum of $4.99 \text{ mol m}^{-2} \text{ d}^{-1}$ in December to a maximum of $64.13 \text{ mol m}^{-2} \text{ d}^{-1}$ in June (Figure 5.4).

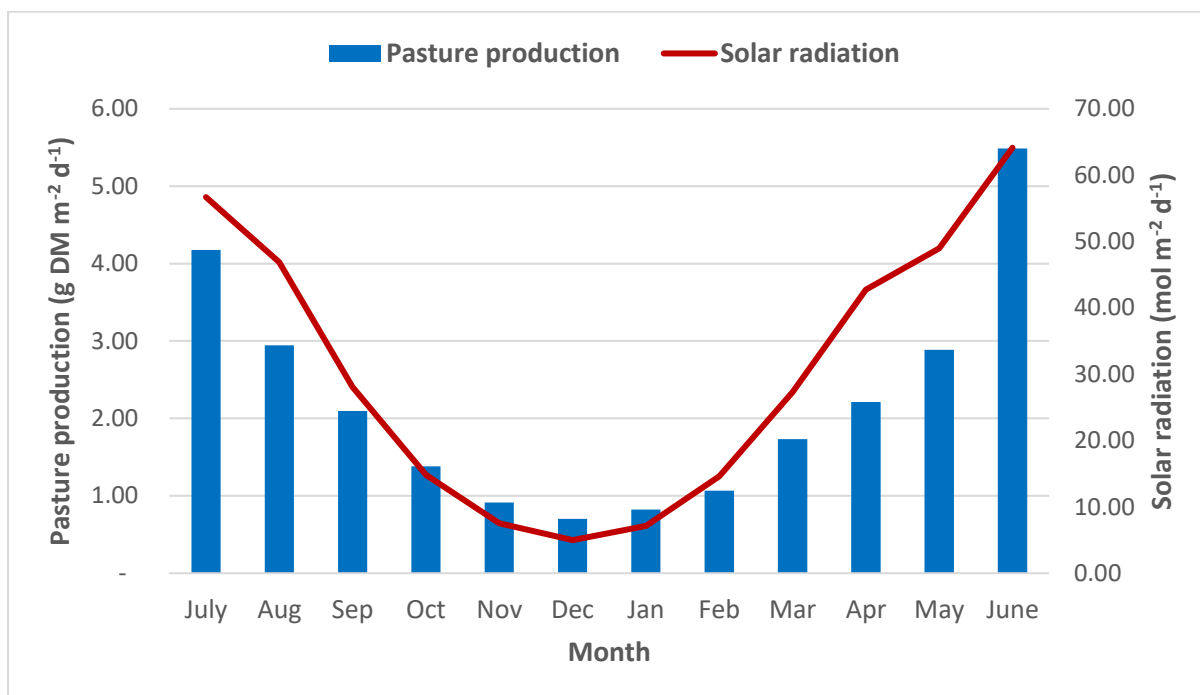


Figure 5-4: Mean monthly levels of solar radiation in relation to pasture productivity.

Pasture yield was significantly different ($p < 0.05$) between months (Figure 5.4 and Appendix 5.2). Mean pasture yield ranged from a daily minimum of $0.70 \text{ g DM m}^{-2} \text{ d}^{-1}$ in December to a daily maximum of $5.49 \text{ g DM m}^{-2} \text{ d}^{-1}$ in June. Higher mean pasture yield was recorded in the months of April, May, June, July, August and September with a peak in June ($5.49 \text{ g DM m}^{-2} \text{ d}^{-1}$) for the three alder blocks, which corresponded to the seasons of spring, summer and early autumn, and the months when the alder tree leaves are in bloom, and temperature and radiation levels are the highest and precipitation is close to moderate level (Figure 5.4). Lower mean pasture yield was obtained from mid-autumn (October) to late winter (February) with a deep in December ($0.70 \text{ g DM m}^{-2} \text{ d}^{-1}$), which corresponded to the seasons when the trees are without leaves and temperature and radiation levels are lower.

Again, pasture yield differed significantly ($p < 0.05$) between seasons. Seasonal herbage productivity model showed that 25.85% of seasonal herbage yield occurred in spring, about 47.71% were obtained in summer, 16.62% in autumn and only 9.82% in winter (Table 5.1) .

Table 5.1: Seasonal distribution of dry matter production at Henfaes SNNE (July 2013 to June 2014)

Season	Month	Pasture productivity (g DM m ⁻² d ⁻¹)				
		Mean \pm Stdev	Min	Max	CV (%)	Percent
Spring	Mar, Apr, May	2.28 ^a \pm 0.93	0.93	5.41	40.78	25.85
Summer	Jun, Jul, Aug	4.20 ^b \pm 1.88	1.36	8.21	44.76	47.71
Autumn	Sep, Oct, Nov	1.46 ^c \pm 0.79	0.33	3.83	54.10	16.62
winter	Dec, Jan, Feb	0.86 ^d \pm 0.48	0.12	2.27	55.81	9.82

^{abcd} Means within a column followed by different letters vary significantly ($P < 0.05$)

Furthermore, significant ($p < 0.05$) relationship was found to exist between pasture production and distance of each grazing exclusion cage to the nearest tree. Cage distance to trees varied from a minimum of 5 m to a maximum of 11.25 m with a mean distance of 8.17 m and standard deviation of 1.90 m (Table 5.2). The proportion of incident solar radiation reaching the cages increased significantly ($R^2 = 0.79$, $p < 0.05$) as the distance between each cage and the nearest

tree increased (Figure 5.5). Similarly, proportion of pasture production increased significantly ($R^2 = 0.81$, $p < 0.05$) with increasing distance from each cage to the nearest tree (Figure 5.6).

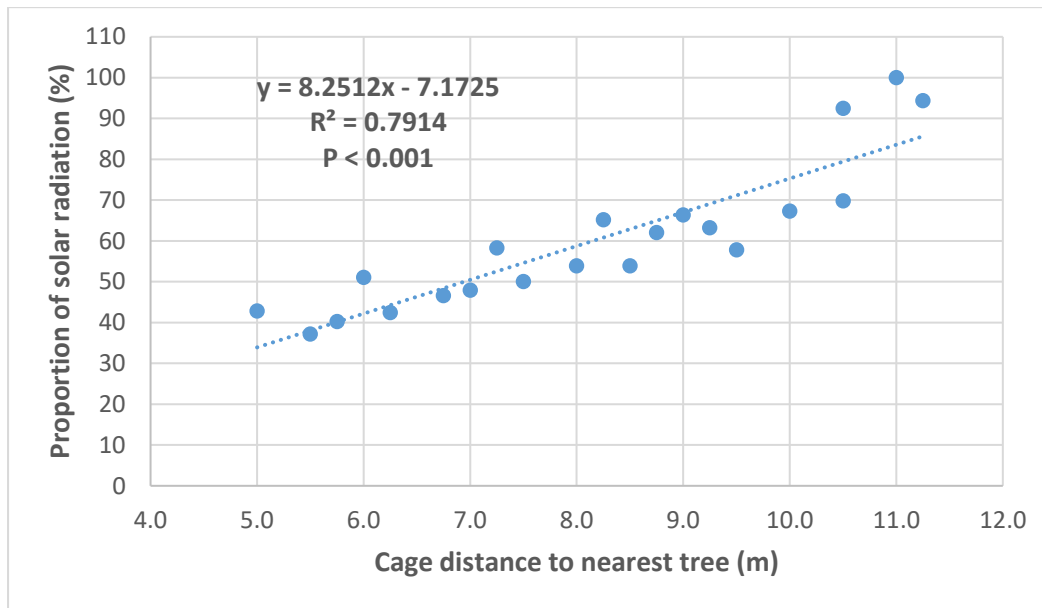


Figure 5-5: Relationship between incident solar radiation and cage distance to nearest trees (July 2013 to June 2014).

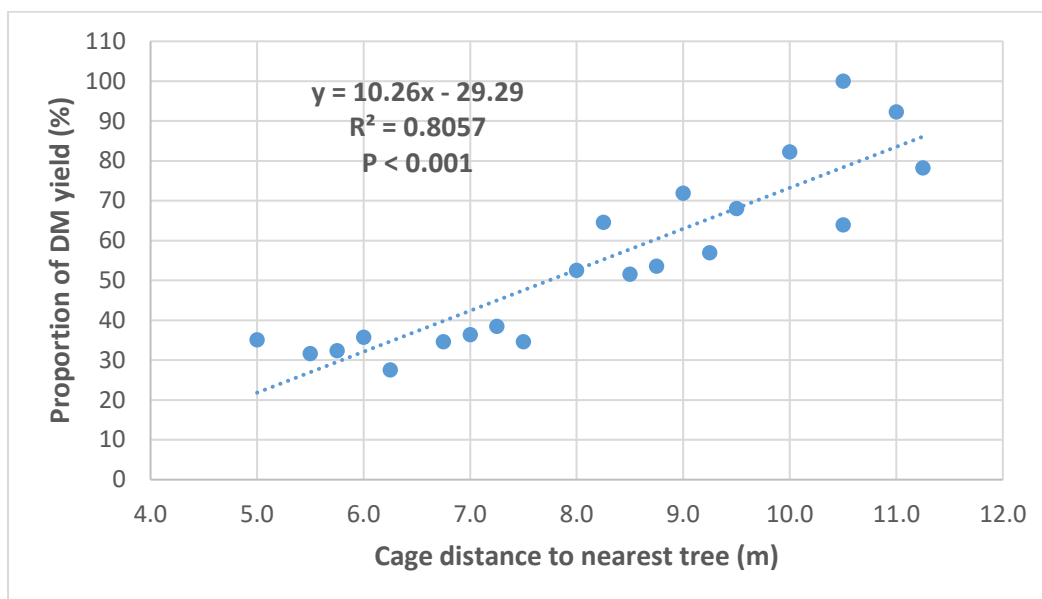


Figure 5-6: Relationship between DM yield and cage distance to nearest trees (July 2013 to June 2014).

5.3.2. Forage quality (nutritive value)

Descriptive statistics for light transmission and the pooled dried and milled forage parameters are presented in Tables 5.2 & 5.3. Results show that solar radiation, DM yield and pasture quality parameters (nutritive values) varied significantly ($p < 0.05$) between the *with-leaves* condition (months of March to September when the trees are with foliage) and the *without-leaves* condition (months of October to February when the trees are without foliage).

Solar radiation reaching the various grazing exclusion cages varied between 12.03 and 41.27 $\text{mol m}^{-2} \text{d}^{-1}$ in *with-leaves* condition and from 7.46 to 11.57 $\text{mol m}^{-2} \text{d}^{-1}$ in *without-leaves* condition (Table 5.2). Mean concentrations of DM, CP, ADF and NDF were 94.43%, 20.07%,

29.15% and 57.23% of dry matter, respectively, in *with-leaves* condition compared to 95.29%, 18.85%, 28.19% and 51.37%, respectively, in *without-leaves* condition. Mean metabolisable energy varied from 8.33 MJ kg DM in *without-leaves* condition to 8.60 MJ kg DM in *with-leaves* condition (Table 5.2).

Table 5.2: Descriptive statistics of light incidence, percentage dry matter (DM), crude protein (CP), acid detergent fibre (ADF), neutral detergent fibre (NDF) and metabolisable energy (ME) concentrations in pasture.

Parameters	Min	Max	Mean	Standard deviation	CV (%)
With-leaves					
Solar radiation (mol m ⁻² d ⁻¹)	12.03	41.27	21.90 ^a	8.37	38.21
Distance of each cage to nearest tree (m)	5.00	11.25	8.17	1.90	23.25
Dry matter (%)	93.80	95.00	94.43 ^a	0.30	0.31
Crude Protein (% of DM)	17.00	23.50	20.08 ^a	1.88	9.36
Acid detergent fibre (% of DM)	26.60	31.20	29.15 ^a	1.17	4.01
Neutral detergent fibre (% of DM)	53.00	60.50	57.23 ^a	1.87	3.26
Metabolisable energy (MJ kg ⁻¹ DM)	7.80	9.70	8.60 ^a	0.42	4.88
Without-leaves					
Solar radiation (mol m ⁻² d ⁻¹)	7.46	11.57	9.64 ^b	0.96	9.95
Distance of each cage to nearest tree (m)	5.00	11.25	8.17	1.90	23.25
Dry matter (%)	94.40	96.60	95.29 ^b	0.64	0.67
Crude Protein (% of DM)	14.80	22.40	18.85 ^b	1.58	8.38
Acid detergent fibre (% of DM)	26.30	30.80	28.17 ^b	1.40	4.96
Neutral detergent fibre (% of DM)	48.30	54.50	51.37 ^b	2.02	3.93
Metabolisable energy (MJ kg ⁻¹ DM)	7.30	8.90	8.33 ^b	0.43	5.16

^{ab} Column means with different letter superscripts are significantly different at P < 0.05.

In *with-leaves* condition, variance for the pasture parameters was highest in CP (CV = 9.36%) and lowest in DM (CV = 0.31%) (Table 5.2). In *without-leaves* condition, CP and DM again showed the highest (CV = 8.38%) and the lowest (CV = 0.67%) variance, respectively. Between seasons, variance in solar radiation was higher in *with-leaves* (CV = 38.21%) than in *without-leaves* (CV = 9.95%). Pastures in *without-leaves* growing conditions had 56% lower solar transmission, <1% higher DM % content, 6% lower crude protein concentration, 3% lower acid detergent fibre concentration, 10% lower neutral detergent fibre concentration, and 3% lower metabolisable energy compared with the results in *with-leaves* growing conditions.

Similarly, DM yield and available CP, NDF, ADF and ME obtained by multiplying their respective concentration values by the DM yield differed significantly ($p < 0.05$) between the *with-leaves* and the *without-leaves* conditions (Table 5.3). Mean daily DM yield was 3.03 g DM m⁻² d⁻¹ in *with-leaves* conditions and 0.98 g DM m⁻² d⁻¹ in *without-leaves* conditions. Mean daily available CP, ADF, NDF, and ME were 0.62 g m⁻² d⁻¹, 0.90 g m⁻² d⁻¹, 1.76 g m⁻² d⁻¹, and 0.26 MJ g m⁻² d⁻¹, respectively, for the *with-leaves* conditions compared to the *without-leaves* conditions. Generally, the pasture parameters show greater variability in *without-leaves* conditions compared to *with-leaves* conditions while light transmission exhibited greater variability in *with-leaves* (CV = 38.21%) than in *without-leaves* (CV = 9.95%).

Table 5.3: Descriptive statistics of dry matter yield (DM), and available crude protein (CP), acid detergent fibre (ADF), neutral detergent fibre (NDF) and metabolisable energy (ME).

Parameter	DM yield (g DM m ⁻² d ⁻¹)	CP (g m ⁻² d ⁻¹)	ADF (g m ⁻² d ⁻¹)	NDF (g m ⁻² d ⁻¹)	ME (MJ g m ⁻² d ⁻¹)
With-leaves					
Minimum	1.73	0.35	0.52	1.02	0.14
Maximum	5.49	1.05	1.44	2.83	0.45
Mean	3.03 ^a	0.62 ^a	0.90 ^a	1.76 ^a	0.26 ^a
Stdev	1.33	0.22	0.30	0.59	0.10
Without-leaves					
Minimum	0.70	0.09	0.12	0.24	0.02
Maximum	1.38	0.36	0.51	0.97	0.15
Mean	0.98 ^b	0.19 ^b	0.28 ^b	0.50 ^b	0.07 ^b
Stdev	0.26	0.09	0.12	0.21	0.03

^{ab} Column means with different letter superscripts are significantly different at $P < 0.05$.

A Pearson product-moment correlation coefficient was computed to assess the relationship between the light transmission, DM yield and available CP, ADF, NDF and ME. All parameters were found to be significantly correlated with each other ($p < 0.001$) (Table 5.3). Again, there was positive correlation between all the variables in both *with-leaves* and *without-leaves* conditions, $r \geq 0.657$, $n = 21$, $p < 0.001$. Based on the range in the light transmission and forage variables, a linear relationship was established between light and the forage parameters in *with-leaves* ($R^2 \geq 0.76$) and in *without-leaves* ($R^2 \leq 0.62$) by adjusting the data to a linear function (Tables 5.3 & 5.4). A scatterplot summarizes the results (Figures 5.7 a-j). Overall, light transmission was strongly correlated with DM yield, CP, ADF, NDF and ME in *with-leaves* ($r \geq 0.872$) but only moderately correlated with these variables in *without-leaves* ($r \leq 0.657$).

Table 5.4: Summary of the linear relationship between solar radiation and forage parameters

Parameters	Solar radiation (%)			
	R ²	SE	F-value	P-value
<i>With-leaves</i>				
DM Yield (g DM m ⁻² d ⁻¹)	0.81	0.44	85.36	.000
CP (g m ⁻² d ⁻¹)	0.86	0.08	112.01	.000
ADF (g m ⁻² d ⁻¹)	0.79	0.14	72.80	.000
NDF (g m ⁻² d ⁻¹)	0.80	0.26	79.80	.000
ME (MJ g m ⁻² d ⁻¹)	0.76	0.05	60.07	.000
<i>Without-leaves</i>				
DM Yield (g DM m ⁻² d ⁻¹)	0.62	0.26	31.02	.001
CP (g m ⁻² d ⁻¹)	0.58	0.05	27.20	.001
ADF (g m ⁻² d ⁻¹)	0.62	0.75	30.70	.001
NDF (g m ⁻² d ⁻¹)	0.61	0.13	30.89	.001
ME (MJ g m ⁻² d ⁻¹)	0.43	0.02	14.44	.001

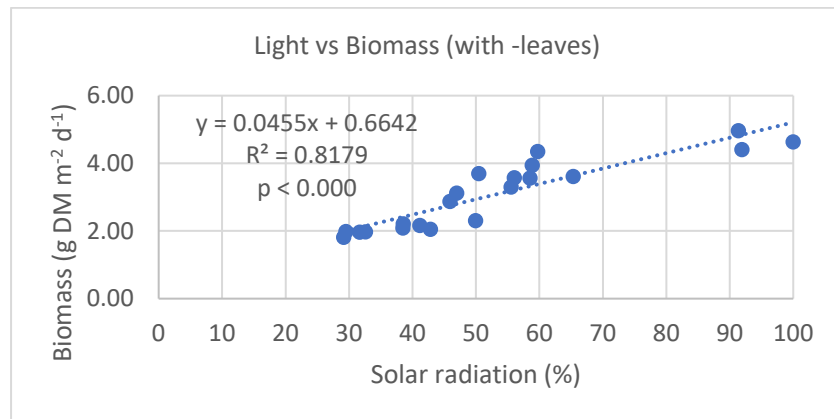


Figure 5.7 (a)

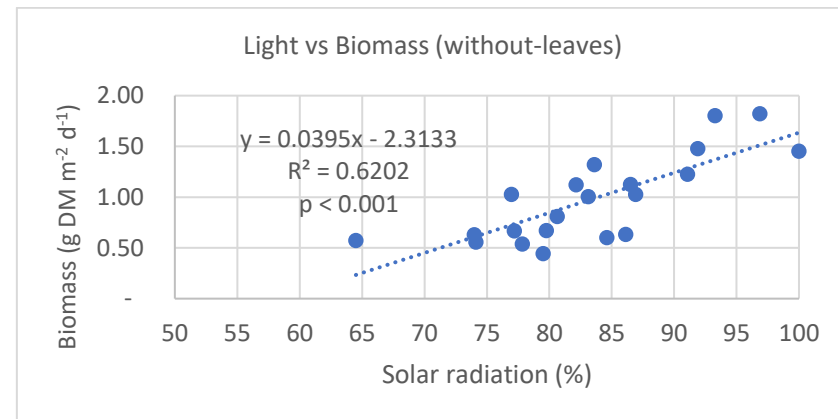


Figure 5.7 (b)

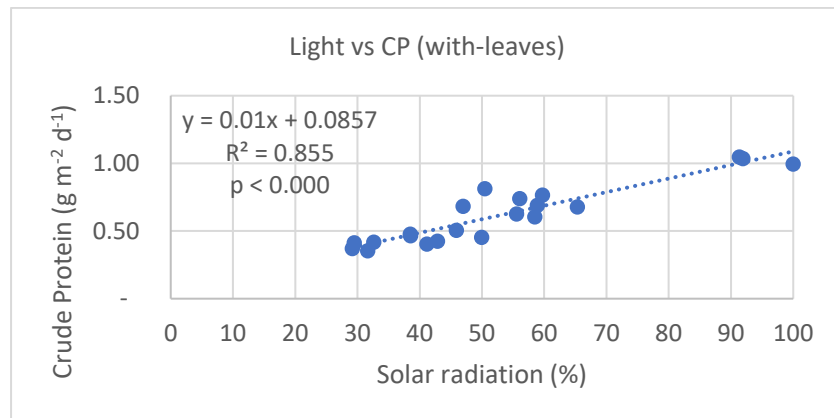


Figure 5.7 (c)

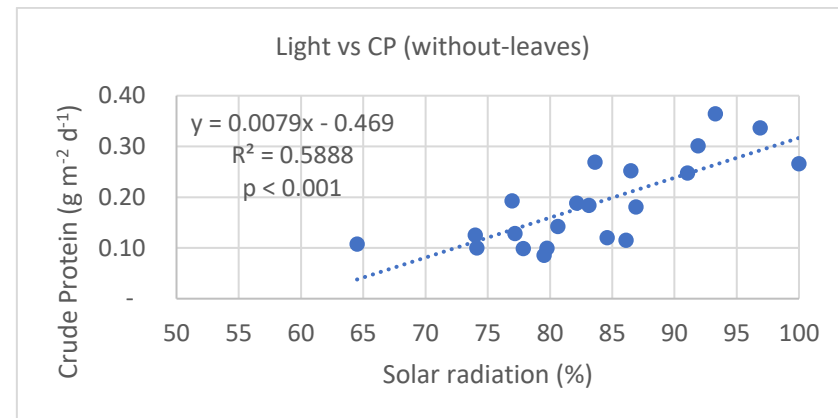


Figure 5.7 (d)

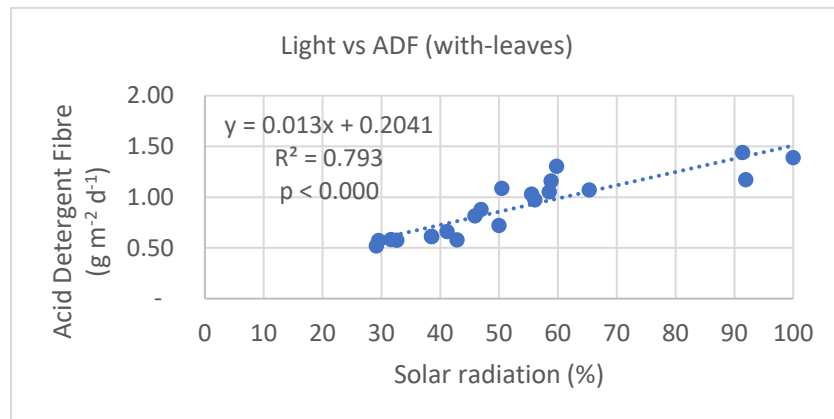


Figure 5.7 (e)

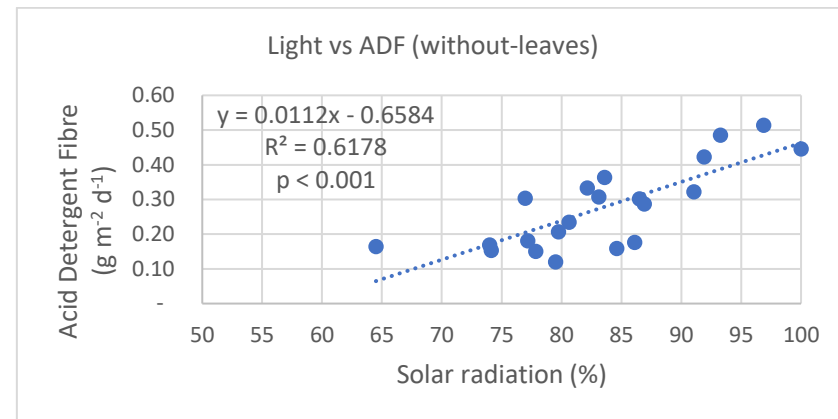


Figure 5.7 (f)

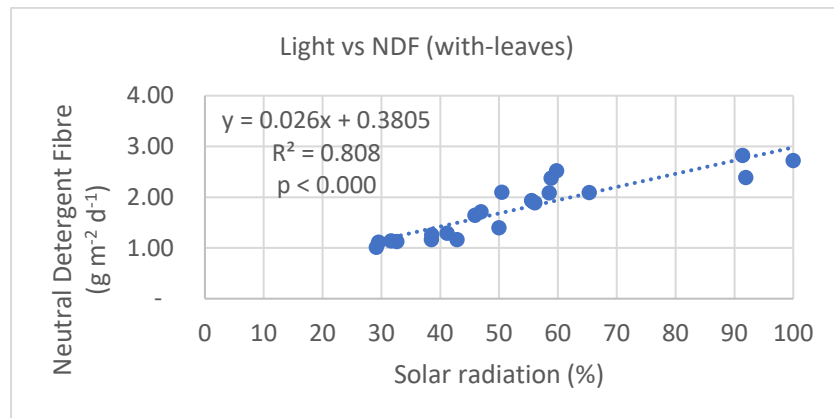


Figure 5.7 (g)

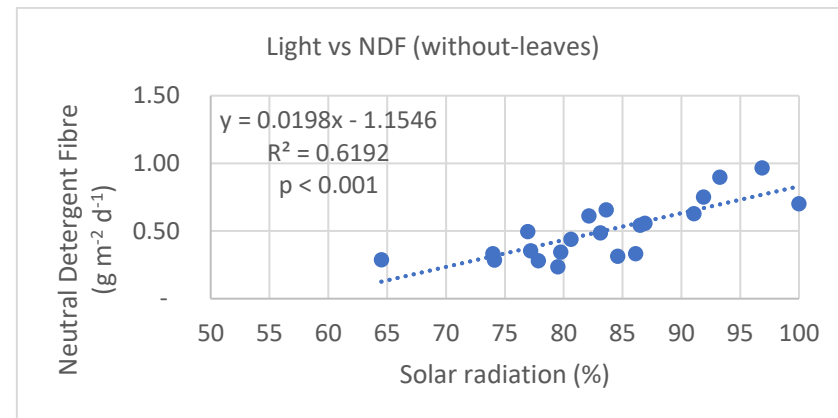


Figure 5.7 (h)

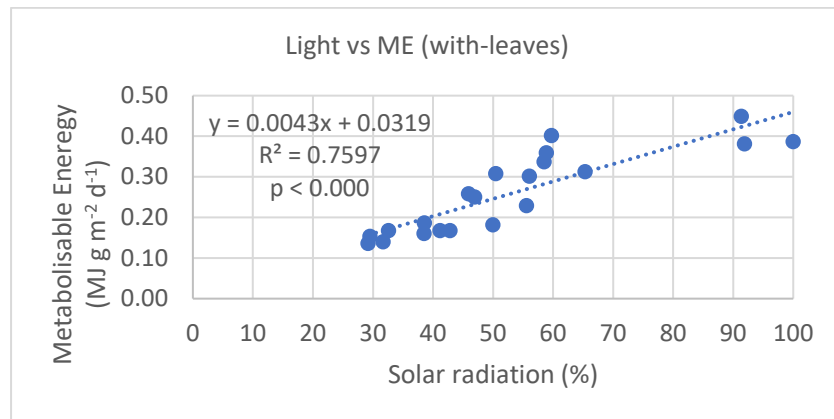


Figure 5.7 (i)

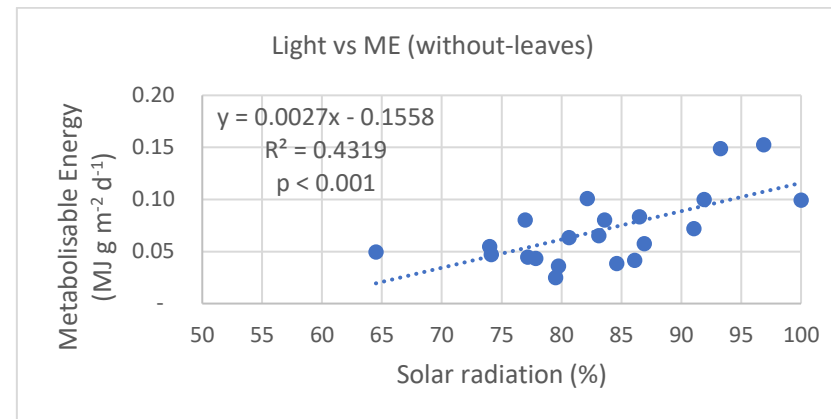


Figure 5.7 (j)

Figure 5-7: (a to j). Scatter plots showing relationships with linear regression between solar radiation incident upon the pasture yield and quality variables as a fraction of above tree canopy in *with-leaves* and in *without-leaves* condition

5.4. DISCUSSION

The light availability at ground level, measured at varying canopy gap levels created by purposive thinning of red alder blocks, correlated significantly ($P < 0.001$) with the observed changes in pasture productivity in the present study. Results show that pasture DM yield increased significantly ($p < 0.05$) with increasing solar transmission (Figures 5.5), and with increasing distance from each grazing exclusion cage to the nearest tree (Figures 5.6). Furthermore, pasture DM yield and pasture quality parameters varied significantly ($p < 0.05$) between the *with-leaves* condition and the *without-leaves* condition (Figure 5.7).

Productivity of pasture have most often been related to competition with the overstory trees for light, soil water or nutrients. Studies have shown that, among those environmental factors, light obstruction by tree canopy has been the main driving factor in most temperate forest ecosystems (Sigurdsson *et al.*, 2005; Anderson *et al.*, 2001; Angelmark *et al.*, 2001; Peterken, 2001; Nygaard and Odygaard, 1999; Stone and Wolfe, 1996). Those authors have observed that more open canopy (less shade) would lead to both increase in pasture productivity and diversity. Again, light availability at ground level, measured as canopy gap fraction, has been found to be the factor that correlated best with the observed changes in ground vegetation biomass and composition (Gonzalez-Hernandez *et al.*, 1998; Sigurdsson *et al.*, 2005).

The variability in the pasture productivity and quality parameters observed in this study has been associated with solar transmission intensity and duration (Photoperiod), and leaf fall (deciduous nature of red alder tree), and the distance of each grazing exclusion cage from the

nearest tree. The with-leaf period corresponded to the summer months (April to October) when the alder trees are with leaves, and temperature and radiation levels are the highest and precipitation is close to moderate level while the without-leaf period corresponded to the winter months (November to March) when the trees are without leaves and temperature and radiation levels are lower (Figures 1.2 & 5.4). Total annual precipitation at the experimental location in 2013 (892 mm) was much higher than that in 2014 (607 mm).

Again, both light intensity and length of day are of importance. In the United Kingdom, photoperiod (the duration of sunshine) varies broadly with the time of the year, with long periods of daylight during the with-leaves condition and short periods of daylight during the without-leaves condition. The length of day is longer in the 7 months of with-leaves condition than in the 5 months of without-leaves condition. Thus, the rate of daily growth is greater in with-leaves condition with longer hours of sunlight, and slower in without-leaves conditions with shorter hours of sunlight. Variation in photoperiod can influence induction of reproductive development of many forage species, which affects forage quality indirectly by decreasing leaf production and increasing production of more stem material in grasses, resulting in higher NDF concentrations (Buxton, 1995). Photoperiod can signal the appropriate time for the transition from vegetative growth to reproductive development, modify the rate of reproductive growth once established, and trigger changes in the rate of leaf area expansion and of dry-matter production which are not necessarily related to reproduction (Hay, 1990; Buxton, 1995). Buxton, (1995) noted that both day length and solar intensity affect morphology, growth, flowering, and maturity of forages, and that at the appropriate photoperiod level, plant

development changes from vegetative growth to reproductive phase. The reproductive expression is enhanced as the photoperiod length is increased. The author gave example by citing Heide (1985) who noted that average stem height of timothy grass increased three-fold as photoperiod increased from 8 to 24 hours. Apart from the effects on flowering, long photoperiods cause high forage quality because of greater photosynthetic activity, which in turn increases soluble sugars that dilute the NDF. Again, long photoperiods usually alter plant morphology, increase yield, increase shoot/root ratios, decrease leaf/stem ratio, and dilutes CP in herbage (Deinum *et al.*, 1981; Juan *et al.*, 1993; Buxton, 1995).

Benavides *et al.* (2008) observed that several studies conducted under deciduous trees have shown seasonal variation in pasture yield because of leaf fall. A study conducted by Douglas *et al.* (2006) over a 3-year period estimated that the average biomass accumulation of swards beneath a stand of *Populus* spp. at 25–100 stems ha⁻¹ and aged 8–11 years was 23% less than open pasture. Benavides *et al.* (2008) noted that this figure nevertheless varied seasonally with differences being greater in spring, summer and early autumn and more similar during the leafless period in late autumn and winter (Benavides *et al.*, 2008).

Ball *et al.* (2001) have well summarized the effects of precipitation and temperature on forage. Schulke (2004) observed that trees can affect the understory pasture as a result of rainfall interception, shading, root competition (for nutrients and moisture), microenvironment changes, effects on soil condition (soil structure), and nutrient cycling (Schulke, 2004). In a trial conducted in north-west Spain, with similar climatic conditions to the United Kingdom,

Rozados-Lorenzo *et al.* (2007) compared pasture production under three evergreen species with that under three broadleaved species. They reported that higher yields were achieved under the broadleaved species because of their lack of foliage at the beginning of spring promoting increases in pasture production.

Therefore, it can be said that level of exposure to solar radiation (the level of shade) is a significant factor determining the productivity of pastures. This result is in agreement with the findings of previous research studies. This result is comparable to that of Knowles *et al.* (1999) where a strong relationship ($R^2 = 0.89$) was shown to occur between measured pasture yield and predicted canopy closure. In a research studies in Scotland, UK Sibbald *et al.* (1994) found that herbage production decreased with increased shading (or attenuation of full sunlight) when precipitation and temperature did not limit herbage growth. Lin *et al.* (1999) also showed a reduction of yield because of increased shade for orchard grass, ryegrass and white clover. This is also in agreement with Peri *et al.* (2007) and Neel *et al.* (2008). Tree shade limits pasture photosynthesis (Rao *et al.*, 1998; Montard *et al.*, 1999; Sharrow, 1999; Esquivel-Mimenza *et al.*, 2013) particularly in C4 species, such as *Brachiaria brizantha* where the rate of photosynthesis tails off at about $1500 \mu\text{mol m}^{-2} \text{s}^{-1}$, which is a bright but not fully sunny day in summer. Pasture grown in areas with lower light levels have been found to be shorter, had fewer numbers of tillers and produced less dry matter compared with treatments with higher levels of radiation (Belesky, 2005; Peri *et al.* (2007). Ehrenreich and Crosby (1960) reported higher production for understory plants within a hardwood forest as crown cover decreased. Production greatly decreased when canopy cover increased. However, the extent of biomass

reduction observed largely depends on the interception of solar radiation caused particularly by the tree species. Decreasing irradiance reduces the growth of pasture species (Smith and Whiteman 1983; Shelton *et al.*, 1987) and influences the outcome of relationships. It has also been noted that light interception by the trees is one of the main factors affecting the productivity of pasture in silvopastoral systems, especially when water and soil nutrients are freely available (Ong *et al.*, 1996; Power *et al.*, 2001).

Furthermore, this result agrees with tree-removal studies that report that herbaceous forage production increases as trees are eliminated (Pratchett 1978; Walker *et al.*, 1986; Harrington and Johns 1990). Thinning trees generally increases biomass productivity of understorey plants especially when pre-treatment stand density is high (McConnel & Smith 1970; Uresk & Severson 1998; Brockway *et al.*, 2002). Ducherer *et al.* (2013) reported that total understorey biomass increased up to 80% within 3 to 4 years after thinning in the Ponderosa pine and Douglas fir forests. Depending on sites and years, biomass production of one or more functional groups, such as forb, shrub, or graminoids, may increase. Uresk and Severson (1998) also reported that eliminating or reducing the overstory in ponderosa pine forests increases understorey biomass production.

Studies of thinning forested environments to create desirable levels of available light have been conducted. While these studies have primarily been designed in an attempt to create favourable light conditions for regeneration of desirable tree species, the information has direct application to creating light levels favourable for the growth of select forages. Garrett *et al.* (2004)

described two popular harvesting practices for creating light levels favourable for the growth of forages in hardwood forests: the group selection and the shelterwood methods. The group selection method creates patches of high light intensity, while the shelterwood method is designed to create a more even distribution of light throughout the forested understory. They further recounted that in young immature stands, release thinning such as timber stand improvement, crop tree, and deferment cuts, will provide increased light levels for forage production while at the same time improving the growth of trees identified for retention (Garrett *et al.*, 2004). Dey and Parker (1996) reported that with a shelterwood harvest in an oak stand, removing 43 and 77% of the basal area increased light intensities to 35% and 65%, respectively. Other studies have discovered that up to 50% of the basal area of hardwood forests may need to be cut to increase light levels to 35- 50% of that found in the open (Sander 1979, Marquis 1988, Dey and Parker 1996).

The issue of whether thinning improve or degrade understory pastures productivity and quality is sometimes contentious. Studies have indicated that forage production is often reduced by trees that compete with understory herbaceous species for water, nutrients, and light (Kay and Leonard 1980; Monk and Gabrielson 1985; Pieper 1990; Burrows *et al.*, 1990; Belsky 1992). As a result, pasturelands are cleared of trees by expensive mechanical and chemical techniques (Beisky 1992). However, other studies have indicated that trees may increase forage production in areas of low tree density, moderate or high soil fertility, and low rainfall (Belsky *et al.*, 1989; 1992; McClaran and Bartolome 1989; Burrows *et al.*, 1990; Wilson *et al.*, 1990; Belsky and Amundson 1992).

In some instances, full sunlight may not be required to maximize pasture growth. Many understory pastures will need only about 10% of full sunlight to reach a state of growth where photosynthesis exceeds respiration, and will reach light saturation at 50% (C3 plants) and 85% (C4 plants) of full sunlight, respectively (Gardner *et al.*, 1985). The light intensity within mature hardwood forests is typically lower than 20% and may be as low as 1% (Dey and MacDonald 2001). While the opening up of some proportion of the canopy can increase the intensity and duration of light reaching the forest floor, and thereby improve the growth of forage crops, the relationship of the canopy opening to available light is not linear, with an estimated residual stocking density of 30% required to provide light levels of 50% of open values (Sanders 1979).

The role of gap size has also been quantified in which direct relationship has been found to exist between gap size/canopy gap level and amount of light penetration (Dey and MacDonald, 2001; Minckler 1961; Garrett *et al.*, 2004)). Dey and MacDonald (2001) reported that gaps have little effect on available light when openings are smaller than 0.04 ha and larger than 0.4 ha in size. Similarly, most studies of light intensity measurement that emphasised the reading of light intensity at the centre of the gap have recognised the decreased availability of light when moving from gap centre towards the edge (Minckler 1961).

Garrett *et al.* (2004) observed that for the silvopasture practice and growth of forages, the uneven distribution of light, such as may be created with gaps, is not desirable, that it is crucial

to understand the important role slope and aspect may play in determining appropriate residual densities when thinning a forest stand. They further noted that south-facing slopes that naturally receive greater solar exposure should, logically, have higher densities of trees than north-facing slopes that are predisposed to less direct sunlight (Garrett *et al.*, 2004).

Light intensity may not be the only limiting factor influencing pasture yield and quality in this study. The addition of extra N from red alder, an N-fixing tree, would certainly have exerted significant influence on the pastures. Other influences such as moisture and nutrient competition, allelopathic effects and smothering are also at work under the trees.

In a trial to compare the impacts of shade duration on pasture production with deciduous and evergreen tree species Power *et al.* (2001) reported that at low levels of shade (<40%), pasture relative yields under a nitrogen-fixing tree, *Acacia melanoxylon*, were greater than relative yields under corresponding levels of artificial shade. They further noted that for a deciduous tree species to be effective in modifying pasture yield during the leaf-free period, they must be leaf-free for longer than 4 months.

The usual practice is to present forage overstory/understory models as linear relationships (Joyce and Mitchell, 1989; Mitchell and Bartling, 1991). However, some studies indicate that the true relationship over the life of a silvopasture is most likely curvilinear with little effect until tree canopy exceeds 30–50% coverage (Krueger, 1981; Joyce and Mitchell, 1989), followed by a fast decline in understory production as tree canopies merge.

It is apparent that environmental conditions in spring and summer enhanced pasture productivity and quality. Development of a balanced diet of silvopastoral systems requires sound understanding of the trends in herbage DM yield and forage quality throughout the year. Trend similar to the model presented in Figure 5.4 was obtained in other studies conducted in other temperate regions (Demanet *et al.*, 2015; Teuber, 2009), though the scale of the changes may differ because of differences in site conditions. Mathematical expression of forage production enables the prediction of periodic growth of pastures, which are useful in the determination stocking rate and the development of feed balances, adjustment of diets to regulate demand and supply processes, conservation programs and surpluses.

The linear model employed in this study made it possible to quantify the trend of daily, monthly, seasonal and locational averages of pasture growth and dry matter yield over time with high level of significance ($p \leq 0.0001$). The frequency of grazing is determined by variations in the seasonal growth of pasture (Holmes *et al.*, 2002). Demanet *et al.* 2015 and Holmes *et al.*, 2002 observed that under optimal conditions of use, rotation lengths may vary between 15 and 35 days in spring and 25 and 90 days, in autumn and winter.

General information on the feed quality of a range of typical forages and animal dietary requirements for ruminant animals are presented in Appendix 5.3 and 5.4, respectively. Forage quality is a direct reflection of the ability of a given forage to meet the nutrient needs of the

consuming animal. The concentrations of the mixed forage quality parameters and the dietary maintenance requirements for sheep are within the acceptable range (AFRC 1993).

Forage quality is a direct reflection of an animal's ability to consume, digest, and assimilate essential nutrients contained within the feed. It has been estimated that about 50-75% of this ability of a given forage to meet the nutrient needs of the consuming animal is related to intake, 25-50% is related to digestibility, and 5-15% is related to metabolic efficiency.

Dry matter represents everything contained in a feed sample except water; this includes protein, fibre, fat, minerals, etc. In practice, it is the percentage of the feed that is not water (moisture). DM increases with plant age when harvested. The lower the DM, the more moisture is present, and the lower is the nutrient density in the fresh feed. Also, high moisture may decrease the storing quality of a feed (through moulding) unless it is made into silage. When fresh forages and grasses make up the bulk of the diet, a large amount of water is consumed, which could limit intake of energy and protein sources.

The crude protein content of a feed sample represents the total nitrogen (N) in the diet, which includes a mixture of true proteins, amino acids, nitrate, and non-protein nitrogen, such as urea and ammonia in a forage. Because N is an integral part of any amino acid, non-protein nitrogen has the potential to be utilized for protein synthesis by rumen microorganisms. The protein in a forage is important since protein contributes energy, and provides essential amino acids for rumen microbes as well as the animal itself. Some protein fractions are more digestible than

others, but in general the higher the protein level, the more digestible is the feed. The more protein that comes from forage, the less supplemental protein is needed. However, most nutritionists consider energy value of forages to be more important than CP. The dietary CP requirement of sheep and lamb are 9 -12% and 11 – 14%, respectively (NRC, 1996) (Appendix 5.4). Across all treatments and seasons, the concentration of CP in herbage in this study exceeds the requirements of growing, finishing and lactating sheep (Table 5.2).

The ADF value refers to the percent of the least digestible parts of cell wall components of cellulose, lignin, silica, insoluble CP, and ash. ADF content increases as the plant matures and is generally higher in legumes than grasses of the same age. Lignin is indigestible, whereas cellulose can be digested by the rumen microbes or bugs. Generally, ADF has been used to predict digestibility and thus energy content of a forage. The lower the ADF content, the higher the digestibility and the higher the energy value of a forage. Forages with higher ADF are lower in digestible energy than forages with lower ADF, which means that as the ADF level increases, digestible energy levels decrease. For any given sample, ADF will be lower than NDF content and the difference between the two reflects the amount of hemicellulose present. In the present study, mean dietary ADF values exceeded recommended levels of 20 - 25% for sheep (Table 5.2 and Appendix 5.4) indicating that voluntary feed intake was not limited by low fibre content.

The NDF value is the percent of total fibre in the feed containing all cell wall components including cellulose, hemicellulose, lignin, silica, insoluble CP, and ash. It is the fibre in the diet that stimulates rumination, chewing, and saliva production. The NDF of a forage is inversely related to the amount that animals are able to consume; thus, as the NDF content of a forage increases, the amount of the forage a ruminant will consume generally decreases. As a result, NDF is often used in formulas to predict the dry matter intake. In the present study, mean dietary NDF values exceeded recommended levels of 25 - 35% for sheep (Table 5.2 and Appendix 5.4) indicating that voluntary feed intake was not limited by low fibre content.

The ME content is the energy in feed, minus energy in faeces, urine and methane that arise from digestion (Waghorn 2007). In ruminants, the ME of a diet is calculated from the heat of combustion of feed eaten, minus the heat of combustion of faeces, urine and methane, derived from the feed eaten. It is the useful energy made available through the process of digestion and the value is expressed as a proportion of the dry matter (MJ/kg). Metabolisable energy of forage is a superior measure to dry matter for estimating animal production potential, because it represents the energy available to the animal for maintenance and production. For a given feed source, the ME declines as the level of feeding increases, due to variation in the amounts of energy lost in faeces, urine and methane. The dietary ME requirement of sheep is 8 -10% (Appendix 5.4). Across all treatments and seasons, the concentration of ME in herbage in this study exceeds the requirements of growing, finishing and lactating sheep (Table 5.2).

5.5. CONCLUSION

Based on the findings of this study it can be concluded that thinning opened the canopy, and decreased canopy area of alder in a silvopastoral system resulting in increased pasture production by increasing transmitted light. Pasture DM production increased with increasing solar transmission as well as with increasing distance between each grazing exclusion cage and the nearest tree (Figure 5.5, 5.6 & 5.7). Changes in the concentration and availability of CP, ADF, NDF and ME with solar transmission can be explained by changes in pasture production. The variability in the pasture productivity and quality parameters observed may also have been influenced by climatic conditions, especially precipitation and air temperature, solar transmission intensity and duration (photoperiod), and leaf fall.

Light intensity and climatic conditions may not be the only limiting factors influencing pasture yield and quality parameters in this study. The addition of extra N from red alder, an N-fixing tree, would certainly have exerted significant influence on the pastures. Other influences such as moisture and nutrient competition, allelopathic effects and smothering are also at work under the trees.

The forage quality parameters, CP, ADF, NDF and ME followed similar trend as the DM yield, decreasing with reduction in solar transmission, canopy gap size and temperature. In addition, concentrations and availability of these parameters were greater in with-leaves than in without-leaves growing seasons in response to variation of photoperiod (the duration of sunshine/day length) in the United Kingdom. The practical significance of these results is that reduced

competition for light after thinning the alder trees could improve the grazing productivity of the understorey pasture.

Chapter 6 : BIOECONOMIC POTENTIAL FOR SILVOPASTORAL AGROFORESTRY SYSTEM IN NORTH WALES

6.1. INTRODUCTION

A silvopastoral system is a farming system that permits the coexistence of trees, pasture and animals on the same piece of land, combining long-term production of timber and fuelwood with yearly production of forage and livestock (Bergez *et al.*, 1999). Silvopastoral systems are usually established to provide both timber and livestock products through the combination of pasture and widely spaced trees. Silvopastoral system integrates trees or shrubs with forage and animal production in the same unit of land for efficient utilisation of space, growing season, and growth factors. If properly managed, silvopastoral processes could improve pasture productivity and quality and provide sustained income with the simultaneous production of trees and grazing livestock.

Silvopastoral systems offer a variety of benefits for livestock management. Trees provide shade and wind protection, which reduce heat stress and wind-chill of livestock; performance is improved and mortality reduced. Economic returns from forage/livestock production continue while creating a sustainable system with environmental benefits. Such benefits of silvopastoral systems have been researched in the UK at the UK's SNNE since 1988. Sibbald and Dalziel (2000) reported that, in the UK's SNNE, no significant differences in lamb growth were observed between silvopastoral treatments and the pasture control until up to ten years after

establishment of the sites (Sibbald and Dalziel, 2000). Results of the research at the silvopastoral system experiment at Henfaes have also shown that there was no significant difference in livestock production between silvopastoral treatments and the pasture control during the first six years of the tree establishment phase (Teklehaimanot *et al.*, 2002). To maintain agricultural production sustainably pruning and thinning may be necessary.

Silvopastoral system may be more advantageous to farmers than non-agroforestry farm management system as the tree-based grazing system could be more resilient to fluctuations in market price and other bioeconomic variables, and could provide the farmer with reasonable level of financial profit and cash flow. Presently, little is known about the economic opportunities and risks associated with operating silvopastoral agroforestry enterprises in Wales. It would be interesting to know whether a silvopastoral agroforestry system could produce better financial and environmental outcomes compared with pure forestry and extensive grazing-only systems.

Although agroforestry experiments in the UK in the 1990s and the success of silvopasture in Northern Ireland have enhanced the biophysical and economic awareness of interactions in agroforestry systems, the commercial acceptance of these systems have remained a mirage, principally because of lack of will on the part of the Forestry Commission and other UK governmental units to promote agroforestry systems and the difficulty on the part of farmers to access some EU agroforestry incentives and research programmes (Burgess, 2011). The financial viability is the major consideration in farmers' adoption of any land use investment

options such as arable farming, agroforestry and forestry, as relative costs and returns are greatly influenced by local differences in yields, prices and government grants (Graves *et al.*, 2010).

This study, therefore, will seek to appraise the bioeconomic potentials of silvopastoral agroforestry systems in the lowlands of North Wales. The results of this research will contribute directly to the goals of the UK's Silvopastoral Network Experiment through a better understanding of the bio-economic potential for silvopastoral agroforestry development in the lowland areas of North Wales. The findings will also help promote greater awareness of the economic value of trees in extensively grazed landscapes in North Wales and should assist future investment decisions by landowners. Furthermore, the study is in line with the Wales "100,000 Hectare Challenge" in which the Welsh Government seek to increase the woodland cover of Wales from the present 14 percent to 20 percent over the next 20 years, with a view to achieving a net increase in carbon sequestration to combat climate change (Osmond and Upton, 2012). To achieve this goal will require planting an average of 5,000 hectares of woodland a year. These goals are particularly relevant in a UK context since there is a general perception amongst landowners that trees compete strongly with pasture and livestock production and are considered an economic liability rather than as a potential asset.

6.1.1. Objectives

The aim of this study is to evaluate the bioeconomic potentials of temperate lowland silvopastoral agroforestry systems in the United Kingdom.

Specifically, the study will conduct a bio-economic analysis to evaluate conventional grazing systems against preferred silvopastoral agroforestry systems for temperate lowland system in North Wales, United Kingdom.

This study is based on the premise that silvopastoral agroforestry system is a viable undertaken compared to a pasture or forestry system,

6.2. MATERIAL AND METHODS

6.2.1. The scenarios

The study compared the economic viability of three farm management systems using discounted cash flow analysis and national costs and prices for both livestock and tree products.

The following three farming enterprise scenarios were considered:

Scenario 1: Conventional lowland sheep spring lambing system - treeless sheep-grazed pasture farm

Scenario 2: Forestry woodland system – 2500 stem ha⁻¹ red alder forestry woodland with no grazing.

Scenario 3: Silvopastoral agroforestry system – 200 stem ha⁻¹ red alder (*Alnus rubra* Bong) trees planted into sheep grazed pasture farm.

These farm management systems are the major components of the UK's National Network Experiments at six sites established across the country investigating the potential of silvopastoral agroforestry on UK farms (Teklehaimanot et al. 2002). Although average farm size in the UK is 57 hectares, this study has chosen one hectare as a basis for comparison in this appraisal. It is pertinent to emphasise that there was a financial optimization of the scenarios under consideration with a view to finding the system that would give the highest net benefit. Major tasks undertaken included the development of the farm budgets and cash flow

plans. These involved detailed listing of all possible sources of revenues, variable costs and fixed costs including their time dimension.

6.2.2. Sources of financial data

Farm establishment and maintenance costs, and tree growth data were secured from the Henfaes Silvopastoral Network Experiment while prevailing costs, returns, and price data were adapted from the John Nix Farm Pocketbook (46th Edition, 2016) (Redman, 2015), the Farm Management Handbook (36th Edition, 2015/16) (SAC, 2014), the Department for Environment, Food & Rural Affairs (DEFRA) statistics, the Forestry Commission, and from various relevant online resources. It is pertinent to note that these data are only approximations of real individual farms present circumstances and conditions.

6.2.3. Phases of economic analysis

The phases of the economic analysis were:

- Definition of costs, revenues
- Development of farm budgets for the scenarios
- Construction of cash flows based on the farm budgets
- Discounting cash flows using prevailing lending discount rate
- Analysis of the discounted cash flow using two measures of project worth, the net present value (NPV) and the annual equivalent value (AEV)
- Sensitivity analysis

The study focused mainly on enterprise budget, financial projection, and valuation measurements framework to arrive at overall evaluation of the systems. A 30-year projection on a one-hectare basis was made for each of the three models. All the costs and revenues expected for each system were detailed in the enterprise budgets which were later combined into cash flow plans with time dimension added. The methods of enterprise budget process suggested by earlier researchers (Godsey 2008; Soeleman *et al.*, 2014) were employed. The comparisons were initially made without consideration for government grant/subsidy.

6.2.4. Farm enterprise budget

The farm budgets presented below describes the costs and returns associated with each of the three scenarios and include information on the returns generated from the enterprises and costs such as livestock/pasture management costs, tree planting, maintenance, harvesting, labour, power and machinery, rent and finance costs, and overhead (Chase, 2006; Doye, 2007).

6.2.5. Scenario 1: Conventional lowland sheep spring lambing system

Analysis of the conventional treeless sheep grazing pasture system exemplified the economic performance of a typical mid-sized commercial family sheep farming enterprise commonly found in the UK. This scenario modelled the economic viability of lowland spring lambing system, organised to maximise the utilisation of pasture and, where lambs were sold off grass. The system generated income from the annual sale of lambs, wool, and cull ewes and rams. Income from lamb sales was also considered as income earned from pasture since pasture was used for grazing the sheep in order to sell the meat of lamb.

6.2.6. Conventional lowland sheep spring lambing system assumptions

The general assumptions of the net cash flow for the conventional lowland sheep grazing system are as follows:

1. The performance data, costs of inputs and outputs, and prices of product sales are known and remain constant over the 30-year projection period.
2. All costs were incurred from year 1 through to year 30 while revenues were spread from year 2 to year 30 (model assumption).
3. Mixed breeds of mature 100 ewes and 2 rams were introduced into improved permanent pasture in year one at a stocking rate of 10 ewes per hectare and 1 ram per 45 ewes (Nix, 2016).
4. Average lamb sale liveweight of 41kg was assumed. 155 lambs were sold per 100 ewes put ram at a market price of £66.7/head (Nix, 2016).
5. 20% of the ewe flock is culled yearly at £65 each, allowing for 4% mortality (Nix, 2016).
6. 23% of ewes are bought or home-raised at £145. Ram was purchased for £480 per head and sold at £75 after 3.5-year life. (Nix, 2016).
7. Variable costs for veterinary and medicine include allowance for wormer, vaccine, treatments for flies and feet.
8. Miscellaneous costs include contract shearing @ £1.49/ewe, scanning £1.00/ewe and ewe and lamb tags £1.49/ewe, carcass disposal £0.56/ewe, straw £1.47/ewe, minerals and licks etc. £2.05/ewe, marketing levy and transport £6.10/ewe (Nix, 2016).

9. Forage costs are based on improved permanent pasture (£147/hectare) (Nix, 2016).
10. Fencing costs include stock proof post and 4-barb @ £5/metre, dug in rabbit proof up to £6.90/metre (per side), and post and 3 rails £13.75/metre (Nix, 2016).
11. Fixed costs include labour (paid and unpaid), running costs of power and machinery, rent and finance costs, general overhead costs and farm maintenance (Nix, 2016).
12. The market is assumed to be perfect, the discount rate of 3.5% (H.M Treasury, 2003) remains constant over the projection period, and cash flows are expressed in British pound sterling.
13. Grants/subsidies were not included in the analysis.

The assumptions and budgetary requirements for livestock production system are summarised in Table 6.1.

Table 6.1: Budget for Livestock Production

	Performance data	
	Farm size (ha)	10
	Number of ewe	100
	Number of ram (1 ram to 45 ewes)	2.22
	Stocking rate (ewes per hectare)	10
	Ewe culling rate	20%
	Ram Replacement Rate	23%
	Fleece weight per ewe (kg)	2
	Lamb sold per 100 ewes put to ram	155
	Average price per lamb (£)	£66.70
Year	Revenue:	Total (£)
2 to 30	Lamb Sales (£66.7 per 1.55 lamb	£29,981.65
2 to 30	Wool (2kg/ewe @ £1.20/kg)	£696.00
2 to 30	Cull Ewes and Ram	£3,915.00
	Total Revenue	£34,592.65
	Variable costs:	
1 to 30	Ewe and Ram Replacement @ 23%	£10,920.00
1 to 30	Concentrates @ £13.30/head	£3,990.00
1 to 30	Vet and Medical Fees @ £10.90/head	£3,270.00
1 to 30	Miscellaneous @ £14.10/head	£4,230.00
1 to 30	Forage Cost (inc. bought in forage and keep)	£4,410.00
	Total Variable Costs	£26,820.00
	Fixed costs:	
1 to 30	Labour (paid)	£1,050.00
1 to 30	Labour (unpaid)	£14,100.00
1 to 30	Power and machinery (running costs)	£6,660.00
1 to 30	Miscellaneous fixed costs (Overheads)	£3,150.00
1 to 30	Rent and Finance Costs	£2,250.00
	Total Fixed Costs	£27,210.00
	Less unpaid labour	£13,110.00
	Total Costs	£39,930.00
	Net Cash Flow	-£5,337.35

6.2.7. Scenario 2: Forestry system

This scenario is reflective of the farm woodland control of the National Network Experiment or the shift from the traditional upland conifers establishment towards the re-establishment of broadleaved woodland in the lowlands of the UK using red alder (Teklehaimanot *et al.*, 2002).

6.2.8. Forestry woodland system assumptions

The general assumptions of the net cash flow for the forestry woodland system are as follows:

1. It is assumed that Red alder was established at 2m x 2m spacing and stocking density of 2500 stem ha⁻¹, growing at a maximum annual increment of 12 m³ per hectare per year, i.e., at yield class of 12, and maintained on a 30-year firewood and sawlog rotation with no grazing.
2. Tree establishment costs include site preparation, tree planting, protection, and maintenance. Harvesting costs include the costs of marking up for thinning and final harvesting. While fixed costs include labour, running costs of power and machinery, general overhead costs, and rent and finance costs.
3. Intermediate thinning was done at age 15 while final harvest occurred in year 30.
4. All operations were assumed to be executed by contractors and all timber sales were done through standing sales (both thinning operations and final harvest).
5. Harvesting revenues were estimated by assigning volume yield to fuelwood and sawlog and multiplying by assumed flat standing sale price of £30/m³ for both fuelwood and sawlog.
6. Grants/subsidies were not included in the analysis.

7. The market is assumed to be perfect, the discount rate of 3.5% (H.M Treasury, 2003) remains constant over the projection period, and cash flows are expressed in British pound sterling.
8. Revenues from the sale of firewood and sawlog, establishment costs, operating costs, and fixed costs are shown in the budgetary requirements for 2500 stem ha⁻¹ red alder block (Table 6.2).

Table 6.2: Budget for 2500 stem ha⁻¹ red alder Block

Year	Phase / Activities	Total (£)
	Revenues:	
15	Thinning (standing sale) (312 m ³ @ £30/m ³)	£9,360.00
30	Final harvest (standing sale) (506 m ³ @ £30/m ³)	£15,180.00
	Total Revenues	£24,540.00
	Variable Costs	
	1. Establishment:	
1	a. Site Preparation	
1	(i). Fencing (materials @ £1.48/m and labour @ £15/hr)	£592.00
1	(ii). Ground preparation @ £99.00/ha	£99.00
1	(iii). Marking out/Staking @ £0.04/spot x 2500 spots	£100.00
1	b. Tree Planting	
1	(i). Purchase of 2500 red alder plants @ £0.40/plant	£1,000.00
1	(ii). Spot spraying of herbicides @ £0.10/spot x 2500 spots	
1	(1-metre-wide spot)	£250.00
1	(iii). Hand planting 2500 plants @ £0.60/plant (including cost	
1	of guards/shelters)	£1,500.00
1	c. Tree Protection	
1	(i). 2500 tree shelters/plastic tubes plus ties @ £0.55 each	£1,375.00
1	(ii). Anchor peg plus tie @ £0.30 each x 2500 trees	£750.00
	2. Maintenance:	
	a. Beating up	
2, 3	(i). Operation @ £300/ha	£600.00
2, 3	(ii). Plant supply @ £200/ha	£400.00
	b. Weeding	
1, 2, 3	(i). Spot weeding with herbicide @ £100/ ha	£300.00
	c. Pruning @ £250/ha	£750.00
	3. Harvesting:	
	(i). Marking-up Thinning (312 m ³ @ £1/m ³)	£312.00
	(ii). Marking-up Clear felling (506 m ³ @ £1/m ³)	£506.00
	Total Variable Costs	£8,534.00
	Fixed Costs	
1 to 30	a. Labour	£750.00
1 to 30	b. Power and machinery (running costs)	£2,700.00
1 to 30	c. Miscellaneous fixed costs (Overheads)	£3,000.00
1 to 30	d. Rent and finance costs	£2,250.00
	Total Fixed Costs	£8,700.00
	Total Costs	£17,234.00
	Net Cash Flow	£7,306.00

6.2.9. Scenario 3: Agroforestry system

The general assumptions of the net cash flow for the agroforestry system were:

1. The agroforestry system was established on existing improved permanent pasture consisting of a mixture of perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) and grazed rotationally. There was therefore no reseeding of the pasture.
2. Red alder (*Alnus rubra* Bong.) was introduced into the improved permanent pasture at 5m x 5m spacing and stocking density of 200 stem ha⁻¹ and maintained on a 30-year firewood and sawlog rotation.
3. The trees were pruned at ages 10, 15, and 20 while commercial thinning was conducted at ages 10 and 20 to open up the canopy for greater solar penetration to understory pasture, to improve the growth and value of the block, and to provide periodic income for the farmer (Sibbald, 2006). The block was clear-felled in year 30.
4. Tree establishment costs included those for site preparation, tree planting, protection, and maintenance. Harvesting costs included the costs of marking up for thinning and final harvesting. While fixed costs included labour, running costs of power and machinery, general overhead costs, and rent and finance costs,
5. It is assumed all operations were carried out by contractors and all timber sales were done through standing sales (both thinning operations and final harvest).

6. Harvesting revenues were estimated by assigning volume yield to fuelwood and sawlog and multiplying by assumed flat standing sale price of £30/m³ for both fuelwood and sawlog.
7. Crown area is assumed to be negligible for the first 10 years with no negative impact on pasture productivity, that from year 11 to year 20, crowns increased to occupy 30% of the area (viewed from above) resulting in a loss of 30% of Area Payment by year 20, and that crown growth and crown area remained constant after year 20.
8. The livestock component for the silvopastoral system was simulated using the information for the conventional pasture system of scenario 1. Sheep were introduced into the system in year two to allow time for forage and tree stabilisation. Revenues from sheep production was received from year 2 to 30. The sheep rearing performance data, pricing, revenues, establishment costs, maintenance costs, and fixed costs remain the same as for the sheep production.
9. Government grants/subsidies were not considered in this analysis.
10. The market is perfect, the discount rate of 3.5% remains constant over the projection period, and cash flows are expressed in British pound sterling.

The assumptions and budgetary requirements for agroforestry system are presented in Table 6.3.

Table 6.3: Budget for Agroforestry System

Year	Phase / Activities	(£)/Unit	Total (£)	Year		(£)/Unit	Total (£)
	Revenues:				2. Maintenance:		
2 to 30	Lamb Sales (£66.70 per 1.55 lamb	£1,033.85	£29,981.65	2, 3	(i). Plant supply @ £120/ha + Beating up @	£137.00	£274.00
2 to 30	Wool (2kg/ewe @ £1.20/kg)	£24.00	£696.00	1, 2, 3	(ii). Spot weeding with herbicide @ £100/ha/yr	£100.00	£300.00
2 to 30	Cull Ewes and ram	£135.00	£3,915.00	10, 15, 20	(iii). Pruning	£100.00	£300.00
10	1st Thinning (standing sale) (22 m ³ @ £30/m ³)	£660.00	£660.00		3. Harvesting:		
20	2nd Thinning (standing sale) (25.20 m ³ @ £30/m ³)	£756.00	£756.00	10	(i). Marking-up 1st thinning (22 m ³ @ £1/m ³)	£22.00	£22.00
30	Final harvest (standing sale) (81 m ³ @ £30/m ³)	£2,430.00	£2,430.00	20	(ii). Marking-up 2nd thinning (25.20 m ³ @ £1/m ³)	£25.20	£25.20
	Total Revenue	£5,038.85	£38,438.65	30	(iii). Marking-up clear fell (81 m ³ @ £1/m ³)	£81.00	£81.00
				1 to 30	4. Livestock/Pasture Management	£894.00	£26,820.00
	Variable Costs:				Total Variable Costs	£2,974.20	£29,437.20
	1. Establishment:						
	a. Site Preparation				Fixed Costs:		
1	(i). Fencing (materials @ £1.48/m	£592.00	£592.00	1 to 30	a. Labour	£35.00	£1,050.00
1	(ii). Ground preparation @ £99.00/ha	£85.00	£85.00	1 to 30	b. Power and machinery (running costs)	£222.00	£6,660.00
1	(iii). Marking out/Staking @ £0.04/spot x 200 spots	£8.00	£8.00	1 to 30	c. Miscellaneous fixed costs (Overheads)	£102.00	£3,060.00
	b. Tree Planting			1 to 30	d. Rent and finance costs	£75.00	£2,250.00
1	(i). Purchase of 200 red alder plants @ £0.60/plant	£120.00	£120.00		Total Fixed Costs	£434.00	£13,020.00
1	(ii). Spot spraying of herbicides @ £0.10/spot x 200 spots	£20.00	£20.00		Total Costs	£3,408.20	£42,457.20
1	(iii). Hand planting 200 plants @ £0.60/plant	£120.00	£120.00				
	c. Tree Protection				Net Cash Flow	£1,630.65	-£4,018.55
1	(i). 200 tree shelters/tubes plus ties @ £0.65 each	£130.00	£130.00		NPV		-£4,254.58
1	(ii). Supporting pressure-treated wooden fence posts	£480.00	£480.00		AEV		-£231.33
1	(iii). Anchor peg plus tie @ £0.30 each x 200 trees	£60.00	£60.00				

6.2.10. Growth and yield data

As yield model specific for red alder especially for a Silvopastoral National Network Experiment setting in the UK is non-existent at present, the best estimate of harvest yields per hectare over the long term were obtained by using inventory data from Henfaes SNNE and applying the biomass equation developed for open-grown red alder trees in chapter 4 of this dissertation. A sensitivity analysis was also conducted to consider what difference to the results this would make if other magnitude above or below the best estimates were assumed. The predicted harvest yields for forestry and agroforestry systems are shown in Tables 6.2 and 6.3, respectively.

6.2.11. Economic modelling

6.2.12. Cash flow budget

Cash flow budgets for the three scenarios were developed based on the annual estimated costs and returns developed in the enterprise budgets. The cash flow budgets indicated when payments and returns occurred over time as well as provided clear pictures of future financial commitments and viability of the enterprises. The net cash flow was determined by subtracting the streams of total costs from the stream of total returns.

6.2.13. Discounted cash flow

Since a future amount of money is worth less in the present a discounting technique is often used to bring future costs and returns to a present day value. To bring the streams of these

future costs and returns to their present value a standard real discount rate of 3.5% was adopted based on recommendations in the Treasury Green Book (HM Treasury, 2003), UK central government's policy evaluation guide book (Davies and Kerr, 2015), which states that:

“Society as a whole prefers to receive goods and services sooner rather than later, and to defer costs to future generations. This is known as ‘social time preference’; the ‘social time preference rate’ (STPR) is the rate at which society values the present compared to the future.” “This guidance recommends that the STPR be used as the standard real discount rate.” (H.M Treasury, 2003).

In addition to the baseline standard discount rate of 3.5%, the rates of 0%, 5%, and 8% were also investigated. Using the adopted discount rates, discount factors were calculated to discount future costs and returns of the enterprises back to the present in pound sterling. The discount factor was calculated using the following formula:

$$DF = \frac{1}{(1 + r)^t}$$

Where r = discount rate and t = number of years money was held

It is customary for the discount factor to decrease rapidly with time, due to the compounding of the discount rate.

6.2.14. Discounted cash flow analysis

Given the information provided in the discounted cash flow budgets, the net present value (NPV) and the annual equivalent value (AEV) measures of enterprise worth were used to measure and compare the economic viability of the three systems. The same starting point of Year 1 and projection duration of 30 years were used for all comparisons.

6.2.15. Net present value (NPV)

Net present value is simply a technique where all future net income streams from the enterprise are discounted to reflect their current or present value (PV). The NPV is the primary criterion adopted by the UK government for deciding whether action can be justified. Since estimates are used, rather than exact costs and returns, NPV can be used as a ranking tool in capital budgeting to analyze and compare the profitability of a projected investment or project alternatives, such as agroforestry, livestock or forestry investments, to see which option is the most economically attractive (Hiley, 1954 & 1956). The NPV was calculated as the sum of the discounted cash flows using the recommended standard discount rate of 3.5% and the formula shown in Equation [1]:

$$NPV = \sum_{t=1}^n \frac{(B_t - C_t)}{(1 + r)^t} \dots\dots\dots [1]$$

where, B_t = Benefits in each project year t , C_t = Costs in each project year t , n = Number of years to the end of project (n ranges from 1 to 30), and r = Discount rate (3.5%)

The formula indicates that the NPV of an investment is determined by discounting all future benefits (B_t) and costs (C_t) to the present with interest rate (r), and the NPV value is calculated as the difference between the present value of cash inflows and the present value of cash outflows.

The calculated NPV may work out to be a negative or positive value or zero. A positive NPV result (where $NPV > 0$) means that, at the discount rate assumed, the present value of the benefit streams is greater than the present value of the cost stream - that is, sufficient to recover investment and should be accepted because it is profitable. On the other hand, a negative NPV value (where $NPV < 0$) indicates that discounted costs exceed discounted benefits. Such an alternative should be rejected because it does not generate enough benefit to offset costs and will result in a financial loss. A zero NPV value (where $NPV = 0$) shows that discounted benefits equal discounted costs and the investment should be accepted because it still generates enough benefits to offset costs – this referred to as the financial break-even point (Godsey *et al.*, 2009)

The formal selection criterion for the NPV measure of project worth is to accept all investments with the higher positive NPV.

6.2.16. Annual equivalent value (AEV)

The AEV is another common discounted measure of economic performance used in capital budgeting to broadly compare investments options that have varying maturity (Godsey 2008).

The calculated NPV can be used to derive the AEV (equivalent yearly income from each investment option) for an investment. AEV has been described as NPV expressed as an annual amount, that is, the property-owner's yearly incomes per hectare that is usually used to compare the economic earnings from various land uses that produce yearly incomes (Bullard and Straka, 1998). It looks at the expected income potentials of alternative investments by estimating and establishing the constant annual cash flow generated by an investment over its lifespan. The generated present value of the constant annual cash flow is equal to the NPV of the investment. Decision criterion is to accept investment with higher AEV.

Annual equivalent value was calculated using the formula as shown in equation [3]:

$$AEV = r (NPV) / (1 - (1 + r)^{-n}) \quad \text{..... [3]}$$

where,

NPV = net present value

r = rate per investment period

n = number of years in investment periods

6.2.17. Sensitivity analysis

Sensitivity analysis was conducted to examine the effects of varying the following factors:

- Lamb price – the baseline lamb price of £66.7/head was increased and decreased by 50%, respectively.
- Fuelwood and sawlog prices - the baseline red alder fuelwood and sawlog prices were increased and decreased by 50%, respectively.
- Tree harvest volume – the baseline yields of harvested red alder in agroforestry and forestry systems were increased and decreased by 50%, respectively.
- Discount rate – the baseline discount rate of 3.5% was substituted for discount rates of 0%, 5%, and 8%, respectively.
- Grant/Subsidy – the NPV, and AEV of the three scenarios were recalculated with the application of the Basic Payment Scheme (BPS) subsidy to both livestock and agroforestry scenarios, and the Welsh Government Glastir New Planting Payment and Woodland Creation Premium for the forestry scenario. These are presented on page 219 (Sensitivity to grant/subsidies).

6.3. RESULTS

6.3.1. Economic indicator analysis

The Net Cash Flow (NCF), Net Present Value (NPV), and Annual Equivalent Value (AEV) measures of investment appraisals were used as financial indicators to quantify and compare the economic viability of Livestock, Forestry, and Agroforestry investment options based on 2016 baseline data. The results of the evaluation are summarised in Table 6.4.

Table 6.4: Summary of present value (PV), net present value (NPV) and annual equivalent value (AEV) of livestock, forestry and agroforestry scenarios at baseline discount rate of 3.5%, on a 10-hectare farm, over a 30-year rotation, assuming no subsidy.

Scenario	PV of Revenue	PV of Cost	NPV	AEV
Livestock	£20,786.44	£24,479.81	-£3,693.37	-£200.81
Forestry	£10,995.20	£12,824.58	-£1,829.38	-£99.47
Agroforestry	£22,500.02	£26,754.60	-£4,254.58	-£231.33

6.3.2. Net cash flow

Using the baseline discount rate of 3.5%, the estimated discounted net margins over a rotation period of 30 years under the present assumptions are presented in Tables 6.5, 6.6, and 6.7 for Livestock, Forestry, and Agroforestry options, respectively. The trends indicate that the livestock option generated negative NCF all through the 30-year rotation with a negative total net margin of -£5,337.35 (Table 6.5 and Figure 6.1). NCF for the forestry option was positive only in year 15 (£8,508.00) and year 30 (£14,384.00) for a positive total net margin of

£7,306.00 (Table 6.6 and Figure 6.1) while the agroforestry option generated positive NCF only in year 10 (£402.85), in year 20 (£495.65), and in year 30 (£2,213.85) for a negative total net margin of -£4,018.55 (Table 6.7 and Figure 6.1).

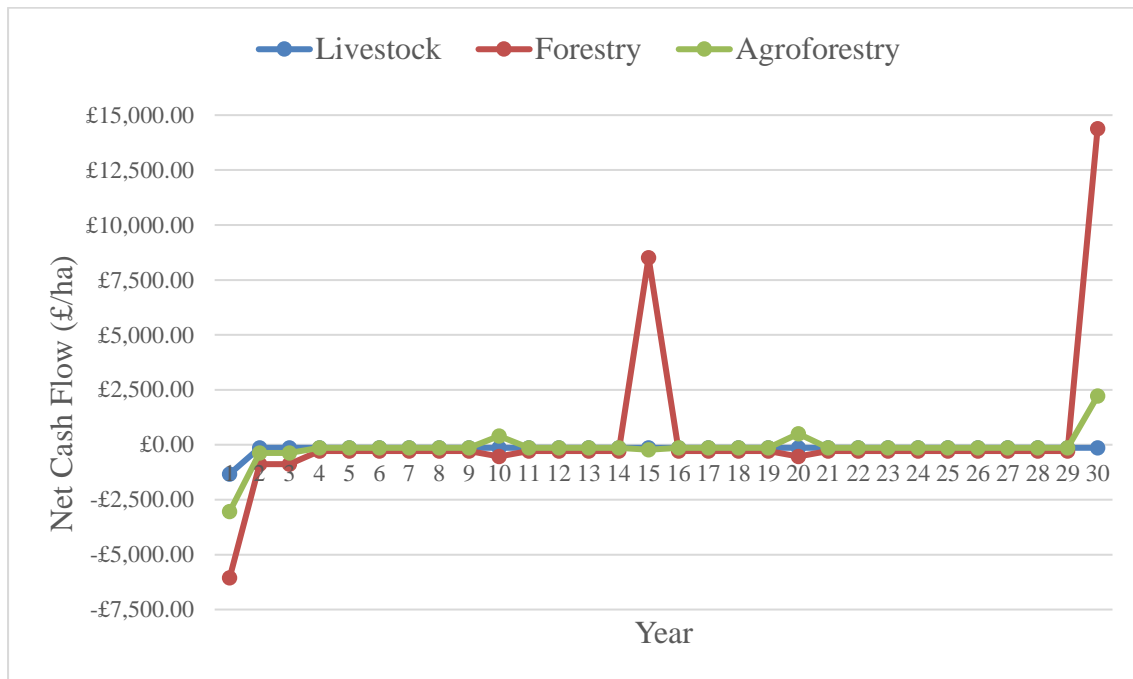


Figure 6-1: Net Cash Flow for livestock, agroforestry, and forestry over a rotation of 30 years

Table 6.5: Discounted benefits and costs for Livestock scenario at 3.5% discount rate on a 10-hectare farm over a 30-year rotation, assuming no subsidy.

Year	Revenue	Cost	Net margin	Discounted net margin
1	£0.00	£1,331.00	-£1,331.00	-£1,285.99
2	£1,192.85	£1,331.00	-£138.15	-£128.96
3	£1,192.85	£1,331.00	-£138.15	-£124.60
4	£1,192.85	£1,331.00	-£138.15	-£120.39
5	£1,192.85	£1,331.00	-£138.15	-£116.32
6	£1,192.85	£1,331.00	-£138.15	-£112.39
7	£1,192.85	£1,331.00	-£138.15	-£108.58
8	£1,192.85	£1,331.00	-£138.15	-£104.91
9	£1,192.85	£1,331.00	-£138.15	-£101.36
10	£1,192.85	£1,331.00	-£138.15	-£97.94
11	£1,192.85	£1,331.00	-£138.15	-£94.63
12	£1,192.85	£1,331.00	-£138.15	-£91.43
13	£1,192.85	£1,331.00	-£138.15	-£88.33
14	£1,192.85	£1,331.00	-£138.15	-£85.35
15	£1,192.85	£1,331.00	-£138.15	-£82.46
16	£1,192.85	£1,331.00	-£138.15	-£79.67
17	£1,192.85	£1,331.00	-£138.15	-£76.98
18	£1,192.85	£1,331.00	-£138.15	-£74.37
19	£1,192.85	£1,331.00	-£138.15	-£71.86
20	£1,192.85	£1,331.00	-£138.15	-£69.43
21	£1,192.85	£1,331.00	-£138.15	-£67.08
22	£1,192.85	£1,331.00	-£138.15	-£64.81
23	£1,192.85	£1,331.00	-£138.15	-£62.62
24	£1,192.85	£1,331.00	-£138.15	-£60.50
25	£1,192.85	£1,331.00	-£138.15	-£58.46
26	£1,192.85	£1,331.00	-£138.15	-£56.48
27	£1,192.85	£1,331.00	-£138.15	-£54.57
28	£1,192.85	£1,331.00	-£138.15	-£52.73
29	£1,192.85	£1,331.00	-£138.15	-£50.94
30	£1,192.85	£1,331.00	-£138.15	-£49.22
Total	£34,592.65	£39,930.00	-£5,337.35	-£3,693.37

Table 6.6: Discounted benefits and costs for Forestry scenario at 3.5% discount rate on a 10-hectare farm over a 30-year rotation, assuming no subsidy.

Year	Revenue	Cost	Net margin	Discounted net margin
1	£0.00	£6,056.00	-£6,056.00	-£5,851.21
2	£0.00	£890.00	-£890.00	-£830.82
3	£0.00	£890.00	-£890.00	-£802.73
4	£0.00	£290.00	-£290.00	-£252.72
5	£0.00	£290.00	-£290.00	-£244.17
6	£0.00	£290.00	-£290.00	-£235.92
7	£0.00	£290.00	-£290.00	-£227.94
8	£0.00	£290.00	-£290.00	-£220.23
9	£0.00	£290.00	-£290.00	-£212.78
10	£0.00	£540.00	-£540.00	-£382.82
11	£0.00	£290.00	-£290.00	-£198.63
12	£0.00	£290.00	-£290.00	-£191.92
13	£0.00	£290.00	-£290.00	-£185.43
14	£0.00	£290.00	-£290.00	-£179.16
15	£9,360.00	£852.00	£8,508.00	£5,078.35
16	£0.00	£290.00	-£290.00	-£167.24
17	£0.00	£290.00	-£290.00	-£161.59
18	£0.00	£290.00	-£290.00	-£156.12
19	£0.00	£290.00	-£290.00	-£150.85
20	£0.00	£540.00	-£540.00	-£271.39
21	£0.00	£290.00	-£290.00	-£140.82
22	£0.00	£290.00	-£290.00	-£136.05
23	£0.00	£290.00	-£290.00	-£131.45
24	£0.00	£290.00	-£290.00	-£127.01
25	£0.00	£290.00	-£290.00	-£122.71
26	£0.00	£290.00	-£290.00	-£118.56
27	£0.00	£290.00	-£290.00	-£114.55
28	£0.00	£290.00	-£290.00	-£110.68
29	£0.00	£290.00	-£290.00	-£106.94
30	£15,180.00	£796.00	£14,384.00	£5,124.71
Total	£24,540.00	£17,234.00	£7,306.00	-£1,829.38

Table 6.7: Discounted benefits and costs for Agroforestry scenario at 3.5% discount on a 10-hectare farm over a 30-year rotation, assuming no subsidy.

Year	Revenue	Cost	Net margin	Discounted net margin
1	£0.00	£3,043.00	-£3,043.00	-£2,940.10
2	£1,192.85	£1,565.00	-£372.15	-£347.41
3	£1,192.85	£1,565.00	-£372.15	-£335.66
4	£1,192.85	£1,328.00	-£135.15	-£117.78
5	£1,192.85	£1,328.00	-£135.15	-£113.79
6	£1,192.85	£1,328.00	-£135.15	-£109.94
7	£1,192.85	£1,328.00	-£135.15	-£106.23
8	£1,192.85	£1,328.00	-£135.15	-£102.63
9	£1,192.85	£1,328.00	-£135.15	-£99.16
10	£1,852.85	£1,450.00	£402.85	£285.59
11	£1,192.85	£1,328.00	-£135.15	-£92.57
12	£1,192.85	£1,328.00	-£135.15	-£89.44
13	£1,192.85	£1,328.00	-£135.15	-£86.42
14	£1,192.85	£1,328.00	-£135.15	-£83.49
15	£1,192.85	£1,428.00	-£235.15	-£140.36
16	£1,192.85	£1,328.00	-£135.15	-£77.94
17	£1,192.85	£1,328.00	-£135.15	-£75.31
18	£1,192.85	£1,328.00	-£135.15	-£72.76
19	£1,192.85	£1,328.00	-£135.15	-£70.30
20	£1,948.85	£1,453.20	£495.65	£249.10
21	£1,192.85	£1,328.00	-£135.15	-£65.62
22	£1,192.85	£1,328.00	-£135.15	-£63.41
23	£1,192.85	£1,328.00	-£135.15	-£61.26
24	£1,192.85	£1,328.00	-£135.15	-£59.19
25	£1,192.85	£1,328.00	-£135.15	-£57.19
26	£1,192.85	£1,328.00	-£135.15	-£55.25
27	£1,192.85	£1,328.00	-£135.15	-£53.39
28	£1,192.85	£1,328.00	-£135.15	-£51.58
29	£1,192.85	£1,328.00	-£135.15	-£49.84
30	£3,622.85	£1,409.00	£2,213.85	£788.75
Total	£38,438.65	£42,457.20	-£4,018.55	-£4,254.58

6.3.3. Present value, net present value, and annual equivalent value

The evaluation of the viability of livestock, forestry, and agroforestry land use scenarios were also compared in terms of present value (PV), NPV and AEV per hectare at the baseline discount rate (DR) of 3.5% as shown in Table 6.4. Results show that under the baseline assumption, the PV of costs for all the three investment options were higher than the PV of their respective revenues resulting in negative values for their respective NPVs and AEVs and are therefore adjudged unviable. Agroforestry option incurred the highest loss with an NPV of -£4,254.58 and an AEV of -£231.33, followed by livestock option with an alternate NPV of -£3,693.37 and an AEV of -£200.81. Forestry option incurred the least loss with an alternate NPV of -£1,829.38 and an AEV of -£99.47 (Table 6.4).

Generally, the three scenarios indicated negative values at the baseline assumptions and are adjudged economically unviable as they all failed to meet the decision rule criteria for investment acceptance. The negative values are indication that the revenues are insufficient to offset the investment costs and therefore engagement in these investments would certainly lead to financial losses.

Negative NPV ($NPV < 0$) indicates the generation of insufficient revenue to offset the costs of establishment and management, which in Agroforestry option include the cost of tree/livestock management. Negative AEV indicates that all the three scenarios would accrue quite significant annual losses per hectare throughout the 30-year rotation period.

6.4. SENSITIVITY ANALYSIS OF RESULTS

6.4.1. Sensitivity to variations in lamb sale price

Changes in investment values as a result of a 50% increase and decrease in the price of lamb on NPV and AEV are presented in Table 6.8. The result shows that both agroforestry and livestock options are sensitive to changes in lamb price. Livestock remains the most preferred option with higher NPV and AEV followed again by agroforestry and forestry options in that order. For instance, a 50% increase in the price of lamb increased the NPV of livestock option from -£3,693.00/hectare to £5,314.00/hectare, while the NPV of agroforestry option increased from -£4,255.00/hectare to £4,753.00/hectare, The NPV and AEV of forestry option remained constant in this analysis.

On the other hand, a 50% reduction in the price of lamb significantly altered the relative economic viability of the investment options by reducing the NPV of livestock from -£3,693.00/hectare to -£12,701.00/hectare and the NPV of agroforestry option from -£4,255.00/hectare to -£21,369.00/hectare thereby rendering both livestock and agroforestry options economically unattractive.

Table 6.8: Sensitivity of net present value (NPV) and annual equivalent value (AEV) per hectare to variation in lamb sale price, wood price, and wood yield at baseline discount rate of 3.5%

Parameters	Livestock		Forestry		Agroforestry	
	NPV	AEV	NPV	AEV	NPV	AEV
Base value	-£3,693.00	-£201.00	-£1,829.00	-£99.00	-£4,255.00	-£231.00
Lamb price +50%	£5,314.00	£289.00	-£1,829.00	-£99.00	£4,753.00	£258.00
Lamb price -50%	-£12,701.00	-£691.00	-£1,829.00	-£99.00	-£21,369.00	-£1,162.00
Wood price +50%	-£3,693.00	-£201.00	£3,668.00	£199.00	-£3,398.00	-£185.00
Wood price -50%	-£3,693.00	-£201.00	-£7,327.00	-£398.00	-£5,111.00	-£278.00
Wood yield +50%	-£3,693.00	-£201.00	£3,485.00	£189.00	-£3,398.00	-£185.00
Wood yield -50%	-£3,693.00	-£201.00	-£7,144.00	-£388.00	-£5,111.00	-£278.00

6.4.2. Sensitivity to changes in fuelwood and sawlog price

Changes in investment values as a result of a 50% increase and decrease in the price of fuelwood and sawlog on NPV and AEV are presented in Table 6.8. The result shows that the performance of forestry option relative to the agroforestry and livestock options was most sensitive to changes in wood price thereby substantially altering the ranking of the investment options. Forestry became the most preferred option. Engagement in either livestock or agroforestry options would result in a financial loss.

For instance, at the baseline discount rate, a 50% increase in the price of fuelwood and sawlog increased the NPV of forestry option from -£1,829.00/hectare to £3,668.00/hectare, while NPV of agroforestry option, though negative, increased from -£4,255.00/hectare to -£3,398.00/hectare (Table 6.8). The NPV and AEV of livestock option remained constant in this analysis.

On the other hand, a 50% reduction in the price of fuelwood and sawlog significantly altered the relative economic viability of the investment options by reducing the NPV of forestry option from -£1,829.00/hectare to -£7,327.00/hectare and the NPV of agroforestry from -£4,255.00/hectare to -£5,111.00/hectare thereby rendering both forestry and agroforestry options economically unattractive (Table 6.8).

6.4.3. Sensitivity to changes in fuelwood and sawlog yield

A sensitivity analysis was also conducted to determine the effect of a 50% increase and decrease in fuelwood and sawlog yield on NPV and AEV. The result shows that the performance of forestry option relative to the agroforestry and livestock options was most sensitive to changes in wood yield thereby substantially altering the ranking of the investment options. Forestry remained the most preferred option followed by agroforestry while the NPV and AEV for livestock option remained constant in this analysis (Table 6.8). For instance, a 50% increase in the of yield of fuelwood and sawlog increased the NPV of forestry option from -£1,829.00/hectare to £3,485.00/hectare, while NPV of agroforestry option increased from -£4,255.00/hectare to -£3,398.00/hectare (Table 6.8).

Conversely, a 50% reduction in the yield of fuelwood and sawlog again altered the relative economic viability of the investment options by reducing the NPV of forestry option from -£1,829.00/hectare to -£7,144.00/hectare and the NPV of agroforestry from -£4,255.00/hectare to -£5,111.00/hectare thereby rendering both forestry and agroforestry options economically unattractive (Table 6.8).

6.4.4. Sensitivity of scenarios to variation in discount rate

The results of sensitivity of net present value (NPV) and annual equivalent value (AEV) per hectare where all management actions were run at 0%, 2%, 5%, and 8% discount rates are presented in table 6.9. The results show that the NPV and the AEV values for livestock and agroforestry scenarios are negative at all discount rates (including the baseline discount rate), indicating financial loss. Forestry option also showed negative values at the baseline and higher discount rates but turned positive at lower discount rates (0% and 2%) implying that investment

in forestry will be profitable at the lower discount rates. On the other hand, it is also evident that the values of the economic indicators increase with decreasing discount rate (Table 6.9). For instance, for forestry option, the estimated NPV (£975.00/hectare) is positive at a lower discount rate of 2% compared to the negative NPV (-£1,829.00/hectare) at higher baseline discount rate of 3.5%.

Table 6.9: Sensitivity of net present value (NPV, and annual equivalent value (AEV) per hectare to variation in discount rate (base rate = 3.5%)

Scenario	Discount Rate	NPV	AEV
Livestock	0%	-£5,337.00	-£178.00
	2%	-£4,264.00	-£190.00
	3.5%	-£3,693.00	-£201.00
	5%	-£3,260.00	-£212.00
	8%	-£2,660.00	-£236.00
Forestry	0%	£7,306.00	£244.00
	2%	£975.00	£44.00
	3.5%	-£1,829.00	-£99.00
	5%	-£3,632.00	-£236.00
	8%	-£5,532.00	-£491.00
Agroforestry	0%	-£4,019.00	-134.00
	2%	-£4,240.00	-189.00
	3.5%	-£4,255.00	-£231.00
	5%	-£4,203.00	-273.00
	8%	-£4,019.00	-357.00

Forestry option remains the preferred and option for showing positive NPV (£7,306.00/hectare) and AEV (£244.00/hectare/year) at 0% compared to livestock and agroforestry options, though forestry option indicated negative NPV and AEV at the baseline discount rate (Table 6.9).

6.4.5. Sensitivity to grants/subsidies

The results of the comparative analysis of the relative economics of the livestock, forestry, and agroforestry scenarios presented so far had been based on the exclusion of the grants/subsidies. A sensitivity analysis was therefore conducted to observe the effect of inclusion of the prevailing grants/subsidies to the three scenarios as follows

[https://naturalresources.wales/...woodland-creation/support-available-for-new-woodla.:](https://naturalresources.wales/...woodland-creation/support-available-for-new-woodla.)

- Livestock scenario: Basic Payment Scheme (BPS) subsidy at £210.10/hectare from year 1 to year 30.
- Forestry scenario: a) Welsh Government Glastir New Planting Payment at £3,600/hectare; b) Glastir Woodland Creation Premium at £350/hectare from year 2 to year 12; and c) Glastir Annual Maintenance Payment at £60/hectare from year 2 to year 13.
- Agroforestry scenario: a) Glastir New Planting Payment at £1,600/ hectare; b) Basic Payment Scheme grant at £210.10/hectare from year 1 to year 30; c) Glastir Annual Maintenance Payment at £30/hectare from year 2 to year 6.
- Deductions were made for the area of the trees which were covered by the tree crowns.

Result shows that the addition of grants significantly altered the relative economic viability as livestock, forestry and agroforestry options registered positive NPV and AEV values at the baseline assumptions and are therefore adjudged economically viable as they all met the decision rule criteria for investment acceptance (Table 6.10). However, at higher discount rates (5% and 8%) livestock options indicated negative values and is therefore unviable while agroforestry remained viable at 5% but unviable at 8% discount rate.

It is pertinent to note that forestry option indicated positive NPV and AEV values with or without grant at very low discount rates (0% and 2%) and thereby being the only viable option at discount rates that are below the baseline rate even in the absence of grant (Table 6.10).

Table 6.10: Sensitivity of net present value (NPV, annual equivalent value (AEV) per hectare to application of grants at varying discount rate (3.5%)

Scenario	Discount Rate	NPV		AEV	
		Without Grant	With Grant	Without Grant	With Grant
Livestock	0%	-£5,337.00	£966.00	-£178.00	£32.00
	2%	-£4,264.00	£442.00	-£190.00	£20.00
	3.5%	-£3,693.00	£171.00	-£201.00	£9.00
	5%	-£3,260.00	-£30.00	-£212.00	-£2.00
	8%	-£2,660.00	-£294.00	-£236.00	-£26.00
Forestry	0%	£7,306.00	£15,826.00	£244.00	£528.00
	2%	£975.00	£8,755.00	£44.00	£391.00
	3.5%	-£1,829.00	£5,477.00	-£99.00	£298.00
	5%	-£3,632.00	£3,257.00	-£236.00	£212.00
	8%	-£5,532.00	£662.00	-£491.00	£59.00
Agroforestry	0%	-£4,019.00	£2,948.00	-134.00	£98.00
	2%	-£4,240.00	£1,473.00	-189.00	£66.00
	3.50%	-£4,255.00	£778.00	-231.00	£42.00
	5%	-£4,203.00	£301.00	-273.00	£20.00
	8%	-£4,019.00	-£268.00	-357.00	-£24.00

6.5. DISCUSSION

The modelling assumption that silvopastoral agroforestry system is a more viable economic undertaking compared to pasture and pure forestry systems is rejected in this study. The results of the bioeconomic analysis show that, in the absence of government subsidies, none of the scenarios is viable at the baseline discount rate. With negative NPV and AEV figures, a decision to engage in any of the three investments based on budgeting information available to investors in 2016 and at the baseline discount rate of 3.5% would certainly result in significant losses to the farmer. This is principally due to output prices being significantly lower than the establishment and running costs, to the effect that even when incomes are discounted to the beginning of the investment period, the NPV and AEV figures per hectare remain negative. Forestry option will incur the least loss in NPV and AEV followed by livestock option and agroforestry option in that order.

The results of the sensitivity analysis suggest that the assumptions used in the analysis are sufficiently realistic. However, significant changes in lamb sale price, wood price, and wood yield greatly influenced the economic viability of the three investment options (Table 6.8). A 50% increase in the price of lamb changed the NPV and AEV of livestock and agroforestry investments from negative to positive figures thereby rendering both options economically profitable at the base line assumptions with livestock as the preferred option. This change in viability and profitability could be explained by the margin between lamb sale prices and establishment and maintenance costs being large enough to overcome the effects of discounting thereby rendering the NPV and AEV positive. On the contrary, a 50% increase in wood price and yield had a mixed influence on the economic viability of forestry and agroforestry. While forestry option became profitable, agroforestry option remained unviable and would likely

require a much higher percentage increase in the price or yield of wood to make it profitable. The question as to whether these increases in prices and yield could be realistic remains a mirage especially with the recent political developments in the United Kingdom, especially the decision to leave the European Union. On the other hand, caution should be exercised here as the argument regarding prices could work both ways. Notably, recent devaluation of the pound has increased, for example, the prices of sheep in pound sterling terms.

The viability of the three investment options in this study decreased with increasing discount rates. This result is in line with the general trend observed in economic investments that shows that high discount rates can significantly reduce the NPV value of a long-term investment where the incomes are accrued late in the lifespan of the investment and often increases the NPV of land use investment where incomes are accrued earlier in the lifespan of the investment. For instant, at a lower discount rate of 2% in the bioeconomic model, the forestry option became economically viable, when it was previously unviable at a higher discount rate of 3.5%.

The observed trend can be explained by looking at the formula used to determine the compound interest factor $(1.0pn)$, which explains the relative effect of discount rate and time between expenditure and incomes. In the formula, p is the rate of interest and n is the number of years of the investment. The rate of interest is raised by the power of the number of years of the investment (Williams, 1988). The number of years of the investment raises the discount factor exponentially and at the higher discount rates, the greater the influence of the number of years is on the investment (Williams, 1988). It can be concluded that at higher discount rates, the period between expenditures and incomes must be short for the investment to remain profitable. In forestry where the investment timescale is much longer than other land use investment, the

discount rate used in the investment analysis has a significant effect on the profitability of the decision.

The finding in the present study disagrees with the results of other studies (Clason, 1995; Sibbald, 1996; McAdam *et al.*, 1999; Thomas and Willis, 2000; Husak and Grado, 2002; Grado and Husak, 2004; Dangerfield and Harwell, 1990) in which silvopastoral agroforestry investments were reported to outperform pure forestry or conventional livestock grazing investments, as well as the popular belief among farmers and landowners that farm forestry is not a viable economic undertaking (Lawrence *et al.*, 2010). Specifically, comparing the financial viability of silvopasture system and pasture system, Sibbald (1996) reported that the net present value for ash (*Fraxinus excelsior* L.) growing in silvopastoral system in lowland UK was greater than the net present value for treeless pastures by 15%. McAdam *et al.* (1999a) and Thomas and Willis (2000) also reported that, under a range of commodity prices and agricultural subsidy support scenarios, silvopasture has a net benefit over open grassland ranging from 34% to 181%, and that even with no farm subsidy support, silvopasture was more profitable (by €20 ha⁻¹) than open grassland because of the additional output of timber from silvopastoral systems. Clason, (1995) reported that silvopasture generated a higher internal rate of return than managed timber or open pasture. Husak and Grado (2002) demonstrated that the adoption of silvopasture is not only economically and biologically feasible but is also more financially attractive when compared with the individual production of soybeans, rice, cattle, and pine plantation.

There is a general perception among farmers that investment in silvopastoral agroforestry will generate little or no economic return and will be unsuccessful. However, it can be argued that

if grant assistance was high enough to make an investment in agroforestry economically viable some farmers would consider a change of land use from conventional livestock farming to agroforestry.

The bio-economic model constructed for this study was modified to accommodate potential grant incomes. Addition of prevailing government grant schemes improved the economic viability of the three investment options (Table 6.10). The eligibility of agroforestry systems for BPS within the UK depended on the nature of the woody component in the system. Where the agroforestry component contains more than 50 trees, appropriate allowances are made for the area taken up by trees, as required by regulation, by deducting areas of land where tree cover prevents growth of vegetation suitable for grazing (Defra, 2006b).

This study has established that pasture productivity, and hence livestock carrying capacity, could not remain constant all through the investment period, even though the trees were pruned and thinned periodically. To make the necessary adjustments, therefore, it is assumed that crown area is negligible for the first 10 years. However, between year 11 and year 20, crowns will increase to occupy 30% of the area (viewed from above) resulting in a loss of 30% of Area Payment by year 20. It is also assumed that crown area will remain constant after year 20 as crowns grow at a constant rate.

Despite the proven financial viability (assuming subsidies are available), and many other positive benefits associated with silvopastoral systems, their acceptance have been highly limited (Dagang and Nair, 2003). This is partly because of high initial investment cost, lack of capital or financial incentives for credit-constrained farmers, lack of awareness among farmers

and landowners of agroforestry practices (Thomas and Willis, 1997) and absence of technical expertise for establishing and running these systems, delayed return on investment, and complexity of the systems. Other barriers to wider adoption include highly limited research studies on temperate agroforestry and absence of demonstrable economic viability and practical management skills, and lack of effective information dissemination scheme and outreach support and extension projects (Smith *et al.*, 2012). Another major impediment to widespread adoption of agroforestry is a lack of cohesive and comprehensive policy support. High limitation on the number of trees to be integrated into agricultural land poses a serious challenge to policy issues concerning forestry and agriculture.

There is a need to provide some form of subsidy to farmers engaged in any of the three land management systems considered in this study. The question is how should that subsidy be applied. Should it be applied on per area basis with the complexity that it entails, or should it be applied on per tree basis, per livestock basis, per millimetre of water infiltrated basis, per tonne of carbon dioxide sequestered, per farm or tree establishment cost, per overhead cost, per labour cost? Payment for the establishment cost would be better because the farmer must pay all the costs upfront and wait for the benefit long-term and so is saddled with cash flow problem. But if the planting is adequately subsidised, cash flow problem is eliminated and what does the farmer do for income subsequently?

6.6. CONCLUSION

In conclusion, this research has found that without significant improvement in markets, an investment in livestock, forestry and agroforestry on a 30-year rotation based on 2016

budgeting information and at 3.5% discount rate is unlikely to be economically viable and profitable unless farmers are provided with grant assistance to receive the associated incomes.

There is therefore the imperative need for policy makers to improve awareness of the benefits of grant incomes and address farmers' concerns about the economic viability of livestock, forestry and agroforestry investments.

Chapter 7 : SYNTHESIS

7.1. INTRODUCTION

Agroforestry system practices have been defined as the deliberate integration of trees with agricultural crops and/or livestock either simultaneously or sequentially on the same unit of land (Nair, 1993), while silvopastoral systems are those agroforestry system practices where trees are combined with forage and livestock production on the same land management unit. Silvopastoral agroforestry systems can provide valuable ecosystem services particularly in terms of timber and pasture/livestock production, climate regulation and carbon sequestration, aesthetic and educational values, and biodiversity, soil improvement and water cycling. The UK's Farm Woodland Forum and its affiliate, the European Agroforestry Federation, have been making concerted efforts to promote the use of trees on farms. Nevertheless, many of these ecosystem services are externalities from the farmer's outlook and therefore tend to be underestimated and undervalued. Again, the approach of farmers, private and public tree planting programmes, and Government subsidised schemes to the application of agroforestry varies between countries. Conflicting policy or funding priorities have often led to intense negotiation or subsequent rejection of vital and well-defined agroforestry research programmes. To provide much needed knowledge, information and experience to farmers in this area, Silvopastoral National Network Experimental farms were established across the United Kingdom. However, there is a scarcity of information on the ecosystem services such a system could provide, and the financial and economic implications of shifting from conventional pasture grazing system to much desired silvopastoral agroforestry system. There

is, therefore, a need to evaluate some of the physical and bioeconomic potentials of the Silvopastoral National Network Experiment in the United Kingdom.

The purpose of this thesis was to investigate the ecosystem service potentials of the Silvopastoral National Network Experiment (SNNE) at Henfaes in North Wales with focus on the nitrogen-fixing red alder (*Alnus rubra* Bong) component. To achieve this objective and based on the premise that silvopastoral agroforestry system can deliver a range of ecosystem services and economic benefits, this research reviewed and synthesised research papers and articles written on ecosystem services of the UK's SNNE with specific focus on the Henfaes's Silvopastoral Systems Experimental Farm (SSEF) of Bangor University, Wales, since establishment in 1992 (Chapter 2); assessed the temporal and spatial changes over time in botanical composition and diversity of pasture species under red alder (Chapter 3); determined aboveground biomass allometric equations and carbon stock of two forms of open-grown red alder (Chapter 4); evaluated the influence of varying solar radiation intensities on pasture productivity and quality in thinned red alder blocks (Chapter 5) and; analysed the bio-economics of conventional grazing system compared to preferred silvopastoral system (Chapter 6). The research studies were conducted from January 2012 to end of 2014. The outputs provide evidence of the physical and bio-economics of ecosystem services from a silvopasture system at different scales and highlight areas of research to address detected knowledge gaps. The present chapter (Chapter 7) puts the preceding research findings presented in Chapters 2 to 6 into perspective and discusses their implications for the establishment and management of silvopastoral agroforestry systems.

7.2. Review of research studies of ecosystem services

The goal of Chapter 2 is to provide an overview of the state of current knowledge of ecosystem services of the UK's SNNE, the Henfaes Silvopastoral Systems Experimental Farm (SSEF) of Bangor University, Wales, and other studies in temperate Europe. This study is based on a systematic review and synthesis of all the scientific literature that has so far been conducted on this topic from 1988 to 2012, using the ecosystem service framework. It summarizes and identifies what has been done so far, the benefits and contributions to our knowledge base, and potential knowledge gaps and priorities for future research from the data of ecosystem services in silvopastoral agroforestry systems.

The result reveals a considerable number of studies on silvopasture and ecosystem services in the United Kingdom and some parts of Europe as well as a clear picture of the data structure, even though this body of literature (n=66) is not large when compared to the vast extent of silvopasture sites in the United Kingdom and temperate Europe. Majority of the reviewed research initiatives were the Henfaes SSEF of Bangor University and the UK's SNNE (Table 2.1), whereas silvopasture systems in other parts of the UK and Europe were given less attention. This limitation on the number of studies is an indication that little research had been done on the topic of ecosystem services in silvopasture systems up to the point, and that this branch of research is only in its infancy.

The review also shows how ecosystem service assessment of the studied silvopasture generally focused on provisioning, regulating, and supporting services, such as timber or fuelwood potential, pasture/livestock management, carbon sequestration, water management, soil

improvement, and biodiversity enhancement, while no attention whatsoever was given to cultural services. Likewise, there was a strong dominance of biophysical assessment approaches and quantifiable indicators, and less attention to monetary approaches and indicators.

A few highlights of contribution of the reviewed research papers to knowledge base include:

- There are negligible effects of the trees on pasture within the first ten years of establishing the Henfaes SSEF, irrespective of the tree species,
- Trees planted in clumps presented better form and growth than the widely spaced trees.
- Trees in agroforestry systems sequestered more carbon per unit area compared to a monoculture field of crop plants or pure pasture.
- There was no reduction in animal production ten years after planting despite interception of up to 10% of total photosynthetically active radiation by the developing tree canopy.
- The poor performance of the wider spaced trees was attributed to exposure to wind and to the effects of animals through browsing or soil compaction.
- Greater biodiversity levels were found in silvopastoral systems than in both open grassland and pure woodland systems.
- Silvopastoral agroforestry system is a more viable economic undertaking compared to conventional livestock grazing and pure forestry systems

Care should be taken when interpreting the results and conclusions of this study. It is most likely that not all relevant scientific literature addressing the research questions of the ecosystem services under review were captured. The search terms might have missed vital

information in relevant publications and grey literatures, such as reports from governments and other institutions or literature published in magazines/journals. Moreover, information from other institutions and experimental sites in the United Kingdom could not be represented because of time and financial constraints. Therefore, there is a clear need to conduct a more exhaustive, holistic and updated review of research papers and articles written on the UK's SNNE silvopasture to include more information from other institutions and experimental sites. For more comprehensive understanding of UK and other European Silvopasture, empirical research should be directed to a wider variety of research approaches and to a wider coverage of ecosystem services to include studies of cultural ecosystem services, as well as studies of the financial and economic implications of silvopasture and the direct contributions of agroforestry to human well-being (e.g. in terms of public health benefits),

7.3. Temporal and spatial changes in pasture species composition

Chapter 3 examined the temporal and spatial changes in pasture species composition and diversity under an alder based silvopastoral system. The focus of attention was on what remained in the pasture in later years after sowing a simple pasture mixture. This was achieved in two experiments by 1) measuring the short-term effects of thinning on understory pasture species composition and diversity on the same red alder plot before and after thinning; (up to 1 year), compared to adjacent open pasture control, using the dry-weight rank method (Mannetje and Haydock, 1963; Jones and Hargraves, 1979); and 2) determining the medium-term changes in pasture species composition and diversity in red alder plots 20 years after sowing a mixed pasture, compared to open pasture control, using the point intercept method (Cook and Stubbendieck 1986).

Results of experiment 1 indicates some inconsistencies in the short-term effects of thinning on understory pasture species composition and diversity. Thinning had some effect on pasture species composition by weight between 2013 and 2014, however, the change was not statistically significant. Thus, the results of experiment 1 rejects the hypothesis that thinning will not change the understory pasture species composition, abundance and diversity in the short-term (up to 1 year) in this study. This result clearly demonstrates the degree of variability which can occur in the short-term following thinning. This is to be expected as it is well known that pruning and thinning debris and tree foliage litter can shade/cover available pasture area or crush and smother understory flora and interfere with grazing (Kellas *et al.*, 1995; Thomas *et al.*, 1999; Benavides *et al.*, 2009).

Though species richness remained the same for the treatments in the present study, there was a decline in the percent composition by weight of the sown species and a slight increase in both grass weeds and forb weeds one year after thinning compared to the adjacent open pastures. The increase in percent composition by weight of grass weeds and forb weeds after thinning was expected, which agrees with the Thysell and Carey (2001) who reported that exotic species were more abundant within 1 and 3 years following thinning in Douglas-fir forests.

Thinning trees did not improve species richness and diversity of under-tree pasture in the short-term in this study even though previous research studies suggested it would. Enhanced species richness and diversity in response to thinning was observed in previous studies more than 3 years following treatment (Bailey *et al.*, 1998, Thomas *et al.*, 1999; Thysell and Carey 2001; Brockway *et al.*, 2002; Ducherer *et al.*, 2013). Therefore, the observed change in the present study may be short impact of tree removal on understory pasture species composition, along

with cold winter season, minimum soil disturbance during thinning using chainsaw, and relatively low tree density before thinning.

The knowledge of the dynamics of species composition and diversity of a pasture is critically important for grazing enterprises as it can help livestock managers make decisions for adjusting stocking rate and managing pastures. The botanical composition of a pasture influences the quality and quantity of herbage available to grazing animals which, in turn, impact on the pasture composition through grazing and nutrient transfer. Managing change is a complex dynamic issue as changes in pasture botanical composition occur over periods of years, in response to climate and soil properties, tree density, tree spacing, as well as livestock and pasture management. Scott *et al.*, (2000) suggested that a desirable botanical composition for a pasture should support profitable livestock production sustainably and should include pastures dominated by deep-rooted, fertiliser responsive perennial grasses combined with a persistent perennial legume and few weeds (such as the composition of pasture in this study).

Experiment 2 showed the pattern of changes in pasture species composition and abundance in the three red alder blocks over the 20-year study period since the establishment of the UK's Silvopastoral Network Experiment at Henfaes in 1992. The pasture species composition and abundance were found to vary considerably across years and between treatments compared to the adjacent open pasture. Compared with open pasture, the contribution of the sown species, *Lolium perenne* and *Trifolium repens*, to total available pasture, though relatively stable over the years (1993 – 2001), had declined considerably by year 2014 in the under-tree pasture at Henfaes, for example, in year 2001 *Lolium perenne* comprised 23% of the total pasture composition in open pasture, whereas in the under-tree pasture *Lolium perenne* contributed

only 10% of total species composition. Conversely, the grass weeds, *Agrostis capillaris*, *Holcus lanatus* and *Poa pratensis*, and the forb weeds, *Urtica dioca* and *Cirsium arvense* provided a greater contribution, remaining relatively low until 2001 after which they increased significantly in 2014 (Table 3.2 and Fig 3.3). Again, both under-tree and open pasture species richness, Shannon-Weiner and Evenness indices declined steadily over the years but rose significantly in 2014. Generally, pasture on the three alder blocks, 20 years after the establishment of the Silvopastoral National Network Experiment at Henfaes, was found to be largely grass weeds (46-48%) followed by forbs or broadleaf weeds while the sown species declined significantly. Therefore, the second hypothesis, that there will be no change in understory pasture species composition, abundance and diversity in the medium-term (up to 20 years), was not supported by the result of this study.

This result is to be expected as it is an established fact that modifications to pasture species composition can be caused by many factors such as the original composition, sowing of introduced species, shade tolerance of pasture species, changes in the microclimate and soil properties, tree density, tree spacing, crown size and magnitude of foliation, fertiliser, livestock grazing and trampling including frequency and intensity, grazing management, silvicultural practices, such as clear-cutting, burning, pruning and thinning, (Benavides *et al.*, 2009; Shakhane *et al.*, 2013).

In the present study, the observed decrease in the sown species and the increase in the unsown species in later years is in agreement with previous study by Snaydon (1987) who reported that, for temperate pasture, sown species in pastures is dominant only for a limited time before the sward is overtaken by unsown species. The result also agrees with other trials reported by

Benavides *et al.*, (2009), which attributed the deterioration of botanical composition of pasture beneath trees over time to a decline in legume and ryegrass contents, and an increase in overall contents of grass weeds litter and dead matter (Appendix 3.2).

Modification to pasture species composition in the temperate region is a continuous process, though stable conditions may appear to be present sometimes (Benavides *et al.*, 2009). The persistence and stability of the sown species for 9 years after establishment in the present study agrees with the result of previous studies which indicated lack of significant difference in pasture production, hence lamb growth rate and livestock carrying capacity, between treatments in UK's SNNE as well as Henfaes SSEF up to ten years after establishment (Sibbald and Dalziel, 2000; Teklehaimanot *et al.*, 2002). The result of the present study also agrees with the results of the research by Hawke (1991) that under-tree content of *L. perenne* and *T. repens* was very similar to that in open pasture during the first 9 years of the stands' establishment. However, there was a significant decrease in the total content of both pasture species by the fifteenth year after planting. Cossens and Hawke (2000) also reported that understorey contents of *L. perenne* and *T. repens* decreased in Year 6, and in Year 9, respectively.

Intermittent forest management intervention practices over the years may have influenced species composition and diversity observed in this study. The pasture has not been reseeded since the establishment of the site in 1992 with a mixture of perennial ryegrass and white clover; the red alder blocks have not been treated with any fertilizer principally because of the capacity of red alder tree and white clover to fix nitrogen; and the weeds were treated with "Grazon 90" only once in early 1993.

The changed pasture species diversity and abundance in the present study is therefore an indication of the proliferation of undesirable weed species, which the farmers could view as deleterious to their livestock production. The replacement of sown species by less valuable species implies that the relative value of such pasture for grazing stock would therefore also be expected to have declined. This effect was more pronounced in the later years of pasture availability and so as a result of both lower production and low nutritional quality, would provide little useful pasture for the land manager. Therefore, it is possible that farmers will be highly concerned about the consequent reduction in pasture production.

The botanical composition of a pasture can be improved most quickly by establishing a new pasture, or renovating a degraded, previously sown pasture. However, frequent pasture renovation may not be financially viable due to the high costs of re-establishing pastures, the limited persistence of current cultivars and the long period to recover costs (Shakhane *et al.*, 2013). Reeve *et al.* (2000) have noted that many producers are often reluctant to invest in more sown pastures because of reported difficulties in getting sown pastures to persist for 10 years or more. Management can clearly have large effects on botanical composition; the challenge now is to learn how to optimize composition to ensure that the feed supply for grazing animals is enhanced over the long-term while optimising profit.

7.4. Allometric equations for estimating biomass and carbon stock of open-grown red alder in Silvopasture

Allometric equations derived from trees grown in forestry systems are most often used in estimating biomass of open-grown trees typically found in agroforestry systems despite the

known differences in their growth forms and possible errors in estimating their biomass and carbon stock potentials. To circumvent these limitations, the objective of Chapter 4 was to develop species specific allometric equations for the estimation of biomass and carbon sequestration potentials for two forms ('good' form and 'poor' form) of open-grown red alder trees in a lowland silvopastoral system in North Wales, UK.

Generally, Chapter 4 showed that the distribution of the timber/logs by size class did not differ significantly ($p > 0.05$) between the good form and the poor form trees. However, the branching process of the two forms of red alder differed significantly ($p < 0.05$). Again, this study demonstrated that biomass and sequestered carbon differed significantly ($p < 0.05$) between the two forms of red alder trees as well as between their components (Tables 4.5 to 4.8). Biomass and sequestered carbon were found to be greater in good form trees than in poor form trees. Furthermore, component-wise, biomass and sequestered carbon were shown to be largest in stems (69.97%), intermediate in branches (23.02%), and lowest in twigs (7.00%) for the two forms of red alder. The hypothesis of no significant difference between the two forms of red alder trees was therefore rejected in this study. This result is expected as it is evident that there is disparity in the trunk shape and crown architecture of both forms of trees.

This study also demonstrated the potential of using stepwise regression analysis to determine the best combination of the independent variables (DBH, HT, CA, BR, WD) that would be required to predict aboveground biomass (AGB) for the two forms of red alder. The results of regression analyses of 14 models (Tables 4.10 and 4.11) established that DBH alone can be a very strong predictor of both tree component biomass and aboveground biomass while crown area was shown to be the most important additional explanatory variable that can improve

considerably the goodness of fit for the models (Table 4.11: models 10, 12, and 14). Moreover, the goodness of fit showed that 94% of the observed variation in AGB was explained just only by DBH for good form trees (Model 9) while for poor form trees, integration of DBH and CA explained 97% of the variability in AGB. This is because of the positive and significant correlation between AGB and DBH ($r = 0.95$; $p < 0.01$) and CA ($r = 0.90$; $p < 0.01$), and between DBH and CA ($r = 0.84$; $p < 0.01$) (Table 4.9).

The observed goodness of fit of the developed models in the present study agreed with previous studies on the relationship between AGB and DBH (Brown *et al.*, 1989; Kettering *et al.*, 2001 Zianis and Mencuccini 2003; Zianis *et al.*, 2005; Alvarez *et al.*, 2012), confirming the prominent effect of diameter on the aboveground biomass prediction. The strong relationship between DBH and CA corresponds to observations by Wirth *et al.* (2004), Ledermann and Neumann (2006), Gschwantner and Schadauer (2006), and Kuyah *et al.* (2012) and indicate that stem diameter is an important indicator of crown size. This suggests that allometric equations relying on both explanatory variables create relatively robust biomass estimates for trees growing openly on farm. Therefore, when DBH is considered as the only explanatory variable, model (9), model (11), and model (13) can be used for predicting good form, poor form, and pooled AGB, respectively (Table 4.11). Furthermore, when only DBH and CA are the explanatory variables, this study recommended the use of model (10) for predicting good form AGB, model (12) for predicting poor form AGB, and model (14) for predicting pooled AGB. These results confirm that the new allometric equations using DBH and CA could be a better predictor of aboveground biomass and can be used as an important tool for predicting carbon stock in agroforestry systems.

Biomass and carbon stock estimates in the present study compared with estimates reported in other studies. For example, the estimated carbon sequestration potential of 12.95 Mg C ha⁻¹ to 24.55 Mg C ha⁻¹ in the present study is within the range of 10 Mg C ha⁻¹ to 208 Mg C ha⁻¹ reported for temperate latitude, and 21 Mg C ha⁻¹ to 240 Mg C ha⁻¹ reported for tropical climate (Dixon 1995; Adesina *et al.*, 1999; Kort and Turnock 1999; Turnock 2001; Montagnini and Nair 2004; Zianis *et al.*, 2005; Peichl *et al.*, 2006). These results imply that the silvopastoral agroforestry ecosystem in the United Kingdom represents a significant carbon sink.

Since this study was the first to develop allometric equations for open-grown red alder trees in a silvopastoral agroforestry system in the United Kingdom, it is hoped that the developed species-specific models will not only assist in quantifying the standing biomass, carbon density and levels of carbon dioxide that could be sequestered by silvopastoral agroforestry systems but may also contribute towards national climate change mitigation goals.

Allometric relationships developed in this study provide unique information that is highly relevant for agroforestry mosaics, and may assist in establishing a new generation of allometric models that use combined DBH and CA variables as primary predictor of aboveground biomass, especially in open-grown trees. However, caution should be exercised in the use of DBH and CA equations, especially in extrapolating beyond the range of the regression data or application in environments greatly different from those found in a silvopastoral national network experiment setting. The relevance for other landscapes with other species, tree configuration and environmental conditions will need to be tested further.

Even though the equations in this study were developed specifically for open-grown red alder

trees in Silvopasture, the models and procedure for these equations have valuable applications to other open-grown tree species and can provide a reference to development of biomass equations for other open-grown tree species in silvopastoral settings. Complimentary studies of similar nature should be conducted on all tree species grown in silvopasture across the wide range of sites represented in the UK's Silvopastoral National Network Experiment using the procedure laid out for open-grown red alder trees in the present study. Equations and predictors presented can be modified accordingly for application with other open-grown tree species in agroforestry settings to develop allometric models that will aid in the effective management of agroforestry systems in general.

7.5. The influence of solar radiation on pasture productivity and quality

Pasture production beneath trees is normally governed by the degree of competition between trees and pasture for light, moisture, and nutrients (Mead 2009; Dodd *et al.*, 2005). Understorey pasture DM yield and quality (nutritive value of herbage) are known to be strongly influenced by tree shading, which is a function of the degree of overstorey canopy closure and available light (Kephart and Buxton, 1993; Sibbald *et al.*, 1994; Knowles *et al.*, 1999; Devkota *et al.*, 2001; Lin *et al.*, 2001; Sigurdsson *et al.*, 2005; Peri *et al.*, 2007; Benavides *et al.* 2009). Their amount and values are useful indicators of the sustainability of silvopastoral farms as they have significant impact on both resource status and economic performance. Understanding of the relationship between canopy closure and understorey pasture DM production is crucial in the development of comprehensive management practice for deciduous tree based silvopastoral systems. The objective of Chapter 5 was to evaluate the influence of solar radiation on the production of DM and the nutritive value of understorey pasture in thinned red alder blocks in

a silvopastoral system with particular attention on how the current tree density treatment of 100-stem ha⁻¹ affected the light at ground-level and how this in turn affected the understorey pasture production and quality.

The results of the present study indicated that level of exposure to solar radiation (the level of shade) was a significant factor determining the productivity and quality (nutritive values) of pastures. A strongly significant ($p < 0.001$) correlation was established between solar transmission at ground level, measured at varying canopy gap levels created by random thinning of red alder blocks, and the observed pasture DM production, pasture quality parameters, and distance from each grazing exclusion cage to the nearest tree. Pasture DM production was found to increase significantly ($p < 0.05$) with increasing solar transmission (decreasing shade) (Figure 5.4), and with increasing distance from each grazing exclusion cage to the nearest tree (Figure 5.6). These results agree with the findings of previous researchers (Sibbald *et al.*, 1994; Knowles *et al.*, 1999; Lin *et al.*, 1999; Power *et al.*, 2001; Belesky, 2005; Douglas *et al.*, 2006; Peri *et al.*, 2007; Neel *et al.*, 2008; Ducherer *et al.*, 2013; Esquivel-Mimenza *et al.*, 2013) who all reported increased pasture DM yield with decreased shading or increased solar radiation. Specifically, Knowles *et al.* (1999) showed a strong relationship ($R^2 = 0.89$) between measured pasture yield and predicted canopy closure. Sibbald *et al.* (1994) showed that herbage production increased with decreased shading when precipitation and temperature favour herbage growth. Belesky (2005) and Peri *et al.* (2007) found that herbage plants grown in areas with lower light levels were smaller, had fewer numbers of tillers and produced less dry matter (DM) compared with treatments with higher levels of radiation. Lin *et al.* (1999) also showed a reduction of yield because of increased shade for orchardgrass, ryegrass and white clover. This is also in agreement with Neel *et al.* (2008).

The present study revealed that the concentrations of CP, ADF, NDF and ME in herbage were greater for the with-leaves condition than for the without-leaves condition. This observed difference may be explained by the variation of photoperiod (the duration of sunshine) in the United Kingdom. The period of day light is longer in the 7 months of with-leaves condition and shorter in the 5 months of without-leaves condition. Consequently, the rate of daily growth is greater in with-leaves condition with longer hours of sunlight, and slower in without-leaves conditions with shorter hours of sunlight.

The present study also linked observed seasonal variation in the pasture yield and quality parameters to leaf fall (deciduous nature of red alder tree). Mean pasture yield was shown to be greater in spring, summer and early autumn when the alder trees have leaves, and lower from mid-autumn to late winter when the trees are without leaves (Table 5.1 and Figure 5.4). Previous studies conducted under deciduous trees have also reported seasonal variation in pasture yield because of leaf fall. Douglas *et al.* (2006) observed that average pasture DM yield beneath a stand of *Populus spp.* varied seasonally with differences being greater in spring, summer and early autumn and more similar during the leafless period in late autumn and winter. The difference in seasonal variation for deciduous trees has been attributed to leaves being responsible for approximately 69% of shaded area (McElwee and Knowles 2000).

It is suggested that factors other than light intensity might have also influenced pasture yield and quality in the present study: The addition of extra N from red alder, a N-fixing tree, moisture and nutrient competition, allelopathic effects, and smothering could also have been at work under the trees.

These results demonstrated that canopy closure is the critical factor to manage if pasture production is to be maintained at an economic level. Both pruning trees or removing trees can decrease canopy closure, reduce light competition, improve the grazing productivity of the understorey pasture, allow the tree leaves to be used as livestock fodder, and improve wood quality. Deciduous tree species with suitable architecture and tolerance to specific site limitations will allow more light to the pasture, particularly when leafless. From the results of this study, reducing tree stocking density to 100-stems ha⁻¹ or below will likely maximise the amount of solar radiation reaching the understorey pasture, allow pasture to persist most of the rotation and enhance sward botanical composition. In conclusion, the use of thinning to reduce the density of red alder to 100-stem ha⁻¹ in a silvopastoral system increased understorey pasture production and nutritive value by increasing transmitted light.

7.6. Bio-economic potentials for silvopastoral agroforestry systems

There is a general perception among farmers that investments in agroforestry or farm forestry will offer little or no economic returns and will be unsuccessful (e.g. Burton and Wilson 2000: Sharpe *et al.*, 2001). Their general acceptance depends on developing and promoting systems that produce financial returns that are at least equal to those obtained from the annual crops they would replace. However, little research has compared returns from silvopastoral agroforestry, farm forestry, and conventional livestock grazing in the United Kingdom. The objective of Chapter 6 was to evaluate the bio-economic potentials of temperate lowland silvopastoral agroforestry systems in North Wales, United Kingdom. The study compared the economic viability of conventional livestock grazing, farm forestry, and silvopastoral

agroforestry investment options at 3.5% discount on a 10-hectare farm over a 30-year rotation using discounted cash flow analysis and national costs and prices for both livestock and tree products based on 2016 baseline data. Base case NPV and AEV were calculated for each production livestock grazing, farm forestry, and silvopastoral agroforestry scenario, assuming no policy interventions. In a sensitivity analyses, the study also demonstrated how the NPV and AEV of the three investment options responded to variations in lamb sale price, fuelwood and sawlog prices, fuelwood and sawlog yield, discount rates, and grants/subsidies.

Generally, results of the economic analyses indicated that under the baseline case, assuming no policy interventions, the PV of costs for all the three investment options were higher than the PV of their respective revenues resulting in negative values for their respective NPVs and AEVs and are therefore adjudged unviable as they all failed to meet the decision rule criteria for investment acceptance (Tables 6.4, 6.5, 6.6, 6.7, and Figure 6.1). Forestry option was shown to incur the least loss in NPV and AEV followed by livestock option and agroforestry option in that order. The negative NPV values implies that, at 3.5% discount rate, the revenues are insufficient to offset the investment costs, which in Agroforestry scenario include the cost of tree/livestock management. Negative AEV is an indication that all the three scenarios would accrue quite significant annual losses per hectare throughout the 30-year rotation period. Therefore, engagement in these investments under the baseline assumption, would certainly lead to financial losses.

This study also showed that changes in lamb sale price, wood price, and wood yield influenced the economic viability of the three investment options significantly. A 50% increase in the prices of lamb changed the NPV and AEV of livestock and agroforestry investments from

negative to positive figures thereby rendering both options economically profitable with livestock more viable than agroforestry (Table 6.8). On the other hand, a 50% increase in the price and yield of wood is shown to render agroforestry option unprofitable and forestry option profitable. A much higher percentage increase in the price and yield of fuelwood and sawlog would likely be required to make agroforestry option profitable.

Again, the viability of the three investment options in this study is shown to decrease with increase in discount rates. For example, the forestry option became economically viable at a lower discount rate of 2%, when it was previously unviable at a higher discount rate of 3.5% (Table 6.9). This result conforms with the general trend observed in economic investments that indicates that high discount rates can significantly reduce the NPV value of a long-term investment where the incomes are accrued late in the lifespan of the investment, and often increases the NPV of investment where incomes are accrued earlier in the lifespan of the investment.

Furthermore, this study disclosed that the application of prevailing government grant schemes significantly improved the economic viability of the three investment options (Table 6.10) as livestock, forestry and agroforestry options showed positive NPV and AEV values at the baseline assumptions and are therefore adjudged economically viable as they all met the decision rule criteria for investment acceptance. Forestry is shown to be the most lucrative option. However, while agroforestry remained viable at a higher discount rate of 5%, both livestock and agroforestry options were rendered unviable at a much higher discount rate of 8% (Table 6.10).

These findings disagree with the results of other studies (Clason, 1995; Sibbald, 1996; McAdam *et al.*, 1999; Thomas and Willis, 2000; Husak and Grado, 2002; Grado and Husak, 2004; Dangerfield and Harwell, 1990) who reported that silvopastoral agroforestry investments were more viable than pure forestry or conventional livestock grazing investments.

Even though farmers planting decisions are driven mainly by short term cashflow considerations, they are barely aware of the economics of agroforestry and farm forestry and this somehow hampers them from applying for grants. It can be argued exclusively that landowner decision-making and conduct is largely a function of finance and economics: land-uses that offer good profits will certainly be adopted. Such profits can originate from “the market” or government grants, or a combination of both. These grant assistance, could raise the bar for potential adoption of agroforestry or forestry, making them more difficult to rationalize economically. More programmes involving direct payments to landowners for ecosystem services could also enhance financial returns and attractiveness for agroforestry systems. Therefore, the present study stressed the need for farmers to be provided with some form of subsidy to enable them engage in any of the three land management systems considered in this study but wondered how the subsidies should be applied. The study further underscored the imperative need for policy makers to improve awareness of the benefits of grant incomes and address farmers’ concerns about the economic viability of livestock, forestry and agroforestry investments.

This research could provide a basis for future comparisons and analysis of farm programs and ecosystem service markets. The results of this research will help promote greater awareness of the economic value of trees in extensively grazed landscapes in the United Kingdom and should

assist future investment decisions by landowners (for example design options for retained trees, tree yields by species) since there is a perception amongst landowners that trees compete strongly with pasture and livestock production and are considered an economic liability rather than as a potential asset.

7.7. Ecosystem Services Valuation

The valued ecosystem services (provisioning and regulating) which are described thoroughly in chapters 3, 4, and 5 were used to generate bioeconomic models (Chapter 6), which shows that at least provisioning services is not enough to provide economically viable land use, but by combining these valued provisioning services with some of the other unvalued ecosystem services, and applying grants/subsidies, perhaps it will be possible to come up with a system that is viable.

Provisioning services such as pasture production, timber production, and possibly increased live weight gain in livestock (resulting from shelter) are easily valued. However, the valued ecosystem services in this study (provisioning) alone are insufficient to produce economically viable systems (pasture/livestock, forestry). Notwithstanding this, there are other unvalued ecosystem services in silvopasture such as carbon sequestration, increased biodiversity, soil improvement (Chapter 2), and cultural ecosystem services that could form the basis of subsidies to make silvopasture economically viable and thereby provide a way forward towards achieving a resilient landscape.

7.8. Drawbacks of silvopasture establishment

Many factors are known to impede the adoption of silvopastoral agroforestry in the United Kingdom. Principal among them is a recurring theme within the literature that many farmers and landowners are put off from engaging with government grant assistance due to concerns surrounding the bureaucracy and complexity of administration, concerns about repayment on crop failure and worries that the grant incomes are not high enough. Grant uptake (and lack of uptake) has been linked to landowner's awareness of / interest in grants; knowledge; and availability of, or particularly lack of, suitable land. Management grants and the additional supplements were viewed with uncertainty and were thought to be badly organized and inadequate; that amount of grant made available to farmers are economically insufficient. Grant application process is viewed as frustrating, inflexible and bureaucratic, and that acceptance of grant money from government will entail a loss of control over their property – particularly when linked to grants for public access (Lawrence *et al.*, 2010).

This research has demonstrated that in the absence of grant income, provisioning services of silvopasture systems is not profitable and markets alone are not sufficient to influence farmers and landowner's perceptions. This raises a critical concern for Government policy makers. There is a need for policy makers to engage with landowners to determine exactly what elements of the scheme administration or rules puts them off from engaging with them.

Policy wise, intensive management of silvopastoral systems can contribute significantly to decrease of livestock grazing pressure, improvement of animal welfare and enriched nutrient management. Silvopastoral agroforestry is in close alignment with current European Union

policy for intensively managed pastures which shifted subsidies from production to area-based payment system with attached strict compliance measures (www.agroforestry.ac.uk/sites/www...ac.../ssm2004_conclusions.pdf). From a global perspective, silvopastoral agroforestry can be a useful instrument to create land use system with higher carbon sequestration potentials than those from pasture land and which can cushion the adverse impacts of climate change.

There is therefore the imperative need for prospective investors and policy makers to fully understand the complexities of biophysical and bio-economic interactions in the silvopastoral systems to be able to position agroforestry within a wider framework of ideal land management. There is a strong argument that future policy needs to reduce the cultural gap between farming and agroforestry. Furthermore, there is a need to increase recognition of integrated land uses and align policy with landowners existing management objectives. With a fuller appreciation of the whole range of benefits provided by silvopasture systems, it is likely that more widespread introduction of these could significantly improve farm resilience to uncertainties in future economic and physical climate.

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APPENDICES

Appendix 2.1: List of references in relation to ecosystem service functions addressed.

S/N	Authors / References	Provisioning services		Regulating services	Supporting services		
		Timber or fuel potential	Tree/Pasture/Lives tock interaction	Carbon sequestration	Soil improvement	Water management	Biodiversity enhancement
1	Agard (2011)			x			
2	Agnew and Sibbald (1996)						x
3	Benjamin (2010)			x			
4	Bergmeier et al (2010)						x
5	Bezkorowajnyj et al., (1993)	x				x	
6	Broadmeadow and Matthews (2003)			x			
7	Burgess, P.J. (1999)						x
8	Crowe and McAdam, (1993)						x
9	Cuthbertson and McAdam, (1996)						x
10	Englund (1995)	x					
11	Follain et al., (2007)			x			
12	Gerety (1998),	x					
13	Granier et al., (1996)					x	
14	Green et al., (1995)	x					
15	Heron (1999)						x
16	Hislop and Claridge (2000)		x				
17	Howe (1997)	x					
18	Islam (2000)	x					

19	Jarju (2000)		x				
20	Kasahun et al., (2011			x			
21	Khanal (2011)			x			
22	Kondziela, (2011)					x	
23	Kreibich and Kern (2000)				x		
24	Kufllu (2000)					x	
25	Lu et al., (1995)					x	
26	Martin (1995)				x		
27	McAdam (1996)						x
28	McAdam and Hoppé (1996)						x
29	McAdam et al., (1999		x				x
30	McAdam et al., (2007a						x
31	McArthur (1991)		x				
32	Mmolotsi (2004)				x		x
33	Mmolotsi and Teklehaimanot (2006)	x					
34	Mmolotsi and Teklehaimanot (2008)				x		
35	Ng'atigwa (1997)	x	x				
36	Nghitoolwa, (2001)		x				
37	Onyeka (1998)		x				
38	Patenaude et al., (2003)			x			
39	Penn et al., (1994)				x	x	
40	Rakkibu (1998)					x	
41	Ramdial (2010)			x			
42	Roberts (1995)	x					
43	Rodwell (2009)			x			
44	Rois-Díaz et al., (2006)						x
45	Sibbald and Dalziel (2000)		x				
46	Sibbald and Sinclair, 1990)						
47	Sibbald et al., (1991)		x				

48	Sibbald et al., (1995)		x				
49	Sibbald et al., (2001)	x	x				
50	Teklehaimanot and Martin (1999)				x		
51	Teklehaimanot and Mmolotsi (2007)				x		
52	Teklehaimanot et al., (1991a)					x	
53	Teklehaimanot et al., (2002)	x	x				
54	Teklehaimanot et al., (1991b)					x	
55	Temba (1999)					x	
56	Toal and McAdam (1995)						x
57	Wang (1999)				x		
58	Winslade (1996)	x					
59	Zapater (1998)	x	x				
		13	13	9	8	10	13

Appendix 3.1: Assessment of pasture species composition and abundance by the Dry-Weight Rank method.



Appendix 3.2: Changes over time in botanical composition of pasture and debris beneath three tree species relative to that of adjacent openpasture

Tree species	Botanical component	Pasture species	Change	Authors
Pinus radiata D. Don.	Legume	Trifolium repens L.	Decrease	Percival et al. (1984a) Percival and Hawke (1985) Hawke (1991) Cossens and Hawke (2000)
		Trifolium subterraneum L.	Decrease	Gillingham (1984) Kellas et al. (1995)
	Grass		Increase	Gillingham (1984) Percival and Hawke (1985) Hawke (1991) Kellas et al. (1995)
		Lolium perenne L.	Decrease	Percival and Hawke (1985) Hawke (1991) Cossens and Hawke (2000)
		Dactylis glomerata L.	Increase	Cossens and Hawke (2000)
	Weed	Holcus lanatus L.	Increase	Kellas et al. (1995)
			Increase	Gillingham (1984)
			Increase	Percival et al. (1984a) Percival and Hawke (1985) Gillingham (1984) Hawke (1991)
Populus spp.	Legume	Trifolium repens L.	Decrease	Douglas et al. (2001, 2006a)
		Lotus uliginosus Schkuhr	Similar	Douglas et al. (2006a)
	Grass		Increase	Douglas et al. (2001, 2006a)
		Lolium perenne L.	Decrease	Crowe and McAdam (1992b) Guevara-Escobar et al. (2007)
			Similar	Douglas et al. (2006a) Similar Wall (2006)
		Holcus lanatus L.	Increase	Crowe and McAdam (1992b) Guevara-Escobar (1999) Wall (2006)
		Dactylis glomerata L.	Increase	Douglas et al. (2006a)
		Agrostis stolonifera L.	Increase	Crowe and McAdam (1992a, 1992b)
		Agrostis capillaris L.	Decrease	Douglas et al. (2006a)
		Poa annua L.	Increase	Crowe and McAdam (1992a) Guevara-Escobar (1999) Wall (2006)
			Decrease	Douglas et al. (2001; 2006a)
	Dead matter and litter		Increase	Guevara-Escobar et al. (2007) Douglas et al. (2006a)
Salix matsudana Koidz.	Legume	Trifolium repens L.	Decrease	Miller et al. (1996)
	Grass	Lolium perenne L.	Decrease	Miller et al. (1996)
		Dactylis glomerata L.	Increase	Miller et al. (1996)

(Adapted from Benavides *et al.*, 2009)

Appendix 4.1: Descriptive statistics of harvested red alder trees for construction of biomass equation

Block	Tree ID	Form	DBH	Height	Basal Area	Volume	CA	WD	BR
			(cm)	(m)	(m ²)	(m ³)	(m ²)	(g/m ³)	
1	P1 F5	1	33.6	12.00	0.09	0.35	56.71	260.50	3.00
1	P1 F11	1	38.0	12.50	0.11	0.47	62.84	359.90	3.07
1	P1 K8	1	26.0	12.00	0.05	0.21	60.09	333.50	3.10
1	P1 L3	1	29.0	11.50	0.07	0.25	53.92	334.30	2.91
2	P2 E9	1	26.0	13.00	0.05	0.23	62.21	751.90	2.74
2	P2 G3	1	25.0	11.00	0.05	0.18	53.41	742.00	2.38
2	P2 L7	1	21.0	11.50	0.03	0.13	21.21	507.40	2.00
3	P3 B12	1	35.0	13.00	0.10	0.42	61.86	323.20	4.71
3	P3 C15	1	33.0	13.50	0.09	0.38	52.92	514.10	3.38
3	P3 J8	1	30.0	12.00	0.07	0.28	49.41	319.80	5.00
1	P1 D11	2	29.0	12.50	0.07	0.28	32.99	491.00	2.35
1	P1 E8	2	25.0	13.00	0.05	0.21	31.67	236.30	2.44
1	P1 K10	2	23.0	12.50	0.04	0.17	40.14	265.30	2.48
2	P2 D13	2	21.0	10.00	0.03	0.12	27.34	455.00	2.38
2	P2 E13	2	23.0	11.00	0.04	0.15	25.06	518.40	1.82
2	P2 L11	2	25.0	10.75	0.05	0.18	35.94	485.70	1.88
2	P2 M9	2	24.0	10.50	0.05	0.16	37.39	624.30	2.13
3	P3 B2	2	26.0	11.60	0.05	0.21	37.78	244.40	3.20
3	P3 C11	2	21.0	10.50	0.03	0.12	39.21	443.60	2.18
3	P3 F6	2	13.0	9.00	0.01	0.04	12.19	152.10	2.67

Form 1 represents Good Form trees while Form 2 stands for Poor Form trees.

CA represents tree crown area.

Appendix 5.1: Harvesting of forages in the grazing exclusion cages to a residual sward height of 2.5 cm on the last day of every month.



Appendix 5.2: Mean monthly levels of solar radiation in relation to mean pasture production

Month	Pasture production (g DM m⁻² d⁻¹)	Solar radiation (mol m⁻² d⁻¹)
July	4.18	56.68
August	2.94	46.91
September	2.10	28.07
October	1.38	14.82
November	0.91	7.52
December	0.70	4.99
January	0.82	7.14
February	1.07	14.62
March	1.73	27.28
April	2.51	42.74
May	2.88	48.95
June	5.49	64.13

Appendix 5.3: Feed Quality for Forage Samples (from AFRC 1993).

Feed Type	Dry Matter (%)	Crude Protein (%)	Acid Detergent Fibre (%)	Neutral Detergent Fibre (%)	Digestibility (%DOMD)	Metabolisable Energy (MJ/kg)
Mixed Pasture	12 – 25	20 – 30	20 – 30	30 – 45	65 – 80	9 – 12
Pasture Silage	25 – 30	14 – 20	20 – 35	30 – 45	65 – 75	9 – 11
Cereal Silage	35 – 40	8 – 12	25 – 40	35 – 60	55 – 65	9 – 10.5
Maize Silage	25 – 35	6 – 9	25 – 35	35 – 50	60 – 70	9.5 - 11
Lucerne Foliage	15 – 25	20 – 30	25 – 30	35 – 45	60 – 70	9 – 12
Lucerne Hay	85 - 90	18 - 25	25 - 35	35 - 45	55 - 65	8 - 11

DOMD = Dry Organic Matter Digestibility

Appendix 5.4: Indicative Feed Requirements for Ruminant Animals (from AFRC 1993).

Animal	Crude Protein (%CP)	Acid Detergent Fibre (%ADF)	Neutral Detergent Fibre (%NDF)	Digestibility (%DOMD)	Metabolisable Energy (MJ/kg)
Cattle (Beef)	>12	19	25	61	9.5 – 10.5
Dairy Cow – Dry	>12	27	35	56	8.6
Dairy Cow - Lactation	>16	21	28	71	11
Calf	>16	>16	23	69	11
Sheep	9 – 12	20 – 25	25 – 35	55 – 65	8 – 10
Lamb	11 – 14	16 – 20	20 – 25	65 – 75	9 – 11