Carbon mitigation from agroforestry in salinized low rainfall farmland landscapes

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DECLARATION

I declare that this thesis is my own account of my research and contains as its main content work which has not previously been submitted for a degree at any tertiary education institution.

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Abstract

Efforts to reduce greenhouse gas emissions have become a global priority and the land sector can contribute significantly to achieving this via a range of mitigation strategies such as the biosequestration of carbon and substitution of fossil fuels through bioenergy. However, the implementation of land sector mitigation is constrained by several uncertainties and knowledge gaps particularly within low rainfall (300 to 400 mm yr^{-1}) farmland environments.

This thesis examines aspects of land sector mitigation through reforestation systems integrated into dryland (300 to 400 mm yr^{-1}) farming systems in Western Australia. The uncertainties that are examined in this thesis include (a) estimation of tree root carbon storage, (b) exploring whether carbon mitigation can be achieved through new agroforestry systems that reduce competitive effects and (c) assessing the sustainability of these new systems in terms of nutrient removal.

Estimates of below ground biomass pools are critical to establishing carbon fluxes on regional scales which can then be applied in global modeling of climate change mitigation strategies. A new methodology for tree below-ground biomass estimation was developed, including a purpose-designed coring machine. Monte Carlo simulation was used to assess the accuracy of a range of sampling regimes through estimates of uncertainty (precision) and bias (error) and these sampling methods were subsequently used to develop allometric relationships to estimate the carbon mitigation potential of tree phases integrated into agricultural systems.

The implications of integrating tree phases into agricultural systems and the effects of

this on the sustainability of existing farming systems were investigated. This included an assessment of potential land use synergies targeting abandoned or marginal land for multiple land use outcomes via landscape rehabilitation and carbon mitigation. The integration of short tree phases (3 years) into low rainfall salinized farmland for the purpose of soil salinity amelioration was shown to have additive environmental benefits as a potential source of biomass feedstock for renewable energy. Allometric relationships were developed for three candidate species (*Eucalyptus globulus*, *E. occidentalis* and *Pinus radiata*) and their carbon storage was assessed based on whole tree destructive sampling, including below ground sampling. The biomass production for different planting density and landscape placement strategies, and for different tree components was estimated to assist in future development of harvesting systems and management of nutrient removal. It was shown that tree phases inserted into farming systems for the purpose of ground water control could potentially serve as a biomass feedstock for renewable energy, either bioenergy for power generation or as feedstock for lignocellulosic (second generation) biofuel, thus offsetting the use of non-renewable fossil fuel.

The sustainability of these systems was investigated to determine their impact on current farming systems and the potential removal of nutrients. Harvesting regimes that remove woody biomass while retaining leaves on site are likely to be more sustainable from a nutrient management perspective. A nutrient assimilation index was developed for these short rotation tree crop systems to aid the management of nutrient removal. The removal of nutrients via a short (3 year) tree phase was less than the cereal cropping systems currently in place and had potential to retrieve leached nutrients from deeper in the soil profile.

Planting of tree and shrub species in severely salinized abandoned farmland was shown to be a potential avenue for carbon mitigation, and a resultant positive land use change. With species selection, management of stand density and landscape position, tree growth and carbon sequestration can be manipulated with rates of sequestration of 1.1 to 2.3 t ha^{-1} yr^{-1} following 8 years growth in the highly saline environment. A combination of shrub (*Atriplex nummularia*) and tree (*Eucalyptus occidentalis*) species were used to mimic natural saline wetland succession and were successful in rehabilitating degraded farmland while effectively sequestering carbon and mitigating atmospheric CO_2.

The challenge remains to integrate these mitigation initiatives and systems into existing economic and social environments and for them to be accepted as typical economic activities. This is not only a challenge from the scientific view point, but encompasses social and political aspects which often makes its application difficult.

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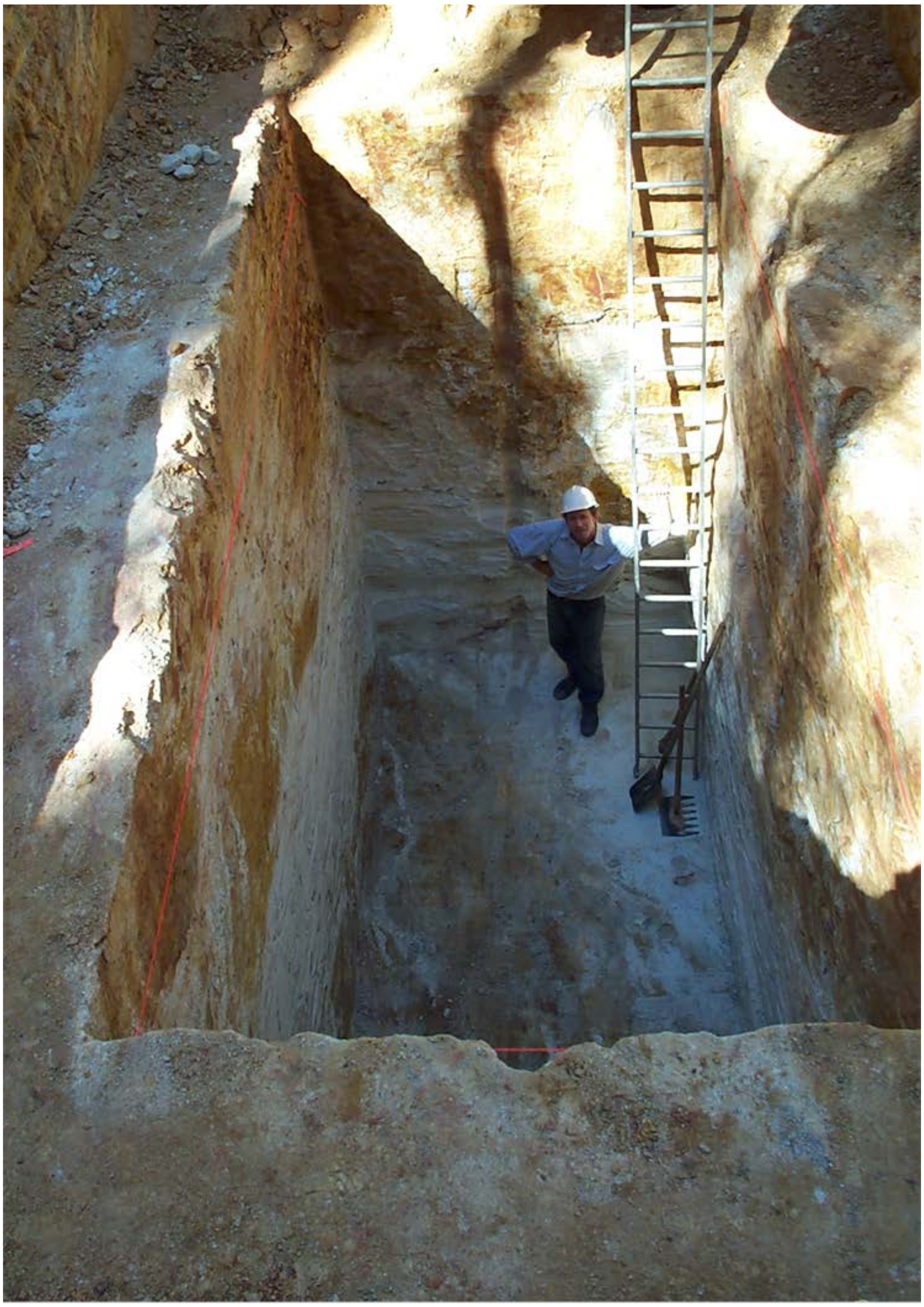

Publications from this thesis

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Acronyms and terms

ACCU	Australian Carbon Credit Units
AFOLU	Agriculture, Forestry and Other Land Use
AGB	Above Ground Biomass
AGO	Australian Greenhouse Office
BGB	Below Ground Biomass
C	Carbon
CEFC	Clean Energy Finance Corporation
CER	Clean Energy Regulator
CFI	Carbon Farming Initiative
CO_2	Carbon dioxide
COP-3	Third Conference of the Parties to the United Nations Framework Convention on Climate Change
CPM	Carbon Pricing Mechanism
CP1	First Commitment Period of the Kyoto Protocol
CP2	Second Commitment Period of the Kyoto Protocol
CV	Coefficient of Variation
CVI	Crown Volume Index

DAP	Direct Action Plan
DBH	Diameter at Breast Height
DOB	Diameter Over Bark
EDM	Edible Dry Matter
ERF	Emissions Reduction Fund
ETS	Emissions Trading Systems
FI	Fit Index
FOLU	Forestry and Other Land Use
GHG	Greenhouse Gas
I	Furnival Index
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
KP	Kyoto Protocol (to the United Nations Framework Convention on Climate Change)
Mg	Mega-grams **(**metric ton)
NAI	Nutrient Assimilation Index
NDCs	Nationally Determined Contributions
NLWRA	National Land and Water Resources Audit

NPP	Net Primary Production
LUC	Land Use Change
LULUCF	Land Use, Land-Use Change and Forestry
PFT	Phase Farming with Trees
R	Energy Efficiency Index
RE	Renewable Energy
REDD +	Reducing Emissions from Deforestation and Forest Degradation
RIRDC	Rural Industries Research and Development Corporation
r:s	Root to Shoot ratio
r^2	Coefficient of Determination
SOC	Soil Organic Carbon
SRF	Short Rotation Forestry
UNFCCC	United Nations Framework Convention on Climate Change

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1 General Introduction

1.1 Preamble

Over the past two decades the awareness and global acceptance of climate change has resulted in a concerted effort to mitigate climate change via a range of international treaties. This process, has been underpinned by the scientific assessments of the Intergovernmental Panel on Climate Change (IPCC) that first reported in 1990 (IPCC, 1990), and commenced in 1992 with the United Nations Framework Convention on Climate Change (UNFCCC) (Wilder and Fitzgerald, 2008). Ratifying parties of the UNFCCC treaty acknowledged global warming as a serious problem and became signatories with the aim to stabilize atmospheric greenhouse gas concentrations.

The Kyoto Protocol (KP), a legally binding agreement conceived during the third Conference of Parties (COP-3) of the UNFCCC in 1997, contains provisions by which signatories to the Protocol can reduce greenhouse gas emissions (UNFCCC, 1997). Articles 3.3, 3.4 and 3.7 of the KP relate to the land sector within the domain of land use, land-use change and forestry (LULUCF), and signatories were able to manage emissions via afforestation, reforestation and deforestation within Article 3.3 and forest management, vegetation, grazing land management and cropland management within Article 3.4. Under the provisions of Article 3.7, often referred to as the Australia clause (Hamilton and Vellen, 1999), Australia was able to meet its targets for the first commitment period (CP1) to the KP (2008 to 2012) largely through avoided deforestation being included into 1990 baseline emissions, despite increases in emissions in other sectors (Howarth and Foxall, 2010). Australia did not elect to include Article 3.4 for CP1 but has done so for the second commitment period (CP2)

(2012 to 2020). In the CP2, LULUCF has become agriculture, forestry and other land uses (AFOLU) with this taking a broader view of the land sector and including agriculture to avoid double counting in national carbon accounts (Smith et al., 2014). Whether emissions reductions can be achieved through land sector mitigation via AFOLU will depend on government implemented policy over this commitment period (van Oosterzee et al., 2014).

Forestry and agriculture play a key role in both adding and removing greenhouse gases from the atmosphere and thus feature in both national and international carbon mitigation strategies (Smith et al., 2014). Forestry and other land use (FOLU) is the most significant non-agricultural greenhouse gas (GHG) flux to the atmosphere and accounted for about a third of anthropogenic emissions from 1750 to 2011 and about 12% from 2000 to 2009 (Smith et al., 2014). The use of forests to mitigate climate change via reforestation or afforestation of agricultural land (carbon sinks or sequestration) and renewable energy systems (bioenergy and biofuels) has gained considerable attention (Canadell and Raupach, 2008; Chum et al., 2011; Malhi et al., 2002). Land sector mitigation, including bioenergy, is considered to have significant potential for climate change mitigation and could contribute 20 to 60% of total cumulative abatement to 2030 (Smith et al., 2014).

Short rotation tree crops have been successfully used for bioenergy production in several regions globally (Chum et al., 2011). However, although there is increasing interest in bioenergy with some 25 years of research and development in Australia, the application of biomass feedstocks for renewable energy, bioenergy production from dedicated tree crops is yet to be realized (Crawford et al., 2012; Mitchell et al., 2012). Carbon mitigation within the land sector via the integration of trees into farmland landscapes is challenging economically and socially. However, there is opportunity for these systems to become more diverse in their application and progress beyond integrated agroforestry (Lefroy and Rydberg, 2003) into "integrated food-energy" systems (Bogdanski et al., 2010). The integration of trees into dryland farming environments of southern Australia has been extensively examined over the past 2 to 3 decades as a response to dryland salinity (Stirzaker et al., 1999; Robinson et al., 2006; Sudmeyer and Hall, 2015) as this is a major water balance problem caused by deforestation in some regions.

Carbon mitigation from the land sector has the potential to address various land degradation issues in some parts of the world, effectively by providing the finance to pay for reforestation (Prance, 2002; Lal, 2004). When applied in the low rainfall region of southwestern Australia this could result in a range of environmental benefits such as a reduction in land salinity (Harper et al., 2007), a decrease in salt-load in reservoirs (Townsend et al., 2012), improved soil quality (Mendham et al., 2003), stabilization of land against erosion or the sustainability of land ecosystems (O'Connell et al., 2009) and enhancement or rebuilding of biodiversity (Steffen et al., 2009). However, reforestation also has the potential to produce a range of adverse impacts including competition for water resources (Jackson et al., 2005), competition with food production (Smith et al., 2014) and the reduction of biodiversity (Lindenmayer et al., 2012).

In Australia, the potential of carbon markets via the forestry sector was seen as an opportunity for abatement measures to help finance reforestation projects on farmland (Shea and Bartle, 1988; Mitchell et al., 2012; Polglase et al., 2013). Prior to Australia ratifying the KP in 2007, carbon offset companies operated in a voluntary carbon market and carbon offsets were accessible to individuals or businesses as means to offset emissions via carbon credits. These offsets were generally in the form of energy efficiency, renewable energy and forestry projects of which forestry was the most popular (Downie, 2007). In Australia, demand for forestry offsets peaked in 2012 in anticipation of an AU$23/t$CO_2$e fixed carbon price (Hamrick and Goldstein, 2015). However, Australia only briefly (2012 to 2014) introduced a Carbon Pricing Mechanism (CPM) which was removed with the repeal of the carbon tax in 2014. In 2015 there were 39 national and 23 sub-national entities globally, pricing carbon via Emissions Trading Systems (ETS) and carbon tax (World Bank Group, 2015), with offsets from forestry and landuse projects accounting for over half of the offsets traded in 2014 (Hamrick and Goldstein, 2015). In Australia, the CPM has been replaced with the Emissions Reduction Fund (ERF) within which abatement is purchased via reverse auction utilizing a $2.55 bn public fund. To date almost half of the fund has been used to purchase abatement with a large proportion of this purchasing avoided deforestation, rather than reforestation projects (Australian Government, 2016).

1.2 Thesis research questions

A range of issues remain before trees can be integrated into low rainfall landscapes to mitigate carbon, and these form the research questions in this thesis:

a) **Can more efficient methods of estimating tree root carbon be developed?**

Carbon mitigation projects often use default values to estimate tree root biomass and carbon storage (Ravindranath and Ostwald, 2008). Methodologies used to measure tree root biomass are varied and the comparative precision of estimates of these different methodologies is unknown. A major limitation is in the cost of gaining field measurements of roots and as a consequence, simple default values are often used to estimate root carbon (Mokany et al., 2006). Not all carbon in forestry projects is accounted for and thus a new apparatus to sample roots was developed (Chapter 3). From detailed measurements of tree roots a range of sampling scenarios were simulated to determine levels of sampling uncertainty (precision) and bias for a range of methodologies for below ground biomass (and thus carbon) estimation (Chapter 4).

b) **How much carbon mitigation can be achieved with new agroforestry production systems that reduce competitive land use?**

Much of the work to mitigate carbon in inland Australia has been with *Eucalyptus* agroforestry (mallee) systems (Wu et al., 2008; Bartle and Abadi, 2010). However, these systems can compete with food production (Sudmeyer et al., 2012) both through direct displacement of land and competitive effects with crops. Two alternative approaches are the use of tree phases in rotation with agriculture (Harper et al., 2000) (Chapter 5) or the use of less productive or abandoned farmland (Chapter 6). The amounts of carbon mitigation for both systems were investigated. For short rotation tree crops, three candidate species were chosen: *Eucalyptus globulus*, *Eucalyptus occidentalis* and *Pinus radiata*; and for abandoned farmland: *Eucalyptus occidentalis* and *Atriplex nummularia*. In both cases, trees were established in replicated experiments and allometric relationships for whole tree (including roots) and tree component biomass developed. The potential for carbon mitigation (either carbon sequestration or bioenergy) as a result of different planting strategies (tree planting density and landscape location) was also assessed.

c) **What is the sustainability of new agroforestry short-rotation production systems in relation to nutrient removal?**

Nutrient export from short rotation tree crops is an important factor for the long term sustainability of tree phases in low rainfall agroforestry systems (O'Connell et al., 2009; Mendham et al., 2014). Component allometric relationships developed for *E. globulus E. occidentalis* and *P. radiata* were applied to determine component nutrient export as a result of biomass removal for potential renewable energy feedstocks (Chapter 7). An index associating biomass yield and nutrient assimilation of whole tree and tree components was developed to apply as a guide for managing nutrient sustainability in biomass systems in low rainfall farmland environments.

This thesis investigates the above questions through three separate but interlinked research projects (Table 1.1) resulting in a series of published manuscripts, which form the majority of the thesis (Figure 1.1). Lastly, Chapter 8 provides and considers the broader context and its application to global climate change on land-based mitigation strategies.

Table 1.1 Research questions, associated research projects and thesis chapters.

Research question	Research project	Thesis chapter
Can more efficient methods of estimating tree root carbon be developed?	***Project one:*** The development of new techniques for tree root biomass sampling and the comparison of the utility of existing below ground tree root biomass sampling methodologies. *Eucalyptus globulus* trees were intensively sampled via volumetric soil coring and bulk excavation and the resultant root mass data were was modelled to simulate different tree root biomass sampling strategies. A range of tree root sampling scenarios were simulated and the outputs generated were used to compare uncertainty (precision) and bias between these sampling strategies.	**Chapter 3:** Developing a specialised soil corer for sampling tree roots. **Chapter 4:** Accuracy of tree root biomass sampling methodologies for carbon mitigation projects.
How much carbon mitigation can be achieved with new agroforestry production systems that reduce competitive land use?	***Project two:*** Tree phases (three to five years) of fast growing tree species have been proposed as a means to ameliorate dryland salinity via the removal of excess soil water. The aim of this trial was to investigate the effect of: a) tree species and planting density in relation to biomass potential for renewable energy feedstocks and b) the nutrient removal was also investigated to determine the sustainability of these tree phase systems.	**Chapter 5**: Estimation of woody biomass production from a short rotation bio-energy system in semi-arid Australia. **Chapter 7**: Nutrient exports from a short rotation energy cropping system.
What is the sustainability of these short-rotation production systems in relation to nutrient removal?	***Project three:*** Various tree species were strategically planted throughout a small sub-catchment including the catchment discharge zone which had become abandoned unproductive farmland. The aim of this trial was to determine the potential of abandoned saline farmland as an option for carbon mitigation via "analogue forestry" with appropriate salt tolerant shrub and tree species.	**Chapter 6**: Bio-mitigation of carbon following reforestation of abandoned salinized farmland.

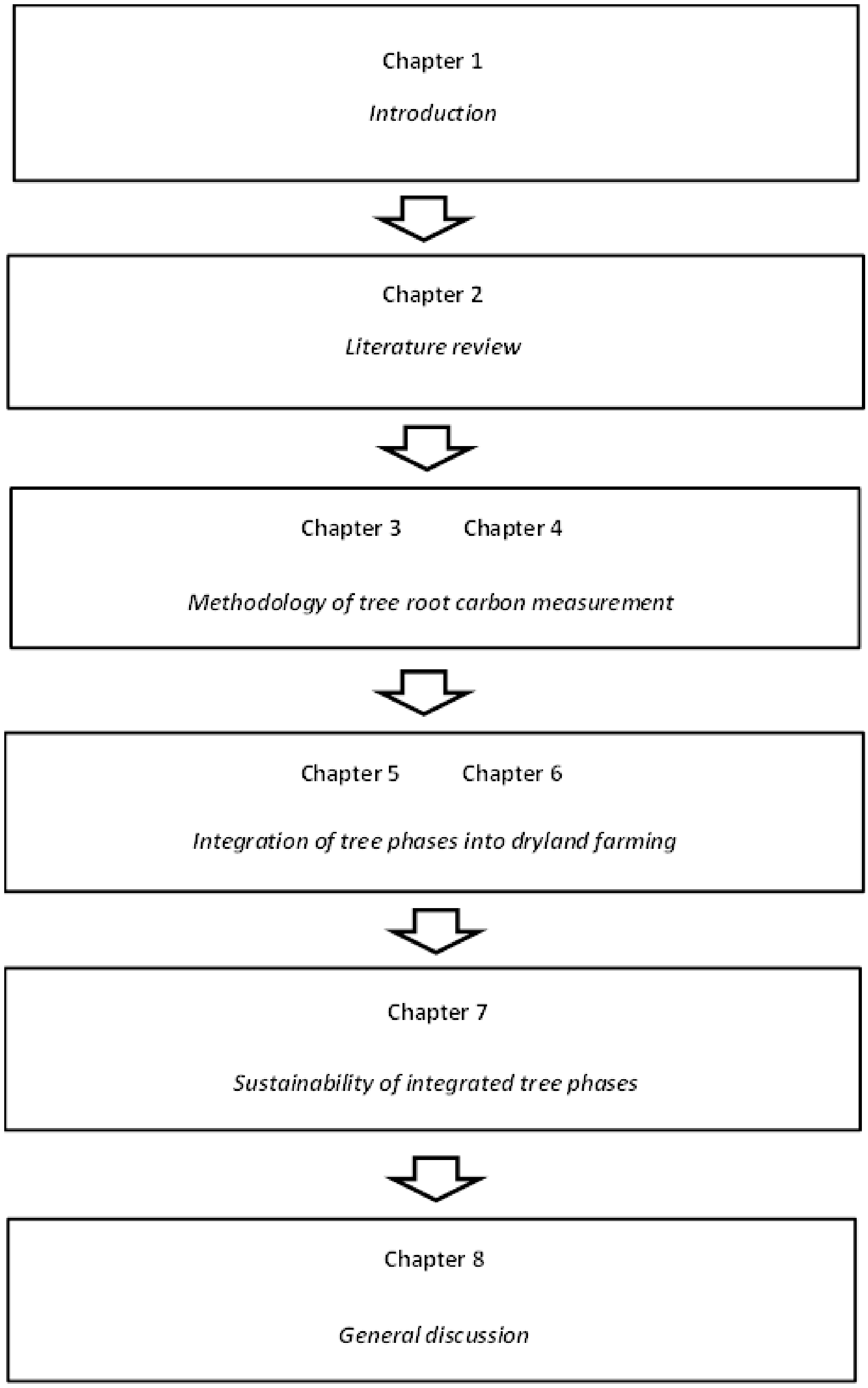

Figure 1.1 Roadmap of thesis chapters and interrelationship of research topics.

2 Literature review

2.1 Introduction

As described in the General Introduction, the land-sector and in particular reforestation is a key component of a range of national and international carbon mitigation strategies (Smith et al., 2014). This literature review will examine a range of issues related to the reforestation of low rainfall farmland landscapes for the purpose of carbon mitigation. In particular, it will examine:

- Carbon mitigation using forests,
- Methods of forest carbon measurement and carbon accounting,
- The application of agroforestry and forestry systems to achieve landscape-scale co-benefits from carbon mitigation, with particular reference to the agricultural systems of southern Australia, and
- Issues related to the sustainability of agroforestry systems used for carbon mitigation.

2.2 Climate change

Anthropogenic emissions have changed the concentrations of a range of atmospheric greenhouse gases resulting in global climate change (IPCC, 2013). Attempts to mitigate climate change has been the focus of a multi-national effort to address this via the UNFCCC. Terrestrial ecosystems sequester approximately 30% of anthropogenic emissions (Luo et al., 2015) and the management of the terrestrial biosphere via reforestation (IPCC, 2006) is seen as an option for mitigation of atmospheric carbon. In Australia the land sector contributes approximately 25% to national emissions (Australian Government, 2013) however, forests integrated with the land sector have the potential to sequester and store significant amounts of this carbon (Mitchell et al., 2012; Polglase et al., 2013).

2.3 Carbon mitigation

Globally, forests cover 30% of the Earth's land surface or approximately 4 billion hectares, and make up 40% of global carbon stores of which 20 to 40% is made up of roots (Brunner and Godbald, 2007; Mokany et al., 2006). Forests contribute significantly to global carbon sinks and fluxes (Pan et al., 2011) and the below ground carbon pool in a forest ecosystem often exceeds the above ground carbon pool (Finér et al., 2011). The potential of forests to sequester carbon is seen as a means to reduce atmospheric CO_2 and to offset fossil fuel emissions by providing a source of biomass for renewable energy (bioenergy) and fuels (biofuels) that will potentially impact on global climate change.

There are three main approaches in using forests to mitigate atmospheric carbon dioxide and thus potentially impact on global climate change: (1) the storage of carbon dioxide in plant biomass and soils (sequestration, "sinks") (Peichl and Arain, 2007), (2) the substitution of fossil fuel use with biomass from forests (bioenergy, biofuels) (Chum et al., 2011) and (3) the preservation of existing carbon stocks in forests (avoided deforestation) (Canadell and Raupach, 2008; Smith et al., 2014). This thesis will examine carbon sequestration and bioenergy.

2.3.1 Carbon sequestration

Carbon sequestration refers to the process of capturing CO_2 from the atmosphere and this can be achieved via several pathways, including bio-sequestration or uptake by vegetation and soil in terrestrial systems. Forest biomass consists of a range of components which are broadly classified as above ground (vegetation) biomass, below ground (roots) biomass and soil organic matter (Pan et al., 2011). Tree biomass consists of above and below ground biomass. These two categories can be divided further into tree components: above ground biomass consisting of stems, branches, twigs, leaves, bark and litter or necromass and below ground biomass consisting of the root bole, large, medium and fine roots. The estimation of carbon within these components is crucial to determining carbon fluxes and balances within a forest bio-sequestration system (Keith et al., 2010). Terrestrial sequestration, the storage of carbon by plants as above and below ground biomass also includes input into the soil carbon pool.

Forests as carbon sinks are typically considered to have a limit to their mitigation potential or "carbon carrying capacity" (Keith et al., 2010; Nabuurs et al., 2013) however, Luyssaert et al. (2008) examined data from 519 study plots for trees ranging in age from 15 to 800 years and found a positive net carbon balance for forests 200 years and above, with an average accumulation rate of 2.4 ± 0.8 C t ha^{-1} yr^{-1}. Mature forests have lower Net Primary Production (NPP) but a greater carbon store in contrast with young forests (reforestation) which tend to have a small overall carbon sink but have greater NPP (He et al., 2012).

2.3.2 Afforestation/reforestation

Afforestation and reforestation both pertain to the establishment of trees on unforested land (IPCC, 1996b). Afforestation refers to land which has historically been without trees, whereas reforestation refers to land which recently supported forests but has been cleared (IPCC, 2007). Both processes result in human induced conversion of non-forested lands to forested lands (UNFCCC, 2008). In addition to afforestation and reforestation, revegetation projects can comprise of species that do not meet the definition of a forest, and thus fall within the Kyoto definition of "revegetation" and encompass grazing plants such as saltbush (*Atriplex* species). The use of perennial shrubs to address environmental degradation in particular, dryland salinity (Norman et al., 2008) is a primary driver for revegetation in southern Australia (Barrett-Lennard, 2002; Barrett-Lennard et al., 2013; Hobbs et al., 2013).

These categories can be used in either block plantings or in agroforestry systems, and as a result agroforestry systems have been incorporated into traditional farming enterprises in anticipation of a potential opportunity for carbon credits (Shea et al., 1998). More recently in Australia, the introduction of the Carbon Farming Initiative (CFI) was seen as an opportunity for land owners to benefit financially from the application of trees for environmental restoration (Australian Government, 2011).

2.3.3 Agroforestry

Agroforestry has been defined by Lundgren and Raint (1983) and also referred to as farm forestry (Powell, 2009) in Australia, and has been introduced into agricultural landscapes in an attempt to support and sustain farming systems, through the deliberate establishment and/or management of trees in agricultural landscapes (Nuberg et al., 2009). Environmental issues as a direct consequence of deforestation for agriculture, in particular soil salinity, have been addressed with the use of perennial shrubs (Lefroy and Stirzaker, 1999; Barrett-Lennard, 2002) and trees (Cooper et al., 2005; Harper et al., 2014). Agroforestry can be undertaken at a range of scales (environmental plantings) and forms (belts or blocks) and is differentiated from industrial plantation schemes (Schirmer and Bull, 2014). These agroforestry systems also have the potential to mitigate carbon via carbon sequestration, adding to forest carbon sinks in national carbon accounts. The evolving carbon market has the potential to supplement budgets for environmental restoration (Harper et al., 2007; Hein et al., 2013) and for mitigation (Bateman et al., 2013).

2.3.4 Bioenergy

Forests can also contribute to carbon mitigation through the production and utilization of biomass ("bioenergy") by substituting for fossil fuel use (Chum et al., 2011; Canadell and Raupach, 2008). Woody biomass is a potential renewable resource with multiple applications; these include timber for construction, feedstock for the paper and pulp industry, feedstock for renewable energy for the bioenergy and biofuel industry and environmental security (Hinchee et al., 2009). The amount of carbon removed from the atmosphere by forests can be increased if biomass is used for renewable purposes, for example, renewable energy to off-set fossil fuel use. Biomass for renewable energy can potentially have a greater mitigating effect of atmospheric CO_2 via "cascade utilization" of biomass (Haberl and Geissler, 2000), where unused or waste biomass is utilized to reduce NPP appropriation

World-wide, photosynthesis produces approximately 220 billion tons (dry weight) of biomass per year and as an energy source, this represents some ten times the world's current energy use (Stucley et al. 2004). The use of short rotation tree crops (SRC) for bioenergy is not new. In the northern hemisphere the use of short rotation tree crops for energy generation is common and is summarized in the Econ Pöyry (2008) report. For example, in Sweden the use of bioenergy increased from 10% of total gross inland energy consumption in the 1980s to 19% in 2006. Swedish bioenergy primarily originates from the forestry sector, which accounts for approximately 90% of the bioenergy used. The development of the bioenergy sector in several northern European countries show strong positive trends. Between 1992 and 2004 bioenergy used in electricity production in Denmark increased by a factor of 7, in Sweden bioenergy increased by a factor of almost 9 between 1992 and 2006 and in Finland bioenergy more than doubled up to 2004 and bioenergy also increased significantly in Norway from 2002 on-wards (Econ Pöyry, 2008).

In the United States and Canada hybrid poplar is the most common species used for short rotation forestry (SRF) typically grown in 10-15 year rotation for the paper and pulp industry (Samson et al., 1999). In the USA, attention has been focused on fast growing short-rotation woody crops such as *Populus*, *Salix*, and *Eucalyptus* and their respective hybrids (Hinchee et al., 2009). Considerable research effort is being devoted to tree genetics and silvicultural practices, including genetic improvement of native Populus species (Polle et al., 2006), introduced hybrid eucalypts for greater biomass yields (Stricker et al., 2000) and altering wood quality to improve feedstock conversion efficiency (Weng et al., 2008; Warden and Haritos, 2008).

In Australia the bioenergy potential from the land sector is yet to be developed. For example, Australia's electricity generation from bioenergy is predicted to be 4% in 2020, well below the 14% benchmark already being achieved by European countries and of this just 2% will be from energy crops (CEC, 2008). Other studies indicate that 20% of current electricity production could be supplied via bioenergy by 2030, including contributions from short rotation tree crops (Farine et al., 2012). The areas of land that could be used for bioenergy crops is potentially vast, if trees are integrated with agriculture to address land degradation associated with clearing for agriculture (Powell, 2009). Additional to this are opportunities of integrated systems with existing agriculture which are yet to be explored. Significant energy returns, or the ratio of energy output/input from mallee crops were calculated by Wu et al. (2008) with energy returns (R) of 41.7, with indications that this system has the potential to offset fossil fuel use and improve GHG balances.

2.4 Forest carbon measurement

The total global forest carbon stocks including soil (to 1 m depth) are estimated at 861 Pg C, and as a net annual (C flux) sink this equates to 1.2 Pg C year^{-1} over the period 2000 to 2007 (Pan et al., 2011). Following the approved methodological guidance by the IPCC (IPCC, 2007), parties to UNFCCC are required to provide periodic estimates of GHG emissions, however, applying these measurement guidelines will be challenging and the estimates made may vary depending on methodology. Petrescu et al. (2012) compared the IPCC GPG 2003 (IPCC, 2003) and the IPCC AFOLU 2006 (IPCC, 2006) estimates of C stock changes in living forest biomass and found at the global level, results obtained with the two sets of IPCC guidance differed by about 40%, due to different assumptions and default factors. The accuracy of C stock estimates will be significantly affected by methodology and the use of vegetation specific data (as opposed to default values) which can only be achieved with actual vegetation measurement and derivation of vegetation specific C stock estimators or functions. Central to any assessment of forest carbon stocks is that the biomass of individual trees is determined in the field and the edifice of any forest carbon inventory is the collection of empirical data (Picard, 2012).

2.4.1 Above ground biomass

In forestry, tree diameter and tree height are commonly measured variables in forest inventory, and biomass predictions have been attempted via relationships between timber volume and biomass, referred to as expansion factors (Snowdon et al., 2002). However, expansion factors can be unreliable and can vary considerably (Johnson and

Sharpe, 1983) with a range of environmental and growth attributes. For example, the ratio of stem mass to above ground biomass for *Eucalyptus grandis* varies with age and therefore an expansion factor for this species would not be constant for trees of different ages (Bradstock, 1981). González-García et al. (2013) found allometric relationships to be more reliable than biomass expansion factors for predicting above ground biomass in a study of *Eucalyptus nitens* stands in northwest Spain. Allometric equations have frequently been applied in forestry and specifically to predict wood production, however, interest in biomass for bioenergy resulted in allometric equations being developed for whole trees and tree components (Picard, 2012).

Allometry for biomass estimation relates independent variables, typically tree diameter and tree height, to predict tree biomass and is based on geometric similitude, where geometry and shape are conserved even though organisms differ in size (Niklas, 1994). Whittaker and Woodwell (1968) first applied this as dimension analysis. Allometric equations are derived by regression analysis using various equation forms (Clutter et al., 1983) and the accuracy of different equation forms (Baskerville, 1971) can account for a high proportion of the observed variance (Madgwick, 1994). Allometric relationships developed for a given tree species need to be robust and should include measurement data from at least 20 trees, a range of tree sizes and site types representative of the species range (Snowdon et al., 2002). Roxburgh et al. (2015) examined the efficiency of various regression forms on the precision of estimates in relation to sample size and found this varied considerably with 17 to 95 sample trees required for the best performing allometric equations whereas for poorer performing allometric forms much larger sample sizes of 25 to 166 trees were required to attain a biomass estimate with a standard deviation of within 5% of the mean. Levels of precision have been prescribed within formal carbon management schemes for carbon abatement projects. In Australia within the Emissions Reduction Fund (ERF) an allometric function or regression relationship must be statistically significant ($p<0.05$) and achieve a coefficient of determination (r^2) no less than 0.75 (Australian Government, 2014a). Given the potential for carbon trading the estimation of carbon to given levels of precision is not only a regulatory requirement but also essential for market confidence.

In Australia an early review (Eamus et al., 2000) of allometric equations for carbon estimation in tropical northern Australia found that a single allometric equation “adequately” described several dominant species within ecosystems, but not across ecosystems. In that review it was shown that for several dominant species *ln* (DBH) and *ln* (biomass) were highly correlated but regression slopes differed for the same species at different sites. Generalized equations across different species gave errors ranging from an under estimation of 11% for *Eucalyptus crebra* to an over estimate of 42% for *Eucalyptus populea*. The effectiveness with which generalized equations predicted biomass over several species varied over different regions, highlighting the need for species and region specific equations and the need for adequate data sets. Large trees can have a significant influence on regression equations and subsequent coefficients of determination and give large weightings to large values (Overman et al., 1994). Paul et al. (2013) examined generalized equation forms incorporating the effects of genus and growth habit and found the percentage error of biomass prediction to be high (45%) for a given site, however, it was relatively low (<11%) when applying generalized equations to regional or estate level estimates across a range of sites. Site and species specific allometric equations for biomass prediction are the most accurate for site based predictions (Paul et al., 2013). More recently Paul et al. (2015) compared generalized biomass models based on approximately 15,000 individual tree and shrub measurements and found generalized model predictions of carbon across eco-regions resulted in mean absolute prediction error of only 13% for stand based biomass estimates.

2.4.2 Below ground biomass

Below ground biomass or tree root mass biomass is challenging to estimate and allometric equations published for root biomass have varied with the methodology employed (Ash and Helman, 1990; Ritson and Sochacki, 2003; Jonson and Freudenberger, 2011; Paul et al., 2014b). The derived allometric relationships can vary in their predictive confidence of root biomass and the methodology applied is also quite variable as highlighted in a recent study by Paul et al. (2014b). The difficulty in obtaining root biomass data and subsequent lack of allometric relationships has resulted in the application of root to shoot ratios (r:s) (Mokany et al., 2006; Kuyah et al., 2012). Mokany et al. (2006) reviewed r:s ratios and found "vegetation" specific r:s ratios were more accurate than generalized relationships for predicting root biomass and the reliance on generalized relationships was not recommended for differing forest and woodland types. Paul et al. (2013) grouped vegetation types into four categories to test generic allometric equations for root biomass and found model efficiencies between 0.64 – 0.90.

2.4.3 Tree root sampling methodology

For the purpose of abatement projects, root mass not accounted for will be a loss of potential income. Similarly, for national carbon accounts of carbon sinks and their potential mitigating effect, accounting for all carbon is crucial. Inconsistences in methodology as a result of different sampling protocols applied in root studies limits the application of data sets for developing generalized equations (Paul et al., 2014b).

Comparison of studies and collation of data sets becomes difficult when attempting to develop larger sample sizes for generalized equations, and inconsistencies relate to either the amount of root mass sampled or the measurement of independent variables that will be applied for regression equations.

The area allocated for a tree plot for excavation differs between studies in two distinct ways; either a set distance from the tree is excavated regardless of the association of the sample trees to the remainder of the stand (Sudmeyer and Daniels, 2010; Jonson and Freudenberger, 2011) or plot sizes are determined by mid-points between trees which takes into account the stand density (Ritson and Sochacki, 2003; Resh et al., 2003). In the former, sampling will only recover a portion of the root mass and will not account for roots extending beyond the sampling zone, however for plots determined by mid points between trees, all the root biomass within a stand is estimated based on a depth limit. Other differences relate to the parameters measured as predictors for the development of regression equations. Measurement height of stem diameter is variable and can be dependent on tree age and tree form. For example, very young trees are often measured below the forestry standard of Diameter at Breast Height (DBH) (Paul et al., 2014a). Another inconsistency relates to the treatment of the" stump" or portion of stem which is typically attached to the root bole following tree harvest. These inconsistences are addressed in a study by Paul et al. (2014b) in which data sets were combined for the development of generalized equations for root biomass.

2.4.4 Soil carbon

Forest carbon stocks represent a significant C sink which removes atmospheric CO_2 and plays an important role in the ability of forest soils to sequester C (Schlesinger, 1990). The available data on tree root biomass and soil carbon is dearth (Zerihun et al., 2006; Pregitzer et al., 2002). For many species used for forestry and energy crops, the long term effect of tree roots on soil carbon has been inadequately researched (Walmsley et al., 2009b). Land use change (LUC) is occurring globally and soil organic carbon (SOC) is a significant component of terrestrial carbon stocks which can potentially be a sink or source of atmospheric carbon (C) (Mendham et al., 2003; Lal, 2004). The depletion of SOC by 20 to 50% has been associated with the removal of forests for the establishment of agricultural land (Post and Kwon, 2000; Davidson and Ackerman, 1993; Lal, 2008). Establishing bioenergy crops on agricultural land is another option for increasing soil carbon (Tolbert et al., 2002) however, the interaction between tree root pools and soil carbon pools is not fully understood and studies have shown contradictory results in ascertaining whether reforestation significantly increases soil carbon pools (Cowie et al., 2006). For example, Akala and Lal (2001) report increases in SOC stocks as a result of reforestation of abandoned marginal land in Ohio, USA and Schauvlieghe and Lust (1999) report an almost two-fold amount of total carbon in a 69 year old forest stand compared to pasture. Yet many paired studies are contrary to the above trends (Guo et al., 2008; Harper et al., 2012). Harper et al. (2012) showed no difference in soil C after 26 years following reforestation of farmland in southwestern Australia. Soil C pools have been shown to decrease following a LUC from pasture to pine plantations (Guo and Gifford, 2002; Turner and Lambert, 2000; Scott et al., 2006). These contrasting findings are further confounded by the lack of standardized sampling protocols for soil C (Lal et al., 2001; McCarty et al., 2010) and the interpretation of bulk density (BD) values when land use and management changes are applied to improve soil carbon stocks (Lee et al., 2009; Throop et al., 2012; Wuest, 2009).

2.5 Application of tree crops

In Australia, economies of scale do not allow for bioenergy systems to be established easily, as a low population density and large distances between populated areas make these systems difficult to implement. The pulp wood industry can be an economic impetus for tree plantations in high rainfall zones (Harper et al., 2009), but in low rainfall areas environmental issues such as dryland salinity have provided some environmental additionality to enable reforestation (George et al., 2012). Over the last two decades pulpwood production in southern Australia has become a recent major industry, with tree crops being established on agricultural land, thus constituting a major LUC from agriculture to forestry (Grove et al., 2001). Although these are not used for bioenergy, they do have some mitigation potential as carbon sinks and are modeled to contribute as much as 23 Mt CO_2–e yr^{-1} emissions reductions by 2030 (CCA, 2014). In southern Australia positive benefits of this LUC have included the reversal of land degradation, improved water quality, diversification of farm income and the potential of carbon mitigation (Harper et al., 2007). The potential of trees for carbon mitigation is now being realized and applied to address environmental issues via reforestation and afforestation (Harper et al., 2007; Mitchell et al., 2012).

Short rotation tree crops have several inherent environmental advantages over conventional agriculture for carbon storage. Greater amounts of carbon can be stored per unit area of land in above ground biomass and in tree root systems. Trees can be grown on marginal lands without fertilizer input and tree crops for bioenergy and biofuel are a renewable source of energy and can displace fossil fuel use and also provide environmental benefits (Harper et al., 2007; Chum et al., 2011). In Australia, avenues for mitigating atmospheric CO_2 via afforestation or reforestation are forced to operate within economic and environmental constraints (Mitchell et al., 2012). The cost and subsequent revenue from tree crops is a key issue and the economic gain to grow trees as a stand-alone crop is often absent in low rainfall regions of southern Australia (Polglase et al., 2008). However, linked with environmental needs and the prospect of carbon markets, mitigation of atmospheric CO_2, via tree crops may become a viable proposition (Bustamante et al., 2014).

2.5.1 Environmental drivers for reforestation

Globally soil salinity is estimated to effect up to 960 M ha across different biomes and agricultural systems (Wicke et al., 2011). In Australia dryland salinity is predicted to affect 17 M ha by 2050, 70% of which is in Western Australia. Clearing of deep-rooted native vegetation for agriculture has contributed to excess groundwater and subsequent soil salinity (Ferdowsian et al., 1996; Clarke et al., 2002), resulting in the loss of productive agricultural land and other adverse environmental consequences, including the threat to conservation reserves and water quality. It is generally accepted that the retention of existing vegetation and the replacement of trees back into the landscape are essential parts of the solution (Stirzaker et al., 2002) although Hatton and George (2000) suspect otherwise. Revegetation with deep-rooted perennials can have positive effects on removing excess soil moisture and subsequent lowering of water tables (George et al., 1999).

The reforestation of large proportions of farmland in Australia for salinity control is unlikely as this would conflict with income-generating activities from farming, and therefore, the application of other regimes of either tree crop rotations or permanent tree alleys or blocks have been recommended (Lefroy and Stirzaker, 1999). The success of these is dependent on a range of factors including species, landscape position and rotation length. While some prognoses for the role of trees in water balance restoration may seem pessimistic (Hatton et al. 2003) other reports have shown positive results (Robinson et al., 2006). Again, the scale of the response to revegetation is dependent on catchment hydrological characteristics (Benyon et al., 2006).

Although it is generally accepted that the application of trees in upslope or recharge zones has a greater effect on groundwater control (George et al., 1999; Harper et al., 2001), plantings on lower slope or discharge zones in low rainfall regions are often affected by water logging and high salinity levels and the use of salt tolerant species is preferred (George et al., 1999; Archibald et al., 2006). The choice of suitable species for the reclamation of discharge zones is necessary to ensure species survival, high growth rates and subsequent water use (Marcar et al., 2003). Although in southern Australia many of the salt affected areas are in low rainfall (~300mm) zones, saline discharge zones have an inherent advantage for tree growth due to excess water, provided the correct species are applied.

In Western Australia oil mallee crops have been incorporated with traditional farming practices in an attempt to ameliorate soil salinity and provide other outputs including eucalyptus oil, activated charcoal and biomass feedstock for energy production (Bartle and Abadi, 2010). However, bioenergy production is yet to come to fruition after some 25 years of research and development (Mitchell et al., 2012).

Phase farming with trees has been proposed (Harper et al., 2000) as a means to restore landscape hydrology, the cause of dryland salinity in the low rainfall zones of southern Australia. The concept relies on the use of high density plantings of high water use species in short (3-5 years) rotations to remove excess soil moisture and also provide biomass for potential bioenergy or biofuel feedstock (Harper et al., 2010).

Unlike permanent mallee belts, tree crop rotations or phases can be incorporated into existing farming systems without the permanent displacement of productive farmland, and this potentially increases the sustainability of present agricultural systems through the lowering of water tables (Harper et al., 2014), removing excess nutrients (Mendham et al., 2012) and improving soil quality via organic input (Lal, 2013). In such a system salinity mitigation and carbon mitigation will be addressed concurrently and with a carbon trading scheme in place, income from carbon credits could potentially underwrite the cost of establishment (Harper et al., 2007).

2.5.2 Competitive effects of reforestation

There is potential for integrating trees into farmland landscapes which can enhance productivity (Lefroy and Rydberg, 2003) and improve the sustainability of land use systems (Mendes et al., 2015). However, reduced yields and competition with adjacent crops and pastures have been reported for oil mallee systems, (Sudmeyer and Flugge, 2005; Sudmeyer et al., 2012) and with other reforestation programs, there are also concerns about the displacement of rural communities (Schirmer et al., 2005) and competition for food (Smith et al., 2013) and water (Jackson et al., 2005). There have also been active political debates related to carbon mitigation projects displacing farming (Mitchell et al., 2012) particularly after analysis by Polglase et al. (2013) suggested that carbon reforestation was possible over several million hectares of Australian farmland.

2.5.3 Sustainability

For bioenergy or biofuel production to be sustainable, these systems should not increase net CO_2 emissions or adversely affect the environment and food security (IPCC, 2011). The use of farmland for carbon mitigation would result in a LUC potentially affecting food production and therefore abandoned or marginal land would be a preferred option (Smith et al., 2013). The use of marginal or abandoned land has already been investigated and a significant potential identified (Gelfand et al., 2013; Gutierrez and Ponti, 2009; Lewis and Kelly, 2014; Ghezehei et al., 2015; Wicke et al., 2011).

A major problem in Australia is the degradation of land as a result of over-clearing native vegetation for the establishment of farmland and unsustainable agricultural practices (SCARM, 1998). The returns from carbon sinks have the potential to financially facilitate revegetation projects for land restoration in relation to soil and water degradation (Harper et al., 2007). Short rotation energy crops have the potential to become a new commercial enterprise and provide sustainable sources of biomass (Harper et al., 2010; O'Connell et al., 2007). Management practices would have to include the recycling of nutrients (Vance et al., 2010). Many studies report the global potential of biomass for bioenergy (Field et al., 2008; Campbell et al., 2008; Hoogwijk et al., 2003) but whether these potentials are sustainable warrants investigation. The sustainability of these systems is increasingly being reviewed with many criteria being considered in relation to sustainability certification (Scarlat, 2011). Sustainability criteria invariably include soil factors such as soil fertility and nutrient removal (Van Stappen et al., 2011; Buchholz et al., 2009; Haberl et al., 2010).

2.5.4 Nutrient removal

The efficient use of nutrients will be paramount to sustainable short rotation energy crops and the predicted global yields in these studies may not be sustainable without nutrient input (Reijnders, 2006). High growth rates can be achieved on agricultural land as a result of a long history of phosphate application. There is evidence that tree crops have the potential to recycle leached nutrients (Mele et al., 2003) however, the sustainability of several rotations is of concern (Reijnders, 2006), particularly tree phases of young trees which have relatively high nutrient concentrations compared to mature forests (Rytter, 2002). Nutrient status of the soil and the retention of harvest residue will need to be monitored to determine the long term effects of multiple crop rotations on N and P and exchangeable cations (Grove et al., 2001).

Nutrient removal by tree crops has received considerable attention along with scrutiny of the C balance of these systems (Grove et al., 2007). Nutrient use efficiency is paramount to the sustainability of energy crops (Safou-Matondo et al., 2005; Wang et al., 1991); in contrast, species with low nutrient use efficiencies have application for the removal of excessive nutrients from effluent sites (Guo et al., 2002; Guo and Sims, 2002). In southwestern Australia, concerns about the sustainability of forest crops have focused on the depletion of soil nutrients in general and more specifically, soil organic matter and nitrogen contents following harvest of pulpwood plantations such as *Eucalyptus globulus* on a 10 year rotation (Mendham et al., 2004).

2.5.5 Energy crops and water use

Opportunities for bioenergy are promoted via agroforestry however, biofuels and associated competition for water resources have received particular attention. Water use for agriculture accounts for 70% of global freshwater use (Otto et al., 2011) and the increasing demand on biomass from food stocks will result in greater pressures on water resources. It is predicted that biofuel production between 2005 and 2030 could increase by four times, with serious implications for water resources (De Fraiture et al., 2008). There have been rapid increases in ethanol and biodiesel production in Brazil and the USA (Warden and Haritos, 2008). Water use for bioenergy is quantified as a means to manage water use (Otto et al., 2011), water volume per unit of bioenergy produced or water efficiency. Water use per litre of fuel produced ranges from 90 litres of water for rain-fed sugarcane in Brazil to 3500 litres of irrigation water in India (De Fraiture et al., 2008). Water quality is also affected as a result of bioenergy production and this can occur throughout the entire production chain beginning during feedstock production via pesticide and fertilizer use (Bioenergy, 2011).

Producing biofuels from perennial crops via lignocellulosic pathways is an option which is not associated with high water demands and requires less pesticide and fertilizer use (IEA, 2011). Perennials are more likely to hold soil in place and reduce the likelihood of erosion and subsequent sedimentation of waterways. Water use efficiency of lignocellulosic feedstocks also surpass those of food-crop ethanol production (NAP, 2007).

Contrary to the above, some research relating to water use in salinized farmland environments indicates that there may be synergistic opportunities between water and biomass production in dryland farming systems (Crosbie et al., 2008). Oil mallee systems (Bartle et al., 2007) and tree phases (Harper et al., 2001) take advantage of excess landscape water in salinized farmland landscapes and could potentially be utilized for biomass production. Despite these areas having low (~300 mm) rainfall, the accumulation of soil water in the profile is an inherent advantage for biomass tree crops and potentially accommodate renewable energy systems without displacing food production.

2.5.6 *Landuse change*

Deforestation for agriculture is responsible for 15% of global emissions of GHG (Berndes et al., 2011) and further deforestation for the purpose of bioenergy crops was found to show a net negative effect in relation to the GHG balance of this scenario (Schulze et al., 2012). Globally, < 1% of agricultural land is utilized for cultivating bioenergy crops however there are concerns over negative LUC as interest mounts in bioenergy (Berndes et al., 2011; Smith et al., 2013). Population growth and consumption trends impose pressure on available land for food production and without policy measures, bioenergy use may impact on biodiversity, soil and water without delivering a net GHG benefit (Gawel and Ludwig, 2011).

Reforestation may however result in negative LUC if competition between carbon mitigation and other land-uses result, in particular if food production is affected (Searchinger et al., 2008; Fischer, 2009). The potential of this to occur is likely to increase with increasing world population and per capita food consumption (Fresco, 2006). LUC can have direct or indirect flow-on effects for GHG balances depending on the systems employed. A LUC from food production to biofuel production is a direct effect, and if this LUC in turn results in more land clearing for food production then this is an indirect effect. Clearly the latter is a negative scenario and would lead to greater GHG emissions as a result of land clearing. To avoid the use of productive agricultural lands for carbon mitigation, focus has been on marginal or abandoned farmland.

2.5.7 *Abandoned land*

Biomass production from abandoned, degraded or marginal agricultural land would help avoid negative LUC and potentially address environmental issues. Globally, it is estimated that 385 to 472 M ha of abandoned agricultural land exist and the potential area weighted mean production of above ground biomass is 4.3 t ha^{-1} yr^{-1} (Campbell et al., 2008). Wicke et al. (2011) explored the global technical biomass potential of all salt affected soils globally, estimating the potential energy yields from 1.1 Gha of saline and sodic soils, although not all of this area was available as it included forests, wetlands and other protected reserves. Abandoned saline land will not support food crops but will support salt tolerant tree crops (Stirzaker et al., 1999; Niknam and McComb, 2000) without displacing food production and generally these do not require fertilizer input (IEA, 2011). Estimates of potential biomass production from abandoned or marginal land are varied as a result of different assumptions and criteria for land use (Dornburg et al., 2010; Erb et al., 2012). Local agricultural systems determine the extent to which energy tree crops may be integrated however, it is generally accepted that this can have a positive effect on the long term sustainability of these agricultural systems (Smith et al., 2014).

2.5.8 Renewable energy potential

Recent studies estimate the potential land globally that could be allocated to bioenergy production without impinging on agriculture production from 0.15 - 2.4 G ha (Hoogwijk et al., 2003) to 0.7 - 3.6 G ha (Smeets and Faaij, 2007). Climate change mitigation via bio-sequestration has the potential to increase the proportion that renewable energy inputs have into the global energy budget. Approximately 10% of global energy use is via the utilization of biomass for energy generation (Bioenergy, 2011). Many publications discuss the global technical potential of bioenergy (Smeets and Faaij, 2007; Dornburg et al., 2008; Campbell et al., 2008; Wicke et al., 2011; Chum et al., 2011) which is defined as the amount of output obtainable by full implementation of demonstrated and likely to develop technologies or practices

(Moomaw et al., 2011). However, mid-range projections or scenarios for modeled global bio-energy potentials in 2050 across different studies differ by a factor of almost fifty (Haberl et al., 2010). Assumptions used in relation to these estimates, for example, future food yields, availability of land and sustainability criteria, result in differing estimates. These discrepancies in global estimates highlight the challenges associated with the measurement of mitigation and its application. The global estimates of land available for the cultivation of bioenergy crops range from 0.6 to 37 million km^2, the largest area given for bioenergy plantations for 2050 being 2.4 times larger than the area currently used for cropland and projected yields for bioenergy ranged widely from 6.9 to 60 MJ m^{-2} yr^{-1} (Haberl et al., 2010).

The transportation sector is a major source of anthropogenic greenhouse gas emissions, representing 23% of CO_2 emissions from fossil fuel combustion and 15% of global greenhouse gas emissions (OECD/ITF, 2010). Policies and sustainability criteria if enforced globally could potentially result in a 10% global biofuel share in the transport sector by 2030 (IEA, 2006). Reducing the dependency on fossil fuels for transport could mitigate significant amounts of atmospheric CO_2, provided balances of emissions in the production and use of biofuels are less than fossil fuels. Currently biofuels (1st generation) are produced from food crops (cereals and sugar cane) however, purpose grown biomass or lignocellulosic feedstocks (2^{nd} generation) offer higher GHG savings in comparison to conventional crops (Berndes et al., 2011).

2.5.9 Carbon neutrality

There are concerns over the assumptions of carbon neutrality of bioenergy from woody biomass (Haberl, 2013; Zanchi et al., 2012). At present it is assumed that the perceived reduction in greenhouse gasses from the substitution of fossil fuels with bioenergy will depend on the source of biomass and net landuse effects (Searchinger et al., 2009). The assumption of "carbon neutrality" is an oversimplification which can result in flaws in emissions accounting (Bird et al., 2012). Clearing mature forests to establish bioenergy crops will incur a carbon debt which may take decades or centuries to offset via fossil fuel substitution (Schulze et al., 2012), whereas biofuels from perennial crops grown on degraded and abandoned land would incur little or no carbon debt (Fargione et al., 2008). The concerns raised by Schulze et al. (2012) do not apply to reforestation of agricultural land for the purpose of energy crops, however, competition for agricultural land would not be desirable and therefore degraded or abandoned land is a preferred option.

2.6 Conclusions

From the literature it is clear that critical issues remain pertaining to the application of reforestation for the purpose of bio-mitigation of climate change and particularly so in dryland environments. The measurement of carbon within forest biomes, the integration of reforestation into farmland landscapes and the sustainability of forest systems if applied to the land sector, requires further research in order to ascertain the full potential of these mitigation systems in dryland environments.

3 Developing a specialized soil corer for sampling tree roots[1]

3.1 Introduction

To develop biomass and carbon prediction equations for whole trees it is necessary to measure both above- and below-ground biomass of sample trees. However, exposing and measuring entire root systems is generally not practical. Therefore, sub-sampling of individual tree root systems is necessary. Common sampling methods involve the use of coring tubes, augers, drilling rigs, or monolith sampling. This chapter discusses the limitations of each of these systems and the development and testing of new equipment to meet root sampling requirements for carbon estimation.

Coring tubes vary from a simple piece of tubing with a sharpened edge to more elaborate systems which dismantle to enable retrieval of the soil core, some having a plastic liner to hold the sample (Prior and Rogers, 1992; Prior and Rogers, 1994). Coring tubes are either hammered into the soil by hand or with the aid of mechanical mass impact (Bohm, 1979). Soil compression within coring tubes may be a problem in some soils (Vogt and Persson, 1991). Generally coring tubes will only cut through small roots, being unsuitable for sampling large roots (Reynolds, 1970). Most rocks are also impenetrable and floating rocks may block the aperture (Vogt et al., 1984). Sampling at depth is difficult and subsequent retrieval can be arduous requiring the use of a tri-pod and winch (Roberts, 1976; Van Rees and Comerford, 1986).

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Auger methods have also been applied to root sampling. These vary from simple hand augers or bucket augers (Farrish, 1991) suitable only for friable soils, to motorized units needing two operators (Ponder and Alley, 1997). Auger methods have also been mechanized by adapting agricultural machinery and the use of hydraulic devices mounted on tractors (Smit et al., 2000). The proline coring device as used by Davis et al. (1983) is another example of a mechanical auger system. While more powerful mechanized equipment will cut through large roots and some rocks, these are not designed to cut cleanly through large roots without damage to the sample or disturbance to the surrounding soil.

Drilling rigs, as used in the mining industry, have also been used to collect soil samples for small and fine root measurement (Carbon et al., 1980). Fitted with hollow augers drilling rigs are able to core through very hard soils and can take soil cores to greater depths. Hollow augers incorporate an inner (non-rotating) tube which is pushed into soil slightly ahead of an outside (rotating) auger. Depending on the soil conditions, undisturbed samples (soil cores) may be retrieved that are suitable for soil profile description. However, these are unable to cut cleanly through large diameter roots and the flytes on the outside of the bore may cause excessive disturbance to the surrounding soil. Other disadvantages of drilling rigs include the operating cost and their physical size. They may be difficult to position close to standing trees and may compact or otherwise damage study plots.

Monoliths involve the excavation and processing of blocks of soil. One method to achieve this is with the aid of a metal box, approximately 20 cm square and 20 - 30 cm deep, with sharpened edges (Vogt and Persson, 1991). The sample box is hammered into the ground and the sample inside is then excavated and processed. Soil compression is negligible with monolith samples compared to coring tubes. However, this technique is labor intensive due to the large sample size that needs to be processed and cannot be easily used for sampling large tree roots or those at depth.

The sampling equipment and techniques described above all have severe limitations with respect to sampling tree root systems. For a proposed study of *Eucalyptus globulus* (Labill) root systems (Chapter 4), sampling equipment was needed which could sample to depth (approximately 6 m) through hard soils with indurated layers or rock. It was also essential that large diameter roots be cut through cleanly and with minimal disturbance to the plot, meaning that the use of an auger system with flytes was unsuitable. The intensity of coring needed to be high and therefore the equipment needed to be light-weight and very maneuverable. With these factors in mind a new apparatus was built specifically for intensive coring of tree root systems and designed to overcome limitations of currently available sampling equipment.

3.2 Materials and Methods

A corer was designed and built that is powered by a 50 cm^3 two-stroke petrol engine adapted from a commercially available post-hole digger (Figure 3.1). The power unit has a reduction box (33:1) and output shaft to which a drive coupling was adapted to drive a purpose built coring head. The coring head is driven via light-weight extension tubes, which have an outside diameter of 25 mm and a wall thickness 2.5 mm (Figure 3.2, Appendix 1). The extension tubes are one meter in length with a threaded hexagonal (24 mm) fitting at each end which enables them to be screwed together. A hexagonal drive lug (male) is connected to the top of the extension tube which in turn is driven by the coupling (female) on the power unit. The hexagonal (male/female) drive allows for quick disengagement of extension tubes.

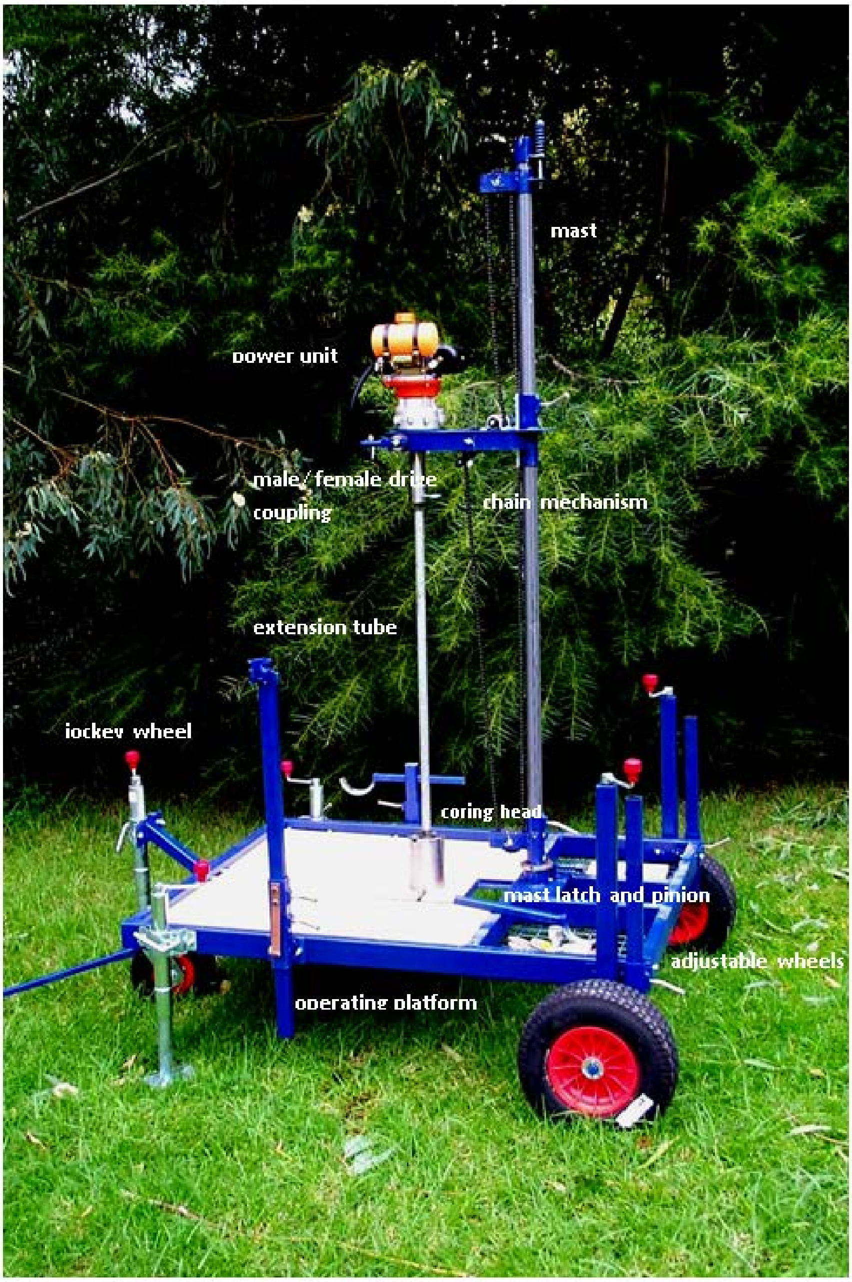


Figure 3.1 Corer with jockey wheel and handle in place as used for maneuvering. The jockey wheel can be replaced with a trailer hitch.

The power unit is guided by a vertical mast attached to an operating platform with a pinion and foot latch. The latch prevents lateral movement of the power unit during coring. When required the latch is disengaged and the power unit swung to one side, the coring head and sample are lifted from the sample hole and placed in a holder

(Figure 3.3) above a sample bag. With the aid of a length of 10 mm diameter stainless steel rod the sample is pushed out the bottom of the coring head into a sample bag.

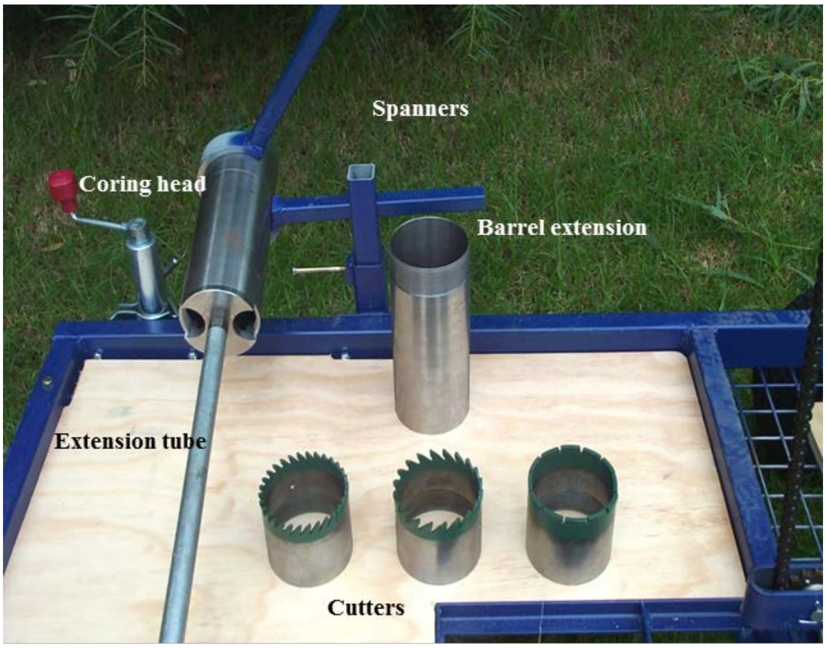


Figure 3.2 Coring head with barrel extension and interchangeable cutters (fine tooth, coarse tooth and diamond matrix).

A feature of the coring head is the interchangeable cutters (103 mm internal diameter) and barrel extension (Figure 3.2). The cutters and barrel extension are threaded to enable the cutters to be removed as required by using two specially made spanners. The length of the coring head (300 mm) minimizes deviation from vertical by hard objects in the soil and can be extended to 500 mm with the barrel extension if necessary.

A coarse-tooth tungsten cutter (15 mm between teeth) is suitable for most soils including compacted clays and some soft rock (e.g. sandstone). For friable soils (e.g. sand) a fine-tooth tungsten cutter (10 mm between teeth) is necessary to minimize tearing of roots. The cutters can be sharpened and the tungsten chips replaced if necessary. For indurated soil layers or rock a diamond matrix cutter can be attached. With the addition of water the diamond matrix cutter will cut through rock of any hardness. The diamond segments can also be replaced if necessary.

Figure 3.3 Coring head in holder with extension tube attached.

Sampling depth is dependent on the number of extension tubes. By connecting sampling tubes together the depth of sampling can be increased. Sampling to date has typically been to 6 m, although 9 m has been achieved.

An operating platform on height-adjustable wheels allows the unit to be positioned and leveled. The unit weighs approximately 150 kg and is easily maneuvered by one person with minimal damage to sample plots. A jockey wheel and handle aid to position the corer. By standing on the platform to operate the corer weight is added for stability and downward pressure. A removable section of the platform facilitates coring within 25 cm of standing trees.

A toe-ball hitch can also be attached to the platform for towing between sampling sites of close proximity. Alternatively, the unit can be dismantled into three parts (platform, mast and engine) for transportation over long distances. Dismantling the unit can be achieved in 10 to 15 minutes.

Three corers have been built and are in operation in Queensland and New South Wales. These have been used in different research studies and have been successful in obtaining soil-root samples from a range of soil types (Barton and Montagu, 2006).

To test the accuracy of the corer, root biomass density estimates obtained by coring were compared to estimates obtained from the same forest area from bulk (25 x 25 x 25 cm monolith) soil samples. Twenty-five monolith samples were taken mid-row in an eight-year old *E. globulus* stand (S1) (34˚ 51’ 05.96” S, 117˚ 45’ 09.82” E) growing on a Eutric Cambisol (McArthur, 1991) comprising of 100 cm duplex sand over clay sub-soil (Figure 3.4). A 25 cm square steel sampler (3mm wall thickness) with cutting edges filed to knife-edge sharpness was driven into the soil to a depth of 25 cm. With the monolith sampler in place, soil was collected using a garden-trowel and care taken to remove only the soil within the sampler.

Around each monolith sample eight core samples were taken equidistant from the monolith sample and also to a depth of 25 cm (Figure 3.5). The fine toothed cutter was used in conjunction with the corer to minimize tearing of small roots. All samples were placed in plastic bags and fresh weight of the sample recorded. Samples were

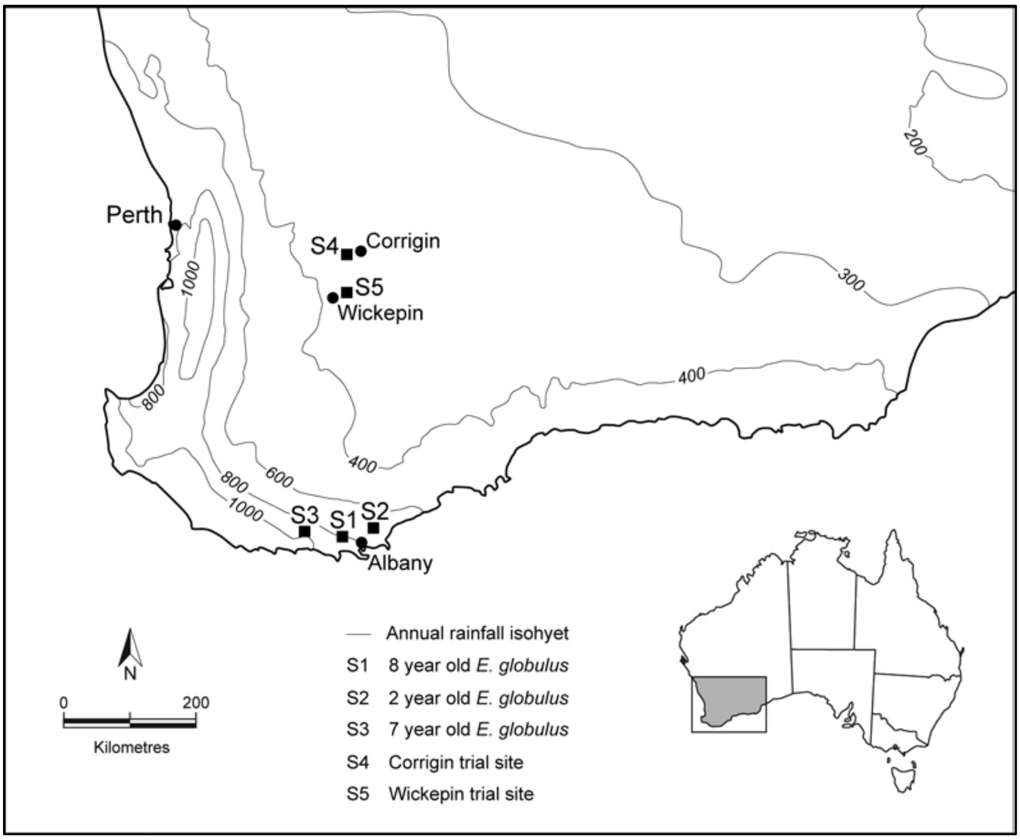


Figure 3.4 Location of sample sites.

stored in a 2000 litre cool box with ice while in the field, then transported to cold storage (-4˚C).

Soil sample dry weights were expressed as mass per unit volume (kg m^{-3}) and these values were compared for both sampling techniques to give an indication of the volume sampled by coring.

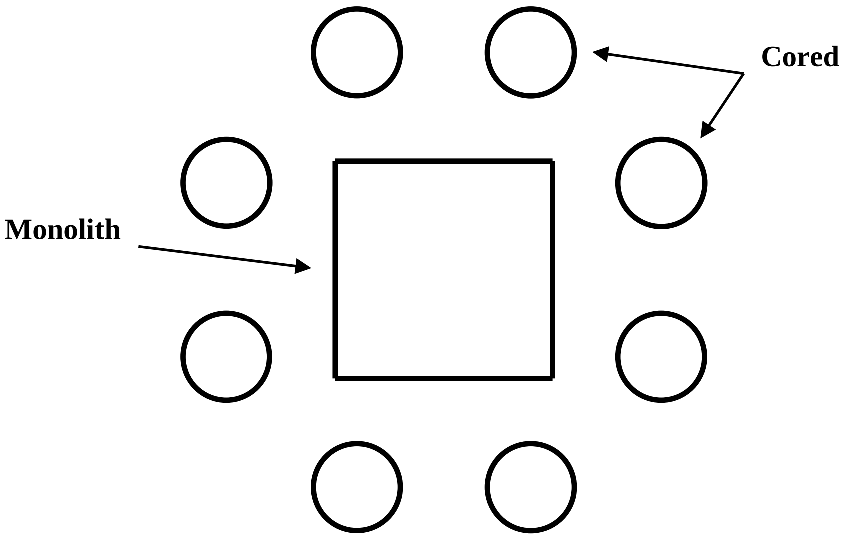


Figure 3.5 Sampling layout for monolith and cored samples.

Two core samples were randomly selected and paired with each monolith sample, the remaining six samples were used for a study investigating the effect of storage of soil samples on root decay. Samples were wet sieved through a 2 mm square aperture sieve, dried at 40°C and sorted to remove debris. Roots were sorted into four diameter classes: <2, 2 to <5, 5 to <10 and 10 to < 20 mm. Dry root mass was then determined after drying to constant weight at 70°C.

3.3 Results

The monolith sampler was very effective and was able to cut through roots without any deformation of roots or compression of sample. The sandy profile did not contain any obstructions (stones) to hinder the effectiveness of the sampler. The largest diameter roots sampled did not exceed 20 mm in diameter.

The monolith method, as applied in this study was similar to taking a large bulk density sample. Comparison of soil fresh weight per volume from the monolith sampler to that from coring, indicated the sample volume from coring was consistent of a volume equivalent to the external diameter of the corer i.e. there was no compaction of soil.

The percentage of roots mass recovered in each size class decreased with increasing root diameter, with approximately 75% of root mass less than 5 mm in diameter for both monolith and cored samples (Table 3.1).

Table 3.1 Proportion of total root biomass of each diameter class.

	Proportion of root diameter class (%)			
Sample	< 2 mm	2 to < 5 mm	5 to < 10 mm	10 to < 20 mm
Monolith	58.8	17.6	9.2	14.4
Cored	55.7	18.3	17.4	8.5

Estimates of root biomass density in the 0 to 25 cm soil layer ranged from approximately 1.2 kg m^{-3} soil (<2 mm diameter roots) to approximately 2.1 kg m^{-3} soil (<20 mm diameter roots), there being only small differences in estimates from the two methods (Figure 3.6). A paired sample t-test indicated the differences by the two methods were not significantly different from zero ($P \geq 0.23$, all cases) for all root size categories (Figure 3.7).

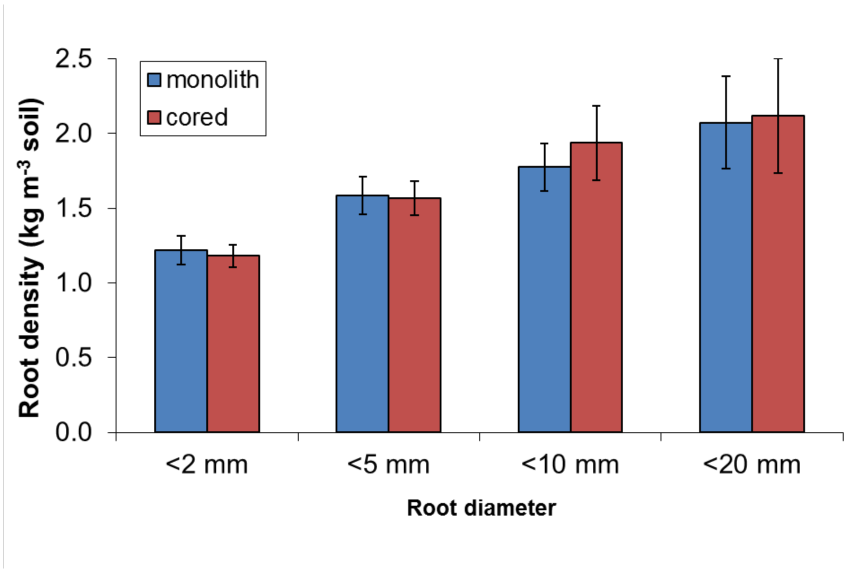


Figure 3.6 Mean root biomass density in paired core and monolith soil samples. Error bars indicate 95% confidence limits.

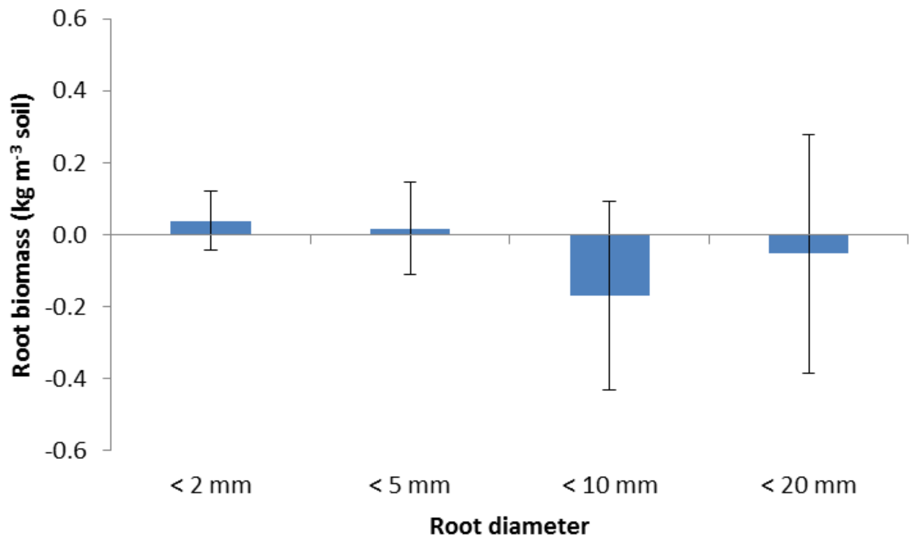


Figure 3.7 Mean difference in root biomass between paired monolith and core samples by root diameter class. Error bars indicate 95% confidence limits.

3.4 Discussion

The cutters designed for the coring head are based on a saw tooth configuration (Appendix 1) and result in a ‘cutting clearance’ of 10 mm. That is, the internal diameter of the cutters is 103 mm and the external diameter is 113 mm. Initially it was assumed all root material coming into contact with the cutters would be sawn through and the subsequent sawdust lost in the sieving process. However, inspections of samples and core holes indicated some tearing of roots occurred, mostly in small (<10 mm diameter) roots. For this diameter class it cannot be assumed that roots would be cut precisely to the internal diameter of the cutters but would tend to break at a point between the internal and external diameter of the cutter. For smaller diameter roots an internal diameter for volumetric calculations would be inappropriate. Medium to large roots (> 10 mm diameter) being more rigid, were sawn through cleanly with only the bark being subject to some tearing. The diameter chosen for volumetric calculations was based on the diameter of roots sampled. The maximum root diameter in this data set was <20 mm with approximately 90% of roots being <10 mm diameter for cored samples (Table 3.1). For this reason the mid-diameter (108 mm) of the coring head was used to calculate soil volume for root density determination. For data sets which include larger diameter roots (>25 mm) an internal diameter would be more appropriate for calculating soil root densities for that size class.

Applying the mid-diameter for calculation of soil volume of the cored samples gave root biomass density estimates consistent with the monolith method. It was assumed the monolith method, as applied in this study was accurate. Therefore, this indicates the coring also gave unbiased estimates of root biomass density. More core samples would be required for the same precision of estimate by the monolith method. However, the monolith method is not practical for routine sampling, being generally too slow, especially at depth, and not suitable for large diameter roots or difficult soil conditions. In contrast the corer is practical to operate under such conditions.

The diameter of sampling tools used for volumetric soil sampling are typically between 2 and 15 cm depending on the application of the sampling (Vogt and Persson, 1991). Corer diameter and subsequent sample volume will have a direct effect on precision of estimates from the same number of samples. Larger core samples will result in lower coefficients of variation but are impractical with regards to storage and processing. A nominal diameter of 10 cm was chosen as a convenient sample size with respect to the number of samples to be stored and processed. If required a larger or smaller coring head can be operated in conjunction with the corer without modification to the apparatus.

The new corer described here was initially designed and built for intensive soil coring of *E. globulus* tree root systems in Western Australia. Soil cores were taken around sample trees which were then excavated and these data used for computer simulated sampling, to help develop future sampling strategies (Chapter 4). Up to 65 cores to six meters were taken in an area of approximately 4 x 2 meters. Large lateral roots were often encountered (~75 mm diameter). The corer was able to cut through roots of any size without disturbance to the surrounding profile or loss of sample. It was important to minimize damage to the sample plot which included a planting mound and therefore sampling equipment had to be light-weight and maneuverable. Soil conditions encountered included loose sand, heavy clays, laterite and silcrete hardpans, and fractured granite. With the selection of interchangeable cutters it was possible to obtain soil cores in these extreme situations.

3.5 Conclusions

The corer described is a cost effective and versatile tool for sampling tree root systems. It can be easily dismantled for transportation or towed over short distances between sample sites and can be maneuvered and operated within highly stocked tree stands. The interchangeable cutters can be re-sharpened or re-built if necessary, and the size, orientation and number of teeth can be customized for specific sampling situations. The coring head used in this study had a nominal diameter of 10 cm, however, smaller or larger diameter coring heads could be manufactured and operated by the same unit. This coring apparatus was developed specifically for the proposed sampling of tree root systems in Chapter 4, and enable intensive soil coring of tree root systems and the sampling of large diameter roots in hard soils to a depth of 6 m.

4 Accuracy of tree root biomass sampling methodologies for carbon mitigation projects[2]

4.1 Introduction

Both the storage of carbon in biomass or production of biomass for bioenergy generation through the afforestation or reforestation of farmland are advocated as a major climate mitigation strategy (Pacala and Socolow, 2004; Canadell and Raupach, 2008; Smith et al., 2014). Globally, renewable energy incentives and greenhouse gas emissions targets are drivers for the development of such systems. However, the efficacy of these mitigation strategies and efforts to restrict the rise of global temperature to 2°C (UNEP, 2014) will rely on acceptable estimations of all carbon pools including tree root systems. Given that tree roots account for 20 to 40% of forest carbon (Brunner and Godbald, 2007; Finér et al., 2011; Mokany et al., 2006) this carbon pool is a significant component of the terrestrial carbon pool as forests contribute significantly to global carbon sinks and fluxes (Eamus et al., 2002).

Carbon in the above ground portion of trees can be relatively easily measured via destructive sampling and the subsequent data used to derive allometric equations (Snowdon et al., 2000). However, below ground biomass in tree roots is considerably more difficult to measure and the methodologies used are varied (Levillain et al., 2011). Estimates of tree root carbon pools based on empirical measurements are lacking; this is a direct reflection of the difficulty of measuring tree root systems and thus an encumbrance in the development of tree root allometric relationships.

[2] In press as: Sochacki, S.J., Ritson, P., Brand, B., Harper, R.J. and Dell, B. (2016) Accuracy of tree root biomass sampling methodologies for carbon mitigation projects. *Ecological Engineering.* http://dx.doi.org/10.1016/j.ecoleng.2016.11.004

Consequently, understanding of tree root systems and their link with above ground biomass and the soil environment (Smithwick et al., 2014) is also limited. On a global scale, the lack of tree root biomass data impedes the understanding of forest biomass carbon and its effect on global carbon pools and fluxes (Vogt et al., 1996). For example, deforestation is one of the major sources of global carbon emissions (Smith et al. 2014) but the estimate of the root carbon stores is invariably based on default estimates (Aalde et al., 2006).

Vogt et al. (1998) suggest the measurement of large tree roots for carbon estimation can be easily derived via allometrics of above ground measurements but this is clearly not the case (Levillain et al., 2011) despite efforts made to develop methodology for tree root sampling (Snowdon et al., 2002). In particular, consistent measurement methodology of coarse roots which may account for over 70% of below ground biomass is lacking (Cairns et al., 1997; Herrero et al., 2014).

Methods for sampling tree roots are varied depending on the needs of specific research, from fine root dynamics (Lopez et al., 1998; Makita et al., 2011) to total tree below ground biomass (Rey de Viñas and Ayanz, 2000). Common techniques used include volumetric sampling of tree roots via coring and soil pit methods (Levillain et al., 2011) monolith sampling (Makita et al., 2011) and voronoi polygons (Saint-Andre et al., 2005; Razakamanarivo et al., 2012), bulk root excavation (Niiyama et al., 2010; Ritson and Sochacki, 2003) and root ball excavation (Miller et al., 2006; Misra et al., 1998; Resh et al., 2003). Given the horizontal and vertical extent of tree root systems, a total estimate of tree root biomass is most likely to be an underestimate (Pinheiro et al., 2016; Stone and Kalisz, 1991).

Many factors affect tree root development and the morphology of tree root systems is determined by species, age, soil and hydrological properties and climatic variables (Tobin et al., 2007). There are limited detailed root biomass data and few species specific allometric relationships exist for below ground biomass (Laclau, 2003; Ritson and Sochacki, 2003; Paul et al., 2014b; Jonson and Freudenberger, 2011) (Chapter 5). Indirect estimates of root biomass have been attempted based on relationships between above ground biomass and root biomass or root: shoot (r:s) ratios (Kuyah et al., 2012; Mokany et al., 2006; Snowdon et al., 2000). Generalized allometric equations based on mixed species data have been applied to circumvent the laborious task of sampling tree roots and developing allometric equations for individual species. However, the variability of estimates can be large when generalized data sets are applied to root biomass (Cairns et al., 1997).

The application of forest systems to mitigate climate change has resulted in the need to quantify carbon for both above and below-ground pools. This information is needed both for carbon offset (abatement) projects and also for national accounting. Given the heterogeneity of tree root systems and the variability of the soil in which roots grow, it is challenging to obtain precise biomass estimates in relation to below-ground tree biomass and consequent carbon storage. Currently, many carbon accounting approaches use default values for root biomass, based on ratios of above ground biomass (Aalde et al., 2006; Mokany et al., 2006).

Typically, sampling for the development of tree root biomass allometric relationships would require sampling many tree root systems to reach acceptable levels of precision (Levillain et al., 2011). An alternative to such extensive sampling is the use of computer simulation. Monte Carlo simulations have been employed in forestry for issues relating to forest fire risk (Carmel et al., 2009), uncertainty of forest carbon flux

(Verbeeck et al., 2006), forest carbon densities and uncertainties (Gonzalez et al., 2010) and forest sustainability (Luxmoore et al., 2002), and more recently by Paul et al. (2014b) in testing allometric relationships for root biomass prediction. The use of Monte Carlo simulation for tree root sampling methodologies has not previously been reported.

A recent comprehensive review by Addo-Danso et al. (2016) on root sampling methods exemplified the range of root sampling methods that have been applied and recommended further studies to directly compare methods of tree root sampling on similar sites. In this study the approach of sampling many tree root systems was substituted with simulated sampling in order to investigate the effectiveness of different tree root sampling methods and sampling regimes. Complete tree root data sets from volumetric coring (Sochacki et al., 2007) and detailed excavation were applied in conjunction with simulated sampling using the Monte Carlo technique. The aim of this study was to compare the precision and bias of tree root biomass estimates from a range of sampling methodologies to ascertain the most effective method of tree root sampling for below ground biomass and carbon estimation. Excavation and coring methods will be examined through a range of sampling regimes to determine the bias and precision of root biomass estimates and compare the effectiveness of these methods for tree root biomass sampling to past studies applying similar methods.

4.2 Methods

4.2.1 Site selection

Eucalyptus globulus (Labill.) is extensively used for reforestation in southern Australia. Two sites (S2 and S3) in southwestern Australia were selected for the purpose of this study (Figure 3.4). The first site (Site A; 34˚ 49’ 01.30” S, 117˚ 59’ 37.93” E) was a 2 year old *E. globulus* stand on a Eutric Cambisol (McArthur, 1991). This soil comprised 150 cm of sand overlying an abrupt boundary to a clay subsoil. The second site (Site B; 34˚ 46’ 07.59” S, 117˚ 22’ 49.81” E) was a 7 year old *E. globulus* stand on a Xanthic Ferralsol (McArthur, 1991), comprising of 30 cm of gravelly loam horizon overlying clay. The gravel was ferricrete. Both stands were planted at 2 m x 4 m spacing resulting in densities of 1250 trees ha^{-1}, and being grown in a 10-year rotation in this region.

The two stand ages were chosen to be representative of a situation where root development was generally not restricted by competition between neighboring trees (2 years) and where the root dynamics were that of a stand nearing the end of a 10 year rotation (7 years).

4.2.2 Tree selection

One sample tree from each stand age was chosen. To characterize the stand around the sample tree a 20 x 20 m plot was demarcated to measure tree parameters of diameter and height. Using allometric equations developed by Brand (1999) the average tree biomass for the plot was calculated and a tree selected with approximately this value. This tree was also selected such that neighboring trees had a similar tree biomass estimate; this being to prevent bias in the sample tree root biomass estimates.

4.2.3 Soil coring

Sampling equipment was needed which could sample to depth (approximately 6 m) through hard soils with indurated layers or rock. It was also essential that large diameter roots be cut through cleanly and with minimal disturbance to the plot. The intensity of coring was high and therefore the equipment needed to be light-weight and maneuverable and not damage the sample plot. With these factors in mind a new apparatus was built specifically for intensive coring of tree root systems and designed to overcome limitations of current sampling equipment. This new apparatus which had an internal core diameter of 103 mm, was able to take soil-root samples of large diameter roots of any size and through soil of any hardness, including rock. A detailed description of this apparatus and testing is described in Chapter 3.

4.2.4 Coring layout

Roots were cored in a circular or Nelder array instead of a square or rectangular grid (Figure 4.1 a). This arrangement of sampling around a tree trunk has been applied to the study of spatial root distribution in orchard trees (Bohm, 1979; Weller, 1971). With this sampling arrangement the intensity of sampling was greatest close to the tree, where the highest root density would be expected, decreasing with distance from the tree. To each core position, a root zone was allocated to which the coring root density values would be applied for sampling simulations (Figure 4.1 b).

Once the sample trees had been measured and the aboveground portion removed, a rectangular plot was allocated to the sample tree and boundaries were designated as midpoints between trees within the sample row and between adjacent tree rows. The plot size and sampling arrangement was the same at each site with a resultant plot size of 4 x 2 m with approximately 50 to 60 cores taken in and around each plot to a depth of 6 m (Figure 4.1). Previous biomass studies (Brand, 1999) on *E. globulus* recovered root material to this depth however, coring beyond 6 m was considered impractical and time consuming. For the purpose of this study adequate data would be acquired to address the research questions.

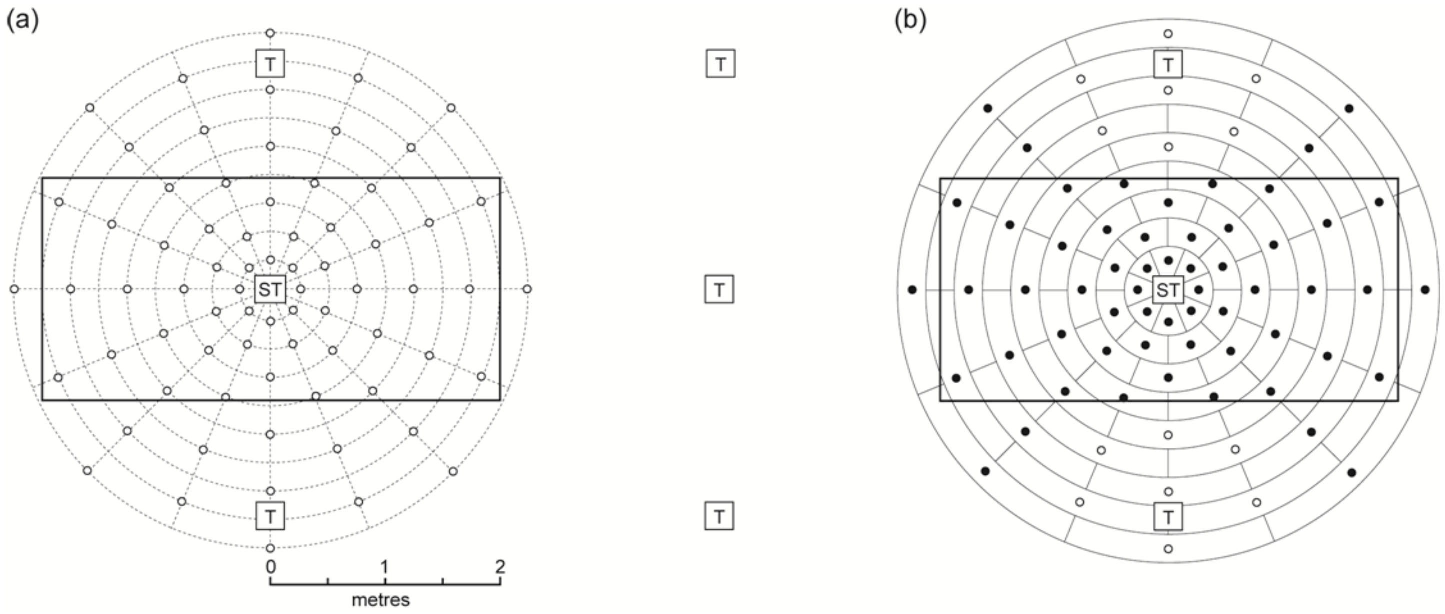


Figure 4.1 Layout of coring positions (○) around the sample tree (ST) with the tree plot designated by the mid point between adjacent trees within the tree row and the mid-point between adjacent tree rows. b) Coring positions (●) allocated to coring zones within the tree plot area. The tree plot was 4 x 2 m in size.

Depth increments for the soil cores were taken relative to the natural land surface being, 0 to 0.25, 0.25 to 0.5 and then at 0.5 m intervals to a depth of 6 m. The planting mound which was raised approximately 0.2 m above the natural land surface was sampled to depth 0 m. Soil cores were collected into heavy duty plastic bags and placed in cold storage at 5^{o}C prior to sieving.

4.2.5 Sieving

Samples were wet sieved with a purpose built sieving apparatus. Soil samples were agitated in a trough with water, allowing the root material to be separated from the soil sample by flotation and then passed over a standard 2 mm laboratory sieve. Some clay samples required soaking overnight to disperse the clay prior to sieving. Sieved root samples were placed into calico bags and oven dried at 40°C to remove excess moisture before sorting into diameter classes: <2, 2 to <5, 5 to <10, 10 to <15, 15 to <20, 20 to <25 and ≥25 mm. After cleaning and sorting the samples were dried to constant weight at 70°C for dry weight determination.

4.2.6 Excavation

Following coring the tree plot area was excavated with a backhoe to a depth of 6 m using the same depth intervals as for coring (Figure 4.2). Soil was placed on the purpose built sieving table which was overlaid with 25 mm square wire mesh (Figure 4.3). Soil clods were broken down by hand to pass through the mesh. Roots were collected from the sieving table and placed into calico bags for drying and sorting.

Soil passing through the sieving table from each excavated depth interval was sub-sampled for fine roots. These ~20 kg samples were wet sieved through a 2 mm sieve and recovered roots treated as for the cores, and used to calculate fine root mass estimates for the whole depth interval. Dead roots from previous land use were not included in this study. The aim was to compare estimates of standing biomass of the current tree stands for the purpose of determining and comparing sampling uncertainty of sampling methodologies.

For this estimate it was necessary to have bulk density values for each depth interval. Knowing the dry mass of the sieved soil sample and the bulk density of the excavated soil layer, fine root mass for that layer could be estimated on a volumetric basis. Bulk density samples were taken spaced uniformly within the excavated layer with one sample taken per square metre. A large volume sampler was built to minimize error when sampling ferricrete gravel layers. This sampler was 100 mm in diameter by 100 mm in height with a wall thickness of 3 mm, resulting in an area ratio (wall thickness:internal area) not exceeding 0.1 as recommended by Greacen et al. (1989). Soil BD samples were oven dried at 105˚C for 48 hours, and the oven dry weight determined.

Bulk density was calculated as

$$BD = \frac{Sd_{105}}{Sv} \qquad (1)$$

where

BD = bulk density (g cm^{-3}),

Sd_{105} = sample dry weight at 105 ^{o}C (g) and

Sv = sample volume (cm^3).

Dry weight of fine roots for each excavated depth interval was calculated as

$$FRwt = \frac{BD*Ev}{Sm} * SRwt \qquad (2)$$

where

FRwt = fine root weight in excavated layer (g),

BD = bulk density of the excavated layer (g cm^{-3}),

Ev = volume of the excavated layer (cm^{3}),

Sm = mass of the sieved soil sub-sample (g) and

SRwt = dry weight of fine roots in sub-sample (g).

Figure 4.2 Excavation of the 7 year tree root system at an excavation depth of approximately 6 m, coring hole visible in the excavation pit wall.

Figure 4.3 Sieving of excavated soil through 25 mm^2 mesh sieve.

4.2.7 Computer simulations

Root sampling scenarios were based on typical methods (Addo-Danso et al., 2016) employed for the measurement of root biomass. Not all methods could be included as the cost of tree root sampling in such detail prohibited this. However, methods deemed most appropriate for tree root sampling in reforestation projects which included complete excavation, as this was the most accurate method of associating tree roots to a single sample trees (Snowdon et al., 2002; Ryan et al., 2011). Soil coring was also chosen as this has been used extensively (Addo-Danso et al., 2016) for tree root studies, in particular for fine root estimates. Although not recommended for coarse roots, this is a relatively easy and effective option to apply in the field for fine root estimates following excavation, and with the newly developed equipment for this study will enable sampling to access roots at depth (6 m) to capture a greater extent of the root system. This resulted in four general modes of sampling being tested. These sampling scenarios were:

1. ***Coring only*** (Saint-Andre et al., 2005; Berhongaray et al., 2015) - The coring only scenario was used to estimate all root diameter classes (excluding the root bole) and was applied as three different sub-scenarios: a) random coring (6 m depth) with sample points applied randomly across the entire tree plot area, b) stratified random coring (6 m depth) where sample areas were chosen in relation to horizontal distance from the tree and c) nested coring (1 and 6 m depth), in which coring was stratified according to depth.

2. ***Bulk excavation plus coring*** (Ritson and Sochacki, 2003; Levillain et al., 2011) - The bulk excavation scenario simulated excavation of the tree plot to a predetermined depth and included stump pulling in conjunction with coring (6 m depth) to estimate mass of distal roots.

3. ***Excavation by root diameter limit plus coring*** (Misra et al., 1998; Peichl and Arain, 2007) - The third method simulated involved excavation of tree roots to a specified diameter limit in conjunction with coring (6 m depth) for distal roots. In this scenario root mass data can be associated to individual sample trees, where sampling depth is not predetermined and roots are removed or traced to a minimum diameter limit. Such an approach can allow excavated root mass to be associated directly to the tree being sampled.

4. ***Root ball excavation plus coring*** (Resh et al., 2003; Miller et al., 2006; Jonson and Freudenberger, 2011) - Root ball excavation is a scenario which attempts to confine sampling to where the greatest concentration of roots occur, that is, in closer proximity to the root bole. Root ball excavation simulated the

removal of all roots in close proximity to the tree root bole and sieving with a fine mesh sieve (2 mm) to collect all roots within a defined radius around the sample tree in conjunction with coring (6 m depth) for distal roots. The radius in the simulations is defined by the root sampling zones designated in Figure 4.1b.

All scenarios were simulated on both the 2 and 7 year old sample trees. Sampling scenarios reported here will be an example of possible scenarios for each of the four categories and will be discussed in terms of their effect on sampling uncertainty.

Root mass values were used to generate uncertainty estimates for different sampling scenarios via the Monte Carlo method (Efron and Tibshirani, 1993). For the purpose of calculating total root mass for any given sampling scenario root mass values from each core were allocated to the area around each core position as indicated in (Figure 4.1b). Using Monte Carlo simulation, different sampling methods were tested and estimated levels of sampling uncertainty generated. Each scenario was run for 1000 iterations being applied to diffrent coring rates in conjunction with a range of excavation regimes such that root mass not retrieved by excavation was accounted for by coring. Levels of sampling uncertainty were generated at a 95% confidence interval where uncertainty is defined as

U(%) = half the 95% confidence interval divided by the
total or mean and expressed as a percentage (IPCC, 2002)

For each of the four sampling methodologies comparisons of simulated root mass estimates with the true (excavated) root mass values determined the bias of tree root sampling methods. Precision of estimates, or sampling uncertainty, for each scenario were compared on the basis of the standard error of the mean. The standard error of the mean and associated confidence intervals vary inversely with the square root of the sample size (Webster and Oliver, 1990) therefore, the standard error can be reduced to any desired value by increasing the sample size. This relationship can be used to extrapolate from the uncertainty estimates generated from the simulated sampling of one sample tree to larger sample sizes.

4.3 Results

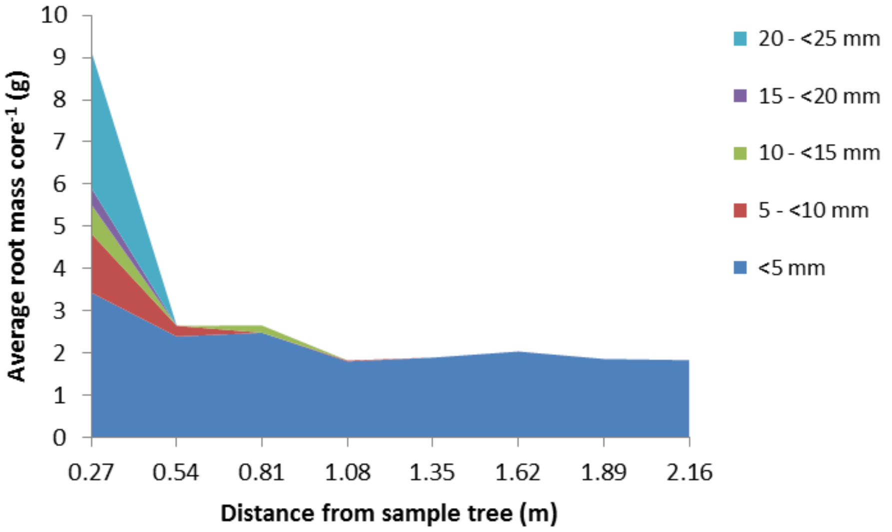
4.3.1 Coring

Root mass decreased with horizontal distance from the sample trees of both ages, and this varied by size class. For the 2 year old tree, the mass of roots >5 mm in size decreased markedly from 0.27 to 0.5 m distance from the tree, with no roots of this size more than 1.0 m from the tree (Figure 4.4a). The 7 year old trees exhibited the same pattern with a large decline in >5 mm diameter roots at 0.72 m distance, and an absence of >5 mm roots after 1.67 m (Figure 4.4b). For trees of both ages roots <5 mm in diameter extended beyond 2 m.

Root mass and diameter diminished systematically with depth for trees of both ages, with roots found throughout the 6 m depth of sampling. Roots >5 mm in diameter were confined to the surface 2 m for the 2 year old tree (Figure 4.5a) whereas roots up to 10 mm in diameter extended to 6 m for the 7 year old tree (Figure 4.5b).

Root mass distribution diminished with depth and with lateral distance from the sample tree, this trend was observed for both trees and is illustrated as a three dimensional surface with a common scale to illustrate the relative root mass distribution for both tree ages (Figure 4.6).

(a)



(b)

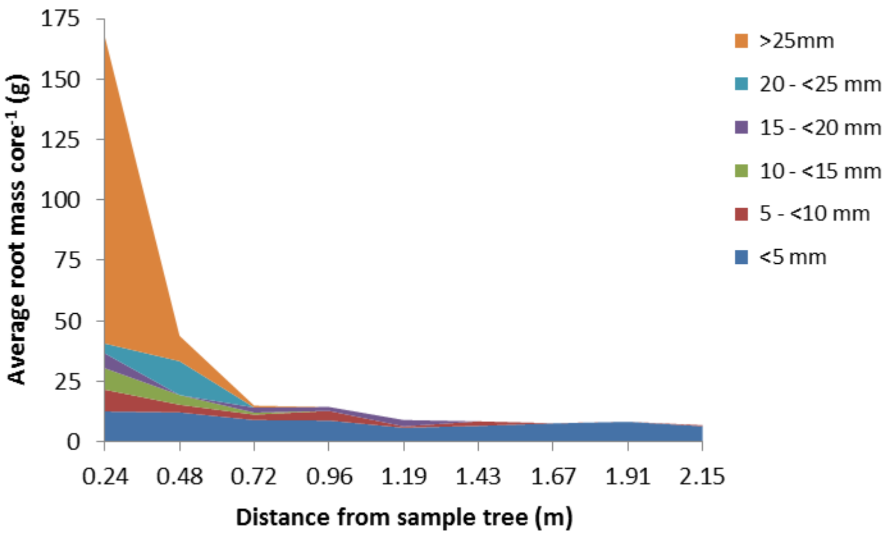
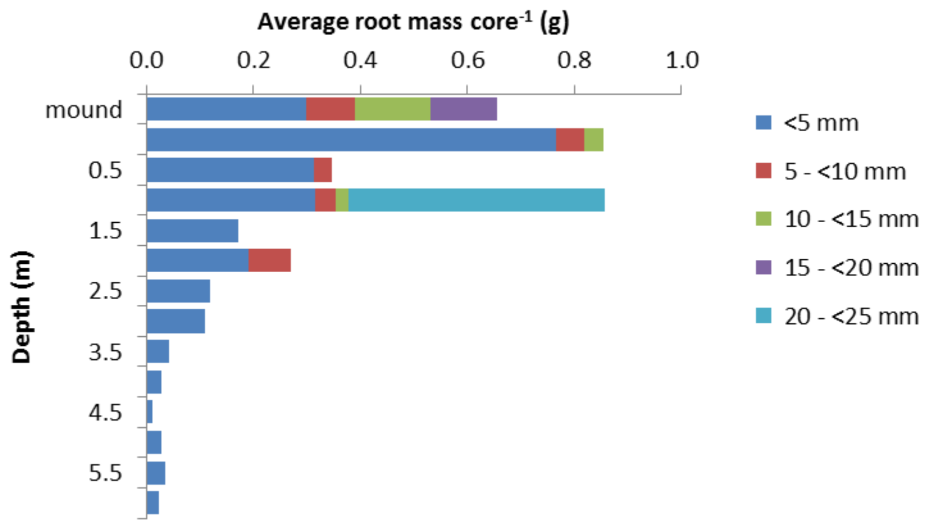


Figure 4.4 Mean root mass sampled in coring for diameter class and distance from the a) 2 year old and b) 7 year old *E. globulus* trees. Distances are actual sampling positions or increments from the sample tree as designated in Figure 4.1.

(a)



(b)

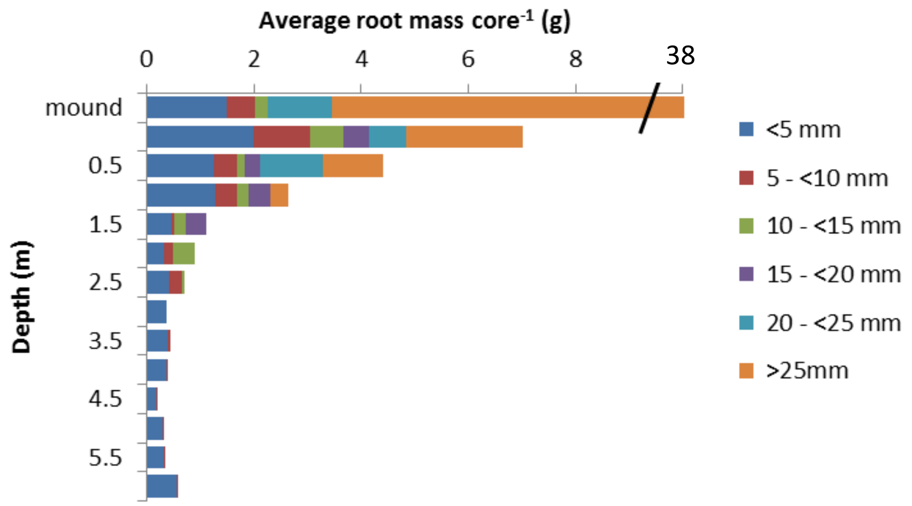
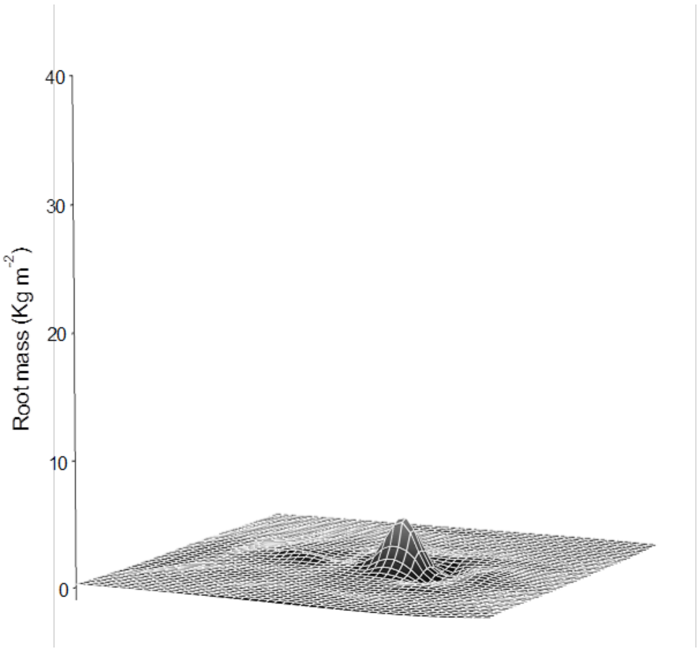


Figure 4.5 Mean root mass sampled in coring for diameter class in relation to sampling depth for a) 2 year old and b) 7 year old *E. globulus* sample trees.

(a)



(b)

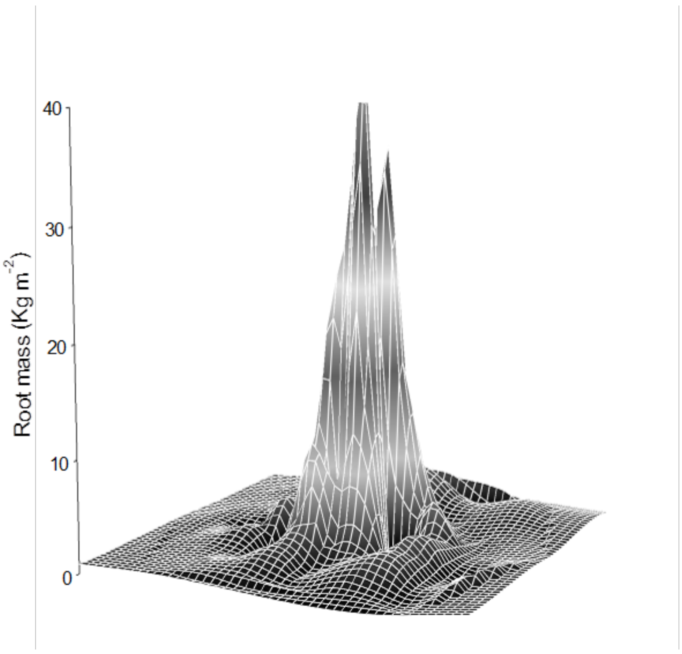


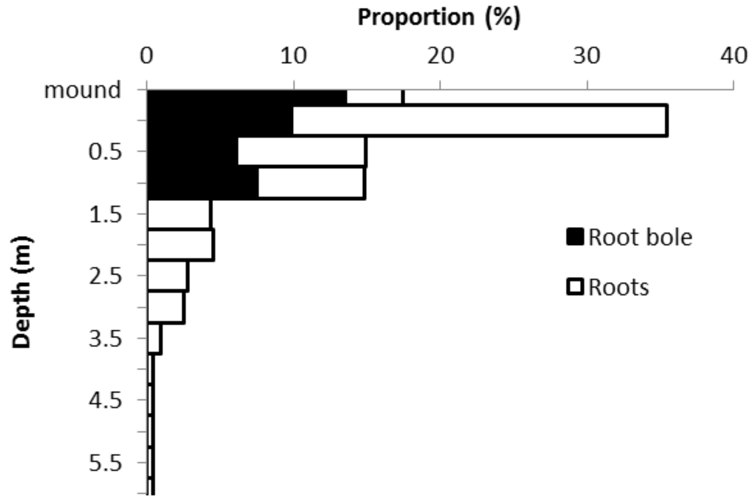
Figure 4.6 A three dimensional rendered surface of root mass density (dry weight) from coring over the sample plot (2 x 4 m) area for the a) 2 and b) 7 year old *E. globulus* sample trees.

4.3.2 Excavation

At both sites approximately half the root mass was recovered from the first 0.5 m of soil depth, with values of 51 and 48% for the 2 and 7 year old trees respectively (

Figure 4.7). The root bole made up 41 and 34% of total root mass for the 2 and 7 year old trees, respectively. A considerable amount (15 to 25%) of the root mass was excavated from the mound even though the mound comprised a relatively small volume of soil. Approximately 85% of root mass exists in the top 1.0 m of soil (including the mound), and this was consistent across the two sites sampled. These values do not concur with that of Schenk and Jackson (2002), who predicted 95% of root mass to occur above a mean depth of 109 cm for Mediterranean shrublands and woodlands. In contrast the depth for 95% of root mass was 200 and 350 cm respectively for the 2 and 7 year old tree.

(a)



(b)

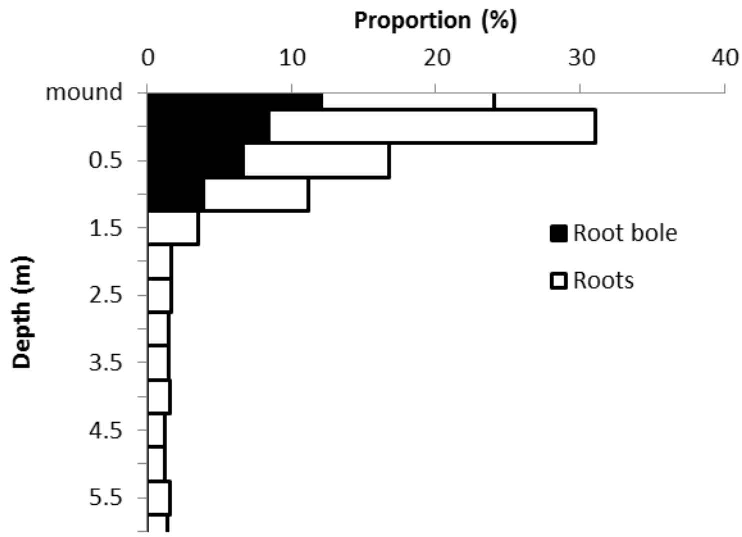


Figure 4.7 Percentage contribution (%) of root bole and roots recovered with depth of excavation for a) 2 and b) 7 year old sample tree.

4.3.3 Simulated sampling scenarios

A range of sampling scenarios were simulated and although many variations were possible, scenarios presented here are those comparable with methods reported in the literature.

4.3.4 Coring only

Uncertainty levels were greatest for the 7 year sample tree for all coring scenarios (Table 4.1). Random coring resulted in uncertainty levels of 47 and 81% for the 2 and 7 year old trees, respectively, from 25 cores. Stratified random coring applied coring to different zones designated radially in terms of distance from the sample tree. When a greater number of cores were placed in close proximity (proximal) to the sample tree and less in the outer zones (distal), uncertainty levels decreased to 38 and 64% for 2 and 7 year old trees, respectively. Nested coring (by depth) improved uncertainty levels marginally as a result of concentrating soil cores to the top 1.0 m soil where the majority of the tree roots occur.

Table 4.1 Summary of sampling uncertainty (U %) associated with different simulated sampling scenarios.

Sampling scenario	Sampling uncertainty U (%)									
	Tree age 2 yr					Tree age 7 yr				
	Number of soil cores									
	5	10	15	20	25	5	10	15	20	25
Coring only										
Random coring	122	86	69	57	47	201	134	109	91	81
Stratified random coring (proximal)	76	58	48	46	38	126	84	78	72	64
Stratified random coring (distal)	189	92	65	59	50	216	137	105	87	82
Nested random coring	83	61	51	37	34	119	93	75	64	59
Bulk excavation + coring	43	33	22	20	19	89	60	46	39	34
Excavation by root diameter limit + coring										
20mm diameter limit	67	46	34	30	28	147	105	83	71	62
10mm diameter limit	57	42	33	30	28	146	97	81	68	62
5mm diameter limit	53	39	33	25	26	140	93	77	70	59
2mm diameter limit	46	31	25	22	20	57	49	43	39	34
Root ball excavation + coring										
Zone 1	57	41	33	29	24	142	98	79	67	61
Zone 2	13	9	7	6	6	49	35	28	25	23
Zone 3	10	7	6	5	5	19	13	11	9	8

To illustrate the effect of coring depth on uncertainty of estimates coring was applied for a range of depths for the 7 year old tree. The uncertainty of estimates decreased as coring depth increased, the greatest change being within the first 1 m depth, beyond which the increasing the depth of sampling had less effect on uncertainty of estimates (Figure 4.8).

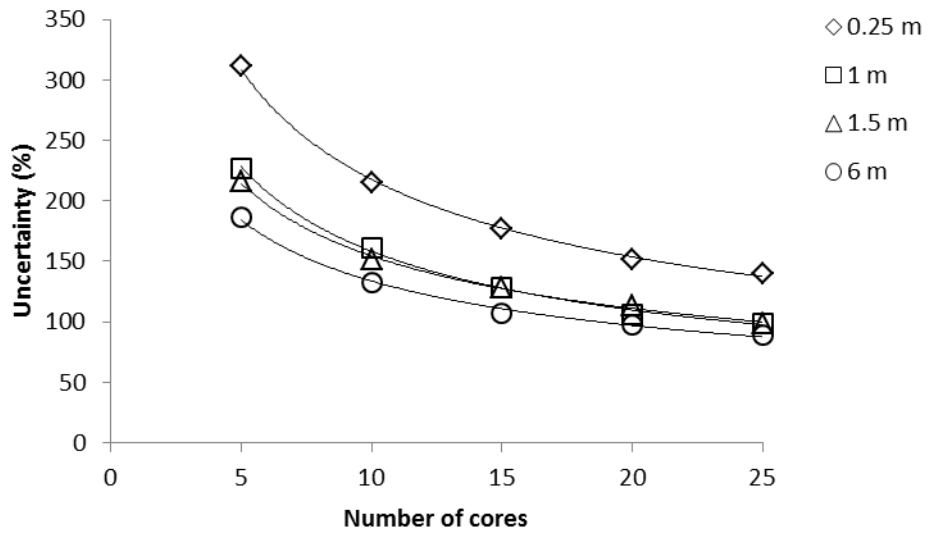


Figure 4.8 The effect of coring depth and number of (random) cores on uncertainty of root biomass estimates for the 7 year old sample tree.

4.3.5 Bulk excavation plus coring

Although this scenario simulates excavation to a predetermined depth, stump pulling in the field recovers coarse and fine roots attached to the root bole and these may extend beyond the depth of excavation. Roots extending beyond the plot boundaries are severed at the plot boundary during excavation. For bulk excavation scenarios, generated uncertainty levels were 19 and 34% with 25 cores for the 2 and 7 year old trees, respectively (Table 4.1).

Incomplete excavation was simulated for the 7 year old sample tree by applying a range of excavation depths beginning with a shallow excavation depth of 0.25 m which simulated the effect of leaving coarse roots in the soil profile. Sampling uncertainty was reduced from 180 to 59% with an increase in excavation depth from 0.25 to 1.0 m with 5 cores (Figure 4.9). With the progressive removal of the large root system sampling uncertainty decreased reducing the number of cores required to attain given levels of uncertainty.

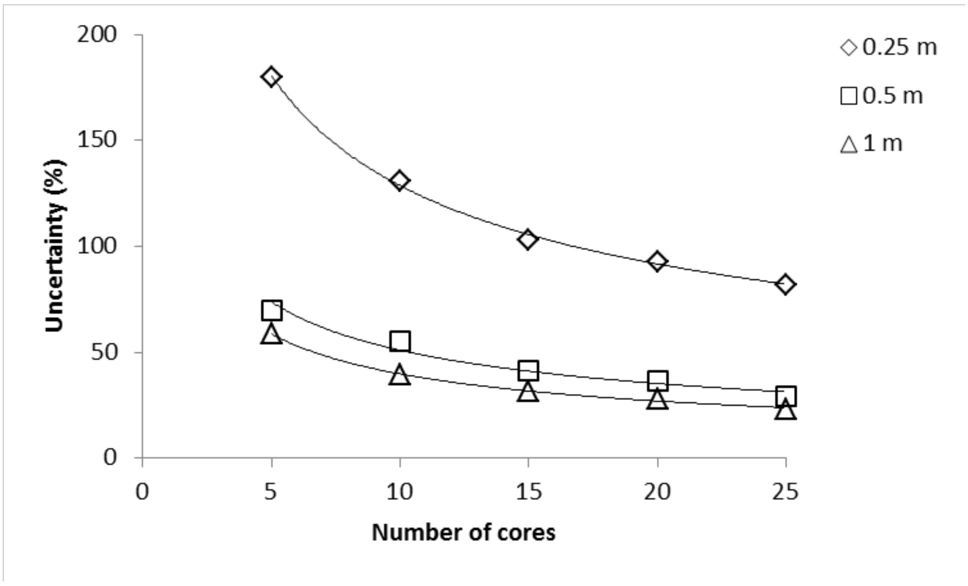


Figure 4.9 The effect of excavation depth and coring (random) on uncertainty estimates for the 7 year old sample tree.

4.3.6 Excavation by root diameter limit plus coring

Excavating to a particular root diameter utilizes techniques that allow roots to be associated directly with the sample tree, for example by excavating with compressed air or water. The effect of excavating to a diameter limit was more pronounced with the 7 year old tree where progressively removing large diameter roots resulted in greater reductions in sampling uncertainty from 147 to 57% for 20 and 2 mm root diameter limits respectively with 5 cores (Table 4.1).

4.3.7 Root ball excavation and coring

Root ball excavation simulated the removal of all roots in close proximity to the tree root bole and sieving with a fine mesh sieve (2 mm) to collect all roots within a defined radius around the sample tree in conjunction with coring for distal roots. The aim of this scenario was to focus sampling more intensively where most root biomass occurs. Excavation to a depth of 1 m in combination with 25 random cores resulted in an uncertainty of 24 and 61% for the 2 and 7 year old trees, respectively for zone 1, an area approximately 1 meter in diameter (Table 4.1). Excavating to include a larger radius around the sample tree (zones 2 and 3) further reduced sampling uncertainty, this being more pronounced for the 7 year old tree. Inclusion of zone 3 for the 2 year old tree only marginally improved sampling uncertainty (Table 4.1).

4.3.8 Bias of sampling scenarios

All simulated sampling scenarios consistently underestimated root mass when compared to the actual root mass recovered for each tree. Overall, bias of sampling was lower for the 7 year old tree for all sampling scenarios, with root mass estimates ranging from -6.0% for coring only to -9.9% for the root ball method (Table 4.2). For the 2 year old tree bias of sampling ranged from -9.8% for bulk excavation to -13.7% for root ball excavation. Root ball excavation had the greatest bias in root mass estimates for both tree ages.

Table 4.2 Sampling bias (%) for different root sampling methods in *E. globulus* trees of two ages.

Root sampling method	Sampling bias (%)	
	Tree age 2 yr	Tree age 7 yr
Coring only	-11.2	-4.7
Bulk excavation	-9.8	-8.2
Excavation to diameter limit	-11.9	-7.4
Root ball excavation	-13.7	-9.9

4.4 Discussion

4.4.1 Sampling methods

This study has resolved several issues raised by Snowdon et al. (2002) in relation to root sampling protocols for carbon accounting. These include sampling methods for defining the root distribution in relation to distance from the sample tree and methodological difficulties in sampling close to trees and coring through roots > 25 mm in diameter. The coring regimes applied in this study also addressed short-falls highlighted by Addo-Danso et al. (2016), that root studies which applied coring often under estimated coarse root biomass directly under the tree stem. For this study a coring apparatus was developed (Chapter 3) which enabled soil coring to be undertaken close to sample trees and through large diameter roots to a depth of 6 m and was capable of sampling tree root systems on sloping ground.

The sampling developed a detailed root distribution dataset and this allowed the testing of a range of sampling scenarios for tree root biomass studies. Based on the statistical relationship between standard error and sample size, sampling uncertainty can be reduced to any desired value by increasing the sample size. For example, for simulations generated here, increasing the sample size by a factor of 4 will reduce the uncertainty level by half for trees of similar size. Thus, with a simulated uncertainty of 50% for a single tree, approximately 25 trees are required to attain a precision of 10% for stand estimates of root biomass. For the development of species specific allometric relationships Snowdon et al. (2002) recommend a sample size of 20 to 40 trees for above ground biomass and a minimum of 20 root system excavations for below ground biomass. Thus, the simulated uncertainty estimates can be a guide to the number of tree root systems which need to be sampled in conjunction with above ground biomass estimates. Although only one species has been used in this evaluation, tree root architecture or morphology has a strong influence on sampling estimates and the observed trends in uncertainty for tree age, root diameter and depth of sampling provide a basis for sampling regimes for root biomass of other tree species.

4.4.2 Soil coring methods

High levels of uncertainty were associated with coring only scenarios and this was influenced primarily by tree age and root distribution. Tree age increased sampling uncertainty as a result of the greater heterogeneity of root mass associated with the presence of large diameter roots (Table 4.1). This was also encountered by Laclau (2003) who found lower coefficients of determination (r^2 values) for coarse roots for 20 year old Ponderosa pine (in Capelco, Argentina) compared to 10 year old stands. The effect of root distribution on sampling uncertainty was demonstrated when coring was stratified and applied at different distances from the sample tree (Table 4.1). Simulations showed that concentrating a greater number of core samples in close proximity to the sample tree resulted in a lower uncertainty of estimates. Often in tree root biomass studies, coring is randomly allocated within tree plots (Barton and Montagu, 2006) or systematically over plot areas (Resh et al., 2003; Saint-Andre et al., 2005). However, from the simulations generated here, precision of estimates can be improved by concentrating coring closer to the root bole. Heterogeneity of root diameter is greatest in proximal roots and decreases with increasing distance from the root bole to a more uniform diameter range or root mass distribution for distal roots. As a result of this reduced heterogeneity, the soil coring intensity has less influence on sampling uncertainty of estimates further away laterally from the root bole and with depth.

Direct comparisons between studies applying coring techniques is compounded by the use of different corer diameters or relative area sampled, which will affect the precision of estimates and the number of cores required (Bohm, 1979; Ping et al., 2010). The corer diameter used in the simulations in this study was 103 mm, which is 65% larger in area than the 80 mm corer used by Levillain et al. (2011) and 15% larger than the corer used by Resh et al. (2003). Coring with small diameter cores can be problematic where there are low root densities (Do Rosário et al., 2000) and this may be indicative of the greater bias and subsequent under-estimation of root mass observed for the 2 year old tree. As reviewed by Addo-Danso et al. (2016) and demonstrated in this paper, coring is not the optimum method for coarse root biomass sampling however, coring was shown to be effective for estimating fine root biomass and is a primary method for sampling tree root systems at depth. As highlighted by Schenk and Jackson (2002) very few comprehensive data sets exist for tree root systems which have been sampled to their full extent or depth within the soil profile.

4.4.3 Excavation methods

Excavation methods are varied in an attempt to minimize the effort required to collect a representative sample of the tree root system for root mass estimates and is typically applied in conjunction with coring (Levillain et al., 2011). Excavation depth is primarily affected by tree age and/or the root morphology of the tree root system. Tree age was shown to increase sampling uncertainty for all excavation regimes, as the presence of large diameter roots increases the heterogeneity of root mass as seen in the diameter distribution of the sample trees. The bulk excavation scenario involved excavating the root bole and sieving all roots within a tree plot area to a predetermined depth as applied by Ritson and Sochacki (2003), however, the depth of excavation is crucial as demonstrated by simulations of inadequate excavation (Figure 4.9). Root morphology and subsequent depth of excavation will govern the effectiveness of this sampling regime and when applying this methodology it is important to ensure all coarse roots are removed in the excavation process as this will affect uncertainty of estimates of subsequent soil coring. Similar results were demonstrated in a study by Levillain et al. (2011) who compared precision of excavation techniques in relation to root diameter and showed that precision of estimates decreased with increasing root diameter and also that precision of estimates increased with depth.

Simulations of excavation by diameter limit also demonstrate the effects of tree age and the presence of large diameter roots. Root diameter limit excavation has been applied in studies in an attempt to improve the efficiency of excavation (Barton and Montagu 2006) and also in an attempt to associate a greater proportion of root mass to the sample tree, for example in young tree stands where root systems have not overlapped with neighboring trees (Brand, 1999). Barton and Montagu (2006) applied a root diameter limit of 15 mm for the excavation of 10 year old *Eucalyptus camaldulensis* in Deniliniquin, New South Wales, Australia and found that coring for roots <15 mm resulted in large margins of error (±59%) for roots 5 to 15 mm in diameter, confirming the results of diameter limit simulations.

The root ball scenario required the least number of cores to attain sampling uncertainty levels below 50% and required less soil coring than the bulk excavation approach however, simulations are based on sieving bulk material through a fine sieve which may be time consuming and problematic in heavy or clay soils. Resh et al. (2003) applied the root ball method to 8 year *E. globulus* and *E. nitens* and estimated between 61 to 85% of the coarse roots to be within the root ball, which measured approximately 1 $m,^2$ and applied coring to estimate the remaining mass of coarse roots. Resh et al. (2003) also found increasing root mass heterogeneity with increasing root diameter and concluded that much higher coring intensity is required to improve the accuracy of estimates primarily due to the presence of >20 mm diameter roots. For mixed species in close proximity the root ball method has been used effectively to account for a high proportion of root mass for specific tree species (Burrows et al., 2000). In comparison, Jonson and Freudenberger (2011) applied root ball methods to an open spaced native woodland of varying ages relying solely on roots recovered from excavating 1 m^2 without soil coring. In this instance, it could be expected that root mass was substantially underestimated, particularly for large trees. Addo-Danso et al. (2016) suggest the root ball method is the most effective excavation method which concurs with the findings here for sampling precision however, concerns have been raised over the coarse roots remaining below the root bole for trees with large tap roots (Addo-Danso et al., 2016) and indeed this was reflected in the accuracy of this method. We recommend the removal of the root bole via excavation followed by soil coring for roots below the root bole.

Root morphology will have some influence in determining sampling strategies and prior knowledge of the root form being sampled would be advantageous. Preliminary excavation of root systems would give some indication of the root form and extent of excavation and coring required. For example, tree stands with widely spaced mature trees were not included in this study, but have been sampled by Ritson and Sochacki (2003) in which coarse root biomass of *Pinus pinaster* was accounted for with extensive excavation to comprehensively capture the entire tree root system. Root systems vary in their lateral and vertical extent and these traits are governed primarily by species, age and soil profile. However, very few studies exist on the depth of tree root systems as documented in a review by Maeght et al. (2013), and further research is required to investigate the below ground ecosystem processes of tree roots and their effect on the dynamics of carbon cycling.

4.4.4 Accuracy of tree sampling methods

As described in Chapter 2, few studies exist on the uncertainty of tree root biomass estimates (Levillain et al., 2011; Paul et al., 2014b) however none address the accuracy of root sampling methodologies. In this study sampling uncertainty (precision) and bias of sampling methods were determined for a range of sampling scenarios via simulations with actual root data sets. The application of Monte Carlo simulation has helped identify aspects of tree root sampling methodologies which can be modified to improve the precision of tree root biomass estimates and demonstrated the accuracy of the sampling regimes simulated. The techniques described here have been used to sample *P. pinaster* grown on farmland in southwestern Australia (Ritson and Sochacki, 2003) and within older stands of a single species similar methodology can be applied. However, stands of mixed species would be more problematic and allocating roots to a particular sample tree may require specialized excavation techniques using air or water excavation (Bohm, 1979).

The lack of standardized root sampling methods and inadequate replication is a familiar theme in relation to tree root biomass data and the contribution of this to the uncertainty of forest carbon accounts (Mokany et al., 2006). Simulated sampling scenarios can be used as a guide for sampling design and as an indication of the number of root systems required to achieve given levels of precision for carbon inventories. It was demonstrated that concentrating a greater number of core samples in close proximity to sample trees by stratification resulted in a 10 to 15% lower sampling uncertainty, this has not been previously reported and should be taken into account when applying coring regimes. Tree age is a crucial factor that should be taken into consideration when sampling tree root systems, and is particularly important if the root ball method is applied. As demonstrated here, bias is greatest for this sampling regime and this has also been noted in other studies which have applied this method (Resh et al., 2003; Jonson and Freudenberger, 2011). Resh et al. (2003) compared coarse root biomass estimates from coring with excavation and found coring underestimated coarse root biomass by 9%, which is within the range of 6.0 to 11.2% for simulated sampling of coring for the 7 and 2 year old trees respectively.

The efforts invested in coring to greater depths will need to be justified against the additional amounts of root biomass that may be recovered, as deep rooted species will be difficult to sample and therefore result in underestimates of tree root carbon stocks in global mitigation strategies. In this study, approximately 85% of the root mass was recovered following excavation to a depth of 1 m for both tree ages. Although the depth of rooting across many species has been examined (Canadell et al., 1996; Jackson et al., 1996; Laclau et al., 2013; Pinheiro et al., 2016), there has not been a systematic review of biomass depth functions in relation to species, age or different site conditions. This is an area of future work and will be a crucial component of better understanding carbon dynamics in forested systems. In a global biogeography of roots Schenk and Jackson (2002) report that sampling depths were often insufficient to characterize root profiles and indeed the results collected here indicate greater proportions of root biomass deeper in the soil profile than extrapolated by Schenk and Jackson (2002).

4.4.5 Global carbon accounts

Improved accuracy of estimates of forest tree root carbon pools will help improve global estimates of forest carbon sinks, an area associated with high levels of uncertainty for national carbon accounting (Heath and Smith, 2000). The lack of tree root biomass data has resulted in a reliance on default r:s ratios to predict root biomass from known shoot biomass (IPCC, 1996a; IPCC, 2003) in estimating national forest carbon stocks. However, the inadequacies of applying these default values has been demonstrated by Mokany et al. (2006), and the reported sensitivity of global carbon stocks to small changes in r:s ratio, further reinforcing the need for adequate tree root sampling methodology and accuracy of tree root biomass or carbon estimates. For tropical forest systems the lack of tree root carbon estimates also extends to emissions estimates for REDD+ (Reducing Emissions from Deforestation and Forest Degradation) initiatives, which are typically estimates of above ground carbon (Sills et al., 2014). These shortfalls have important implications for forest carbon sink projects, bioenergy systems and ultimately, the effectiveness of climate change mitigation programs and associated carbon models.

4.4.6 Conclusions

The measurement of major tree root systems for the purpose of biomass and carbon estimation is crucially important, however, access to complete root system data sets is rare. Comparisons between studies is difficult as different sampling techniques have been used and the inherent heterogeneity of tree root systems requires extensive sampling to achieve results of sufficient precision. Excavation techniques for the measurement of below ground biomass are a means of associating root mass directly to individual sample trees, from which allometric relationships can be developed for future non-destructive estimates of carbon stocks. The results presented here can be used as a guide in designing appropriate tree root sampling regimes to improve precision and accuracy of root biomass estimation. Precision of coring estimates can be improved by 10 to 15% by focusing coring in closer proximity to sample trees where large diameter roots predominantly occur. Excavation of whole (plot) tree root systems can result in greater accuracy of estimates compared to root ball methods by up to 4%. These data are critical for the development of allometric relationships for carbon stock estimation and data for global carbon models. In Chapter 5 the results from these sampling simulations were taken into account in designing sampling regimes for biomass sampling of tree phases and subsequent development of biomass equations.

5 Estimation of woody biomass production from a short rotation bio-energy system in semi-arid Australia[3]

5.1 Introduction

Prior to the development of large areas of south-western Australia for farming, salinity was confined to natural playa systems in valley floors (Harper and Gilkes, 2004). The replacement of deep rooted natural vegetation with dryland agricultural crops and pastures has led to a hydrologic imbalance, with rising water tables remobilizing salts stored in soil profiles that are often many metres deep (Peck and Hatton, 2003). This in turn has resulted in the widespread salinization of land and water supplies. In Australia the area of land at risk to secondary salinity is estimated to be 17 million hectares by 2050 (National Land and Water Resources Audit, 2001).

Studies of both large (Bari et al., 2004) and small (Clarke et al., 2002) watersheds support the proposition that re-inserting deep-rooted perennials into the farming systems can slow, halt or even reverse dryland salinity. Similarly, studies of the water content under trees indicate that the roots of Eucalypts in particular can penetrate to depths of >10 m after 7 years, thus preventing recharge to ground waters (Robinson et al., 2006). The control of salinity will thus require the establishment of perennial plants extensively and strategically distributed across the landscape and integrated into the existing agricultural systems (Harper et al., 2001).

[3] Published as: Sochacki S.J., Harper RJ and Smettem, K.R.J. (2007). Estimation of woody biomass for short rotation bio-energy species in south-western Australia. *Biomass & Bioenergy* **31**, 608-616.

It has been suggested, however, that the proportion of the landscape requiring treatment to regain hydrological control could be up to 80% (George et al., 1999). This presents the conundrum that the treatment required to stabilize the farming systems may in fact displace them. Although trees planted in strips, integrated with farming, may stop recharge to groundwater they also obtain water from adjacent farmland (Robinson et al., 2006) and are therefore competitive with crops. Similarly, trees placed permanently across farmland at cover levels proposed by George et al. (1999) can often lead to uneconomical farm returns due to the yield that must be forgone by crop displacement. In addition, the area where much salinity occurs (300 to 600 mm yr^{-1} rainfall) is also outside the zone where trees are traditionally grown for wood production due to low yields and large transport distances to markets.

To resolve these issues, *viz*. integrating the benefits of trees into dryland farming systems, whilst still allowing farming to occur over an area that is sufficiently large to be economic, Harper et al. (2000) proposed inserting short rotations (3 to 5 years) of high water use tree species to restore landscape hydrology in a system termed phase farming with trees (PFT). Resultant benefits could include not only the cessation of recharge, by creation of dry soil buffer zones over the depth of the tree roots, but also improvements in soil structure, fertility and the control of herbicide resistant weeds in cropping systems. The trees themselves would represent a source of wood fibre or biomass for bioenergy production (Harper et al., 2000).

Widespread adoption of phase farming with trees would produce large amounts of biomass from small trees. There are however no published yield data for potential biomass crops grown in Western Australia and particularly in the 300 to 600 mm rainfall zone where the PFT system has potential. This Chapter is designed to provide this information for several prospective species by development of allometric equations to estimate the biomass yield that can be achieved after 36 months and determine the effects of site conditions and stocking on yield.

5.2 Materials and Methods

5.2.1 Location

The study site (S4) was located near Corrigin, Western Australia, approximately 240 km east of Perth (Figure 3.4), (117°41’47.13”E; 32°23’24.67”S) the State capital. This site was selected as having soils and landforms representative of the general region (McArthur, 1991). It has a semi-arid Mediterranean climate, with a seasonal drought from November to April, a mean annual rainfall (1889-2001) of 365 mm yr^{-1} and mean annual pan evaporation of 1789 mm yr^{-1}. The mean rainfall during the three years of the experiment was 304 mm yr^{-1}.

Conventional farming involves annual rotations of cereal (*Triticum aestivum*, *Hordeum vulgare*) or legume (*Lupinus angustifolius*) crops with improved annual legume (*Trifolium subterraneum*) and grass (*Lolium rigidum*) pastures, grown during the winter rainfall season. It is thus similar to the farming systems of broad areas of southern Australia (Squires and Tow, 1991).

5.2.2 Experimental design

An experiment was established in August 2001 (i.e. winter) to determine the potential of short rotation tree crops to remove excess soil water to depth (6 to 8 m) and create a buffer of dry soil to capture the leakage that occurs below the shallow root zone of subsequent annual crops (Harper et al., 2008). The experiment was designed to determine whether water use and biomass production could be manipulated by species selection, planting density or fertilizer application. The trial design consisted of three replicate blocks, each with 25 treatments comprising five species; *Eucalyptus globulus*, *Eucalyptus occidentalis, Pinus radiata, Allocasuarina huegeliana* and *Acacia celastrifolia*, planted at 500, 1000, 2000 and 4000 trees ha^{-1}, as well as 500 trees ha^{-1} plus nitrogen fertilizer applied at 100 kg N ha^{-1}. There was no response to the fertilizer treatment, thus these plots are not considered further here. Trees were planted by hand, following treatment of the site with herbicides in 50 m x 50 m plots.

The three blocks were situated in the same field, but arrayed in different landscape positions. Block 1 (upper-slope) was on a gravelly ridge, Block 2 (mid-slope) in a concavity with a sandy duplex profile and Block 3 (lower-slope) with a sandy duplex profile with a moderately saline water table at 2-3 m. All sites had deeply weathered profiles, typical of the region, to at least 10 m depth.

The *Allocasuarina huegeliana* and *Acacia celastrifolia* treatments failed in terms of both growth and water depletion and have thus been discounted as candidate species for phase farming. This Chapter reports on the three remaining species, *E. globulus, E. occidentalis* and *P. radiata*. Water use of these species and final yield are described in (Harper et al., 2014).

5.2.3 Biomass sampling

5.2.3.1 Sample tree selection

Trees were sampled at 36 months of age in August 2004, with the sample including a range of tree sizes and form representative of the stand. Ten trees were selected from each treatment (500, 1000, 2000 and 4000 trees ha^{-1}) at each of the 3 landscape positions, with 110 in total from *E. globulus* and *E. occidentalis* and 120 from *P. radiata*.

5.2.3.2 *Measurement of predictor variables*

Predictor variables measured were tree height, crown volume and diameter over bark (DOB) at 10, 50 and 130 cm above ground level. Due to the young age of the trees, range of tree heights and variation in tree form, three diameter measurements were taken. *E. occidentalis* often had more than one stem at 130 cm and a diameter equivalent was calculated when more than one stem was measured (Avery and Burkhart, 1983). The diameter equivalent is the diameter of a circle of area equal to the sum of the cross-sectional areas of all stems measured and was calculated as

$$D_h = \left(\frac{4\sum_{i=1}^{n} A_{hi}}{\pi} \right)^{1/2} = \left(\sum_{i=1}^{n} D_{hi}^2 \right)^{1/2} \tag{3}$$

where

Dh = diameter equivalent for the measurement height,

Ahi = cross-section area of the *i*th stem at measurement height and

Dhi is the diameter of the *i*th stem at the measurement height.

A crown volume index (*CVI*) was derived from crown measurements as

$$CVI = dlw \tag{4}$$

where

d = crown depth measured from the crown base to the tree top,

l = crown length along tree row and

w = crown width across the row.

It was not always possible to measure stem diameter at the nominated measurement height due to branches or forking at that point. Where this occurred *Dh* was estimated assuming a conical stem shape (D $\propto$ height) as

$$D_h = \frac{(Ht - h)}{(Ht - Hpom)} Dm \tag{5}$$

where

Ht = tree height,

h = measurement height (10, 50 and 130cm),

Hpom = height of point of measurement and

Dm measured diameter at the *pom*.

5.2.4 Destructive sampling

5.2.4.1 Above-ground

The procedures of Snowdon et al. (2002) were used for biomass sampling. Following the measurement of predictor variables, trees were felled and total fresh weight was measured using a purpose-built system of bi-pod frame (Figure 5.1), pulleys and scales. Measurements were made to an accuracy of 0.1 kg.

Four above-ground tree component categories were measured: (1) leaf and twig (<15 mm diameter); (2) stems and branches (>15 mm diameter); (3) dead branches; and (4) ground litter. A nominal branch diameter of 15 mm was used to differentiate components (1) and (2) and attain more homogenous material and therefore more consistent moisture values.

Figure 5.1 Determining component fresh weights with a purpose built bi-pod and weighing scales.

To determine the dry biomass without having to dry and weigh the whole tree, sub-sampling of tree components was undertaken to derive moisture ratios. Component sub-samples of 300 to 1000 g were collected and weighed at the time of sampling, then placed into calico bags for oven drying at 70ºC to constant weight. Dry weights of sub-samples were used to calculate moisture ratios which were then applied to the tree component fresh weights. Following drying, sub-samples were selected to determine leaf:twig and bark:stem ratios. These were then applied to the total weights for each treatment and site.

Ground litter (leaf litter) beneath sample trees was typically within an area of approximately 1 m in diameter and was collected from around the base of trees with the use of a lawn rake. Quantities of litter collected were small and were dried as a whole sample.

5.2.4.2 Below-ground

Sampling of below-ground biomass was informed by the excavation and analysis of root mass distribution determined in Chapter 4. Based on tree root mass distributions root excavation to diameter limit was applied however, coring was not applied and the focus of sampling was on the major root system and did not include sampling the distal and fine root systems.

Tree proximal roots were excavated with an excavator and collected by hand to a nominal root diameter limit of 5 mm, which corresponded to a maximum soil depth of approximately 0.35 m. Specht and West (2003) also excavated roots by hand to a minimum of 5 mm diameter with the aid of water. Excavation took place within two weeks of the above-ground sampling under dry soil conditions, which enabled any adhering soil to be removed by agitating the root system with the excavator bucket. Other detailed techniques such as bulk sieving and coring that are typically used to sample distal roots were not employed. Most of the root systems excavated were dried and weighed whole, using the procedure described above, with only very large roots weighed fresh and sub-sampled.

The aim was to collect root material in a manner similar to harvesting methods that may be employed if short rotation tree crops were integrated into dryland farming systems. It is envisaged that the whole tree would be harvested for biomass fuel and the tree roots would be removed in a manner to allow for resumption of cropping (Harper et al., 2000). The procedure used here thus provides an estimate of the tree root biomass that would be recovered in an operational system. A total of seven partitioned categories were thus derived; (1) stem, (2) bark, (3) twigs and branches, (4) leaves, (5) dead branches, (6) ground litter and (7) roots.

5.2.4.3 Allometric (prediction) equations

Allometric equations relate the growth of one part of an organism to another part or the whole organism, and in the case of tree biomass are typically related to easily measured predictor variables, for example diameter, height or crown volume (Keith et al., 1999). Tree size, age or form will affect what parameters are measured as predictor variables. For example, although crown volume has been used in other studies, it could not be used to develop allometric equations here due to crown closure at 4000 trees ha^{-1}.

Techniques for developing prediction equations are described in Clutter et al. (1983). These equations are used to predict stem volume but the same principals can be applied to develop equations for predicting tree biomass. Of the equations given by (Clutter et al., 1983) the following were used:

non-linear:

$$\mathrm{B}t = \mathrm{b}_0 + \mathrm{b}_1 d^{\mathrm{b}2} ht^{\mathrm{b}3} + \epsilon \qquad (6)$$

logarithmic:

$$\ln\mathrm{B}t = \mathrm{b}_0 + \mathrm{b}_1 \ln(d) + \mathrm{b}_2 \ln(ht) + \ln\epsilon \qquad (7)$$

$$\mathrm{B}t = e^{\mathrm{b}0} + d^{\mathrm{b}1} + ht^{\mathrm{b}2} + \mathrm{CF} \qquad (8)$$

weighted non-linear:

$$\mathrm{B}t\, x^{-k} = (\mathrm{b}_0 + \mathrm{b}_1 d^{\mathrm{b}2} ht^{\mathrm{b}3} + \epsilon)\, x^{-k} \qquad (9)$$

where

Bt = tree biomass,

d = stem diameter ($d130$, $d50$ or $d10$),

ht tree height,

$b0$,$b1$,$b2$ and $b3$ the parameters to be estimated and

ϵ = error term.

Equation (8) was log-transformed to reduce heteroscedascity and facilitate fitting by linear regression. A correction factor (CF) was applied to remove bias in the estimate of B*t* through back-transformation (Snowdon, 1991).

To satisfy the condition of constant variance for regression analysis (1983), equation (9) is a weighted form of equation (6) where x^{-k} is a weighting factor, either d^{-k} or $(d^2ht)^{-k}$ if both diameter and height were used as predictor variables. The optimum value of k was selected on the basis of lowest Furnival index (I) (Furnival, 1961) which is used to evaluate and compare biomass models. When the dependent variable is some function of biomass, Furnival's index is an average standard error transformed to the units of biomass (Parresol, 1999).

5.2.4.4 Measurement of stand biomass

Following the development of allometric equations, measurements were made of all treatments to obtain estimates of biomass. Predictor variables for derived allometric equations were measured on all trees within 20 m by 20 m permanent measurement plots. The allometric relationships were used to develop estimates of the biomass of different tree components and total biomass. Yield data were analyzed by analysis of variance using XLSTAT software (Addinsoft, 2005).

No estimates were made of the energy content of the materials produced in this study.

5.3 Results

5.3.1 Tree growth

Tree heights 36 months after establishment varied from 0.9 to 5.8 m, and diameters at breast height from 0.9 to 9.7 cm for *E. globulus* (n = 110), 1.2 to 4.9 m and 0.8 to 9.3 cm for *E. occidentalis* (n = 110) and 1.2 to 3.7 m and 1.0 to 7.4 cm for *P. radiata* (n = 120) (Table 5.1). The largest trees occurred in the 500 trees ha^{-1} treatments with ranges in tree mass for the three species being, 0.2 to 32.1 kg $tree^{-1}$ for *E. globulus,* 0.3 to 31.1 kg tree-1 for *E. occidentalis* and 0.8 to 10.7 kg $tree^{-1}$ for *P. radiata* (Table 5.1).

Table 5.1 Ranges in tree height, stem diameter at breast height (1.3 m) and total (above and below ground) tree biomass for the trees used to develop the allometric equations at Corrigin.

Species	n	Tree height (m)	Stem diameter at 1.3 m (cm)	Total tree biomass (kg)
E. globulus	110	0.9 - 5.8	0.9 - 9.7	0.2 – 32.1
E. occidentalis	110	1.2 - 4.9	0.8 - 9.3	0.3 – 31.1
P. radiata	120	1.2 - 3.7	1.0 - 7.4	0.82 – 10.7

5.3.2 Variation in partitioning between tree components for different species, planting densities and slope position

There were only small amounts of ground litter for *E. globulus* (1.7%) and *E. occidentalis* (1.1%) and none for *P. radiata* (Table 5.2). Only *E. globulus* had any dead branches, but these comprised only a small amount (0.1%) of the total biomass. The two eucalypts comprised around 12% bark, compared to 6% for *P. radiata*. The proportion of stem-wood varied from 17% for *E. occidentalis* to 21 and 23% for *E. globulus* and *P. radiata* respectively. The proportion of leaves varied from around 23% for the two eucalypt species to 30% for *P. radiata*.

The proportion of roots was similar for both *E. globulus* and *P. radiata*, with mean values of 23 and 24% respectively, whereas the values for *E. occidentalis* were much higher at 33%. Expressed as root:shoot (r:s) ratios, these represent values of 0.31 and 0.51 for *E. globulus* and *E. occidentalis* respectively.

Table 5.2 Mean proportion in relation to total biomass for each tree component for each tree species. The root:shoot (r:s) ratio is derived from the total below ground and total above ground biomass.

Species	Proportion of tree components (%)							
	Ground litter	Dead branches	Leaf	Branches and twigs	Stem-wood	Bark	Roots	r:s ratio
E. globulus	1.7	0.1	23.5	17.7	21.2	12.1	23.8	0.31
E. occidentalis	1.1	0.0	22.8	12.0	17.4	12.9	33.9	0.51
P. radiata	0.0	0.0	30.5	15.5	23.4	6.1	24.6	0.33

5.3.3 Allometric equations

Total biomass data were combined for all planting densities and compared with the three stem diameter measurements. The diameter at 10 cm had the highest correlation with biomass, with 96, 92 and 88% of the variation explained for *E. globulus*, *E. occidentalis* and *P. radiata*, respectively. There was no indication of different trends for different planting densities (Figure 5.2), thus it is considered that a single relationship can be used for all trees of a particular species. Height was not as strongly correlated with biomass, with 77, 73 and 70% of the variation explained for *E. globulus, E. occidentalis* and *P. radiata,* respectively. This is consistent with other studies; Burrows et al. (2000) found that stem circumference accounted for 99% of the variation of total above ground biomass for *E. crebra* and *E. melanophloia*, 96% in *E. melanophloia* regrowth, and tree height, 95%, 91% and 92% of the variation, respectively.

Prediction equations for each species are shown in Table 5.3a along with goodness of fit statistics. For all three species equation (8) gave the best result for total biomass based on Furnival's index. Allometric equations were also derived for separate tree components using the logarithmic form, and these are presented in Table 5.3b-d.

5.3.4 Variation in biomass yield, with slope position and density

The surviving tree densities at the end of 12 and 36 months, as a proportion of the nominal planting density, are presented in Table 5.4. Although the experiment was planted at a series of set planting densities (500 to 4000 trees ha^{-1}), weed competition and insect attack resulted in variable survival. For all treatments, apart from the 2000 and 4000 trees ha^{-1} *E. occidentalis* treatments and the 4000 trees ha^{-1} *E. globulus* treatment, all trees that had survived to 12 months were also alive at 36 months.

Estimates of biomass yield were calculated for each treatment by applying the allometric relationships to individual tree data from within the permanent sampling plots (Table 5.5). This procedure took into account the variable final stocking in the plots. Similarly, for those plots with many dead trees, which had died recently (within the previous 6 months), the yield was calculated on stem diameters as for the live trees. Average 3-year cumulative biomass yields ranged from 0.5 to 16.6 t ha^{-1} 3 yr^{-1} for *E. globulus,* 4.0 to 22.2 t ha^{-1} 3 yr^{-1} for *E. occidentalis* and 1.6 to 15.4 t ha^{-1} 3 yr^{-1} for *P. radiata*.

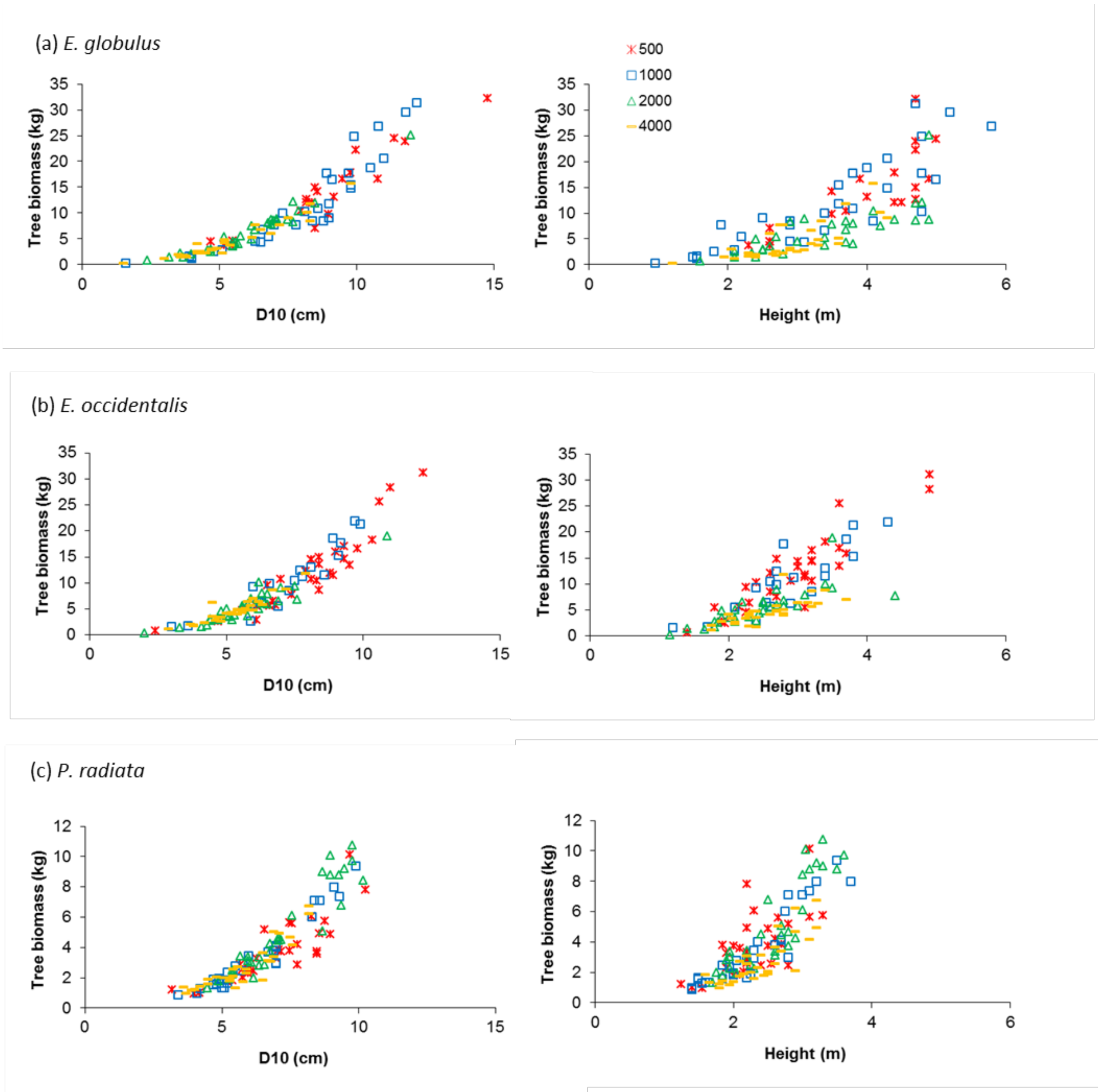


Figure 5.2 Relationships between diameter (cm) at 10 cm and tree height (m) and biomass (kg tree^{-1}) for (a) *E. globulus*, (b) *E. occidentalis* and (c) *P. radiata* for each planting density.

Table 5.3 Prediction equations for (a) whole tree biomass (kg tree^{-1}), (b) leaves (kg tree^{-1}), (c) stems (kg tree^{-1}) and (d) roots (kg tree^{-1}) derived from measurements of stem diameter and tree height for *E. globulus*, *E. occidentalis* and *P. radiata*.

Species	**Prediction equation**	***n***	***r*²**	***FI***	***s.e. (kg)***	***CV (%)***
(a) Whole tree						
E. globulus	$\mathrm{B}t=1.013e^{-2.523}d_{10}^{1.92}ht^{0.642}$	110	0.97	0.95	1.6	18
E. occidentalis	$\mathrm{B}t=1.01e^{-2.429}d_{10}^{1.948}ht^{0.702}$	110	0.93	0.93	1.6	20
P. radiata	$\mathrm{B}t=1.024e^{-2.665}d_{10}^{1.689}ht^{0.804}$	120	0.92	0.92	0.7	18
(b) Leaf						
E. globulus	$\mathrm{B}l=1.022e\text{-}^{3.553}d_{10}^{1.806}ht^{0.533}$	110	0.94	0.84	0.7	31
E. occidentalis	$\mathrm{B}l=1.032e^{-3.483}d_{10}^{1.856}ht^{0.505}$	110	0.87	0.78	0.6	33
P. radiata	$\mathrm{B}l=1.027e^{-3.69}d_{10}^{1.74}ht^{0.576}$	120	0.85	0.86	0.3	22
(c) Stem						
E. globulus	$\mathrm{B}s=1.018e^{-4.078}d_{10}^{1.920}ht^{0.963}$	110	0.98	0.96	0.5	18
E. occidentalis	$\mathrm{B}s=1.013e^{-4.067}d_{10}^{1.932}ht^{1.112}$	110	0.96	0.97	0.4	17
P. radiata	$\mathrm{B}s=1.032e^{-3.954}d_{10}^{1.328}ht^{1.582}$	120	0.91	0.89	0.3	24
(d) Roots						
E. globulus	$\mathrm{B}r=1.076e^{-3.938}d_{10}^{2.206}ht^{0.099}$	110	0.77	0.91	0.5	26
E. occidentalis	$\mathrm{B}r=1.029e^{-3.434}d_{10}^{1.923}ht^{0.635}$	110	0.87	0.87	0.7	27
P. radiata	$\mathrm{B}r=1.032e^{-4.082}d_{10}^{1.777}ht^{0.548}$	120	0.88	0.86	0.2	23

B*t* total biomass, B*l* leaf biomass, B*s* stem biomass, B*r* root biomass, *n* number of sample trees, r^2 proportion of variation explained, *FI* Fit index, *se* standard error of estimate and *CV* coefficient of variation.

Table 5.4 Survival of trees at 12 and 36 months (%) as a percentage of the initial planting density for each species, treatment and site.

Species	Slope position	Tree survival (%)							
		500 trees ha^{-1}		1000 trees ha^{-1}		2000 trees ha^{-1}		4000 trees ha^{-1}	
		12 m	36 m	12 m	36 m	12 m	36 m	12 m	36 m
E.globulus	upper-slope	30	30	50	48	51	50	46	43
	mid-slope	95	95	88	88	100	76	89	51
	lower-slope	95	95	98	93	96	75	100	82
E.occidentalis	upper-slope	80	80	85	85	73	70	79	6
	mid-slope	95	95	*	*	98	15	99	1
	lower-slope	100	100	93	90	96	95	99	97
P. radiata	upper-slope	100	95	53	53	98	98	98	98
	mid-slope	100	100	100	100	99	99	96	96
	lower-slope	85	85	83	83	94	94	85	85

* data unavailable for these treatments

Table 5.5 Estimates of total biomass produced (Bt ha^{-1} 3 yr^{-1}) at 36 months, with actual stocking (trees ha^{-1}) at 12 months in parenthesis, for each species, treatment and site.

Species	Slope position	Total biomass (Bt ha^{-1} 3 yr^{-1}) 500 trees ha^{-1}		1000 trees ha^{-1}		2000 trees ha^{-1}		4000 trees ha^{-1}	
E. globulus	upper-slope	0.5	(150)	1.8	(500)	4.3	(1020)	8.5	(1840)
	mid-slope	6.2	(475)	12.0	(880)	10.6	(2000)	9.8	(3560)
	lower-slope	8.9	(475)	9.6	(980)	11.0	(1920)	16.6	(4000)
E. occidentalis	upper-slope	4.0	(400)	5.7	(850)	5.6	(1460)	9.1	(3160)
	mid-slope	5.7	(475)	*	*	8.3	(1960)	9.2	(3960)
	lower-slope	10.9	(500)	14.1	(930)	12.0	(1920)	22.2	(3960)
P. radiata	upper-slope	1.5	(500)	1.7	(530)	5.6	(1960)	15.4	(3920)
	mid-slope	2.1	(500)	3.1	(1000)	12.6	(1980)	14.8	(3840)
	lower-slope	2.3	(425)	10.5	(830)	14.3	(1880)	5.4	(3400)

* data unavailable for this treatment

For each species there was an increase in yield with planting density, and slope position with this being highly significant ($P<0.001$). The highest biomass yields for *E. globulus* and *E. occidentalis* were from 4000 trees ha^{-1} treatments located on the lower-slope site (Table 5.5). For a density at 12 months of around 2000 trees ha^{-1}, *E. globulus* yields were 8.5, 10.6 and 11.0 t ha^{-1} 3 yr^{-1} for upper, mid and lower slope sites, respectively (Table 5.5). For *P. radiata*, the highest yields were observed at 4000 trees ha^{-1} in the upper-slope site. Mean yields of the three species, in the high planting density plots, were not significantly different and ranged from 12 to 14 t ha^{-1} 3 yr^{-1}.

5.4 Discussion

Both planting density and the position of trees in the landscape had a strong influence on biomass yield. There was a consistent increase in biomass yield with increasing planting density, indicating the benefits of using high stocking to rapidly occupy the sites. If it is assumed that the three landscape positions comprise similar proportions across the landscape, mean total biomass yields at the highest planting density were 11.8, 13.5 and 11.9 t ha^{-1} 3 yr^{-1}, for *E. globulus*, *E. occidentalis* and *P. radiata*, respectively. The experimental site experienced relatively dry conditions during the experiment (287 mm yr^{-1} rainfall vs. a long-term mean of 375 mm yr^{-1}) and assuming that productivity is generally related to rainfall (Harper et al., 2005), higher yields will be expected to be achieved under average conditions.

Trees planted in lower landscape positions had greater yields and this is likely to be the result of greater water availability, either through the accumulation of run-off from up-slope areas or access to groundwater. For the highest (4000 trees ha^{-1}) planting density *E. globulus* achieved a yield of 16.6 t ha^{-1} 3 yr^{-1} in this setting and *E. occidentalis* 22.2 t ha^{-1} 3 yr^{-1}. As groundwater in this environment is semi-saline, the utilization of this water for tree growth will vary between species, and their natural tolerance of salinity. *E. globulus* is a species from high rainfall regions and did not perform well on the upper-slope site where soil water was less available, however the lower yield on these sites was partially due to poor post planting survival (Table 5.4). *E. occidentalis* is a species with some salt tolerance as it naturally grows adjacent to saline playas. In contrast, the largest yield of *P. radiata* (15.4 t ha^{-1} 3 yr^{-1}) was achieved in the upper landscape site from a planting density of 4000 trees ha^{-1} with yields at this density in the lower landscape being relatively small. An obvious question relates to whether yields can be further promoted using even higher planting densities.

These contrasting results between species are fortunate and indicate that different species may be required for different hydrological settings. Variations in landscape characteristics and subsequent water availability can be accounted for by using appropriate species in those landscapes. High mortality after 32 months in some high density, upper landscape plots indicates that the available soil moisture has been exhausted and that a limit of biomass production has been achieved.

Of the allometric equations the logarithmic form Eqn.7 was the best with predictors of diameter at 10 cm (D10) and tree height (Ht) for all three species. The relationship between diameter and height often varies over the range of a species and it is preferable to include both of these parameters when developing general equations (Wharton and Griffith, 1993). The coefficients of variation (approximately 20%) for the whole tree equations compare well with values from allometric equations used in biomass prediction of other tree species (Snowdon et al., 2002). For inventory purposes single allometric equations are preferable as opposed to separate allometric equations based on planting density on the grounds of convenience.

This Chapter presents total biomass estimates for different tree components for the three candidate species and such estimates, derived from allometry, will be critical for bioenergy process plant efficiency. Biofuel quality and subsequent energy value will be determined by the proportions of leaf, twig, bark and wood and the subsequent moisture and ash content of these components. For example, wood fuels with a 4% total ash will have 3% less energy than biomass with 1% total ash (Food and Agriculture Organization, 2004). The ability to estimate the biomass of tree components and energy potential will be invaluable for determining the economic viability and potential of short rotation tree crops for bioenergy plants.

Although the phase farming with trees system is envisaged as comprising a whole tree harvest, where the whole tree equation will be applicable, the other tree component equations (Table 5.3) allow the testing of alternative harvesting scenarios. These scenarios could, for example, include the separation and retention of leaves to maintain site fertility through nutrient cycling. The retention of the root systems in the ground for farmland returned to annual pasture rather than cereal cropping may be another scenario. In this case, *P. radiata*, a softwood species would be preferred as decay rates would be higher than for hardwood species allowing for cereal cropping to take place sooner. The nutrient contents of the different tree components have been determined and the effects of short rotation forestry systems on nutrient cycling are described in Chapter 7.

Methods to harvest tree crops together with their root systems have not yet been developed but alternative methods involving blade ploughs and reel rakes will be tested. The harvesting methods employed may be affected by tree size and their respective root systems. Trees grown at 500 trees ha^{-1} were 32 kg tree^{-1}, some of the largest sampled and may be difficult to harvest with typical tree crop harvesters (Mitchell et al., 1999).

Root:shoot (r:s) ratios did not vary significantly between planting density and slope position but were significantly different between species, with *E. occidentalis* having a higher proportion of root biomass (0.51) compared to *E. globulus* (0.31) or *P. radiata* (0.33). This has implications for harvesting systems, both in terms of recovery of material, and for the removal of stumps in preparation for a return to cropping. It is envisaged that the whole tree would be harvested for biomass fuel and the tree roots would be removed in a manner to allow for resumption of cropping (Harper et al., 2000).

Although there were no significant differences in r:s with planting density, this differs to other studies on trees of similar age which showed varying responses of r:s to planting density. For example, Eastham et al. (1990) sampled 2.5 year old *Eucalyptus grandis* planted at densities from 100 to 2000 trees ha^{-1} and found the r:s ratio decreased with increasing planting density. It is difficult to compare r:s ratios derived here to other studies as the roots in this study were excavated to simulate harvesting for biomass. If the distal root system and fine roots were sampled, differences in planting densities may have been evident.

The suitability of root material as a biomass fuel is uncertain, due to soil contamination as this reduces heat exchange efficiency and results in down-time for maintenance and cleaning of furnaces. If the tree roots are not utilized for biomass fuel then they will need to be removed prior to the resumption of cereal cropping. In this case species with high r:s ratios, such as *E. occidentalis,* may not be desirable as relative recoveries will be lower.

5.5 Conclusions

This Chapter aimed to determine the rates of biomass production possible from 3 year old trees in an ultra-short agroforestry rotation. It was shown that with high stocking densities and optimal slope position biomass yields of 15 to 22 t ha^{-1} 3 yr^{-1} were possible, dependent on species. When averaged across the landscape, yields were more modest and ranged from 12 to 14 t ha^{-1} 3 yr^{-1}. These were achieved in lower than normal rainfall conditions however, with a return to normal conditions and higher planting densities it may be feasible to produce greater yields. Also, other as yet untested species may have faster growth rates under these conditions. To maximize biomass production and water use, a balance is required between planting density and water availability and thus matching species to site is important. This Chapter does not consider the hydrological effects of the system, these are discussed elsewhere (Harper et al., 2014). Similarly it does not consider harvesting and planting systems or the economics that take into account the environmental benefits of reducing salinization. Component biomass determined in this chapter will be applied in Chapter 7 to determine the nutrient export of these tree phase systems.

6 Bio-mitigation of carbon following reforestation of abandoned salinized farmland[4]

6.1 Introduction

Reforestation of farmland represents a major method of mitigating rising atmospheric carbon dioxide contents, either through carbon sequestration or via the substitution of fossil fuels with bioenergy (Canadell and Raupach, 2008; Schlamadinger and Karjalainen, 2000). Although reforestation alone is unlikely to allow resolution of global carbon imbalance (Pacala and Socolow, 2004) it nonetheless represents a useful contribution. Current interest and developments with second generation biofuel technologies may further increase the demand for land used for carbon mitigation. For example, biofuels developed from cellulosic feedstocks show promise, not only with respect to yield, but also in relation to fossil fuel displacement and subsequent CO_2 removal (Schmer et al., 2008).

Widespread reforestation may, however, result in competition between carbon mitigation and other land-uses, in particular food production (Gunther, 2009). This conflict in land-use is likely to increase with increasing world population and per capita food consumption. An alternative approach is to consider biomass production from abandoned or poorly productive agricultural land. To date, there are few data to support the cost-effectiveness, or otherwise, of this proposition. Studies in relation to global biomass energy and the use of abandoned farmland show Australia as having potentially large areas of abandoned farmland (Fischer and Schrattenholzer, 2001;

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Campbell et al., 2008; Wicke et al., 2011) however, these are based on generalized models rather than specific regional data sets. Similarly, several research and development gaps were identified for Australia’s second generation biofuels industry by Warden and Haritos (2008). Of these, provision of biomass feedstock was identified as an area that required particular attention.

Globally, many regions experience salinization, with soil salinity being prevalent in more than one hundred countries (Rengasamy, 2006) and across a range of climates (Marcar and Khanna, 1997). Globally, the impact of human land-use has resulted in an estimated 74 Mha of salinized agricultural land (Dregne et al., 1991), of this area 43 Mha is irrigated land and 31 Mha is secondary salinization of non-irrigated land.

Across major regions of southern Australia the removal of deep-rooted native vegetation (perennials) and replacement with shallow rooted agricultural crop and pasture species (annuals) has resulted in increased recharge to groundwater. A subsequent rise in water tables and mobilization of salt stores has led to the development of dryland or secondary salinity (National Land and Water Resources Audit, 2001; George et al., 1999). In Australia, it is projected that around 17 Mha of agricultural land will be salt-affected by 2050 (National Land and Water Resources Audit, 2001).

Two general plant-based approaches have been used to treat salinized farmland in Australia, these being either reforestation using salt tolerant trees or revegetation with forage shrubs. Both approaches attempt to gain some economic return from salt affected farmland, while removing excess soil moisture and restoring landscape hydrological balances (George et al., 1999). On salinized land the replanted species must be tolerant to both water logging and high levels of soil salinity (Barrett-

Lennard, 2003). Although there have been many studies to identify suitable tree species for reclamation of saline farmland (Marcar et al., 1995; Benyon et al., 1999; Niknam and McComb, 2000), and some have pointed out the benefits of using salinized lands for fuel-wood production (e.g. El-Lakany, 1986), there are few data on biomass production for carbon mitigation potential. Of potential halophytic shrubs, saltbush (*Atriplex* spp.) has been the most extensively examined due to its salt tolerance and its nutritional potential as an alternative fodder for livestock (Norman et al., 2004). Biomass estimates reported from many of these studies relate mostly to edible dry matter (EDM) and not total biomass.

This Chapter examines the carbon mitigation potential from abandoned salinized farmland, (or saltland) treated with *Eucalyptus occidentalis* and *Atriplex nummularia,* and the influence of both site (e.g. salinity) and silvicultural factors on yield to eight years of age. These data are discussed in relation to both potential feedstock for second generation biofuel production and production of salt tolerant fodder crops.

6.2 Materials and Methods

6.2.1 Location

The study site (S5), was located near Wickepin, Western Australia, approximately 240 km east of Perth (117°39’59.95”E; 32°43’50.47”S) the State capital (Figure 3.4). This site was selected as having soils and landforms representative of the general region (McArthur, 1991), which has a semi-arid Mediterranean climate, with a seasonal drought from November to April, a mean annual rainfall (1889-2001) of 365 mm yr^{-1} and mean annual pan evaporation of 1789 mm yr^{-1}. The mean rainfall during the eight years of the experiment was 303 mm yr^{-1}.

6.2.2 Experimental design

A salt scald had developed in a valley floor during the previous two decades on land originally cleared for farming in the early 1900s, and this consisted of a bare, hypersaline area which has been extensively eroded and fringing less-saline areas (Figure 6.1). *E. occidentalis, A. nummularia, Allocasuarina huegeliana* and *Acacia celastrifolia* were planted adjacent to the salt scald in June 2001 at planting densities of 500 and 2000 trees ha^{-1} in a randomized complete block design, consisting of two replicate blocks (one either side of a salt scald), each with eight treatments and three replicates. Plants were germinated and grown in containers for approximately 6 months, then transplanted into 50 cm high mounds within 40 x 40 m treatment plots.

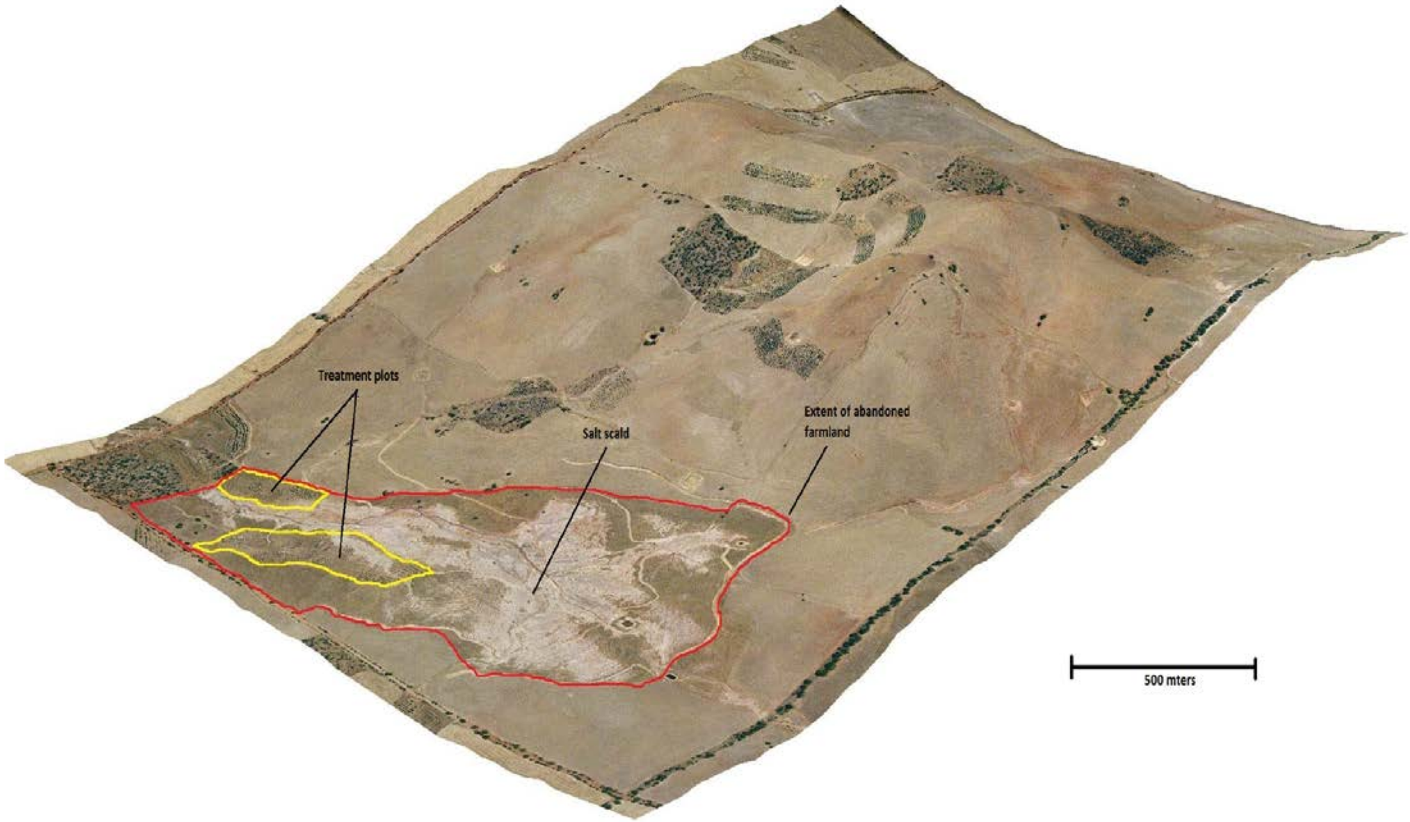

Figure 6.1 Aerial photograph of the Wickepin experimental site (117°39’59.95”E; 32°43’50.47”S), overlaid on a digital elevation model, showing the salt scald and location of the experimental plots.

A. huegeliana and *A. celastrifolia* performed very poorly and were not subsequently measured throughout the trial. *A. celastrifolia* was grazed by stock and some plots were completely destroyed. *A. huegeliana* survived grazing but growth was very poor in comparison to *E. occidentalis* and *A. nummularia*. This Chapter therefore focuses on the performance of the latter two species.

6.2.3 Biomass estimation

6.2.3.1 Measurements

Measurements were made of all treatments on an annual basis, to obtain estimates of biomass. Predictor variables used in the development of allometric relationships were measured on all *E. occidentalis* trees and *A. nummularia* shrubs within 20 x 20 m permanent measurement plots. Plots of this size were considered unlikely to be affected by edge effects between contrasting treatments. For *E. occidentalis,* attributes measured included diameter over bark at 10 and 130 cm above ground, total tree height; and for *A. nummularia* crown length and width measured perpendicular to each other, height and crown base height. The measurement was carried out at the same time each year (May - July) to ensure there was time for crown regrowth of *A. nummularia* following late summer grazing. Survival was estimated as a proportion of the plants alive at the time of measurement compared to the number initially planted.

6.2.3.2 Allometric equations

Species specific allometric equations for *E. occidentalis* were applied from a previous study (Chapter 5) in the same farming region and on plantings with the same planting density. These equations predicted whole tree, leaf, stems and root biomass from tree height and diameter over bark at 10 cm above ground. For *A. nummularia*, allometric equations were developed from biomass sampling carried out in June 2005. Equations were derived using the same procedure as in Chapter 5 with respect to equation fitting and goodness of fit tests.

6.2.3.3 Above ground biomass sampling for Atriplex nummularia

A total of 12 permanent measurement plots were sampled, with the five plants sampled from within each plot covering the dynamic range of sizes. Crown width, length, shrub height and crown base height were measured to calculate a crown volume index (CVI) for regression analysis. Stem diameter was also measured at 10 cm above ground height (D10) and a diameter equivalent calculated (Avery and Burkhart, 1983). The fresh weight of each saltbush was measured and a whole branch sub-sample collected for moisture determination and to determine wood and leaf proportions. Sub-samples were placed in calico bags and oven dried at 70˚C to constant weight. Moisture ratios were derived to convert total saltbush fresh weight to oven dry weight. A subset of the oven dry samples were stripped of leaves and small stems <2 mm with this fraction representing the proportion of edible dry matter (EDM) as described by Andrew et al. (1976).

6.2.3.4 Below ground sampling for Atriplex nummularia

For the estimation of below ground biomass the roots of two plants from each plot were excavated to a depth of 30-50 cm, using a 1.5 ton excavator. Soil was placed on a sieving table having a mesh size of 50 mm and roots collected as described by Ritson and Sochacki (2003) and in Chapter 4. Roots were excavated to a minimum diameter of approximately 5 mm, roots of lesser diameter were not sampled as it was difficult to associate these to the individual trees being sampled.

6.2.3.5 *Estimation of stand carbon sequestration*

Carbon sequestration was estimated from dry biomass by (a) assuming a carbon content of the dry biomass of 50% (Gifford, 2000) and (b) converting this to carbon dioxide equivalents on the basis of molecular weight as

$$CO_2\text{-e} = Db \times Cb \times 3.67 \qquad (11)$$

where

D*b* = oven dry biomass,

C*b* = default carbon proportion of dry biomass (50%) and

3.67 = atomic mass ratio of carbon dioxide to carbon for CO_2 (44/12).

6.2.3.6 *Soil salinity measurements*

Apparent soil electrical conductivity (ECa) was measured using a Geonics EM38 electromagnetic induction meter. The EM38 values were used in a relative manner to compare treatments. The apparent electrical conductivity of the salt scald was measured in early winter 2005 and again in the winter of 2009. Measurements were made within the measurement plots both on the tree mound and in the adjacent alley. At each measurement location the EM38 was used in both horizontal (EM38 H) and vertical mode (EM38 V). In 2005 EM38 measurements were also taken along a 100 m transect starting at the salt scald fringe and extending across the treatment plots in order to identify if a salinity gradient was present.

6.2.4 *Statistical analysis*

Plot data were investigated by analysis of variance and Pearson’s correlations using Xlstat statistical software.

6.3 Results

6.3.1 Allometry for Atriplex nummularia

A total of 54 shrubs were sampled for the development of allometric equations for *A. nummularia*, of which 22 were sampled for below ground biomass. Shrubs sampled ranged from 1 to 45 kg total dry weight with an average of 19% EDM and a root to shoot ratio of 0.22 (Table 6.1).

Table 6.1 Characteristics of *A. nummularia* shrub biomass (kg plant^{-1}) sampled for allometric equations. *n* is the number of samples, s.e. is the standard error.

Attribute	*n*	Biomass (kg plant^{-1})		
		Range	Mean	s.e.
Above Ground (B_{ag})	54	0.6-42.9	10.5	1.4
Below ground (B_{bg})	22	0.1-9.2	3.0	0.6
Total (B_t)	22	0.9-45.0	13.5	2.9

Above ground biomass followed a linear trend in relation to CVI with very similar relationships being observed for both the 500 and 2000 trees ha^{-1} treatments (Figure 6.2) The data for the 500 and 2000 trees ha^{-1} treatments were combined for the development of allometric equations, for both above ground and below ground biomass (Table 6.2). Only above ground biomass was estimated annually following recovery from grazing and is assumed to provide an estimate of crown volume unaffected by grazing. Estimation of below ground biomass using CVI as a predictor variable would only be applicable to the initial sampling year therefore an allometric equation for below ground biomass was also developed using D10 as a predictor (Table 6.2).

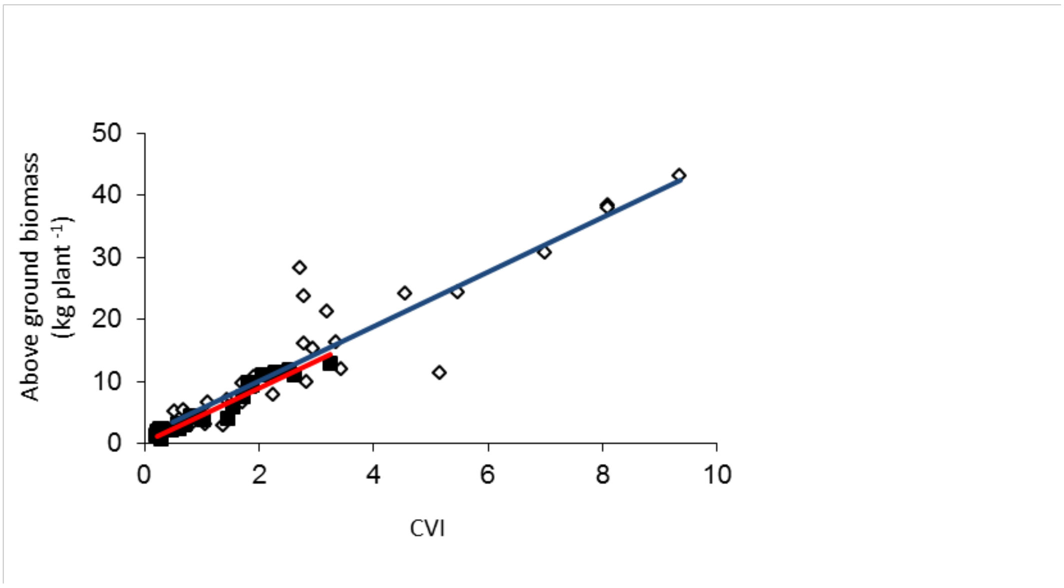

Figure 6.2 Relationship between Crown Volume Index (CVI) and above ground biomass (kg plant^{-1}) for the *A. nummularia* shrubs sampled from the 500 (◊) and 2000 (■) trees ha^{-1} treatments for the development of allometric equations.

Table 6.2 Allometric equations for the prediction of above ground (B_{ag}) and below ground biomass (B_{bg}) in *A. nummularia*, including a range of goodness of fit indices as in Chapter 5. These were based on the crown volume index (CVI) and diameter of the stem at 10 cm above ground (D_{10}). *n* is the number of samples, r^2 is Pearson correlation coefficient, FI is fit index, s.e. standard error of estimate and CV (%) is the coefficient of variation.

Component	*n*	pred.var.	Allometric eqn.	r^2	FI	se	CV %
Above ground biomass	54	CVI	$B_{ag}=0.494+4.6069*CVI$	0.97	0.97	1.34	14
Below ground biomass	22	D_{10}	$B_{bg}=0.0874*D_{10}^{1.6163}$	0.56	0.47	0.87	28

*r:s ratio = 0.39

6.3.2 Survival and tree growth

Survival of *E. occidentalis* was consistently around 85% for both the 500 and 2000 trees ha^{-1} treatments (Figure 6.3a, b). For *A. nummularia* survival was between 86%

and 89% for the 500 and 2000 trees ha^{-1} treatments respectively (Figure 6.3c, d). The highest mean total biomass yields for *E. occidentalis* were 18 and 37 t ha^{-1} for the 500

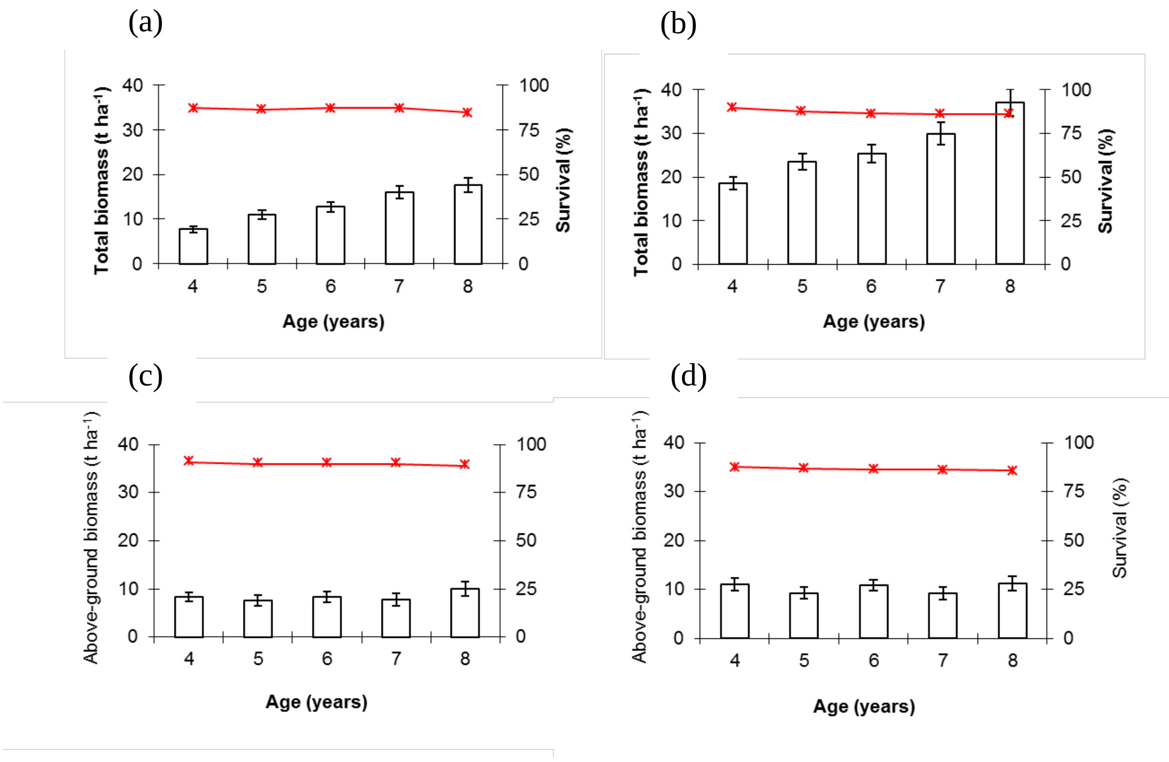

Figure 6.3 Total biomass (t ha^{-1}) and survival (%) of *E. occidentalis* planted at (a) 500 trees ha^{-1} and (b) 2000 trees ha^{-1}. Above ground biomass and survival of *A. nummularia* planted at (c) 500 trees ha^{-1} and (d) 2000 trees ha^{-1}. Error bars are standard error.

and 2000 trees ha^{-1} treatments respectively at 8 years (Figure 6.3a, b). Total average biomass yields were significantly greater ($P<0.05$) for the 2000 trees ha^{-1} compared to the 500 trees ha^{-1} treatments for all years measured (4 to 8 years of age).

Above ground biomass yields for *A. nummularia* did not increase over time (Figure 6.3c, d) possibly as these plots were grazed by sheep each summer. The greatest above ground biomass yields were achieved from the 2000 trees ha^{-1} treatment, with yields of 11 t ha^{-1} at 4, 6 and 8 years. The greatest mean above ground biomass yield for the 500 trees ha^{-1} treatment was 10 t ha^{-1} at 8 years. There were no significant differences in mean total biomass yield between the two planting densities for any given year measured (2005 to 2009). Below ground biomass yield for *A. nummularia* was estimated for the first measurement year (4 years) at 1.9 and 3.7 t ha^{-1} for 500 trees ha^{-1} and 2000 trees ha^{-1} respectively, for subsequent years below ground biomass was not estimated.

Total biomass yields of *E. occidentalis* for the 2000 trees ha^{-1} treatments were significantly correlated with soil apparent electrical conductivity as measured with the EM38 H at 4 years ($r = -0.9$, $P < 0.05$) and 8 years after establishment ($r = -0.93$, $P < 0.05$). A negative linear relationship is evident for both the 500 and 2000 trees ha^{-1} treatments (Figure 6.4a, b).

EM38 H (mS m^{-1}) readings taken along a 100 m transect extending across treatment plots from the fringe of the salt scald were negatively correlated with distance from the salt scald for both tree mounds ($r=-0.75$, $P<0.0001$) and for measurements taken in the tree alley ($r= -0.84$, $P<0.0001$) (Figure 6.5). EM38 H measurements from the alley were also significantly ($P<0.0001$) higher than the mounds.

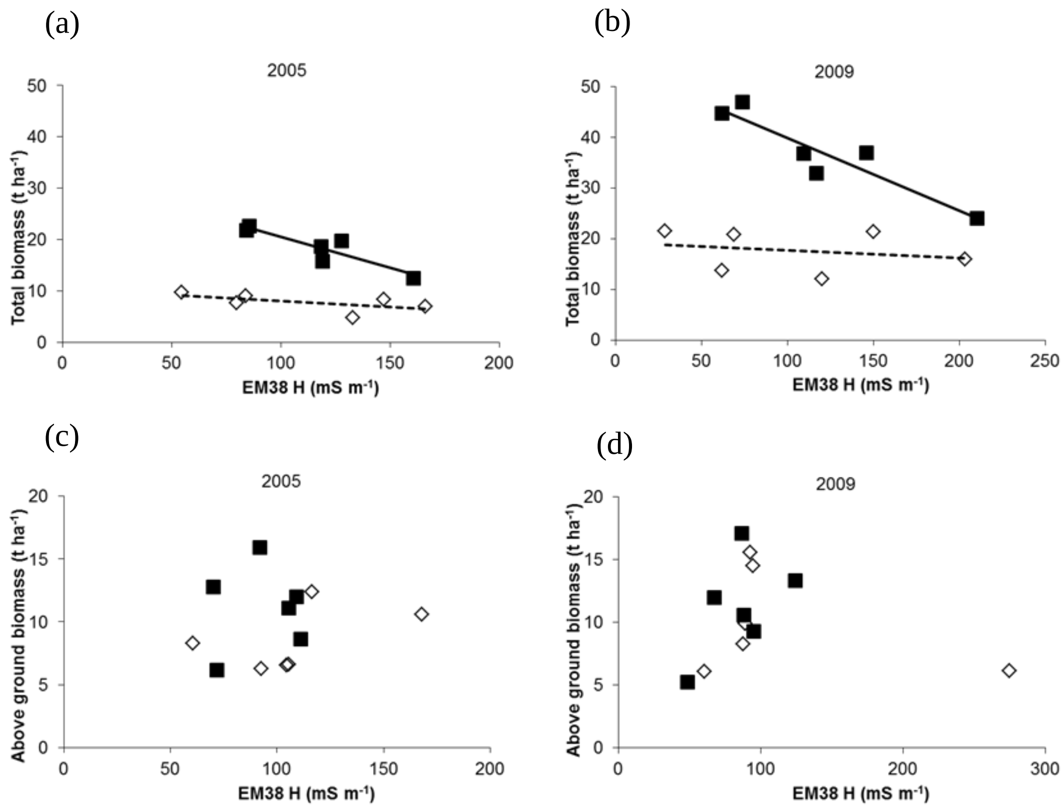

Figure 6.4 Relationships between total biomass and soil conductivity in 2005 and 2009 for *E. occidentalis* (a and b) and *A. nummularia* (c and d) for 500 (◊) and 2000 (■) trees ha^{-1} treatments.

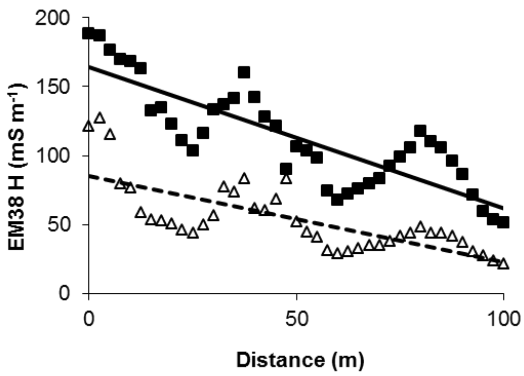

Figure 6.5 Soil conductivity along a 100 m transect extending from the salt scald fringe, across treatment plots as measured with an EM38 (8 years after establishment) for mounds (Δ) and tree alleys (■).

6.4 Discussion

6.4.1 Biomass production

The areas adjacent to the salt scald had been effectively abandoned to agriculture but *E. occidentalis* produced a mean of 4.6 t ha^{-1} yr^{-1} (8.5 t CO_2 -e ha^{-1} yr^{-1}) after eight years, with plot values ranging as high as 5.9 t ha^{-1} yr^{-1} (10.8 t CO_2 -e ha^{-1} yr^{-1}). This is within the range of 1 to 10 t ha^{-1} yr^{-1} reported on poor soils unsuitable for agriculture in other studies (Hoogwijk et al., 2003). *Atriplex nummularia* was planted into large mounds in close proximity to the salt scald, with total standing above ground biomass of between 9 and 11 t ha^{-1} between years 4 and 8, of which 19% was fodder. Fodder production from this species has received considerable attention as an alternate and complementary animal feed during the typically dry summer period this region experiences. In conjunction with other pastures, *Atriplex* plantings have also provided additional nutrition for sheep during autumn (Barson et al., 1994). Approximately 1.9 t ha^{-1} yr^{-1} of edible fodder was produced on this site and this is comparable to other studies (Abu-zanat et al., 2004; Benjamin et al., 1995).

Biomass production was affected by both stand density and soil salinity. Biomass production by *E. occidentalis* was higher at 2000 trees ha^{-1} than at 500 trees ha^{-1}. However, at 2000 trees ha^{-1} biomass production was negatively affected by soil salinity levels but not at 500 trees ha^{-1}. The response of *E. occidentalis* to soil salinity at 500 trees ha^{-1} differs from the observations of Benyon et al. (1999) and Marcar et al. (2003). Benyon et al. (1999) reported a 10% reduction in height of *E. occidentalis* at a root zone salinity of 10 dS m^{-1} while Marcar et al. (2003) reported no growth decline up to an ECe of 10 dS m^{-1}. In this study published calibration curves for the EM38 developed for soils of this region (Bennett and George, 1995) were used to produce estimates of soil salinity. Based on the EM38 values, the soil salinity was in excess of 10 dS m^{-1}, yet no significant response was observed for the 500 trees ha^{-1} *E. occidentalis* treatments. Marcar et al. (1995) reported marked variations in the response of different provenances of *E. occidentalis* to soil salinity and this may explain these results.

Although some halophytes show improved growth under mildly saline conditions (Stirzaker et al., 2002), there was no significant effect of soil salinity levels on biomass production of *A. nummularia.* Soil electrical conductivity levels indicate a reduced level of salinity in mounds compared with the inter-row region and this may have reduced any effects of soil salinity. Salt concentrations are reduced following winter rains due to leaching of salt from the mounds back into the subsoil, an effect apparent in comparisons of salinity in the mounds and adjacent alleys in Figure 6.5. Mounding has been used in other saltland planting studies and has been effective in leaching salt from the seedling root zone (Marcar et al., 1995). The good growth of *A. nummularia* on these sites and the lack of a salinity response implies that this species may tolerate being planted in closer proximity to the hypersaline salt-scald and thus utilize more of the abandoned farmland.

6.4.2 Future plantings

There are many reports of different growth rates for different species along salinity gradients (van der Moezel et al., 1988; Niknam and McComb, 2000) and these are commonly associated with ecological successions such as in local saline wetlands. *Atriplex* species are reputed to have much greater salt tolerance that *Eucalyptus*, and indeed, this is the observation under natural conditions in south-western Australia, where *Atriplex* species occur on the beds of relict playas with fringing salt-tolerant eucalypts (Harper and Gilkes, 2004). It may thus be possible to design land treatments for salinized sites that take such gradients into account and replicate the ecological successions that occurs in wetland ecosystems. *Atriplex nummularia* could for example be planted into mounds in close proximity to salt scalds in conjunction with *E. occidentalis* adjacent and upslope to these plantings. *Atriplex nummularia* has a limited effect on water table levels (Slavich et al., 1999) but is able to tolerate high salinity levels, coupled with *E. occidentalis,* a high water use species. The combined application of these two species would produce a) fodder for livestock and b) biomass for bio-energy or biofuel feedstock, c) be treated as a carbon sequestration option and potentially utilize excess soil water and help reduce water logging. Coppicing of harvested trees may be an option to minimize turn-around between rotations and eliminate the need for further soil disturbance through replanting or the stand could be left unharvested to sequester carbon. Marcar et al. (2003) report good coppice re-growth for *E. occidentalis* from cut stumps three years after thinning.

A key factor for carbon mitigation will be whether trees can persist on such sites and not be affected by salt accumulation (Stolte et al., 1997). Archibald et al. (2006) report on the growth and survival of 25 year old trees adjacent to salt scalds in this region, with these still persisting despite some salt accumulation in their root zones, however these results may be quite site specific. If such salt affected areas are planted as short rotation tree crops (Harper et al., 2010) as opposed to long term stands, salt accumulation will be less critical to mitigation performance. The sustainability of plantations on saline discharge sites is dependent on many interacting factors, which can only be resolved through further investigation. In this landscape position reforestation is most likely to have a minimal, if any, effect on ground water levels and landscape hydrology (George et al., 1999). Reforestation however, will help to stabilize degraded parts of the landscape and has the potential to positively improve soil quality through the addition of soil carbon, which is typically reduced on salinized soils as a result of reduced vegetative cover (Wong et al., 2010).

6.4.3 Biofuel feedstock potential

With predictions of salinity in Australia affecting up to 17 million hectares (National Land and Water Resources Audit, 2001) and poor economic returns from moderately saline farmland areas, the use of these marginal or abandoned farmland areas for carbon mitigation warrants further investigation. If the results presented in this Chapter are indicative of abandoned saline farmland in Australia then there is potential to produce considerable amounts of feedstock for lignocellulosic or second generation biofuel without competing with food production. Of the 17 Mha predicted to become affected by salinity, there will be areas that will not be suitable for carbon mitigation by reforestation, for example hypersaline areas and forest reserves. At the average growth rates reported in this study, and assuming production from half of the salinized area, and a conversion rate of 95 liters t^{-1} of biomass (Sims et al., 2010) an eight year rotation of *E. occidentalis* would potentially produce 3.5 million tons of liquid biofuel per year. This is approximately 8% of Australia’s 2009 fossil fuel use (BP, 2010). Australia’s current biofuel production is approximately 0.5% of total transport fuel and does not impinge on food production, however, if biofuel mandates are introduced, feedstocks required for biofuel production may compete with food production (O’Connell et al., 2009).

6.4.4 Global potential

The potential of applying *E. occidentalis* in conjunction with *A. nummularia* to salinized areas in other regions is yet to be investigated. *Atriplex* species have been trialed in many countries and regions including North Africa (El Aich, 1992), Egypt (Fayed et al., 2010) and Israel (Benjamin et al., 1995). *Atriplex nummularia* has been selected due its high growth rate and palatability for livestock. Reported yields range from 11 t ha^{-1} (Benjamin et al., 1995) in arid regions (northern Negev, Israel) to 23 t ha^{-1} (Barrett-Lennard and Malcolm, 1995) for irrigated plots in Australia. In the San Joaquin Valley, California, *A. nummularia* was irrigated with saline water (18 dS m^{-1}) and produced 15.6 t ha^{-1} biomass from four successive total harvests (Watson and O'Leary, 1993). The performance of species in field trials result from a range of soil and climatic conditions and management scenarios and these need to be taken into account in the development of regional or global estimates. For example, mounding, fertilizer application and local conditions such as waterlogging will all have an effect on yield. In comparison, the use of salt tolerant eucalypt species on salinized farmland in regions outside Australia is limited and data relate predominantly to *E. camaldulensis* Dehnh (Marcar et al., 1995). The use of *E. occidentalis* for saltland plantings is less well documented, despite this species having a higher tolerance to soil salinity than *E. camaldulensis* (Marcar et al., 1995). Benyon et al. (1999) reported a 10% reduction in height growth of *E. camaldulensis* with soil salinity as low as 2 dSm^{-1}. However, for *E. occidentalis*, a 10% reduction in height was only evident at 10 dS m^{-1} with similar responses observed for stem diameter and crown volume.

This highlights the need for further studies and the synthesis of data from different regions globally. The performance of species in relation to site-specific conditions will provide invaluable input data for modeling global estimates of biomass yield. Salt affected regions globally cover a range of soil types and levels of salinity and therefore yield from reforestation would be affected accordingly. Wicke et al. (2011) report on global potential of bioenergy from salt affected land based on the

Harmonized World Soil Database (HWSD) and reported an average global biomass yield of 3.1 oven dry t ha^{-1} yr^{-1}, based on yields from three species including *E. camaldulensis*. In that study, Australia is reported as having 169 Mha of salt affected land and the average biomass yield for Australia is estimated at 7.6 oven dry t ha^{-1} yr^{-1}, twice the global average. Clearly this estimate of salt affected land is very different to local predictions of salinity in Australia of 17 Mha (National Land and Water Resources Audit, 2001). The lack of data for Australia and the accuracy of the HWSD is questionable, thus highlighting the need for species specific regional data sets and accurate global soil data for modeling of global bioenergy potential. To estimate the global bioenergy potential of *E. occidentalis* and *A. nummularia* would require modeling of soil and climatic inputs, regional yield data and management regimes and species attributes that are beyond the scope of this study.

Nonetheless, it is obvious from this and other studies (Dornburg et al., 2010; Wicke et al., 2011) that considerable potential exists for bioenergy production from salinized farmland and given the threat increasing soil salinity has on global food production, further research is warranted. Similarly, given the tolerance of *A. nummularia* to salinity, the utilization of this species in bioenergy production could also be usefully examined.

6.5 Conclusions

This chapter has demonstrated the potential of salinized abandoned farmland in providing carbon mitigation through biomass for bioenergy or as a future lignocellulosic feedstock. The commercial production of second generation biofuels from lignocellulosic biomass is promising and supplies of biomass feedstocks are obviously crucial to the success of commercial production; abandoned salinized farmland is therefore a potential feedstock source for this future industry. Mitigation can thus occur without displacing food production, and using combinations of *Eucalyptus* and *Atriplex* species may not only provide some mitigation but also produce fodder for livestock. Further research is required to extend the results of this study across broader areas of salinized land, determine the long-term viability of such land treatments and examine the greenhouse gas balances of harvested and grazed systems on abandoned farmland. It will be imperative that species be tested in different soil and climatic regimes in order to improve regional data sets for global estimates.

7 Nutrient exports from a short rotation energy cropping system[5]

7.1 Introduction

Short rotation energy crops have the potential to provide sustainable sources of biomass (Harper et al., 2010) (Chapter 5; Chapter 6) provided management practices include the recycling of nutrients (Vance et al., 2010). Many studies report the global potential of biomass for bioenergy (Hoogwijk et al., 2003; Campbell et al., 2008; Field et al., 2008) and the sustainability of these systems is increasingly being reviewed with many criteria being considered in relation to sustainability certification (Scarlat and Dallemand, 2011). Sustainability criteria invariably include soil factors such as soil fertility and nutrient removal (Buchholz et al., 2009; Van Stappen et al., 2011; Haberl et al., 2010). The efficient use of nutrients will be paramount to sustainable short rotation energy crops and the predicted global yields in these studies may not be sustainable without nutrient input.

Phase farming with trees (PFT) trees has been advocated as a method of producing biomass for bioenergy applications in the dryland farming systems of southern Australia (Harper et al., 2010; Harper et al., 2000) with similar "green fallow" systems advocated for Africa (Sanchez, 2002) and willow biomass crops in the United States (Heller et al., 2003). A major concern with the increasing interest in bioenergy and biofuel crops is that it involves land use change (LUC) and the competition for agricultural land (Van Stappen et al., 2011). Instead of permanently setting aside areas

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for tree crops, PFT offers an opportunity to produce a renewable energy source as well as address environmental issues that may be specific to particular agricultural systems.

PFT uses short rotation (3 to 5 years) tree species with high water use to produce bioenergy and restore landscape hydrology (Harper et al., 2010). The removal of deep rooted native vegetation, followed by replacement with shallow rooted agricultural plants in south western Australia has led to groundwater rise and widespread salinization (Peck and Hatton, 2003). Under the PFT system excess soil moisture that has accumulated below the shallow root zone of annual crops is transpired by the perennial system, creating a dry soil buffer that reduces recharge to groundwater (Harper et al., 2010). Biomass produced by the system could represent a major potential feedstock for stationary bioenergy or liquid biofuels produced by cellulosic or pyrolitic processes. For example, in Chapter 5 yields of 16 to 22 t dry matter ha^{-1} 3 yr^{-1} are feasible in an area with only 300 mm yr^{-1} rainfall, using high density PFT plantings (4000 trees ha^{-1}) of *E. occidentalis*. PFT could also be applied to marginal lands which are not suited for intensive agriculture and a woody biomass crop would be more sustainable.

PFT will involve the complete removal of above ground biomass and the partial removal of root systems (Chapter 5). If the removal of the tree crop substantially reduces the nutrient status of the soil then the return to agricultural farming systems after the tree phase may be compromised. There are many reports of nutrient removal in forestry and bioenergy production systems, particularly N, P, K, S, Ca and Mg (Grove et al., 2007). Other studies have related nutrient exports to nutrient use efficiency, either in relation to nutrient efficient species for sustainable energy crops (Wang et al., 1991; Safou-Matondo et al., 2005) or species with low nutrient use efficiencies for the removal of excessive nutrients from effluent sites (Guo et al., 2006b; Guo and Sims, 2002) In southwestern Australia, concerns about the sustainability of forest crops have focused on the depletion of soil nutrients in general and more specifically, soil organic matter and nitrogen contents following harvest of pulpwood plantations (O'Connell et al., 2003; Mendham et al., 2004), such as *E. globulus* which is grown in 10 year rotations. Similarly, intensive farming activities such as continuous cropping with cereals or harvesting pastures for hay have been associated with potassium deficiencies in this region (Cox, 1980).

The sustainability of soil nutrient removals by PFT plantings in lower rainfall areas (300 mm mean annual rainfall) with short growing periods of 3-5 years has thus been raised as a concern (Harper et al., 2008). For PFT to be implemented over large areas nutrient use needs to be sustainable and ideally, losses should not exceed those from current farming practices. This Chapter reports nutrient losses following harvesting of an experimental PFT planting in comparison to other farming practices.

This chapter investigates the implications of species selection on biomass production and compares subsequent nutrient use efficiency of tree components and proposes a nutrient assimilation index (NAI) as a sustainability index for the comparison of nutrient use efficiency across different studies and species.

7.2 Materials and Methods

7.2.1 Location

The study site (S4) was located near Corrigin, Western Australia, approximately 240 km east of Perth (Figure 3.4) , (117°41'47.13"E; 32°23'24.67"S) the State capital and has been previously described in Chapter 5.

7.2.2 Experimental design

An experiment was established in August 2001 (winter) to evaluate the potential of short rotation tree crops to transpire excess soil water to depth (6-8 m) and create a buffer of dry soil to capture the leakage that occurs below the shallow root zone of subsequent annual crops (Harper et al., 2008). This has been previously described in Chapter 5.

7.2.3 Nutrient history of the site

At this site conventional farming is similar to that practiced across broad areas of southern Australia with annual rotations of cereal (*Triticum aestivum*, *Hordeum vulgare*) or legume (*Lupinus angustifolius*) crops with improved annual legume (*Trifolium subterraneum*) and grass (*Lolium rigidum*) pastures, grown during the winter rainfall season (Squires and Tow, 1991). The property was originally cleared from natural bush-land in 1920, and typically had superphosphate (9.1% P) applied at an annual rate of 100 kg $ha^{-1}yr^{-1}$ and was subsequently cropped or grazed on an annual rotational basis. Nitrogen in these cropping systems is derived from either leguminous crops and pastures or through applications of nitrogen fertilizer during cropping phases at rates of up to 50 kg N $ha^{-1}yr^{-1}$.

7.2.4 Soil sampling and analysis

Soil samples were collected to compare soil nutrient stores and tree nutrient exports at the completion of the experiment. Three soil pits were excavated in the field adjacent to each treatment block. Three samples were taken at 0 to 0.1, 0.1 to 0.2 and 0.2 to 0.3 m depth in each pit using a 0.1 m internal diameter bulk density sampler. The soil profile was sampled to this depth as previous studies have shown this depth as having the highest fine root density of tree species and this would also be the rooting depth for nutrient uptake of annual crops (Turner and Kelly, 1977). Samples were oven dried at 40°C to constant weight and sieved through a 2 mm sieve to allow calculation of gravel content. The sieved portion of the samples was bulked for each depth interval for each sample pit.

Total C and N were determined by combusting soil samples at 950ºC in oxygen using a Leco FP-428 Nitrogen Analyzer (Sweeney and Rexroad, 1987). Available phosphorus and potassium were measured using the Colwell method (Rayment and Higginson, 1992). Electrical conductivity, pH in water and pH in calcium chloride were determined in a soil:solution ratio of 1:5 using deionized water (Rayment and Higginson, 1992). Exchangeable cations (Ca, Mg, Na and K) were determined using the method of Gilman and Sumpter (Rayment and Higginson, 1992).

The soil nutrient store, on a mass density basis, was calculated for the three depth intervals by using mean nutrient concentrations, gravel content and bulk density.

7.2.5 Allometric relationships and component biomass

The development of allometric tree component relationships for the three tree species at this site was reported previously (Chapter 5). These were developed from the sampling of trees at 36 months of age, with the sample including a range of tree size and form representative of the stand. Roots were excavated to include all material to a minimum diameter of 5 mm. Results were calculated on the basis of oven-dry (70°C) biomass. In this chapter, these relationships were applied to predictor variables (diameter at 0.1 m above ground and tree height) measured on all trees within 20 m by 20 m permanent measurement plots to derive estimates of tree component biomass.

7.2.6 Nutrient analysis

Material collected and dried for biomass sampling (Chapter 5) was sub-sampled for nutrient analysis. Nutrient analysis was performed on leaf, twig, stem-wood, bark and roots. Wood from roots and stem samples was cut into smaller pieces in preparation for milling. Nitrogen was determined by combusting finely ground samples at 950^{o}C using a Leco FP-428 Nitrogen Analyzer (Sweeney and Rexroad, 1987). For determination of phosphorus, potassium, sulfur, sodium, calcium and magnesium, samples were digested in nitric acid and subjected to ICP-AES (McQuaker et al., 1979).

7.2.7 Nutrient assimilation index and nutrient export

A Nutrient Assimilation Index (NAI) was proposed

$$NAI = B\mathit{c}/N\mathit{c} \qquad (10)$$

where

NAI = nutrient assimilation index (Mg kg^{-1}),

B*c* = oven dry component biomass yield (Mg ha^{-1}) and

N*c* = component nutrients assimilated (kg ha^{-1}).

This approach allows for the standardization of measurements in determining the sustainability of tree crops, with values having the same units (Mg) typical of biomass measurement. Larger values indicating a good nutrient use efficiency and lower values indicating poorly performing species. Other studies have used yield (kg) divided by assimilation (kg), which results in large unwieldy values for the sake of comparison (for example, Adegbidi et al., 2001; Safou-Matondo et al., 2005).

Component nutrient export was estimated from mean component nutrient values for each treatment multiplied by the component biomass of that treatment and are presented on an oven-dry basis.

7.2.8 Statistical analysis

Nutrient foliar analysis data were examined using XLSTAT and SigmaPlot 12 statistical software, a balanced design ANOVA model for factors landscape position, species and tree component was accessed. Data were transformed to ensure homogeneity and normality to satisfy assumptions for ANOVA. Normality was assessed using Shapiro-Wilk's test. The effect of qualitative variables on dependent variables was related by the F statistic and comparison procedures using the Holm-Sidak method.

7.3 Results

7.3.1 Soil properties and nutrient store

The proportion of gravel ranged from 1% (0 to 0.1 m depth) in the lower landscape treatments to 58% (0.2 to 0.3 m depth) in the upper landscape treatments (Table 7.1), on an oven dry basis. All soils were mildly acidic with pH ranging between 4.6 and 5.0, for all three depth intervals. Soil carbon concentrations were greatest in the surface 0.1 m layer for all three landscape positions at approximately1.0% for the upper and mid-slope and 0.8% for the lower slope. Contents of N, P, K and soil exchangeable cations were generally greatest in the top 0.1 m layer and decreased with depth. Total contents of soil nutrients (kg ha^{-1}) are presented in Table 7.2 for the top 0.3 m of the soil profile.

Table 7.1 Soil properties and average element concentrations of soils in the paddock at each landscape position (n = 9).

Landscape position	Depth	Bulk density	Gravel	Total C	Total N	Bic-P	Bic-K	EC	pH ($CaCl_2$)	Exchangeable Cations Ca	Mg	K	Na
	(m)	*(Mg m^{-3})*	*(%)*	*(%)*	*(%)*	*(mg kg^{-1})*	*(mg kg^{-1})*	*(dS m^{-1})*		*(cmol kg^{-1})*			
Lower	0-0.1	1.59	1	0.86	0.08	13.7	37.0	0.09	4.7	1.71	0.34	0.10	0.22
	0.1-0.2	1.72	2	0.32	0.02	15.0	25.3	0.03	4.6	0.35	0.08	0.06	0.11
	0.2-0.3	1.73	2	0.23	0.02	11.7	26.0	0.02	4.8	0.32	0.08	0.06	0.10
Mid	0-0.1	1.56	6	1.04	0.09	24.7	100.7	0.09	4.6	1.63	0.41	0.24	0.18
	0.1-0.2	1.66	7	0.37	0.03	11.0	85.0	0.04	4.6	0.80	0.21	0.19	0.17
	0.2-0.3	1.67	23	0.67	0.05	6.0	87.3	0.04	5.0	0.93	0.27	0.19	0.13
Upper	0-0.1	1.54	15	1.17	0.11	28.0	71.7	0.08	4.6	1.31	0.36	0.18	0.18
	0.1-0.2	1.76	49	0.40	0.02	12.0	71.7	0.04	4.6	1.13	0.31	0.20	0.16
	0.2-0.3	1.76	58	0.37	0.02	7.0	85.0	0.04	5.0	1.65	0.43	0.23	0.29

Table 7.2 Mean soil nutrient contents (kg ha^{-1}), across the three landscape positions, for different soil depths.

Depth (m)	Nutrient mass (kg ha^{-1})						
	Total C	Total N	Bic-P	Bic-K	Exch-Ca	Exch-Mg	Exch-K
0-0.1	14 866	1 303	32	101	485	69	105
0.1-0.2	4 980	336	17	83	261	41	100
0.2-0.3	5 243	384	10	82	334	54	107
Total	25 089	2 024	60	266	1081	165	313

7.3.2 Concentrations of elements for tree components

Concentrations of elements for different tree components are presented in Table 7.3. Significant variations in macro-nutrient concentrations were observed between tree components for N, P, K and S with these being significantly (P<0.001) higher in the leaf component of all three species. *E. globulus* and *E. occidentalis* had significantly (P<0.001) higher concentrations of Ca in all components than *P. radiata.*

Table 7.3 Mean element concentrations (with associated standard deviation) of tree components for *E. globulus* (*Eg*), *E. occidentalis* (*Eo*) and *P. radiata* (*Pr*).

	N			P			K			S			Ca			Mg		
	Eg	*Eo*	*Pr*	*Eg*	*Eo*	*Pr*	*Eg*	*Eo*	*Pr*	*Eg*	*Eo*	*Pr*	*Eg*	*Eo*	*Pr*	*Eg*	*Eo*	*Pr*
	(% w/w)																	
Leaf	1.32^{a}	1.28^{a}	1.62^{a}	0.10^{a}	0.09^{a}	0.18^{a}	0.68^{a}	0.52^{a}	0.81^{a}	0.14^{a}	0.14^{a}	0.15^{a}	1.67^{a}	1.30^{a}	0.50^{a}	0.27^{a}	0.24^{a}	0.23^{a}
	0.18	*0.19*	*0.25*	*0.01*	*0.02*	*0.05*	*0.10*	*0.07*	*0.23*	*0.02*	*0.01*	*0.02*	*0.41*	*0.46*	*0.07*	*0.05*	*0.04*	*0.07*
Twig	0.53^{b}	0.57^{b}	0.62^{b}	0.07^{b}	0.05^{b}	0.11^{b}	0.44^{b}	0.38^{b}	0.76^{a}	0.06^{b}	0.06^{b}	0.07^{b}	1.40^{a}	1.38^{a}	0.26^{b}	0.23^{a}	0.25^{a}	0.17^{b}
	0.07	*0.11*	*0.13*	*0.02*	*0.01*	*0.03*	*0.10*	*0.11*	*0.21*	*0.01*	*0.01*	*0.01*	*0.34*	*0.32*	*0.04*	*0.05*	*0.08*	*0.03*
Stem-wood	0.22^{c}	0.25^{c}	0.22^{c}	0.03^{c}	0.02^{c}	0.03^{c}	0.20^{c}	0.18^{c}	0.20^{b}	0.02^{c}	0.03^{c}	0.03^{c}	0.13^{b}	0.14^{b}	0.08^{c}	0.05^{c}	0.05^{b}	0.05^{c}
	0.03	*0.06*	*0.04*	*0.01*	*0.00*	*0.01*	*0.05*	*0.05*	*0.05*	*0.01*	*0.01*	*0.00*	*0.03*	*0.03*	*0.01*	*0.01*	*0.02*	*0.01*
Bark	0.47^{b}	0.43^{d}	0.65^{b}	0.05^{d}	0.03^{c}	0.07^{d}	0.32^{d}	0.32^{b}	0.50^{c}	0.04^{b}	0.04^{d}	0.07^{b}	2.06^{a}	1.85^{c}	0.31^{b}	0.29^{a}	0.34^{c}	0.16^{b}
	0.06	*0.05*	*0.09*	*0.03*	*0.00*	*0.01*	*0.08*	*0.09*	*0.11*	*0.00*	*0.00*	*0.01*	*0.39*	*0.36*	*0.10*	*0.05*	*0.05*	*0.02*
Roots	0.43^{b}	0.35^{e}	0.44^{d}	0.04^{e}	0.03^{c}	0.07^{d}	0.32^{d}	0.29^{b}	0.43^{c}	0.05^{b}	0.04^{d}	0.05^{b}	0.94^{c}	0.65^{d}	0.13^{d}	0.14^{d}	0.10^{d}	0.08^{d}
	0.09	*0.06*	*0.07*	*0.01*	*0.01*	*0.02*	*0.06*	*0.06*	*0.12*	*0.02*	*0.01*	*0.01*	*0.15*	*0.16*	*0.02*	*0.03*	*0.02*	*0.01*

** n=6, means in columns with the same letter are not significantly different, P<0.001.*

7.3.3 Biomass yield

For each species there was a significant (P<0.001) increase in yield with planting density, and slope position. All species yielded significantly more biomass at 4000 trees ha^{-1} (Chapter 5), and the values for that planting density are used here. The highest biomass yields for *E. globulus* and *E. occidentalis* were from 4000 trees ha^{-1} treatments located on the lower-slope site with respective values of 16 and 22 Mg ha^{-1} 3 year^{-1} respectively, whereas *P. radiata* had the highest yields in both mid and upper slope treatments with approximately 14 Mg ha^{-1} 3 year^{-1} of total biomass Figure 7.1.

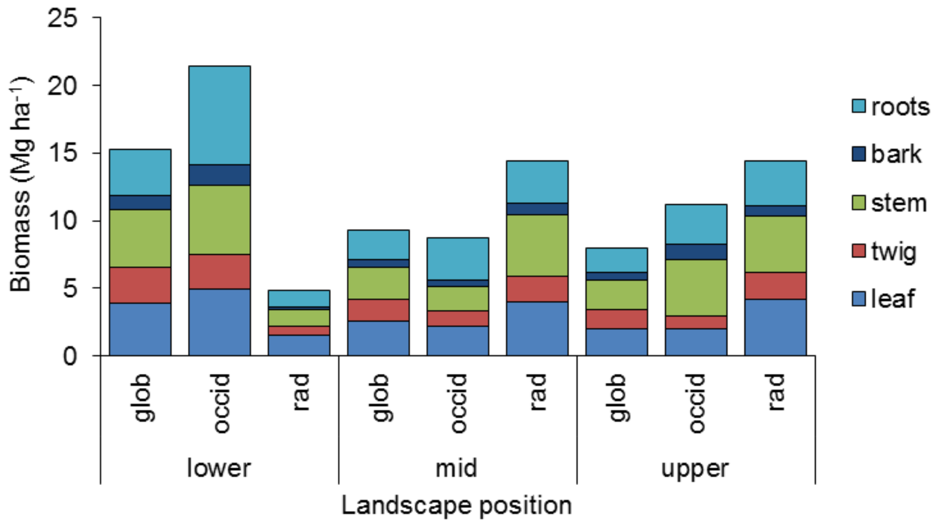


Figure 7.1 Biomass yield (Mg ha^{-1}) after three years growth for each tree component for *E. globulus* (glob), *E. occidentalis* (occid) and *P. radiata* (rad) planted at 4000 trees ha^{-1}, for lower, mid- and upper slope positions.

7.3.4 Nutrient export

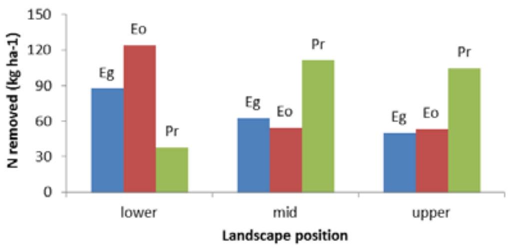
Leaf and stem-wood accounted for about half of tree biomass for all three species Table 7.4. Leaves made up approximately 25% of biomass for *E. globulus* and *E. occidentalis* and 30% for *P. radiata*. Stem-wood was approximately 25% for all three species. Stem-bark was similar for *E. globulus* and *E. occidentalis* at 7% and slightly less for *P. radiata* at 4%. *E. globulus* and *P. radiata* both had approximately 25% of total biomass in the root component. This increased to 34% for *E. occidentalis*.

Table 7.4 Partitioning of tree components leaf, twig, stem-wood, stem-bark and roots for each species.

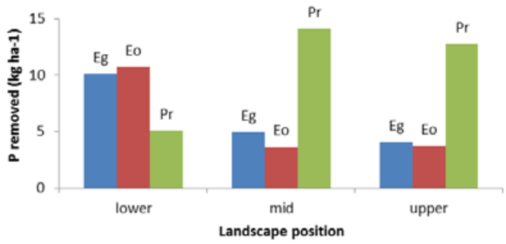
	Tree component				
	leaf	twig	stem-wood	stem-bark	roots
	(%)				
E. globulus	25.1	17.8	26.3	7.0	23.8
E. occidentalis	23.8	12.0	23.5	6.7	33.9
P. radiata	30.5	15.5	25.2	4.3	24.6

The total nutrient export for each species and landscape position, calculated from biomass yield and nutrient concentrations, was estimated for N, P, K, S, Ca and Mg (Figure 7.2). Nitrogen export was highest for *E. occidentalis* at 124 kg ha^{-1} followed by *P. radiata* with approximately 110 kg ha^{-1} for both mid and upper-slope positions (Figure 7.2). N export for *E. globulus* was 87 kg ha^{-1} in the lower slope position. *Pinus radiata* had the highest levels of P (14 and 13 kg ha^{-1}) and K (87 and 80 kg ha^{-1}) export at both mid and upper-slope positions respectively. All three species had similar levels of Mg export for the mid and upper-slope of about 15 kg ha^{-1} and over 25 kg ha^{-1} for *E. globulus* and *E. occidentalis* in the lower slope position. S export was highest for *E. occidentalis* at 14 kg ha^{-1} followed by *P. radiata* with 11 kg ha^{-1} for at both mid and upper-slope positions. Ca export was similar for both *E. globulus* and *E. occidentalis* at 160 kg ha^{-1} for lower slope treatments, these two species having significantly (P<0.001) higher levels of Ca export for all three landscape positions in comparison to *P. radiata* with a maximum Ca export of 35 kg ha^{-1} for mid and upper slope treatments.

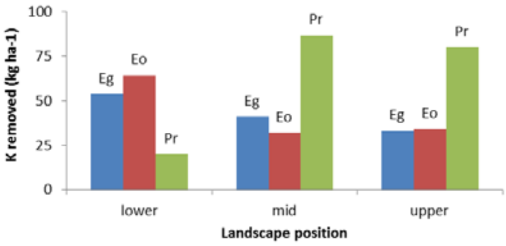
(a)



(b)



(c)



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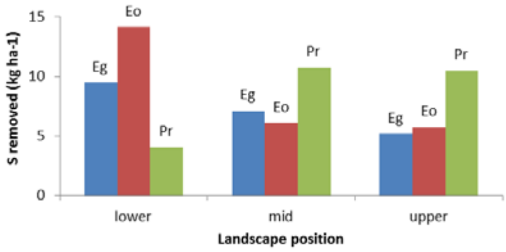
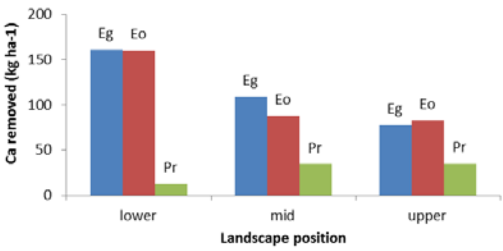


(e)



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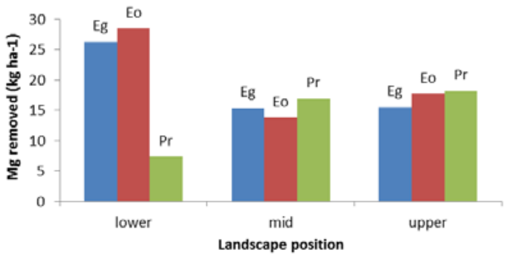


Figure 7.2 Total amounts of (a) nitrogen, (b) phosphorus, (c) potassium, (d) sulfur, (e) calcium and (f) magnesium (kg ha^{-1}) removed in each landscape position with harvest at three years for *E.globulus* (Eg), *E. occidentalis* (Eo) and *P. radiata* (Pr).

7.3.5 Nutrient export of tree components

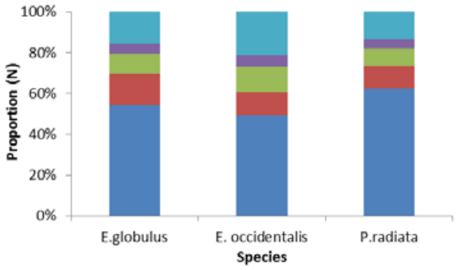
The leaf component of all three species had significantly (*P*<0.001) greater proportions of nutrients compared with all other components for each of the nutrients

N, P, K, S, Ca and Mg. In each case it accounted for 40 to 60% of nutrient export for all three species (Figure 7.3). Bark contributed approximately 5% of N, P, K and S and approximately 10% of Ca and Mg for *E. globulus* and *E. occidentalis*. For *P. radiata* bark nutrient export was approximately 5% for all of the above nutrients. Roots and twigs contributed to nutrient export in a similar manner with proportions in the range of 10 to 20%.

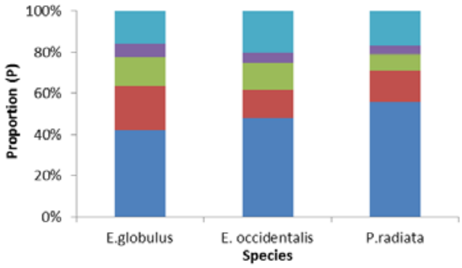
7.3.6 Mean landscape nutrient export

Total nutrient export for each species in each of the three landscape positions is shown in (Figure 7.4). *P. radiata* had the highest amounts of N, P and K export (85 kg ha^{-1}, 11kg ha^{-1} and 62 kg ha^{-1}, respectively) followed by *E. occidentalis* (77 kg N ha^{-1}, 6 kg P ha^{-1} and 43 kg ha^{-1}) and *E. globulus* (67 kg N ha^{-1}, 6 kg P ha^{-1} and 43 kg ha^{-1}). S export was similar for all three species between 7 to 9 kg ha^{-1}. Ca export was highest for both *E. globulus* and *E. occidentalis* at approximately 115 kg ha^{-1} compared to 28 kg ha^{-1} for *P. radiata*. Mg export ranged from 14 kg ha^{-1} for *P. radiata*, to approximately 20 kg ha^{-1} for *E. globulus* and *E. occidentalis*.

(a)



(b)



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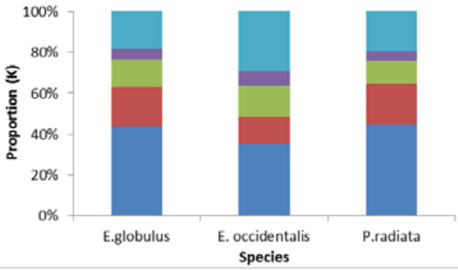
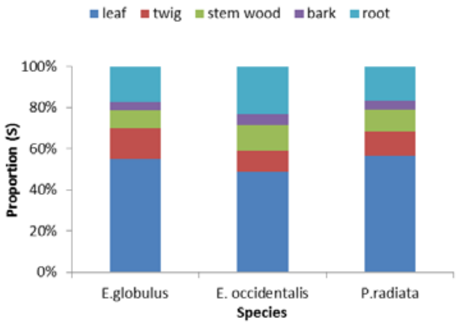
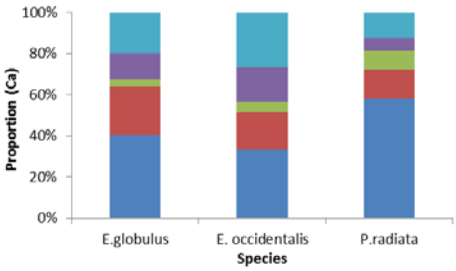


(d)



(e)



(f)

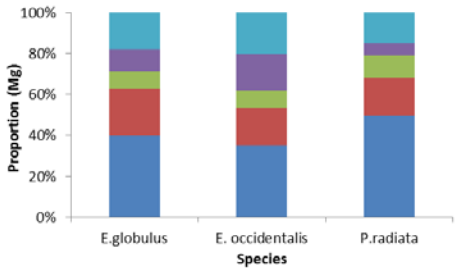


Figure 7.3 The proportion (%) (a) nitrogen, (b) phosphorus, (c) potassium, (d) sulfur, (e) calcium and (f) magnesium contained in different plant components for each species combined across all landscape positions.

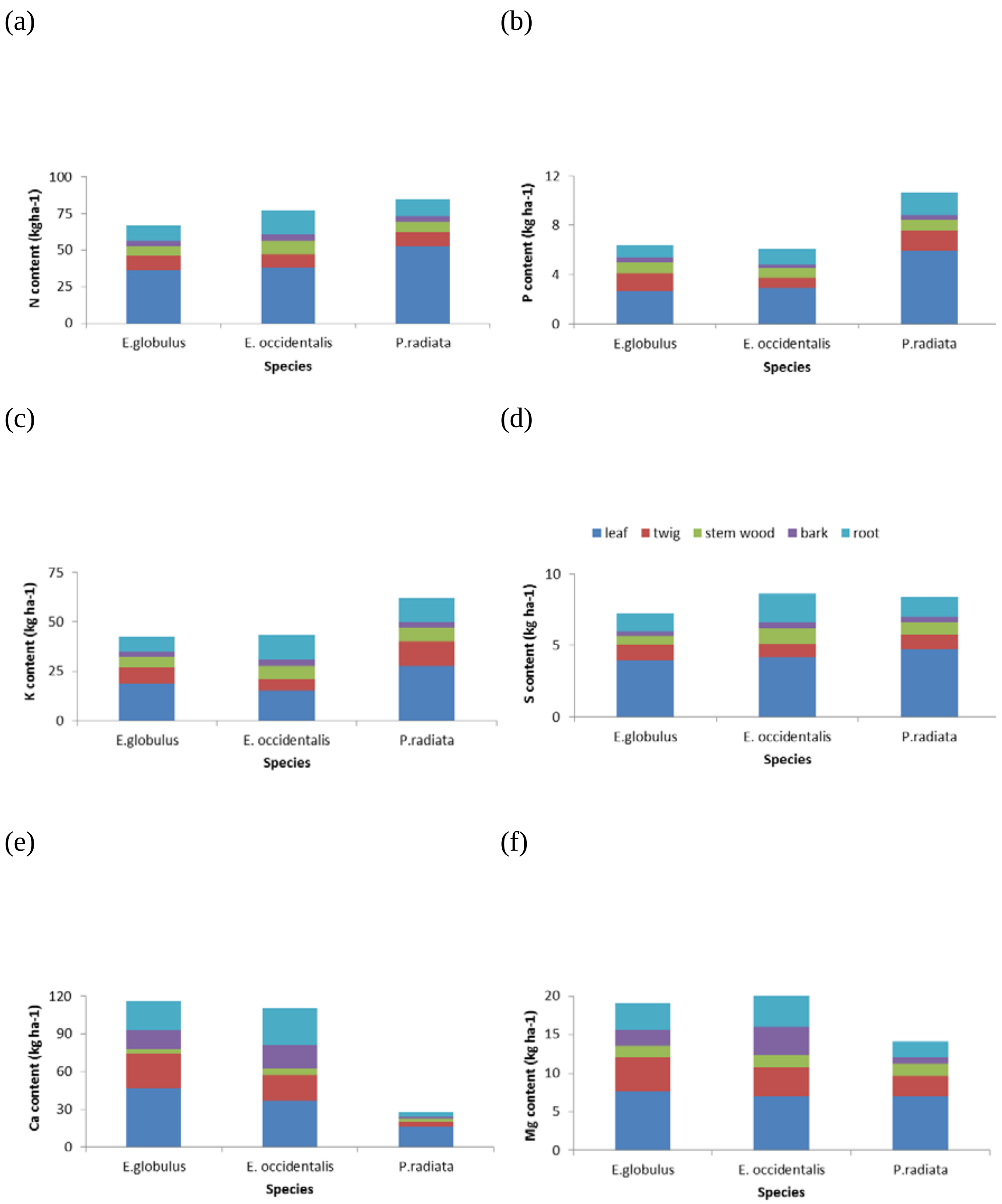


Figure 7.4 The amount (kg ha^{-1}) of (a) nitrogen, (b) phosphorus, (c) potassium, (d) sulfur, (e) calcium and (f) magnesium contained in different plant components for each species combined across all landscape positions.

7.3.7 Nutrient assimilation index (NAI)

The nutrient assimilation index (Eqn. 10) was used to compare component nutrient use efficiency for each species (Figure 7.5) for all components NAI generally followed the order of leaf<twig<bark<root<stem. Similarly, for all three species leaves had the lowest NAI for most nutrients and stem-wood had the highest NAI (Figure 7.5). For leaves, the lowest NAI values occurred for N. *E. globulus* and *E. occidentalis* had low NAI values for Ca for leaf, twig, bark and root components.

Table 7.5 Comparison of export of nitrogen, phosphorus and potassium between wheat cropping and phase farming with trees.

	Nutrient export (kg ha^{-1} yr^{-1})		
	N	P	K
Wheat			
Grain*	42.0	5.5	10.0
Straw**	18.0	5.1	33.0
Trees			
Maximum	40.0	4.6	26.6
Mean	23.0	3.3	20.0

* 2 t ha^{-1} for wheat ** 3 t ha^{-1} for straw, nutrient values derived from Bolland et al. (2000).

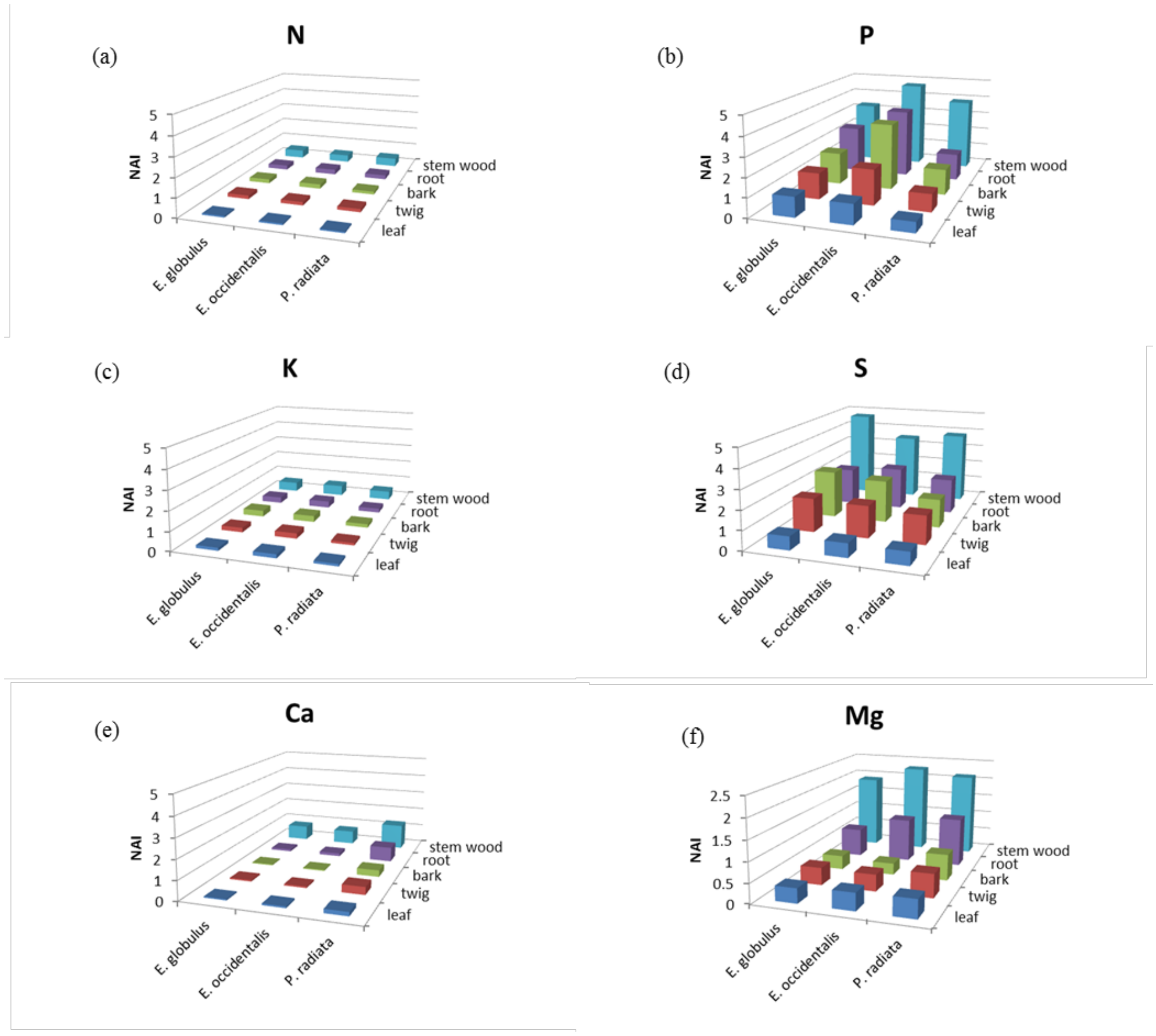


Figure 7.5 Nutrient Assimilation Index for (a) nitrogen, (b) phosphorus, (c) potassium, (d) sulfur, (e) calcium and (f) magnesium for each component for *E. occidentalis*, *E. globulus* and *P. radiata*.

7.4 *Discussion*

Differences in biomass production were not entirely reflected in nutrient export. For example, although *E. occidentalis* produced 14% more biomass than *P. radiata*, the differences in tree component nutrient content between species resulted in *P. radiata* exporting greater levels of several nutrients. When component nutrient exports were averaged across the three landscape positions (Figure 7.4), *P. radiata* had the highest amounts of N, P and K export, 85 kg ha^{-1}, 11 kg ha^{-1} and 62 kg ha^{-1} respectively, even though this species did not have the highest biomass yield. The highest overall nutrient export for N, P and K occurred in situations with higher concentrations of nutrients, partitioning of tree components and relatively high biomass yield. For example, *P. radiata* had 5% more leaf component (nutrient rich) and 10% less root component (nutrient poor) than *E. occidentalis*. *E. occidentalis* (77 kg N ha^{-1}, 6 kg P ha^{-1} and 43 kg K ha^{-1}) and *E. globulus* (67 kg N ha^{-1}, 6 kg P ha^{-1} and 43 kg K ha^{-1}) had lower levels of nutrient export when calculated as an average landscape value.

7.4.1 Nutrient export

The amount of nutrients assimilated by the PFT system is small in comparison to longer rotation tree crops, such as *E. globulus* (Mendham et al., 2004), due to the short rotation length, and lower yields at this lower rainfall site compared to those sites examined by Mendham et al. (2004). Similarly, the export of different nutrients by the PFT system can be compared to nutrient removal in agricultural production in the same region. Here, the farming practices comprise cropping with cereal grain, grazing of introduced pasture plants or the cutting and removal of hay crops. These practices generally rely on annual nutrient input via fertilizers to maintain yields.

The typical nutrient export for wheat grain and wheat straw are compared to a PFT crop in (Table 7.5). The removal by the trees is clearly less for N and P when wheat grain only is removed, and potentially less for K if straw is removed. Wise and Pitman (1981) compared nutrient export of agricultural crops to six species of eucalypts grown in ten year rotations on the north coast of New South Wales, with nutrient export for cereal crops of 100 kg N ha^{-1}, 30 kg P ha^{-1} and 60 kg K ha^{-1} for a single year, which were not as conservative as the values estimated in this study. Compared to cereal cropping with one fallow year every four years, the nutrient removal by eucalypt plantations was much less. Other studies have compared cereal crops to tree crops and found similar levels of nutrient accumulation. Wang et al. (1991) showed N accumulation in 5.5 year rotations of *Casuarina equisetifolia* and *Albizia procera* calculated on an annual basis was comparable to maize and sorghum. Harmand *et al*. (2004) report that harvesting of grass for fodder could potentially be a more serious risk to site fertility than 5 to 7 year tree rotations.

7.4.2 Nutrient assimilation index (NAI)

The NAI proposed in this study will enable the comparison of tree and component nutrient use across different studies for differing species. Nutrient use by tree crops is often expressed as kg ha^{-1} of nutrient export, which does not immediately relate to biomass yield (Merino et al., 2005; Safou-Matondo et al., 2005). When nutrient use is related to biomass yield as nutrient use efficiency, the units have been expressed either as kg biomass per kg nutrient (Adegbidi et al., 2001) or kg biomass per g nutrient (Kimaro et al., 2008), however neither of these relate to typical units of biomass yield (Mg). In this study, the use of a NAI with units of Mg kg^{-1} enable nutrient use to be associated with typical units for biomass yield and can be more easily related to yield associated with tree crop species. For example, for a tree crop species with a NAI of 2 for N that yields 100 Mg ha^{-1} of biomass, the export of N would be 50 kg ha^{-1} and this could then be related to a demand of 50 kg ha^{-1} on soil nutrient store.

In this study the NAI was used to compare tree components to determine nutrient use and potential harvest manipulation to retain nutrients on site. Leaves typically had the lowest NAI and the retention of leaf biomass on site is therefore an option to minimize nutrient loss. The NAI of tree components followed similar trends to previous studies relating to nutrient use efficiency (Kumar et al., 1998). The order of nutrient assimilation among components was leaf<twig<bark<root<stem-wood, which was similar to the results of Wang et al. (1991) with the most nutrient efficient component being stem-wood, having a NAI of 4.7 Mg kg^{-1} for P and S. Wang et al. (1991) showed that nutrient use efficiency varied widely between five tropical taxa including *Eucalyptus robusta*. Nutrient use efficiency increases with tree age and consequently short tree rotations are nutrient expensive and the retention of nutrients on site is important to ensure sustainable tree crop management.

The NAI is thus a valuable guide to the efficiency of a tree crop in relation to nutrient use efficiency and biomass yield. For sustainable biomass production, site nutrient levels need to be maintained to avoid site degradation and a NAI can be used as a basis for comparing different management strategies. One approach is via species selection; in this study *P. radiata* was not as nutrient efficient as *E. globulus* and *E. occidentalis*, assimilating more nutrients despite producing less biomass. The NAI may also have applicability in phytoremediation where trees are being used to strip nutrients from the soil to prevent eutrophication of groundwater (Rockwood et al., 2004); in this case, species with a low NAI would be purposely selected.

7.4.3 Soil protection

Soil protection via retention of nutrients and minimizing nutrient export will be crucial for the sustainability of PFT systems. Studies have shown that N is immobilized by eucalypt leaf residues that could result in short term limitations in N supply (Aggangan et al., 1999; Corbeels et al., 2003). This reduction in N mineralization is a result of an increase in lignified litter but it may also represent an opportunity to improve soil structure and soil carbon storage through the addition of organic matter. Guo et al. (2006a) reported that up to 24% of total N uptake by 3 year old eucalypt species is returned to the forest floor via litter fall. This recycling of N would also be associated with organic carbon input into the soil.

The mean amount of P and K removed from the site represented 16 and 22% of the soil available nutrients, respectively. Soil P is a function of the quantity of labile P and an inverse function of the soil buffering capacity (Holford, 1997). The availability of P in the soil is a complex and dynamic interaction between P in the solid phase and the concentration of P in solution (available P). Phosphorus in solution is absorbed by roots and is replenished from the solid phase which is affected by how strongly P is sorbed by the soil (Holford, 1997). Continuous application of superphosphate, as is the case in agriculture, has the effect of eventually saturating the soil with sorbed P, resulting in more available P for crop growth (Bolland et al., 2003). In this study, more detailed analysis of soil P is warranted in order to determine the effect trees have on available P in the cropping root zone and on recycling leached P from deeper in the soil profile as a result of many years of phosphate application.

The amount of K removal may be a concern for the lower landscape position which had the lowest soil K concentrations (Table 7.1). Brennan et al. (2004) investigated the effect of soil K levels on wheat yields in sandy soils in south-western Australia and reported the critical levels of Colwell K as <50 mg kg^{-1}. The concentrations for the lower landscape range from 25 to 37 mg kg^{-1} for the soil depths sampled and such areas may require the addition of K if returned to conventional wheat cropping. Grove et al. (2007) reported amounts of biomass removal similar to those reported in this study for five year old eucalypt mallee trees, and recommended the application of supplementary fertilizer to sustain growth rates over several harvesting cycles. In Brazil, where some soils are low in K content, K fertilizer inputs managed routinely across rotations to ensure deficiency does not occur (Stape et al., 2010).

7.4.4 Nutrient management

Nutrient export could also be manipulated through employing different harvest strategies, such as only removing selected components of the trees, or through the return of processing residues following bio-energy production. For example, approximately half of the exported N, P, K, S, Ca and Mg are in the leaf component (Figure 7.3) and inducing leaf senescence would result in the retention of nutrients in the farming system and potentially improve soil carbon content by incorporating leaf material back into the soil. The obvious trade-off is the loss of 25 to 30% of biomass, available for bioenergy production, and it may be more economically viable to replace the removed nutrients with mineral fertilizers. Re-spreading ash from bio-energy plants would recycle some nutrients, such as P and K in particular and possibly have a liming effect on the soils as ash can contain carbonates (Harper et al., 1982). Depending on the bioenergy process, biochar could be produced from this material with possible beneficial effects on soil fertility (Steiner et al., 2007).

Further research is required to determine the effectiveness of strategies to either replace these removed nutrients via the application of fertilizer, modifying harvesting practices to only remove select components of the trees, or through the return of processing residues from bio-energy production. The proportion of agricultural land under PFT would be dependent on individual farm budgets and the proportion of land set aside would vary depending on farm income and management.

7.5 Conclusions

In this study, the export of nutrients for a three-year rotation bioenergy system did not detrimentally affect the nutrient status of the dryland farming system. The application of PFT in short rotations on agricultural land was not only sustainable but potentially enabled the capture and recycling of leached nutrients. The NAI provides an objective basis for optimizing nutrient management in bioenergy systems, with quite different values between tree species and tree components. Despite high nutrient demands (low NAI), nutrients exported do not exceed typical farming practices and would have minimal impact on soil nutrient stores. Nutrient removal could be alleviated through the employment of different harvesting strategies, or the replacement of lost nutrients either by reapplication of biomass wastes after processing or through rotations of legume crops for N input.

8 General Discussion

8.1 Introduction

Research outcomes from this thesis have been described in previous chapters and subsequent journal publications, hence in this general discussion, the interplay of research outcomes will be applied in a broader context.

In Australia, avenues for mitigating atmospheric CO_2 via the land sector are limited by economies of scale, and are forced to operate within economic and environmental constraints (Bryan et al., 2014). These are in turn governed by a plethora of ever changing legislation and policy (Macintosh, 2013; Mitchell et al., 2012). The most recent changes include removing the carbon price established in 2014 (revoking the carbon tax) and adopting the Carbon Farming Initiative (CFI) with the Emissions Reduction Fund (ERF) (Australian Government, 2014b). However, climate change is a global issue, and Australia’s efforts are a part of a concerted global effort, brought together by the international treaty of the United Nations Framework Convention on Climate Change and the legally binding Kyoto Protocol. Fragmentation is a term used to describe global climate policy actions as countries implement their own policy agendas with diverse levels of ambition with respect to climate change mitigation (Schwanitz et al., 2015) and Australia is no exception. In the Kyoto Protocol’s first commitment period, Australia was given what some authors consider a “free ride” in meeting emission targets via avoided land clearing through Article 3.7 (the Australia clause), while increasing emissions in other sectors (Hamilton and Vellen, 1999; Howarth and Foxall, 2010). This allowed Australia to meet its commitment of an 8% increase in emissions.

Nonetheless, the land sector continues to make a large contribution to Australia's mitigation ambitions. Articles 3.3 and 3.4 of the Kyoto Protocol provide avenues not only for carbon mitigation within the land sector but also the potential to address environmental issues at an extensive scale (Swingland et al., 2002; Harper et al., 2007; George et al., 2012). Extensive soil degradation, loss of agricultural production, declining water quality, biodiversity decline and dryland salinity, could potentially be addressed through afforestation and reforestation and help meet emission targets.

In this thesis three major themes were evaluated which relate to climate change mitigation via afforestation/reforestation, specific to Article 3.3 and bioenergy production: 1) the methodology of biomass estimation and associated uncertainty of tree biomass estimates, 2) integration of trees into agricultural systems for climate change mitigation and the achievement of environmental co-benefits, and 3) a reduction of competitive effects increasing the sustainability of integrated tree phase systems in agriculture. These all have implications and application, not only in Australia's efforts for climate change mitigation, but are also applicable in other regions.

8.2 Methodology

Globally, forest ecosystems make up the third largest terrestrial carbon pool (Mahli et al., 2002), and the use of forests to mitigate climate change has been well recognized (Luyssaert et al., 2008). For land sector carbon mitigation strategies, the ability to estimate various carbon pools is crucial to quantifying global climate change mitigation and with interest in carbon as a market commodity, there is a financial incentive to gather precise verifiable estimates of carbon stocks generated by carbon offset projects.

Above-ground forest biomass is relatively simple to estimate however, below-ground root biomass is more challenging. As discussed in Chapter 4, methodologies are varied and estimates can have high levels of uncertainty and thus represent an obvious deficiency in the estimation of forest carbon pools. Methods of root sampling have essentially remained unchanged for the last five decades (Maeght et al., 2013) with excavation and soil coring the preferred methods. Very few studies have attempted to quantify the precision of root biomass sampling methods (Levillain et al., 2011).

Estimates of forest carbon inventories on global scales can be wide ranging. Waggoner (2009) reported on the uncertainty and discrepancy of forest inventories, highlighting the challenges in quantifying carbon on a global scale and reported 30 to 40% differences in carbon density depending on the IPCC methodology used. Tree root carbon in particular has been difficult to estimate resulting in the reliance on r:s ratios as advocated in the IPCC (IPCC, 1996b; IPCC, 2003) guidelines where tree root data are unavailable. However, these have been shown to inadequately describe root biomass based on above ground biomass (Mokany et al., 2006). Given that tree roots account for a significant proportion of forest carbon and that roots interact with soil carbon, there is a need to improve methods for estimating tree root biomass in afforestation/reforestation projects for climate change mitigation, not only as described here for the reforestation of salinized farmland, but more broadly to forest systems, such as for example, the carbon budgets of avoided deforestation and degradation in REDD+ initiatives (FAO et al., 2008). Here again, carbon loss from tropical deforestation and degradation are estimated as above ground carbon stocks only and do not include tree root carbon (Sills et al., 2014). Within REDD+ initiates, monitoring is categorized into deforestation, degradation and regrowth, all of which have a below-ground component. Technological advances enable remote sensing of carbon stocks via satellite and more recently drone technology, however, ground-truthing is crucial for any remote sensing of forest biomass and carbon stocks.

In Chapter 4, a novel approach using Monte Carlo techniques was applied to tree root data sets to determine how the precision of root mass estimates can be improved. Sampling regimes were simulated to test different combinations of excavation and coring and determine the sampling uncertainty associated with each methodology. The outputs resulted in new sampling methods to improve the precision of root biomass estimates and these were achieved without costly replicated field sampling. For example, sampling uncertainty of soil coring can be improved by as much as 10 to 15% if coring is concentrated in close proximity to sample trees. Completely excavating root systems reduces the reliance on soil coring to estimate coarse roots thus improving the precision of root biomass estimates while reducing the amount of soil coring. These are aspects of tree root biomass sampling which will help improve the efficiency of sampling and reduce the cost involved, a major constraint in tree root studies. Computer simulations can be used as a guide to improve precision and reduce bias when sampling tree root systems and help in the design of sampling regimes for root biomass estimates, empirical datasets which are lacking for regional and global carbon models.

8.3 Integration of trees into farming systems

The mitigation of climate change via tree crops falls into two general categories; biomass feedstocks for bioenergy and biofuel and carbon sequestration. The production of biofuels from second generation lignocellulosic feedstock is being promoted as an alternative to first generation biofuels in an attempt to address concerns over land use change and the competition between food and fuel crops

(Holland et al., 2015). Studies in the EU have shown that short rotation woody crops grown on agricultural land (with low organic carbon) over a two year rotation have a 52 to 54% saving in GHG emissions and show strong prospect as a source of renewable energy (Njakou Djomo et al., 2013). Globally, bioenergy is the largest source of renewable energy (Eisentraut, 2013), however in Australia this is largely unexploited, contributing two thirds of national (5%) renewable energy (Penney et al., 2012).

In low rainfall agricultural regions of Australia, opportunities exist to address environmental issues with the placement of trees and shrub species into agricultural landscapes. Much of Australia's 100 million hectare wheat-sheep zone has been over cleared of native vegetation, giving rise to environmental degradation, in particular, dryland salinity (Nuberg et al., 2009). However, Australia's low rainfall agricultural zone has failed to attract any appreciable investment for agroforestry (Huth et al., 2002). There have been efforts to integrate trees into the landscape via permanent alleys of mallee eucalypts for the purpose of ground water control, and the potential supply of biomass feedstock (Bartle and Abadi 2010, Wildy et al. 2004, Robinson et al. 2006) or carbon (Harper et al., 2011; Mitchell et al., 2012) however, after more than 25 years of research and development the bioenergy proponent of this has yet to come to fruition. Similarly, there have been concerns raised about competitive effects between these trees and crops (Sudmeyer et al. 2012), both from reduced yields and the displacement of productive land.

Land use change to forestry for the purpose of climate change mitigation is a concern if productive agricultural land is displaced (Smith et al., 2014). However, some woody crops can be grown on marginal lands without displacing food production, an issue of concern as energy tree crops gain momentum (Perez-Cruzado et al. 2011). The challenge addressed in Chapters 5 and 6 was the integration of tree crops with minimal displacement of productive land. This has been facilitated in two different ways to avoid negative land use change.

Firstly, the application of short rotation tree phases for the amelioration of soil salinity was shown to produce a potential biomass crop. Biomass production was affected by landscape position, species and planting rate and was applied as a temporary land use change with other added benefits including possible remediation of leached nutrients. In contrast to alley tree crops, this land use change is not permanent and potentially impacts less on regular farming enterprises. Secondly, the reforestation of abandoned or marginal land can occur via 'analogue forestry' (Senanayake and Jack, 1998). With the correct species and planting density, salinized farmland can be rehabilitated to provide additive environmental benefits and carbon mitigation. Globally, it is estimated that 385 to 472 million hectares of abandoned agricultural land exist and the potential area weighted mean production of above ground biomass is 4.3 t ha^{-1} yr^{-1} (Campbell et al., 2008). In Chapter 6 similar yields were obtained from abandoned salinized farmland, a positive landscape change from otherwise unproductive land. Given the extent of dryland salinity across southern Australia (NLWRA 2001), salinized farmland could potentially be rehabilitated and become opportunities for carbon farming (Gianatti, 2012). The results in this thesis are site specific and further investigations are required to determine the full potential of these systems.

The challenge is not only the development and application of these systems for bioenergy and biofuel but the integration with existing farming enterprises, to complement and enhance land use outputs, while avoiding the removal of productive agriculture. At the landscape level, implementing tree phase rotations will potentially result in increasing trends of carbon stocks, particularly if abandoned land with carbon-poor soils are reforested. The application of reforestation can also have long term mitigation advantages over other approaches such as management of native forest carbon stocks. Harvesting of mature native forests can potentially incur a carbon debt which may take decades to recover (McKechnie et al., 2011) despite an initial offset of fossil fuel emissions (Holtsmark, 2012). In contrast, reforestation of farmland for renewable energy has the potential for immediate carbon sequestration (or mitigation) during the growth phase and then serve to offset fossil fuel emissions via bioenergy or biofuel. On a global scale, an integration of mitigation avenues from forests is required, taking advantage of low carbon density farmland landscapes to generate immediate mitigation responses, while native forest stands are adequately managed to avoid incurring long term carbon debt. Land availability will be a crucial factor and therefore to avoid land competition, abandoned land or land with low productivity may be an option.

8.4 Sustainability

Global energy forecasts consider bioenergy inputs in climate change mitigation strategies (Smith et al., 2014) and as a result sustainability of renewable energy systems from forestry are being increasingly assessed as a result of greater emphasis placed on forestry for long term climate change mitigation (IPCC, 2011). Sustainability of short rotation woody crops are being scrutinized in relation to nutrient removals (Dimitriou et al., 2009; Upham et al., 2011), harvest regimes (Walmsley and Godbold, 2010; Walmsley et al., 2009a) and water use (Mendes et al., 2015). Water use concerns relate to competition for water within agricultural systems and the effects of bioenergy feedstock production on environmental contamination via fertilizers, pesticides and sedimentation, not only in Australia (George, 2013) but in other regions (GEA, 2012; UNEP, 2011; Berndes, 2002).

In Chapter 5 the biomass potential of three candidate species was demonstrated and this indicates the potential should tree phases be integrated into agricultural landscapes for salinity amelioration. However, if tree phases are applied, the sustainability of these systems should also be demonstrated. In Chapter 7 the nutrient removal was determined for whole tree and tree components for a three year tree phase. The amount of nutrient removal was not greater than the typical cropping systems that were in place prior to the tree phase. This may be attributable to the short growing phase and the adaption of *Eucalyptus* species to infertile soils. Trees grown on agricultural landscapes have access to leached nutrients from annual fertilizer application which are stored at depth, typical in the poor infertile soils of these regions. In contrast, mallee eucalypts planted in alleys in the same regions of southwestern Australia and harvested continuously in 5 year rotations, removed significant levels of labile N and P and results indicated that these nutrients may need to be replaced for successive long term harvests (Grove et al., 2007). Both SRF and mallee eucalypts have been successfully applied to ameliorate soil salinity and have potential for carbon mitigation, however the long term sustainability of the latter may depend on the addition of supplementary nutrients. Although nutrients can be replenished with synthetic fertilizers, the status of soil organic matter is dependent on the return of plant residues (Lal, 2009). An understanding of nutrient removal of different species and tree components will contribute to developing management options which may be specific to tree crops and regions. To further this understanding of nutrient removal a sustainability index was derived to compare species and tree components (Chapter 7). This can assist in strategies to aid the management of site nutrient removal which may be manipulated via harvest regimes. For example, the leaf component is nutrient rich containing approximately 50% of nutrients, therefore, retaining leaf biomass on site through induced leaf senescence would be an option in recycling of nutrients and organic matter.

The growing concerns related to land and water degradation (Noble, 2012) are influencing traditional farming enterprises which are now changing to accommodate the integration of trees in an attempt to adopt more sustainable farming practices. The resultant “food or fuel” debate over the use of food crops for biofuels has raised concerns not only over the use of food crops for ethanol production (de Souza Ferreira Filho and Horridge, 2014) but also carbon balances, fertilizer input (in particular nitrogen (NAP, 2007)) and whether or not these systems are in fact carbon neutral (Haberl, 2013). These concerns are giving momentum to a growing interest in second generation biofuels. Application of perennials for lignocellulosic feedstocks for bioenergy have some inherent advantages, not only in their efficiency of nutrient use, but also the ability to occupy less productive (marginal) land than is required for food production (Karp and Shield, 2008), and therefore provide more sustainable systems for renewable energy.

Water use efficiency of ethanol (1st generation) biofuels are resource demanding whereas production of lignocellulosic feedstocks are potentially more efficient (NAP, 2007). Across southern Australia, dryland salinity is characterized by a landscape level hydrological imbalance, resulting in both soil water storage and increase in ground water pressures. The effect of tree phases on the removal of excess soil water has been demonstrated by Harper et al. (2014). Contrary to recent literature (PMSEIC, 2010; George, 2013; Bioenergy, 2011) which raises concerns over bioenergy production and water use, the potential of bioenergy-water synergies in salinized farmland landscapes were demonstrated in Chapters 5 and 6, where trees have been applied to address landscape hydrological imbalance, an extensive problem in agricultural regions across southern Australia. Biomass yields are positive indications of the potential for biomass feedstocks, however, further research is required to establish the water use efficiencies of these systems. The reforestation of degraded and abandoned farmland (Chapter 6) can have positive LUC effects without impinging on water resources, synergies which as yet are not being exploited for carbon mitigation.

8.5 Future research directions

Determining tree root biomass remains a challenge and tree root studies in general will remain inherently difficult simply due to the fact that tree roots grow in a soil medium. A recent review (Addo-Danso et al., 2016) on root sampling methods, exemplified the range of root sampling methods applied and recommended further studies to directly compare methods of tree root sampling on similar sites. This has been addressed in this thesis and should be extended to other sampling methods. Further studies are required to build the knowledge base of tree root-soil ecosystem interactions, in particular, biomass depth functions of tree root biomass for species and climatic regions and the response of root systems to LUC, in particular afforestation/reforestation and deforestation. Estimations of carbon emissions from deforestation and forest degradation are 15% of the global total (van der Werf et al., 2009) and these do not incorporate measurements of root responses (Houghton, 2005) but rather, estimates based on default values. The lack of empirical data for tree root systems will limit the confidence with which model outputs can be applied in predicting mitigation potential of mitigation strategies.

For reforestation of farmland to play an integral part in mitigating climate change, these systems will require further development and integration within agro-ecological systems. A potential biomass supply chain has been established, however, further research is required to determine the feasibility of bioenergy systems being incorporated into current energy systems and the potential of this feedstock for lignocellulosic biofuels.

Water use efficiency and nutrient sustainability will govern the long term sustainability of mitigation systems from the land sector. Nutrient removal will require further evaluation for proposed reforestation systems and assessment within different agricultural landscapes. The potential of water-biomass synergies in saline landscapes will require further investigation to determine how these opportunities can be used to advantage where degraded land has the potential for bio-mitigation.

8.6 Final conclusions

Despite the global acceptance of climate change, a major challenge remains in applying scientific outcomes which rely on government policy and socio-economic acceptance, particularly when change may incur a financial cost or burden. In Australia, climate change policy is continually debated with major policy changes occurring with changes of government. This is evident in recent climate change policy, for example, in the repeal of a carbon tax and the reluctance to commit to strategies via a carbon tax or carbon trading schemes. The lack of policy for biomass based renewable energy is not conducive to investment in new renewable energy technologies. Given the reduction of renewable energy targets and attempts to abolish the Clean Energy Finance Corporation (CEFC), confidence for investment in renewable energy has been diminished (CEFC, 2015). In the future, industry may be disadvantaged internationally if Australia is not seen as a “green” economy. Australia is lagging behind, naively relying on promises of cleaner coal power generation, when other nations are investing heavily in renewable energy, progressing with the touted next industrial revolution (Bloem et al., 2014).

In a recent review (Crawford et al., 2015) of the potential biomass for bioenergy in Australia, short rotation tree crops were identified as a significant source of future biomass, however, the challenge remains in progressing this feedstock via government intervention and business investment. The integration of trees into farmland landscapes in southwestern Australia has waned in the last decade. Soil salinity, a major environmental problem across southern Australia, in particular Western Australia, was addressed with the integration of trees into farmland landscapes, with a significant uptake in the planting of oil mallees, pre-empting the potential of carbon credits to fund landscape rehabilitation. However, despite large areas of reforestation in Western Australia’s low rainfall region and potential biomass feedstocks (Clean Energy Council, 2011), development of bioenergy systems utilizing biomass from dedicated tree crops has not progressed beyond a small (de-commissioned) bioenergy pilot plant, and opportunities for aviation biofuel from these tree crops is still in infancy (Australian Government, 2012).

Despite the potential to address environmental degradation with reforestation, ecosystem restoration on a major scale in low rainfall farmland regions has yet to attract monetary value. The CFI was introduced in 2011 to facilitate carbon offsets and potentially provide an opportunity for land owners to trade carbon within a carbon pricing scheme utilizing a range of land management options to increase carbon in land systems. However, following the repeal of the carbon tax and the introduction of the ERF, carbon sequestered from CFI activities (Australian Carbon Credit Units (ACCU)) will be auctioned within the ERF via the Clean Energy Regulator (CER).

The relatively high cost associated with some land sector abatement may not be conducive to investment.

Despite obvious environmental issues as a result land degradation and the potential for carbon mitigation, Australia is not involved in major reforestation. In contrast, China has made massive commitments to reforestation, with 13 million hectares planned for reforestation between 2001 and 2015 (Turnbill, 2007), described as one of the largest ecological engineering projects worldwide. The world's largest emitter of greenhouse gases is taking a lead in climate change mitigation strategies via reforestation.

China is also trialing carbon trading schemes in seven provinces and cities before going forward on a national level and in the US sub-national action has been effective where 10 states operate their own carbon pricing schemes despite the lack of a national price on carbon (Australian Government, 2014c). Despite the lack of clear federal climate change policy, state governments can play a role, and have taken the lead in climate change in the past. For example, the world's first mandatory emissions trading scheme, the Greenhouse Gas Abatement Scheme was introduced in 2003 in NSW (The Climate Council Australia, 2014). Given Australia is the highest emitter of greenhouse gases on a per capita basis, perhaps more is expected of Australia's commitment to climate change. The reliance on the ERF within the Direct Action Plan (DAP) to achieve pledges for emission reductions based on a $2.55 bn public funds has drawn criticism (Clarke et al., 2014; Hawkins, 2014). The scheme has not engaged major emitters in the industrial sectors and a large portion of emissions auctioned have been to the land sector for avoided deforestation, which may have alternatively been achieved at no cost via government regulation. The mechanisms of the ERF may not be adequate to promote ideological change towards renewable energy or directly address the source of emissions. Accessing the performance of the ERF to date, indications are that this scheme will not be sustainable over the long term and Australia's formal UN climate pledge of 26 to 28% below 2005 levels by 2030 will require further policy measures to achieve its target. Ironically, despite the need to address environmental issues with reforestation and opportunities for climate change mitigation in the land sector via reforestation, extensive land clearing continues in Australia (Bulinski et al., 2015)!

At the 21st Conference of the Parties (COP21) of the United Nations Framework Convention on Climate Change (UNFCCC), the importance of forests as carbon sinks was a key outcome of this meeting held in Paris, France. A major outcome of the Paris agreement was the strong recognition of the value of reducing emissions through forest protection. These build on the Warsaw Framework for REDD+ which was an outcome of the 2013 UNFCCC meetings. Mitigation strategies to reduce deforestation is seen as a mitigation option which can have an immediate effect on emissions and a significantly large impact on carbon stocks (IPCC, 2007).

The importance of sinks and reservoirs in the agreement are recognized with specific reference to the role of forests in climate change mitigation, sending a "strong political signal" as to the importance of forest protection, management and restoration. Support for sustainable management of all types of forests will not only mitigate carbon but result in improved lifestyle in many developing nations.

The agreement also addresses the implementation of a "global stocktake", with the first stocktake taking place in 2023 and every five years thereafter. Given that almost 90 countries have identified forestry within their action plans, the ability to estimate carbon stocks in existing forest sinks and new reforestation projects will be a key issue and a challenge for a global stocktake. Carbon accounting systems are not in place for agroforestry-reforestation systems and there are many associated challenges for landscape scale reforestation for carbon mitigation. The research outputs from this thesis can be applied to help facilitate the integration of tree-crop systems into salinized low rainfall farmland landscapes, and take advantage of climate change mitigation opportunities in the land sector.

9 References

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10 Appendix I Schematic diagram of coring head