Abstract

This thesis examines the effects of species, rainfall and soil type on tree biomass regressions, as well as the effects of stand dominance and structure on stand biomass regressions in north-east Australian woodlands. This was achieved by examining tree characteristics and biomass relationships for a series of woodland monitoring sites throughout the study area. This study utilised a modified data set from this permanent monitoring site network to provide structural attributes for trees and communities of varying composition in the grazed woodlands. These data were supplemented with environmental data and tree harvest data sets.

Initially, the research reported in this thesis developed allometric and stand biomass regressions for *Callitris glaucophylla* communities. This research also demonstrated that changes in tree-form were not reflected in changes in the environment, nor did such changes reflect changes in tree biomass regressions for three eucalypt species. As a result, a common regression provides a robust estimate of total aboveground biomass of eucalypt trees in the study area. Thus expensive destructive harvesting can generally be avoided for minor eucalypt species. Finally, this study demonstrated a successful methodology that described the stand structure of all the grazed woodland sites based on tree heights. This methodology was developed to allow the expansion of a single stand regression to estimate stand biomass across the entire north-east Australian woodlands.

The findings demonstrated in this study, combined with the long-term data from the permanent monitoring network sites, should enhance the estimation of carbon flux within eucalypt communities of north-east Australia's grazed woodlands.

Application of tree and stand allometrics to the determination of biomass and its flux in some north-east Australian woodlands.

Tree and stand allometrics

by

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Table of contents

List of tables

List of figures

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Declaration of Originality

This thesis reports the original work of the author, except as otherwise stated in the acknowledgements and listing of the associated publications. It has not been presented to this or any other university for the award of another degree.

Madonna B. Hoffmann

Publications

Data used during the preparation of this thesis also formed part of the contribution made by the candidate as a co-author in the following publications:

Burrows, W.H., Hoffmann, M.B., Compton, J.F., Back, P.V. and Tait, L.J. (2000). Allometric relationships and community biomass estimates for some dominant eucalypts in Central Queensland woodlands. *Australian Journal of Botany*. **48**: 707-714.

Burrows W.H., Hoffmann, M.B. Compton, J.F. and Back, P.V. (2001). Allometric relationships and community biomass stocks in White Cypress Pine (*Callitris glaucophylla*) and associated eucalypts of the Carnarvon area – South central Queensland (with additional data for Scrub Leopardwood – *Flindersia dissosperma*). National Carbon Accounting System Technical Report No. 33. Australian Greenhouse Office: Canberra.

Williams, R.J., Zerihun, A., Montagu, K.D., Hoffmann, M., Hutley, L.B. and Chen, X. (2005). Allometry for estimating aboveground tree biomass in tropical and subtropical eucalypt woodlands: towards general predictive equations. *Australian Journal of Botany*. **53:** 607-619.

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Definition of terms

MAR - Mean annual rainfall

QDPI&F- Queensland Department of Primary Industries & Fisheries

DBH, D, DBHOB - Diameter breast height (1.3m from ground) over bark

TRAPS - **T**ransect **R**ecording And **P**rocessing **S**ystem. Each monitoring site consists of 5 belted transects (100 m × 4 m) 25 m apart in a relatively uniform community. Data describing location, species, circumference at 30 cm from ground, height, burn ratings and comments on plant health (dead, sick etc.) were recorded for every plant within the transect belt following the TRAPS methodology. These monitoring sites were established on rural landholdings and represent a range of vegetation communities, dominant species and geographical locations. Full description sites and methodology in Back *et al.* (1997).

Chapter 1. Introduction

Woodlands and savannas are a major resource used for extensive grazing in Australia due the vast area that is grazed by domestic stock (Figure 1-1). Since the introduction of domestic stock into savannas and woodlands, woody plant encroachment and vegetation thickening have been reported worldwide (Archer *et al*. 1995; Van Auken, 2000). Australian woodlands and savannas are thickening, as well as, encroaching into previously treeless areas since the introduction of domestic stock (Tothill *et al*. 1985; Pyne, 1990; Burrows *et al*. 1998). Such changes are a result of the dynamic nature of these communities and alterations in land management practices since European settlement.

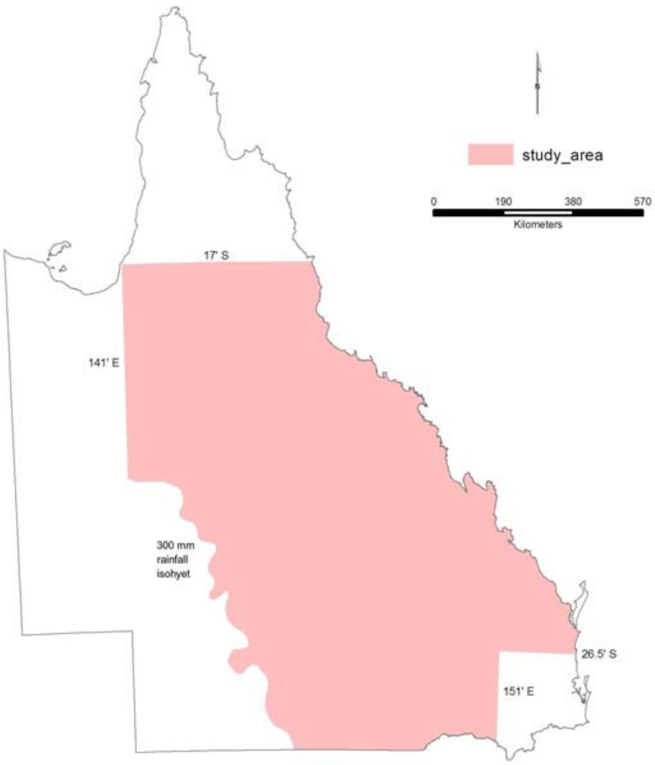

Figure 1-1. Map of study area enclosing the grazed woodlands of north-east Australia.

Changes in woodland structure include encroachment by another species and changes in the understorey structure, both resulting in a change of dominant species. Woodland structures are capable of altering without a change of dominant species. In particular, eucalypt stands can increase plant numbers by recruitment and regrowth from lignotubers. Furthermore, vegetation thickening occurs when existing individuals are released from suppressive conditions, for example, existing eucalypt seedlings undergoing increased growth rates due to an advantageous change in competition.

Increases in woody vegetation within these communities may play a major role in carbon accounting by acting as a carbon sink and reducing the levels of greenhouse gases. Carbon sinks could result in a reduction of economic costs to meeting international commitments to net emission reduction. Given that woodlands occupy a large part of the north-east Australia region any carbon sequestration in this area would be substantial (Burrows *et al*. 2002).

Accurate measures of any potential carbon sink are required for calculating carbon fluxes. The application of allometric regressions is an accepted and widely used method to estimate tree biomass, because the development of individual species-specific relationships is time consuming and very expensive. The use of common biomass regressions have been demonstrated for a limited range of species and environments elsewhere in Australia (Burrows *et al*., 2000; Eamus *et al*., 2000). Additionally, the establishment of stand biomass regressions that can be applied to woodland communities have been limited by the lack of availability of appropriate regressions.

Australian woodlands are subject to a range of management regimes such as grazing, fire and clearing. Changes in woodland structure following the introduction of domestic livestock include dominant species changes, encroachment and structural changes. However traditional forestry techniques for quantifying stand structure appear to be inappropriate in these communities. Furthermore, the effect these changes in woodland structure have on potential stand biomass regressions is unknown.

This study examined the effects of species, rainfall and soil type on tree biomass regressions, as well as the effects of stand dominance and structure on stand biomass regressions in north-east Australian woodlands. This was achieved by examining stand characteristics and biomass relationships from a range of permanent woodland monitoring sites throughout the study area.

Chapter 2. Review of Literature

2.1 North-east Australian Woodlands

Woodlands in Australia are often classified as a sub-type of forest and there is considerable overlap in the usage of both terms (National Forest Inventory, 1998; Hauenschild and Smith, 1999). Additionally, the terms savannas and woodlands can also be used intermittently. The distinctions between forests and woodlands, as between woodlands and savannas, are often clouded and a different term is often used for the same community.

Australian eucalypt forests are often found within the narrow coastal zones, where rainfall is high and reliable compared with inland zones (Florence, 1996). Projected canopy cover usually covers approximately 30 to 70% of the ground area (Florence, 1996). Queensland is reported to have over 20 million hectares of medium eucalyptus forests and three quarters of these have been classified as woodlands with the remaining portion classed as open forests (National Forest Inventory, 1998).

Woodlands are commonly defined by stand characteristics, the most usual definition being communities of single stemmed woody plants between 10 m and 30 m in height with separate crowns. Woodlands typically have an open structure with a discontinuous grassy understorey (West, 1999 cited by Landis and Bailey, 2005) and canopy covers of approximately 10 to 30% (Florence, 1996). Woodlands are also described as having a canopy cover between 20% and 50% (National Forest Inventory, 1998), or between 5% and 60% (Gillison, 1984), providing tree height was greater than 2 m. Adding to the ambiguity of definitions, open forests have been analogously classified as "….*a community dominated by trees and which had a canopy cover of between 30 and 70%….*"(Gill, 1984). However, Walker and Hopkins (1990) report that 60% crown cover occurred when tree canopies were touching, not overlapping, hence an open forest may also have separated crowns.

North Australian woodlands are also commonly included in the international classification of savannas. Savannas typically have a continuous grass understorey with a scattered woody overstorey (West, 1999 cited by Landis and Bailey, 2005). Additionally, Mott and Tothill (1984) describe Australian savanna communities as an open woodland (usually *Eucalyptus* spp.) with a more or less continuous grass dominant understorey. As a result of inconsistent definitions of stand characteristics, the separation of the open forest communities, woodland communities and savanna communities is indistinct.

Woodland[1] distribution in north-east Australia usually occurs east of the 400 mm mean annual rainfall isohyet (National Forest Inventory, 1998; Hauenschild and Smith, 1999). In this study woodland distribution occurs east of the 300 mm mean annual rainfall isohyet not to exclude savanna communities in this extreme rainfall distribution. The more open woodlands (savannas) usually occur in the western region of this distribution, and the remaining woodlands have a tall through to medium understorey, depending on environmental conditions. Woodlands contain at least 80% of Australia's 500 plus *Eucalyptus* spp. (Gillison, 1984). Variation of species and structure occurs along several environmental gradients, such as mean

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[1] With the boundaries between forests, woodlands and savannas being indistinct and with increasing changes in these communities due to management, in this study the term woodlands includes savanna, woodland and forest communities that have the potential to be grazed by livestock.

annual rainfall, rainfall patterns and temperature, resulting in woodlands that are diverse and variable.

Physical features of the woodland environment in north-east Australia include a topography that is flat, or gently sloping, and usually associated with a primary mountain range, usually the Great Dividing Range. This woodland environment consists of four geomorphic divisions, the coastal plains, north-west uplands, eastern highlands and the plains (2/3 of the state) (Weston, 1988). Rainfall variability of 25% to 50% is higher than the world average (Weston, 1988). Queensland's highest mean annual rainfall (c. 4000 mm) occurs on the northern coast and the lowest mean annual rainfall (c. 150 mm) occurs in south-western region. Two distinct rainfall patterns determine precipitation events. The summer rainfall pattern (isohyets are parallel to the coast) contributes up to 90% of total annual rainfall, while for winter rainfall, isohyets decrease from south to north (Weston, 1988). The intersection of summer and winter rainfall isohyets forms a grid pattern over eastern Australia and results in most centres having unique rainfall histories. Boland *et al.* (1992) divided eucalypt distribution into two main groups that are dependent on rainfall season, i.e. summer rainfall and winter rainfall eucalypts, and noted that some species occurred across both divisions, in varying forms. Maximum and minimum temperatures in north east Australia range from greater than 38°C in summer to less than 0°C in winter. Maximum temperatures tend to decrease towards the coast from the inland and as latitude increases and minimum temperatures tend to increase towards the coast from the inland and as latitude increases.

North-east Australian soils are not considered highly fertile, and according to Weston (1988), only one third are clay, fertile loams and friable earths, while the remaining soils are duplex, massive earths and sands. The more fertile areas have sodic and alkaline sodic clay soils, such as the Darling Downs, Central Highlands, Brigalow Belt and Mitchell Grass Downs. Alkaline sodic duplex soils are less widespread and have limited cropping potential (Weston, 1988) and are usually the more fertile of the woodlands. The duplex, massive earths and sands usually support only grazing and forestry.

Woodlands and savannas are a major resource for extensive grazing in Australia and worldwide (Gillison, 1984; Hadley, 1985; Florence, 1996). Within Australia, over 111 million hectares is grazed by domestic livestock (National Forestry Inventory, 1998). Different management regimes have been imposed on these communities following the introduction of domestic livestock and as a result the structure has changed (Gillison, 1984; Archer *et al*., 1988; Burrows *et al*., 2002; Landis and Bailey, 2005). Today's woodlands bear little resemblance to their pre-European structure.

Changes in woodland structure include changes in dominant species from, encroachment by another species, and changes in the structure of understorey population. Understorey classifications of woodlands include tall understorey and medium understorey (National Forest Inventory, 1998). Eucalypt woodlands with a tall understorey, often consisting of cycads, banksias, acacias and casuarinas, are found in northern Australia and inland eastern Australia where they experience a high rainfall. The drier inland regions support box and ironbark woodlands (*Eucalyptus populnea*[2], *E. melanophloia*, *E. crebra*, *Corymbia polycarpa*), with shrubby to tall woody native species understorey (such as *Acacia* spp., *Eremophila* spp., *Cassia* spp.

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[2] All species authorities throughout this thesis are listed in Appendix 8-3)

etc.). In some communities, no woody understorey is present, and a grassy layer exists in its place. Woodlands are not dominated exclusively by the eucalypts, and co-dominants may include *Melaleuca* spp. in eastern central Queensland (Burrows *et al.,* 2000), *Acacia* spp. in the brigalow belt and *Callitris* spp. in the Injune–Mitchell region of Queensland.

2.2 White cypress pine communities

Callitris glaucophylla, or white cypress pine or cypress pine, is a native conifer belonging to the Cupressaceae family. Mature *C. glaucophylla* usually has an average height of approximately 15-20 m tall, but can reach 30 m. Similarly, average trunk diameter at breast height (1.3m)(DBH) is 0.45 m, but can reach 0.9 m (Boland *et al.,* 1992; National Forestry Inventory, 1998). *C. glaucophylla* typically has a single straight trunk, however branching patterns vary according to tree stand density (Boland *et al.,* 1992). The tree has a typical dense conical shape in open woodlands, whereas in dense stands branching occurs on the upper trunk forming a relatively flat top (Cronin, 1988). Grey green leaves are reduced to scales occurring in whorls on the green branchlets (Cronin, 1988; Hauenschild and Smith, 1999). The *Callitris* genus regenerate from seed, not from lignotubers (Parsons, 1981), and the winged seeds are enclosed within spherical, woody, valved cones which fall to the ground after maturity (Cronin, 1988).

C. glaucophylla is usually found in areas of relatively harsh environments which are characterised by soils with low fertility and poor water holding capacity, particularly sandy and light clay soils, sandy podzolic soils and solodic soils (Johnston, 1975). Pure stands are commonly found on deep sands and co-dominant stands extend to the previously mentioned soil types (Hauenschild and Smith, 1999), often allied with *Eucalyptus* and *Angophora* species. *C. glaucophylla* is often found in areas where maximum and minimum temperatures can exceed 38°C and be below 0°C, respectively (Johnston, 1975) while annual rainfall is usually between 300-650 mm (National Forestry Inventory, 1998). Thus, *C. glaucophylla* is both drought resistant and frost tolerant.

Carron (1984) reported that there were 1.5 million ha of *C. glaucophylla* in Australia, of which 80 000 ha was state forest. In 1998 the National Forest Inventory noted that 1.5 million ha of pure *C. glaucophylla* stands still existed in Australia and co-dominant *C. glaucophylla* woodlands occupied an additional area of 7 million ha. Eastern Australia has the largest area of *C. glaucophylla* in Australia, with approximately two thirds of the total *C. glaucophylla* distribution (National Forest Inventory, 1998). Distribution is usually confined to inland New South Wales and inland south Queensland (Johnston, 1975; Cronin, 1988) usually on topography that forms rolling hills (Cronin, 1988). Within this area it forms a discontinuous belt from the Great Dividing Range north of Taroom, south to the NSW border, and west of Dalby to Augathella (Dale, 1979; National Forestry Inventory, 1998).

Rainfall events favouring seedling establishment are very irregular so that stands are usually even-aged (Johnston, 1975). *C. glaucophylla* belongs to a group of native species that are heat sensitive, non-sprouting and long lived, and an irregular fire pattern facilitates their existence in some Australian communities (Harrington *et al.,* 1984). As individuals trees *C. glaucophylla* are not fire resistant (Johnston, 1975), but conversely, as a community they are resistant to a single fire event (Harrington *et al.,* 1984; Kirkpatrick, 1994). *C. glaucophylla* produce numerous seeds enclosed in cones, which protect the seeds from fires, thus there is a seed bank existing when adults are killed by fire (Kirkpatrick, 1994). Therefore, fire also contributes to the development of even-aged stands, but increases in fire frequency will cause decline and eventual local extinction of the community, as regenerating plants are destroyed before maturity and establishment of a seedbank (Harrington *et al.,* 1984; Kirkpatrick, 1994).

C. glaucophylla is able to invade forests (replacing the original dominant species) and pastures (Hauenschild and Smith, 1999) in the absence of regular fires, coupled with the ability to produce numerous, easily dispersed seeds. Kirkpatrick, (1994) reports that *C. glaucophylla* occurs in dense thickets where fire frequencies have decreased or ceased. Pictorial evidence of encroaching *C. glaucophylla* into adjacent open woodlands (Figure 2-1) as well as historical accounts of *C. glaucophylla* thickening are available. One hundred years ago, the 1901 Royal Commission into the condition of crown lands revealed that *C. glaucophylla* was invading open woodlands and that western NSW land was originally *"……open box-forest country with currajong and an occasional pine tree upon it. The overstocking of the country, coupled with the rabbits, prevented the growth of grass to anything like its former extent, and so causes a cessation of bush fires which formerly had occurred periodically. This afforded the noxious scrub a chance of making headway.*" (Royal Commission, 1901).

Binnington (1997) maintains that the *C. glaucophylla* has increased in area since European settlement due to management changes of these ecosystems. Rolls (1981) quotes historical sources which note that *"The pines came up ten thousand to the hectare. 'One year the stockmen saw the little pines just up to the top of the horses hooves', one man told me. 'The next year the pine tops brushed their boots as they* *rode. And a year or two after that – those old stockmen used to ride at ten past ten, knees cocked up from the saddle like wings – well they had to jam their knees in hard behind the pads or the pines would have pushed them backwards out of the saddle. Soon they just mustered their stock and got out. There was no more room for grass to grow'.*" These reports support increasing evidence of *C. glaucophylla* encroaching into areas that were previously open woodlands or grasslands.

Figure 2-1 *C. glaucophylla* thickening in open woodland area. Kogan, Queensland. a (1936), b (1989) (Photos provided by QDPI&F, Forest Service).

C. glaucophylla is resistant to pests and diseases and is classified as a softwood, although it is physically very hard, often harder than some hardwoods, such as American oak (Lewis, 1999). *C. glaucophylla* has significant commercial value as a multi-purpose building timber (Johnson, 1975). According to Lewis (1999), one mill in south west Queensland exported 900 cubic metres to Japan as structural timber and

3500 cubic metres to USA as flooring. Queensland Forestry Department harvested more than 130 000 cubic metres of *C. glaucophylla* wood in 1996-97 (Hauenschild and Smith, 1999) and 120 000 – 130 000 cubic metres in 1999 (Lewis, 1999).

Although there is a demand for *C. glaucophylla* as a harvestable timber, this demand seems not to be increasing, and is met by current Department of Primary Industries Forestry Service managed forests. Therefore the documented expansion of *C. glaucophylla* communities has the potential to add to the carbon sink in Queensland's woodlands. However there are no known allometrics available for estimating total above-ground biomass for these communities in the grazed woodlands of north-east Australia.

2.3 Woodlands as a carbon source and sink

Many researchers believe that forests and woodlands have the potential to reduce net carbon emissions (Scholes and Bailey, 1996; Cannel, 1999; Beil, 1997; Trinus *et al.*, 1997). There has been an increased emphasis worldwide on using commercial forestry and agroforestry as a means of increasing carbon sinks (Brown, 1997; Trinus *et al.*, 1997). Additionally, Brown (1997) also recognises there are significant amounts of woody biomass in non-commercial lands.

Biomass is the term commonly applied to organic matter (oven dried to a constant weight) and measured as mass or weight. Traditionally forest biomass has been recorded for commercial and research purposes as a dryweight and has occasionally been determined as a 'wet weight' (Finlayson *et al.,* 1993). In such cases the results are not comparable with any other estimation due to the variation of moisture content of plants (which is species and seasonally dependent). Plants accumulate organic matter through growth by removing CO_2 from the atmosphere, utilising the carbon and releasing oxygen as a by-product. The approximate carbon content stored in a plant is about 50% of the plant biomass (Brown, 1997; National Greenhouse Gas Inventory Committee, 1997). Therefore forest carbon stocks can be determined by simply estimating the forest biomass, usually achieved by means of an allometric regression that relates an easily measured variable to aboveground biomass. The carbon sink within a forest is obtained by measuring the change in stocks over a nominated time interval.

Woody plant encroachment and vegetation thickening in savannas and woodlands have been reported worldwide. Woody biomass has increased following the introduction of domestic livestock into savannas and woodlands (Walker *et al.,* 1981; Tothill *et al.,* 1985; Archer *et al.,* 1995 and Van Auken, 2000). Furthermore, Pyne (1991) and Burrows *et al.* (1998) noted that the grazed woodlands in Australia are thickening and even encroaching in to previously treeless areas. This increase in vegetation (woody biomass) could play an important role in reducing the levels of greenhouse gases by acting as a carbon sink. The attraction of carbon sinks to achieve net emission reductions is that they could result in a lowering of economic costs to meet international commitments. Woodlands occupy a large proportion of the north-east Australia region and thus any carbon sequestration in this area would be considerable (Burrows *et al.,* 2002). However, an accurate measure of this sink is essential for any estimation of carbon sinks, and subsequent calculations of carbon fluxes.

2.4 Methods of Estimating Biomass in Tree Stands

Simple ocular and educated estimations were once the common method of measuring forest biomass stocks. These two methods are easily obtained although results can be variable if an operator isn't experienced, although some operators are surprisingly accurate (Brack, 1996). Modern methods commonly used to estimate biomass include stand table and mean tree values, volume equations, and randomised branch and importance sampling (West, 2004). Another accepted and widely used method is the application of allometric relationships to stand descriptors (dimensional analysis).

2.4.1 Volume equations

Foresters have traditionally used volume to estimate the merchantable timber within a trunk (Binkley *et al*., 1997). Most volume equations are used according to Homer's (1964) application for predicting tree volumes (cited by Parresol, 1999). However, volume equations are limited by some assumptions (such as a tree trunk which is considered to be a conical shape). In trees that have a uniform trunk taper, this assumption is valid (e.g. "pine trees" and some plantation species) and a reasonable estimate of trunk biomass is obtainable. For estimating total tree biomass, volume equations were adapted, by applying wood densities to volumes (which are measured as tonnes/m^3) and then applying an expansion factor to account for branches, bark and leaves to estimate total tree biomass. Expansion factors are commonly calculated by using the ratio of trunk volume biomass to total tree biomass. Although volume equations are readily available for commercial forestry species in Australia (Vanclay and Shepherd, 1983; Preston and Vanclay, 1988) they are rarely available for native woodland/savanna species.

Wood density data are lacking for many forest species of the world (Brown, 1997). However, information for Australian native species are well represented (Cause *et al.,* 1989; Boland *et al.,* 1992). Nevertheless, there is a dearth of available ratio/expansion factors for the estimation of total biomass for Australian species. Nearly all of the existing relationships have been developed for forestry or agroforestry plantations, and therefore ideally such data should not be applied to communities that are not forestry or agroforestry plantations (Brown, 1997).

2.4.2 Stand tables and mean tree relationships

Stand tables are used by foresters to summarise a sample population of a forest stand. Biomass relationships are applied to stand tables in the manner described by Brown (1997). This method is similar to the mean tree methodology used by researchers to estimate forest biomass (Attiwill and Ovington, 1968; Crow and Laidly, 1980) and assumes that the population can be represented by a sample within a size class.

Stand tables apply a biomass regression to an average tree that represents each size class (Brown, 1997), whereas the mean tree method harvests and calculates the dryweight for each sample tree within a size class (Harrington, 1979). The sample biomass is then multiplied by the number of trees in the respective size class. The dryweight sum of the size classes is the estimate for the sample total dryweight. For both methods, estimates become inaccurate when the size classes are large, or the sample within a size class is skewed. The method of selecting the tree to represent any size class is also important. Brown (1997) states that small and even diameter classes give the most unbiased results (specifically classes <10 cm) although, this method underestimates biomass, especially when large trees are in the stand (Brown, 1997). If tree form (diameter × height relationship) within a community is uniform and there are no other methods available then stand tables or mean tree methods can be satisfactory, and have been used in such circumstances for understorey species (Moore *et al.,* 1967; Burrows *et al.,* 2000). Westman and Rogers (1977) also used mean tree methods of similar form shrubs to estimate the biomass of those species for which they didn't have any values or samples. Again, the latter application was in the understorey, which comprised only a minor component of total community biomass. Hence, mean tree and stand table estimations have their problems, especially where assumptions are not correctly understood (Attiwill, 1966; Ovington *et al*., 1967).

Such concerns have led to allometric relationships being applied to individual trees in the stand (Brown, 1997). This should require no extra data collection (once such regressions are developed) as the stand table and mean tree summary are based on an inventory of the community. Thus, the data for individual trees are readily available. Computer technology now makes it simple to apply biomass regressions to individual tree predictor variables, and thus overcome most of the bias involved with stand tables estimates and mean tree approaches.

2.4.3 Allometric Relationships

Parresol (1999) defined allometry as "*the measure and study of growth or size of a part in relation to an entire organism*". The variation of these relationships relates to the natural variation of form within a plant species and/or genus (Attiwill, 1966; Keith *et al.,* 2000; Lott *et al.,* 2000). Allometric relationships are developed by the destructive harvesting of selected individuals within a range of dimensions. A conveniently measured predictor variable (e.g. stem diameter at 'breast height') is then regressed against the variable to be estimated (e.g. total above-ground biomass).

Harvested tree samples need to represent the population being studied, however because destructive sampling is costly in both time and labour, the size of the sample is often limited in range and numbers. One method to decrease the expense of destructive harvesting is to utilise a double regression technique, which does not require the destructive sampling of the whole tree (Harrington, 1979). In some cases destructive harvesting is not practical, such as in the cases of endangered species, so allometric relationships can be developed with selective measurements and restricted subsampling (Vann *et al.,* 1998). Often these methods can be tested on a similar species that are not endangered. Sampled data can be used to develop regression models which are usually applied to predictor variables in stand tables or for individual trees to estimate biomass.

2.4.4 Allometric models and variables

The tree attributes most commonly utilised as predictor variables include diameter at breast height over bark, tree height, stem circumference at 30 cm from ground, branch diameters and age. Diameter was reported to be an accurate variable to predict wet weight by Finlayson *et al.* (1993), Nelson *et al.* (1999) and Lott *et al.* (2000); but the latter suggested that cross sectional area (basal area) gave better results than other variables. Ibrahim (1995) measured branch basal diameter and observed that when a branch site was blocked by foliage then locating the branch after felling was difficult. Lott *et al.* (2000) also reported difficulties measuring individual branches. In eucalypt woodlands measuring high branches is impractical, and several branch diameters are needed for a single tree, however this variable would be ideal for coppicing plantations or regrowth as branching is usually close to ground level.

Regressions need to utilise a variable that is readily measured and commonly used (Keith *et al.,* 2000). However, Senelwa and Sims (1998) reported that there was a paucity of allometric relationships based on total dry mass and readily measured variables available, while Brown (1997) observed that there are very few relationships that encompass large ranges or sample numbers. Published regression models generally fit into two groups, linear or non-linear, and utilise different tree parameters (either singly or in combination) as predictor variables.

2.4.4.1 Linear additive

Allometric linear regressions have been developed using combinations of variables such as those developed by Madgwick and Satoo (1975), Feller (1980), Deans *et al.* (1996) and Senelwa and Sims (1998). Deans *et al.* (1996) reported clear linear relationships between D^2H and biomass for a general species equation of regeneration plots, but noted that trees with a small diameter at breast height were estimated to have a negative biomass. Parresol (1999) stated that the linear model was reliable, but only when there was a small range of tree sizes, such as those of plantation or regeneration plots. Parresol (1999) further noted that to fit biomass data to a linear relationship a weighted analysis was required, provided that all other regression assumptions are reached. However, Nelson *et al.* (1999) reported that tree biomass data doesn't comply with the assumptions for linear regressions, as the residuals are usually not random.

Baskerville (1972) suggested that regressions developed by avoiding the use of a logarithmic transformation "*…. may be dangerous when it leads to violation of necessary assumptions of regression analysis*". Therefore, linear relationships should be avoided on untransformed biomass data. Senelwa and Sims (1998) report a high correlation between stump basal area and tree biomass, as well as a "clear" curvi-linear relationship between stump diameter and tree dry weight. Nevertheless, these authors used a linear regression to avoid bias produced by back transformation of logarithmic values however, their best biomass predictor (D^2H) had errors up to 18% and larger trees were underestimated. Hierro *et al.* (2000) compared natural log-log models with power equations for eight species of shrubs and found the log-log transformation model gave the best fit.

2.4.4.2 Non linear

Non-linear regressions are usually transformed by taking the natural logarithm (ln) of both independent and dependent variables. This transformation ensures the independence of variance and the mean; it also ensures that the residuals are consistent with a normal distribution (Keith *et al.,* 2000). The variance becomes constant with increasing tree size and the parameters of the equation can be estimated by the least squares procedure (Parresol, 1999).

Many researchers have used a log transformation of data e.g. Westman and Rogers (1977), Harrington (1979), Bradstock (1981), Nelson *et al.* (1999) and Burrows *et al.* (2000). Both Parresol (1999) and Ibrahim (1995) report that a logarithmic transformation gave the best fit to biomass data when compared with linear models using the same variables.

Zar (1984) reported that simply taking the antilogarithmic transformation to obtain an arithmetic value of the dependent variable gives a biased estimate. This bias is a result of the transformation giving the geometric mean (median) instead of the arithmetic mean (Flewelling and Pienaar, 1981; Keith *et al.,* 2000). The necessity and method to correct the bias is acknowledged by many researchers, including Zar (1968), Madgwick and Satoo (1975), Senelwa and Sims (1998) and Eamus *et al.* (2000). Despite this knowledge, some researchers (Ward and Pickersgill, 1985; Finlayson *et al.,* 1993; Nelson *et al.,* 1999; Hierro *et al*., 2000) either avoid transformation or don't implement a correction.

2.4.5 Back transformation correction factors

Nelson *et al.* (1999) used transformed data and avoided the bias by not performing the back transformation, so weights are ln dryweight (kg), which has no useful meaning when determining biomass of a tree. Several different methods have been used to apply a correction factor to back transformed logarithmic yields, such as those applied by Baskerville (1972), Beauchamp and Olson (1973), Flewelling & Pienaar (1981), Snowdon (1991) and Snowdon (1992).

Baskerville (1972) suggests the use of his equation: $\hat{Y} = e^{(\mu + (\sigma^{\wedge}2)/2)}$, where $\hat{Y}$ = estimated arithmetic mean of Y distribution at X, $\sigma^{\wedge}2$ = variance of the regression equation, and μ = regression equation, $\ln(y) = a + b \ln(x)$. This correction is widely utilised and is simple to calculate. For example, Westman and Rogers (1977) adopted this correction for *E. signata* and similarly, Scanlan (1991) used this correction for *Acacia harpophylla*. But Parresol (1999) reports evidence that this correction tends to overestimate the true bias. Beauchamp and Olson (1973), Madgwick and Satoo (1975), Flewelling and Pienarr (1981) and Eamus *et al.* (2000) also reached the same conclusion. This bias is very small when variance is small, but the bias increases if variance is large, due to the bias being proportional to the variance (Snowdon, 1992). Nevertheless, if a fast, simple and easy to apply correction is required then the

Baskerville (1972) correction would suffice in most situations (Beauchamp and Olson, 1973; Flewelling and Pienaar, 1981; Eamus, 2000).

Another correction method, the ratio estimator, was developed by Snowdon (1991, 1992) and used by O'Brien *et al.* (1995) and Medhurst *et al.* (1999). This method is unbiased provided several assumptions are adhered to, including that the relationship passes through the origin. Most published relationships fail this assumption. Snowdon (1991) compared this method with Baskerville's (1972) simple correction and Finney's (1941) correction and reported the ratio method produced better results (providing his assumptions were adhered to). Flewelling and Pienaar (1981) reported that Finney's (1941) correction was unbiased, providing the slope co-efficient was known. Beauchamp and Olson (1973) suggest the use of a range of complex corrections based on Finney's (1941) methods. Although these equations are complex, modern day computer capacities would facilitate the solution of such equations (Flewelling and Pienaar, 1981; Brown, 1997) and this correction method is also preferred by Eamus *et al.* (2000). Burrows (1976) and Burrows *et al.* (2000) routinely used one of the Beauchamp and Olson (1973) corrections to correct estimated dryweights in their woodland studies.

2.4.6 Estimating biomass for Allometric regressions

2.4.6.1 Destructive Harvesting

Traditional destructive harvesting involves obtaining the fresh weight of trees, drying subsamples, calculating wet to dryweight ratios and applying these ratios to estimate the total dryweight of trees. Smaller trees are easier to measure, both total wet tree weight and components as entire trees can be measured (Parresol, 1999). However, as trees increase in size it becomes more expensive and less practical to dry the whole tree and it is more economical and sensible to take subsamples to estimate the total tree dryweight. The physical requirements of destructive harvesting have encouraged the development of newer methods for estimating tree dryweight, such as Randomised Branch Sampling.

2.4.6.2 Randomised Branch Sampling

Random branch sampling methodology uses ratio and probability theories to estimate tree biomass, and is often combined with importance sampling methodologies. This produces unbiased estimates and variances, provided that assumptions are not violated (Valentine *et al.,* 1984). Following the methodology of these authors probabilities are calculated with an initial ocular estimation of tree biomass. Therefore, two people estimating the same tree, if inexperienced, may have very different estimates of biomass. Tree biomass can be estimated from a single sampling procedure (Valentine *et al*., 1984) which reduces labour costs and time. If variance estimates are required then more than one sampling procedure is needed (Valentine *et al.,* 1984).

Valentine *et al.* (1984) and Gregoire *et al.* (1995) note that randomised branch sampling also gives an unbiased sample. To estimate biomass within an area using random branch sampling, many sample sites are needed together with a systematic sampling method to select sample trees. Gregoire *et al.* (1995) stated that '….. *the larger number of degrees of freedom, obtained because of the greater number of trees would reduce the mean square residual error of the regression, thereby increasing the precision of values predicted from it*'. This method works very well in forested areas where conditions for random branch sampling are ideal. However, Gregoire *et al.* (1995) report that this sampling would be less effective in areas where the pre-determined sampling unit is less than average inter tree distance. In areas such as open woodlands, a larger sampling unit would be required, thus covering a larger area of land. The larger area increases stratification problems, thus decreasing the number of trees sampled and reducing precision. When tested in the field on individual trees, the resultant biomass estimates produced errors of 5.6% to 14.4% of the actual weights (Valentine *et al.,* 1984). Additionally, disk measurements obtained as part of the randomised branch procedure and the cutting of subsamples, must be very accurate to prevent further increases of errors in biomass estimation.

2.4.6.3 Other Methods

Harrington and Johns (1990) used a technique which involved taking a fresh sample of a species and then use this as a ‘scale’ to approximate proportions taken up by actual species in quadrats. These fresh samples were then dried, weighed and the ratio applied to estimate the quadrat samples dryweight. Estimator bias was corrected by similar procedures to those adopted with Botanal (Tothill *et al*., 1992) which is used to measure pasture yield and species composition.

In some situations, volume equations have been used together with sampling to estimate biomass and develop relationships (Deans *et al.,* 1996). In the latter case small trees were harvested and sampled while the biomass of large trees was estimated by a combination of volume and estimation of branch biomass from cross-sectional area relationships. The component yields are then added to estimate total tree biomass and develop a relationship. Feller (1980) used a combination of ratio sampling and volume density to estimate weight for use in allometric equations.

The literature reports numerous methods to estimate plant biomass (Catchpole and Wheeler, 1992; Parresol, 1999). Some of these methods use complex procedures requiring numerous accurate measurements to determine biomass. This contrasts with minimal measurements required for destructive harvesting.

2.4.7 Variability

Regardless of the method used to estimate tree biomass, bias is introduced whenever destructive sampling occurs. Brown (1999), Parresol (1999) and Keith *et al.* (2000) all state that the trees selected for sampling should represent the whole of the population that is being measured. Research conducted on the biomass of eucalypts suggest only a limited range/number of trees have been harvested, e.g. see Ward and Pickersgill (1985), Westman and Rogers (1977), Bradstock (1981) and Ward and Koch (1996). Bradstock (1981) obtained a highly significant relationship after using a log transformation, but the sample number was small, less than seven. Likewise, while a "perfect" regression can be fitted to two points, this relationship may not represent the population from which it was sampled. Therefore, one should be very careful when applying regressions to predict biomass of trees outside the range (size class) for which they were originally developed. This point is emphasised by the fact that a regression of tree circumference (or dbh) against aboveground biomass commonly produces a positive exponential curve. Hence Brown (1999) suggests that if a predictor parameter for a tree exceeded the range of values in the regression then the actual biomass could deviate appreciably from the estimated biomass.

When using allometrics to estimate stand biomass, the parameters of the allometric regression need to be considered. It is also important to know that the allometric regression is representative of the data, in both size and form. Some harvest samples may cover a large range of the population, while other methods limit the sample by targeting the mean of a population (Deans *et al.,* 1996) and, thus harvest only a very limited number of trees. The characteristics of the harvested trees (i.e. the species, age, life form and environmental habitat) determine the range of trees that should be estimated by such data.

The applicability of an existing relationship can be increased by an additional sampling of trees outside of the size range from which the regression was derived, or by increasing samples across environmental gradients, or by including more variables (Lott *et al.,* 2000). However Ter-Mikaelian and Korzukhin (1997) report that the addition of extra variables didn't markedly improve the accuracy of their regressions. Another approach to improving predictions was to include the data for primary branches (basal diameters and the associated estimate of branch biomass) to increase sample numbers (Westman and Rogers, 1977). However, these samples were not independent, since they were from the same tree (branch diameter correlated to trunk diameter), and misleading results could be implied (Parresol, 1999).

Burrows *et al.* (2000) limited the size range of the data being applied to the regression to the size of the largest tree that was used to derive the regression. The biomass ascribed to those trees outside the range of the regression was constrained to that estimated for the largest tree of the data set. To avoid such problems it may have been an advantage to have fewer trees representing a wide range of class sizes when deriving regressions, than harvesting more samples representing a small range (Young, 1976 cited by Feller, 1980).

2.5 Using a common regression to estimate tree biomass

Ideally, a unique allometric regression for each species in each environment should be utilized, however developing species-site-specific relationships is both time consuming and very expensive. The overall costs may far outweigh the research benefits. To develop individual allometric relationships for every species of tree in Australia is a formidable task; however, by contrast, a single relationship to cover all forest trees in Australia is also impractical. The solution to determining community biomass lies between these extremes.

Two conflicting schools of thought exist about the use of allometric regressions. One school holds that allometric regressions should be used only on the same species within the original environment for which they were established. However, the second school proposes that allometric regressions can accurately predict the biomass of similar species outside the original environment for which they were established.

Biomass regressions developed for a given tree species either over or underestimate biomass when applied to another species (Brown, 1997; Keith *et al.,* 2000). The authors partly attribute this error to differences in tree form between the calibration and prediction species. Regressions derived from primary forests overestimate biomass when applied to secondary forests, by 10% to 60%, in the Amazon (Nelson *et al*., 1999). Brown (1997) notes that native woodland/forest trees are likely to have differing branch patterns than plantation trees and would probably have a higher biomass than similar sized (trunk diameter) plantation trees. O'Brien *et al.* (1995) suggests that canopy structures differed in closed and open canopy trees. Biomass relationships for trees in a closed canopy forest (plantations) were independent of site, age and silvicultural components, however, open canopy forests were dependent on these parameters (Medhurst *et al.,* 1999). Additionally, trees in stands have higher rates of height growth relative to circumference growth compared with trees in the open (Niklas, 1995). Therefore regressions developed for plantation stock should only be cautiously applied to open forest or woodlands of the same species. Such observations led Nelson *et al.* (1999) to suggest that regressions/allometric relationships should be restricted to tree species that have similar form or growth patterns, otherwise very significant errors can occur.

Ter-Mikaelian and Korzukhin (1997) determined three methods for using regressions not specific to a site: (i) choose regressions from a geographically similar site, (ii) use several estimates to obtain one estimate, or (iii) use several similar relationships to generate data then develop a new regression on the generated data. These methods were suggested to reduce errors associated with using allometrics outside the calibrated range.

Keith *et al.* (2000) believe that current allometric equations should be limited to the range of trees and the environments on which they were based on, and shouldn't be expected to apply to communities/stands outside these ranges (tree and environmental) without introducing some bias. Brown (1997) suggests that regressions can be used on sites differing from those on which they were developed, if there are no other data available, even so the estimated biomass needs to be treated cautiously. Keith *et al.* (2000) also stated that if the bias of a general equation is acceptable, then it can be used. Thus, when species-specific allometrics are available they should be used within the range of values on which the relationship was derived. For species that have no such relationship, equations for other species should be used carefully, and only where no other data are available. Using species-specific regressions is usually not possible in practice as it implies that allometric regressions should be developed for every ecosystem/community.

Medhurst *et al.* (1999) believe that a regression may be used outside of the original study area, but only after a validation is completed. Feller (1980) quoted it is of '*…doubtful validity to use, for a forest, regressions developed for a different forest…*'. This author also stated that any estimate is unlikely to be inaccurate if the range and form of species in the target community are similar to the range and form of species used to derive the relationship. A general equation may possibly suit a range of species of similar form, yet a general equation is clearly not as accurate as estimating tree biomass based on species-specific relationships (Senelwa and Sims, 1998; Vann *et al.,* 1998). Nevertheless O'Brien *et al.* (1995) reported that a common relationship can be used for more than one species provided the species have similar forms. In this case, they defined forms by using growth curves (tree age × height). Biomass regressions can be applied to communities without being too inaccurate provided the tree-form of the targeted species is the same as the tree-form of the species contributing to the biomass regression utilised (Feller, 1980). In these cases, it was assumed that tree-form will vary with environmental change, and that the accuracy of the biomass regressions in prediction will be effected by environment.

Brown (1997) and Hassal and Associates (1999) suggested that as rainfall increased in southern Australia so did potential biomass and tree circumference. Tree stem diameter increases with the application of fertilisers (Misra *et al.,* 1998) and forest biomass increases in stands with increased inherent soil fertility (Laurence *et al*., 1999). However, none of this supports the hypothesis that biomass increased for trees of the same circumference when grown in higher rainfall regions, or fertilised. It simply states that trees grow larger when watered or fertilised. For instance, increases in aboveground biomass, when irrigated stands are compared with non-irrigated stands, are due to increases in diameter and height growth rates, not from changes in tree allometric relationships (Reed and Tomé, 1998).

Diameter and height relationships are usually represented by diameter vs height regression lines on a log-log scale (Henry and Aarssen, 1999). Diameter-height regressions are steeper in young even-aged stands and become flatter as the stand ages (Eerikainen, 2002). However, it is not valid to compare the slope of diameter-height regressions of different height ranges (Henry and Aarssen, 1999) as diameter and height growth rates change as stands age (Niklas, 1995). Dominant height, or top height, of a stand can be modelled by height diameter relationships (Eerikainen, 2002) and can indicate the potential growth of a stand. Hence, dominant height is used as a measure of site quality (Eerikainen, 2002; Tewari *et al.*, 2002). Increased growth rates, in both diameter and height, were reported at better sites when compared with poorer sites (less rainfall) (Tewari *et al.*, 2002). Once again, this indicates that trees simply grow larger under more favourable environments and does not indicate that trees increase in height whilst circumference remains unchanged.

In contrast, biomass regressions have been found to be robust estimators (i.e. regressions are statistically strong enough to accurately estimate biomass) over a range of species in plantation and rainforest communities (Senelwa & Sims, 1998; Coomes & Grubb, 1998; Brown *et al.*, 1989; Chambers *et al.*, 2001). O'Brien *et al.* (1995) also suggest that an allometric regression can be used to estimate the biomass of several species. Within a limited environmental gradient, biomass regressions can be used to estimate biomass with accuracy. For example, Burrows *et al.* (2000) reported that biomass regressions for three eucalypt species in Central Queensland did not differ significantly from one another other; despite being developed for *E.*

populnea, *E. crebra* and *E. melanophloia* separately. Likewise, Hingston *et al.* (1981) determined no significant differences in the biomass regressions derived for two eucalypt species (*Eucalyptus marginata* and *E. calophylla*) in Western Australia. Senelwa and Sims (1998) stated that biomass allometric relationships within five species of eucalypt were not significantly different. Likewise Lott *et al.* (2000) noted that their species and site specific regressions weren't significantly different from a single species (combined) regression for *Grevillea robusta* stands.

A suite of allometric regressions has been developed for some Australian forest trees, and these have recently been collated in two reports (Eamus *et al*., 2000; Keith *et al.,* 2000) prepared for Australia's National Carbon Accounting Scheme (NCAS). However, the effect of using these existing allometrics to estimate aboveground biomass of stands with different species and environments is unknown. Some researchers believe that differences in tree-form result in an over-, or under-, estimation of aboveground biomass when regressions not developed for a specific stand are used. Tree-form is the relationship between diameter (or circumference) and height and is usually represented by a log-log scale (Henry and Aarssen, 1999). There is no evidence of changes in tree-form relationships due to increases in rainfall or fertility. Allometric biomass relationships have been found to be accurate over a range of species and communities throughout the world. Aboveground biomass regressions for some eucalypt species have been found to be statistically similar in limited environmental gradients. One common biomass regression for similar species can be used in similar locations; however, it is still not known if existing biomass regressions can be applied to the larger tree species mix and range in north-east Australia's woodlands.

2.6 Scaling Up (Stand Biomass)

One common method for estimating stand biomass is to sum the biomass estimates of individual trees within the stand of specified area. All methods for estimating biomass density must involve the estimation of individual trees and the summation of these amounts to acquire a stand biomass density (Parresol, 1999). When measuring biomass density, there is an assumption that the plots used to estimate biomass are an accurate sample of the strata they represent. However, plots are often placed with a bias towards what ecologists believe to be a 'mature' stand (Brown, 1997). Madgwick and Satoo (1975) report less variability in plantation stands than in natural stands, an observation also supported by Brown (1997) and, not surprising, as plantation stands are usually even-aged stands that have been managed to ensure tree size is maximised and relatively uniform throughout the stand.

Parresol (1999) suggests that errors can occur in the sampling of any forest and can be influenced by sample size, sampling methodology and the variables measured, as well as the natural variation between the trees within the stand. For similar reasons, Baker *et al.* (1984) state that a continental relationship to estimate biomass for *Pinus radiata* was not viable, however the use of a regional based relationship was viable. Keith *et al.* (2000) suggest that the development of regional relationships may avoid allometric relationships being used for purposes other than those for which they were originally developed.

Vegetation often changes with varying landscape, and this heterogeneity can be a source of errors in regional biomass estimation. If heterogeneous regions are subdivided into homogeneous strata, then the sampling efficiency of the inventory will increase (Brack, 1996). Just as individual trees should be representative of a population, the sample plots selected must be representative of the region it represents. This is achieved by having an acceptable number and distribution of sample plots that are based on a statistically sound sample selection. Errors in biomass inventory estimations can result from a biased estimation method, e.g. the biomass relationships, or from inaccurate forest inventory. Additionally, care should be taken to ensure that large trees are measured as accurately as possible (especially in mature forests) as they contribute a larger proportion of biomass on a site than the smaller trees (Brown, 1997).

Regressions should use predictor variables that are readily measured and used in most forest inventories (Keith *et al.,* 2000) to facilitate their widest application. The use of pre-existing biomass regressions reported in the literature can reduce costs and time. However, this assumes that the regressions used are derived from trees similar to trees within the forest inventory that it is being applied to, otherwise errors can be substantial. Above-ground carbon stocks of forests are readily determined as forest inventories have been reported for large tracts of commercial forests.

Most Australian inventories and allometric data have been developed for commercial forestry plots, however many of the woodlands and savanna areas are not monitored by foresters. Brown (1997) believes that the general lack of inventory data for open forests and woodlands is because they have limited commercial value, and there is a need to develop both inventories and allometric relationships for these communities. Keith *et al.* (2000) also recognise a need to develop an inventory and allometric relationships for non-commercial forest types.

An inventory of the grazed mature eucalypt woodlands in north-east Australia was established by QDPI&F for which some allometric relationships have been developed. QDPI&F maintains a geographically diverse network of permanent monitoring sites throughout Queensland's woodlands. A detailed data set for over 100 sites is maintained for communities in north-east Australia. These monitoring sites are recorded by the Transect Recording And Processing System (TRAPS) (Back *et al*., 1997; Back *et al.,* 1999) and have been used to investigate population dynamics (Back, 2005). Circumference data recorded using TRAPS are measured in 1 mm increments, with each tree relocatable at successive recordings. Thus the data set is sensitive to the smallest basal area change and will also demonstrate clear evidence of population changes. A subset of this data set has undergone several tests of representativeness (Burrows *et al.,* 2002) and were reported to be an acceptable representation of Queensland's eucalypt woodlands across rainfall, temperature and soil types.

Above-ground carbon stocks are reported to be dependent on the successional stage of the stand (Brown, 1997; Clark and Clark, 2000; Hoshizaki *et al*., 2004). Additionally, stand composition and how components alter with disturbances over time (natural and anthropogenic) can influence the accuracy of biomass estimation (Brown, 1997). Also, trees are not necessarily the only vegetation that contributes to savanna, woodland or forest community biomass. Woody understorey species can make a significant contribution to total community biomass. Grove and Malajczuk (1985) suggest that although understorey varies at sites, it can contribute a 'significant proportion' of above ground biomass. Ward and Pickersgill (1985) reported similar biomass at two sites, but the distribution of the biomass differed. One site had a significant amount of understorey, and the other site had relatively little understorey.

This is also noted by Brown (1997), who suggested that ignoring the understorey biomass can underestimate total above-ground biomass by up to 50%. Consequently, stand components vary with stand type and within stand types, often due to disturbances (natural and anthropogenic) and different management regimes.

In an earlier study (Burrows *et al.,* 2000), allometric regressions were used to develop a ratio between stand basal areas and stand biomass. However, this study was limited to sites that were dominated by *Eucalyptus* spp., of known stand history, such that each community had not been disturbed in the 30 years previous to monitoring. The effects of differing stand histories on total aboveground biomass are unknown and hence, a reliable method of determining stand structure in the north-eastern woodlands of Australia is required.

2.7 Old growth forest biomass

Forest systems have been studied for many decades (de Liocourt, 1898 cited by Leak, 1964; Harper, 1977; Oliver and Larson, 1990; Florence, 1996). One assumption is that forests achieve a state of equilibrium, in which new individuals replace dead individuals in a continuous cycle (Leak, 1964; Oliver and Larson, 1990; Kohyama, 1991). These forests are described as balanced uneven-aged forests, or old growth forests, and are defined as having a diameter size distribution, or volume distribution, that is essentially constant over long time periods (Leak, 1964). Any changes to mortality and/or growth rates, caused by disturbances, will alter the size class distribution and until a new equilibrium is re-established the forest is described as unbalanced (Leak, 1964).

A second assumption is that forest and woodland populations often existed in a stable state prior to European settlement, such as the North American forests (Oliver and Larson, 1990). However, most old-growth forests have developed from one extensive recruitment event that occurred after a past catastrophic disturbance (Harper, 1977). Once established, the old-growth forests have been maintained by smaller disturbances, both natural and anthropogenic, influencing mortality and/or recruitment rates (Oliver and Larson, 1990). In eucalypt forests, this type of forest development can be cyclic over hundreds of years, and usually consists of an ‘upgrade phase’ and ‘downgrade phase’ (Florence, 1996). As stand age increases, during the upgrade phase, the total stand biomass also increases. Then, as the stand matures, stand biomass asymptotes and then biomass decreases, as the stand declines during the downgrade phase (Hoshizaki *et al.,* 2004; Shibuya *et al.,* 2004). However, recent studies (Hoshizaki *et al.,* 2004) report that some forests, which were previously thought to be mature and stable, have been increasing in biomass over the last few decades. Hence, forests that appear to be at a climax may actually be in an upgrade phase, indicating that the point of asymptote is not readily determined by short term studies.

Old-growth forests have relatively few large trees with an increasing number of smaller trees (Brown *et al*., 1997; Poage and Tappeiner, 2005; Zenner, 2005). In terms of biomass, these few large individuals contribute to the majority of stand biomass (Weiner and Solbrig, 1984; Hutchings, 1986 cited by Rouvinen and Kuuluvainen, 2005; Brown *et al*., 1997). Additionally, the death of a single large tree in an old-growth stand will have a greater effect on total stand biomass than the death of several smaller trees, and conversely, the death of a single large tree in a regrowth stand, (where there are more large trees) will have little effect on stand biomass

(Hoshizaki *et al.,* 2004). Therefore, the contribution of large trees to aboveground biomass can be variable in different forest systems (Cummings *et al.,* 2002).

2.8 Forest Structure

Forest populations are described in numerous ways, such as size class distribution, succession stages and development stage or stand structure (Zenner, 2005). Weiner and Solbrig (1984) define size hierarchy as the frequency distribution of individual plant sizes where a few large individuals contribute the most of the population's biomass and most of the individuals are relatively small. They prefer to use this term to describe the stand structure of forests. Age class distributions have also been used to describe stand structure (Hitimana *et al.,* 2004; Shibuya *et al.,* 2004), however, the measurement and interpretation of age is precarious in uneven-aged stands (Moser, 1972). Tree rings are an indicator of growing season (wet season) rather than chronological year (Eshete and Stahl, 1999). The ability to count tree rings to assume age is not accurate in African savannas (Walker *et al.,* 1986; Lilly, 1977 cited by Brown and Bredenkamp, 2004) and extreme weather events, such as droughts, can affect the presence of tree rings (Detienne, 1989 cited by Eshete and Stahl, 1999; Stokes and Smiley, 1968 cited by Eshete and Stahl, 1999).

In Australian woodlands and savannas, dry seasons can be extended over several years in extreme droughts thus making age difficult to estimate from tree rings. Additionally, the age of seedlings and saplings are difficult to assess in American rangelands (Landis and Bailey, 2005). This is due to the ability of seedlings to survive for many years as suppressed seedlings, and evidence from the grazed woodlands of Queensland indicate that some individuals have persisted for over 20 years (P.V. Back pers comm., 2005). Tree diameters have substantial phenotypic plasticity, allowing a suppressed individual to respond when suppression is released (Knowles and Grant, 1983). This trait has been noticed in several forest communities. For example, pine seedlings were suppressed in a grazed environment for many years, however once grazing was removed the seedlings were released from suppression and became a source of very rapid sapling regeneration (Darwin, 1859 cited by Harper, 1977). Similarly, spruce seedlings remained suppressed for periods of 20-30 years while the canopy of mature trees remains in tact. Spruce budworm damage to mature trees released suppressed seedlings which then rapidly grow to re-establish canopy cover (Harper, 1977). In mature and collapsing stands, emergent seedling density increases (Shibuya *et al.,* 2004) as competition from senescing trees decreases and the long established seedlings are being released from competition.

The most popular method for describing the properties of a forest stand is by analysing the diameter size class distribution, which is more commonly known as size class distribution (De Liocourt, 1898, cited by Goodburn and Lorimer, 1999; Leak, 1964; Bailey and Dell, 1973; Moser, 1976; Harper, 1977; Walker *et al.,* 1986; Knowles and Grant, 1983; Vanclay, 1995; Rouvinen and Kuuluvainen, 2005). A size class distribution is the relationship describing the frequency of trees within a diameter class plotted over the mean of each diameter class (Leak, 1964; Schmelz and Lindsey, 1965; Moser, 1976). Guidelines for developing balanced stand structures in managed forests are based on the diameter distribution curves of old-growth stands (Goodburn and Lorimer, 1999). Uneven-aged forests across the world have been described as having a size class distribution resembling a “reversed-J” shape curve or negative exponential curve (Leak, 1964; Schmelz and Lindsey, 1965; Moser, 1976; Murphy and Farrar, 1982). Similarly, age class distributions of uneven-age forests also resemble a “reverse-J curve” (Hitimana *et al.,* 2004). Conversely, even-aged stands (plantations or stands that have had a recent catastrophic disturbance) are reported to have a normally distributed size class distribution (Schmelz and Lindsey, 1965). The “reverse-J” curve size class distribution is also typical of species that have a good rejuvenation rate and hence, a continuous replacement within the stand (Lykke, 1998).

Size class distributions are also utilised to develop stand tables, which predict the number of individuals within a size class (Leak, 1964; Moser, 1976) and thus determine stand volumes of commercial forestry stands. Murphy and Farrar (1982) further developed the size class distribution by calculating class basal area using a “*doubly truncated exponential probability density function*”. The limits for each size class enabled the authors to increase the accuracy of the estimated basal area for each size class, rather than simply multiplying the basal area (calculated from the size class midpoint) by the number of trees in the class. Murphy and Farrar’s (1982) calculation is complex and recent literature indicates that most prefer the traditional size class distribution (Lykke, 1998; Hitimana *et al.,* 2004; Rouvinen and Kuuluvainen, 2005).

2.9 Size Class Distribution Curves

The size class distributions, of both balanced and unbalanced stands, have been described in numerous ways, such as reverse-j curves (Leak, 1965), negative power curves (Muller, 1982) or combinations of these curves (Coomes *et al.,* 2003). Reverse-j curves, or negatively exponential curves, are reported to be typical of uneven-aged stands (Leak, 1965). In these stands the number of plants in each diameter class decreases as the diameter classes increases. It is recommended that diameter classes are kept small to ensure that diameter is measured on an essentially continuous scale. This allows the relationship between diameter size class and frequency to be expressed as a natural logarithmic function (Leak, 1965). Reverse-J curves assume a constant mortality rate (Leak, 1965; Meyer and Stevenson, 1943 cited by Muller, 1982; Goodburn and Lorimer, 1999) and some researchers prefer to use the negative power curve, which assumes a declining mortality with increasing size (Muller, 1982).

Within a single species population, the decreasing plant density within size classes can be used to predict the increasing biomass within size classes (Pickard, 1983); this is also known as 'self-thinning rule'. This relationship was further developed, so that plant density scales as the -2 power of the diameter size class, or -3/4 power of aboveground biomass (Enquist and Niklas, 2001). However, size class distributions often vary among communities and correlate with community plant density (Niklas *et al.,* 2003). The scaling factor (-2 power) barely differs with changing species diversity, stand biomass, latitude and geographic sampling areas (Enquist and Niklas, 2001; Niklas *et al.,* 2003). A later study reports that the scaling factor can vary. A departure from the -2 scaling factor indicates those communities that are either under utilising the available space (low plant density) or over utilising available space (high plant density) (Niklas *et al.,* 2003).

Both functions, the scaling factor and the reverse-J curve, can be used to describe different sized stems within size class distributions. The scaling factor of Enquist and Niklas (2001) accurately describes smaller stems, while the more traditionally used reverse-J curve accurately describes the larger stems in mixed species uneven-aged stands (Coomes *et al.,* 2003). However, in that study it was reported that browsing by feral deer resulted in a reduction in recruitment (Coomes *et al.,* 2003), which would be likely to have an influence on the size class distribution curves.

The use of negative exponential sine curves have been suggested by Hett and Loucks (1976), which appear to have the best fit in old-growth stands when compared with both reverse-j curves and negative power functions. However, in this study, the distribution investigated was an age-class distribution, and it was noted that more data were required for a complete analysis (Hett and Loucks, 1976).

Different parameters of size class distribution curves have also been used to describe the structure of forest communities. Large variations in the slope of size class distributions are common between different communities (Pickard, 1983; Niklas *et al.,* 2003). Differences in slope can indicate the size of individuals dominating a stand (Niklas *et al.,* 2003), or the rejuvenation status of a species (Lykke, 1998). Survival and growth rates of stands have also been described by size class distributions (Condit *et al.,* 1998). Variations in the slope of size class distributions are reported to correlate with community plant density rather than stand structure (Niklas *et al.,* 2003). The appearance that a wide range of stand characteristics can be described by a single size class distribution appears to be unrealistic. Size class distributions describe a single parameter of a stand at a single point in time, however stand structure is influenced by many dynamic factors, such as recruitment, growth rates and mortality (Coomes *et al.,* 2003).

2.10 Changes in Size Class Distributions

Communities can have large variations in structure due to differing responses to disturbances, such as fire and harvesting (Talamo and Caziani, 2003; Brown and

Bredenkamp, 2004), especially in multi-use woodlands and savannas where a number of disturbances can occur to the same community. Some species have very rapid establishment events after disturbances, while other species build up a population of seedlings over a number of years which respond after disturbances (Harper, 1977; Florence, 1996). Alternatively, species that are long-lived maintain stable populations without regular recruitment events (Condit *et al.,* 1998). For example, the dry tropical forests of India maintain populations through irregular recruitment and have few juveniles (Sukumar *et al.,* 1992 cited by Condit *et al.,* 1998). Similarly, Australian eucalypt seed banks and newly germinated seedlings are very sensitive to a range of environmental conditions and survival rates can be very low (Florence, 1996). Under ideal conditions, seeds will germinate rapidly and seedlings will grow vigorously (Florence, 1996). Eucalypt woodland communities in north east Australia are maintained by sporadic recruitment events. These characteristics of different population strategies can influence changes in size class distributions in different ways.

Regeneration, or release from suppression events, appears within size class distributions as pulses, or the curve appears bi-modal (Landis and Bailey, 2005; Rouvinen and Kuuluvainen, 2005). However, a bi-modal curve can also represent a mature stand in mixed species communities where species have very different growth and recruitment rates (Hitimana *et al.,* 2004), especially if one species of the community belongs to the understorey, or is suppressed (Goodburn and Lormier, 1999). Size class distributions have been used an indication of past disturbances (Schmelz and Lindsey, 1965) and to assess disturbance effects (Hitimana *et al.,* 2004) within forests because they are altered by environmental factors, both natural and man-made. However, disturbance has also been indicated by average stem diameter, which is skewed to the right in older stands, and to the left for younger stands (Niklas *et al.,* 2003).

In contrast, regrowth stands have been reported not to demonstrate a normal distribution curve that typically develops after a disturbance (Muller, 1982). The basal area of both, old growth and regrowth stands, did not differ significantly after 35 years (Muller, 1982); in addition species composition and size class distributions were similar in these old growth and regrowth stands (Muller, 1982). Despite significantly different densities between regrowth and old growth forests after 10 years, the basal areas of each stand were not significantly different (Talamo and Caziani, 2003). However, the distribution of the basal area between the two stands was different; the regrowth forest had a greater proportion of sapling basal area than the old growth forest (Talamo and Caziani, 2003). Additionally, differences in sapling densities of regrowth and old growth stands were no longer evident after 25 years (Talamo and Caziani, 2003). Although complete recovery of old-growth forest stands to catastrophic disturbances is slow (Talamo and Cazianai, 2003), the evidence of past disturbance, such as sapling densities and size class distributions, can be masked between 10-25 years of regrowth.

Different population strategies can influence size class distributions in many ways due to different responses to disturbances. Traditional size class distributions can be interpreted in numerous ways, so that the same distribution can represent very different stand structures in communities with different population strategies or stand disturbance histories. Other methods of describing stand structures in forests, for example, average stem diameter, have been used successfully. Identifying stand disturbances from single size class distributions and densities appears unreliable as regrowth can be masked after 10-25 years of regrowth.

2.11 Application of size class distributions to determine stand structure changes.

Size class distributions of mixed-species communities can often be misinterpreted and produce ambiguous conclusions about stand structural changes. This is especially true for even-aged stands, where pooled size distributions often approach a reverse-J curve (Hough, 1932 cited by Goodburn and Lorimer, 1999; Oliver, 1978 cited by Goodburn and Lorimer, 1999; Muller, 1982). Similarly, even-aged stands with suppressed trees can also have a size distribution approaching a reverse-J curve (Goodburn and Lorimer, 1999) and some mixed-species communities are not uni-modal and are difficult to describe by standard distribution functions (Zhao *et al.,* 2004), usually as a result of different regeneration and growth rates of different species (Hitimana *et al.,* 2004). Hence, size distributions of individual species should be examined separately (Condit *et al.,* 1998; Goodburn and Lorimer, 1999; Hitimana *et al.,* 2004)

Reverse-j and negative power curves have been fitted to old-growth, regrowth, and mixed species stands (Muller, 1982). Reverse-J curves have also been fitted to even-aged mixed species stands and stands that have been heavily logged in the early 1900's (Goodburn and Lorimer, 1999). Other factors besides population health affect size class distributions (Condit *et al.* 1998). Stands with very different species diversity, with each species having very different densities and biomass, can have very similar size class distributions and total standing biomass (Enquist and Niklas, 2001). The shape of size class distributions in determining past disturbances and stand dynamics is not without some reservations (Condit *et al.,* 1998; Niklas *et al.,* 2003).

Old-growth communities can be detected by the presence of some large old individuals, as no recent catastrophic disturbance has removed them from the communities. However, changes in the height of plants dominating a stand may also indicate past disturbances, as previously suppressed seedlings and young saplings are released, just as changes in diameter size distributions can indicate seedling release events (Landis and Bailey, 2005). Rouvenen and Kuuluvainen (2005) reported that, in natural forests, diameter classes of large trees have low densities, making it difficult to characterise stands by diameter size class distribution functions. This would be exaggerated in the woodlands, which are considered to be more open in structure, having less trees contributing to canopy cover than forests. Moreover, increasing class sizes is an inappropriate method of increasing the number of plants within each diameter class, as size class increments are required to be small to allow analysis of distributions curves, which assume that data are continuous (Leak, 1965). Brown and Bredenkamp (2004) utilised three height classes to successfully identify structural classes of woody species in South Africa. Hence, the application of height classes as an indicator of stand structure in eucalypt communities needs to be assessed and the subsequent effect of woodland structure on basal area to stand biomass ratios investigated in the eucalypt woodlands of north-east Australia.

2.12 Summary

Increases in woody vegetation may play an important role in reducing greenhouse gas levels by acting as a carbon sink. However, accurate measures of any potential sink are essential for calculations of carbon fluxes. There are few allometric regressions predicting aboveground biomass for Australian woodland species such as *C. glaucophylla*.

The use of common biomass regressions is documented worldwide for plantation and rainforest communities. Common biomass regressions have also been demonstrated for a limited range of species and environments in Australia. However, it is unknown if common biomass regressions are robust enough to be applied to the larger tree species mix and range in north-east Australia's woodlands.

Australian woodlands are diverse and variable, and have been subject to a range of management regimes since European settlement. Traditional forestry techniques for quantifying stand structure appear inappropriate for use in Australian woodlands. Additionally, the effects of differing stand structures on total aboveground biomass are unknown and hence, a reliable method of determining stand structure in the north-eastern woodlands of Australia is required.

Chapter 3. Above-ground standing biomass in *C. glaucophylla* communities of south west Queensland.

3.1 Introduction

Callitris glaucophylla is a native conifer used for commercial timber production and is distributed discontinuously through Eastern Australia. It is usually found on well drained soils especially sands in semi arid areas. Since European settlement, changes in land management (particularly reduced fire incidence) have enabled *C. glaucophylla* to encroach and thicken into grasslands, other native forests and woodlands. This ongoing thickening suggests that *C. glaucophylla* is potentially a carbon sink.

An accurate measure of biomass is essential for any estimation of carbon sinks and fluxes. Allometric regressions have been used worldwide for more than 50 years to estimate the aboveground components of tree biomass using an easily measured tree component as the predictor variable. However, allometric regressions are lacking for *C. glaucophylla* in eastern Australian woodlands.

One common method for estimating total stand biomass is to sum the biomass estimates of individual trees within the stand of specified area. All methods for estimating biomass density must involve the estimation of individual trees and the summation of these amounts to acquire a stand biomass density (Parresol, 1999). This can be tedious and time consuming when estimating stand biomass for large areas, or numerous sites. Burrows *et al.* (2000), developed a ratio between stand basal area and stand biomass using allometric regressions. This enabled stand biomass estimations to be determined quickly and accurately. Burrows *et al.* (2002) further developed this relationship to determine carbon fluxes in eucalypt woodland communities.

The aim of this chapter[3] was to determine and compare a suite of allometric regressions for native *C. glaucophylla* and to then use these regressions to develop stand based biomass relationships for *C. glaucophylla* communities.

3.2 Materials and Methods

3.2.1 Location of sites

The study area was located over an extensive area of *C. glaucophylla* woodlands and included State Forests, Leasehold and Freehold land in the Mitchell district (approximately 25° 40'S latitude and 147° 28'E longitude). The Queensland Department of Primary Industries and Fisheries' Forestry Site, Hillside, was included within the study area and is located approximately 125 km north west of Mitchell, and 80 km east of Augathella (Figure 3-1). In 1979, the area adjoining Hillside was a virgin stand (Dale, 1979), since then normal forestry management practices have commenced, such as thinning stands and harvesting. The culling of trees in excess of 80 cm diameter over bark is practiced as the commercial wood quality of these individuals is normally unacceptable. However the Queensland Department of Primary Industries Forestry limit *C. glaucophylla* to 40 cm Diameter Breast Height Over Bark (DBHOB) which is equivalent to approximately 125.66 cm circumference at breast height (Johnston, 1975).

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[3] The harvest data in Chapter 3 was published in Burrows *et al*. (2001). See Publications (page xii) for reference.

Two data sets of *C. glaucophylla* communities were used in this study to investigate aboveground biomass. The first data set was comprised of communities in the Hillside study area, covering approximately 40 km × 60 km. There were 150 primary plots (500 m × 150 m) arranged in a 4 km sampling grid across this study area established by a team led by Dr Richard Lucas, University of New South Wales. A sub-sample of the primary plots which comprised of 25 secondary plots (50 m × 50 m) are presented in the present study. Data were recorded for every tree in secondary plots (tree species, location within plot, circumference at 30 cm above ground, circumference at breast height (1.3 m above ground), height and canopy dimensions).

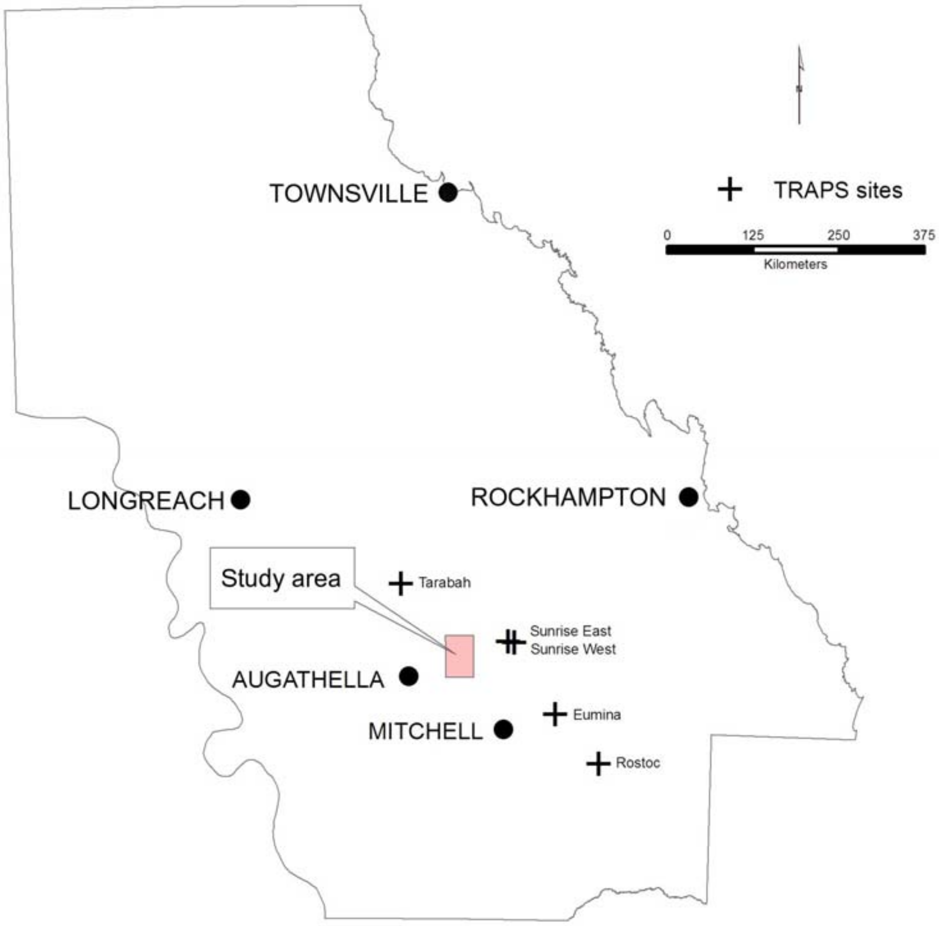

Figure 3-1. Distribution map of *Callitris glaucophylla* sites throughout Queensland. Shaded area represents the Hillside study area and crosses represent TRAPS monitoring sites from QPDI&F's permanent monitoring network.

The second data set encompassed five communities from the Department of Primary Industries and Fisheries (DPI&F) permanent monitoring network (the Transect Recording And Processing System (TRAPS) data set). These TRAPS sites were included to increase the geographical range of the study communities. TRAPS sites are located throughout Queensland's grazed woodlands (Burrows *et al*. 2002) and consist of a sample area of 0.2 ha, comprising of five 100 m x 4 m permanent transects located 25 m apart, where a detailed community inventory is recorded. These additional sites increased the total number of censused sites to 30 located across southern Queensland (Figure 3-1).

3.2.2 C. glaucophylla harvesting

The harvest sites for the *C. glaucophylla* were located on a deep sandy soil, adjacent to two of the secondary plots (C01, C02). Methods for selection and harvesting of trees were similar to those outlined by Burrows *et al.* (2000). Twenty *C. glaucophylla* trees were harvested ranging in circumference from 7.5 – 135 cm. Tree stem circumferences selected for harvest were chosen in a stratified manner to cover the range of circumferences recorded from the plot stand analysis. Each tree harvested was determined by visual assessment to be of 'average' form, with respect to shape, vigour, foliage cover etc for its particular size class.

Harvest tree characteristics were measured (height, circumference at 30 cm, circumference at 1.30 m and canopy) before felling and each tree was individually numbered and tagged. Tarpaulins and hessian were spread over the predicted fall area to minimise the loss of leaf, small stem and fruit and to isolate tree components from the ground litter. Stumps were re-cut as close as practically possible to ground level after felling.

The felled tree was then sorted on the tarpaulins into the following compartments:

- Leaf (included both leaf and photosynthetic branch tips)
- Live branches (<1 cm, 1-4 cm, 4-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, etc)
- Live trunk (<1 cm, 1-4 cm, 4-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, etc)
- Dead branches
- Capsules

Fresh weight of tree compartments were weighed immediately in the field. Mass of the components determined how each component was weighed in the field. Small samples were measured using electronic scales with a portable power source, medium samples required a tripod and tarpaulin set up with spring scales (Figure 3-2), while large samples were weighed on a set of portable cattle scales.

Sub-samples were taken for moisture determination and re-weighed immediately. However, if the sample was small enough, the whole compartment was taken as a sample. Sub-samples of each components were returned to laboratory for drying at approximately 65°C in a dehydrator until constant weight was obtained over consecutive days.

Figure 3-2 Equipment used to harvest trees and measure light and medium size biomass components – example is from a eucalypt woodland stand.

3.2.3 Treatment of the data

Number and circumference at 30cm from ground level data were used to estimate basal area (m^2) per hectare for each sub-plot. The sites extracted from the Queensland Department of Primary Industries TRAPS database were analysed by the use of the TRAPS program (Back *et al.,* 1997).

Wet to dry ratios of the sub-samples were determined using field and laboratory data. Wet to dry ratios were applied to wet component weights to determine component dryweight and consequently total tree dryweight. Regressions between dryweight and the four independent variables were investigated[4]. The dryweight data were transformed by taking the natural logarithm and these regressions were similarly investigated.

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[4] All regression and statistical analysis in this thesis used Genstat for Windows 6th Edition (Chapter 3) or 8th Edition (Chapter 4 & 5), (Copyright 2005, Lawes Agricultural Trust (Rothamsted Experimental Station). Default probabilities were accepted.

The lognormal regressions had the form:

- $\ln y = a + b \ln x$, where y = tree component biomass, (leaf, stem, etc.) and total tree biomass, x = independent variable, a = intercept, and b = slope. The independent variables were represented by:
- trunk circumference at 30 cm from ground level (circ30) (cm),
- trunk circumference at breast height, 130 cm from ground level (circ130) (cm),
- perpendicular height of tree (H) (m), and
- canopy area (CA - length and width of canopy used to calculate area) (m^2).

The circ30 regression was then applied to individual tree data from the 30 censused sites (Hillside and TRAPS sites). However, communities that were not managed for timber had not been subjected to the forestry practice of culling larger trees and therefore, consisted of trees outside the range of the allometric relationships. These trees were limited to the maximum circumference measured for the harvested regression trees, i.e. 135 cm to ensure that the results were not exaggerated. Estimates of biomass for these trees were underestimated and the number of large trees within monitoring sites will influence the amount of bias. This bias was considered to be small, as the majority of trees were less than 135cm.

In estimating biomass, the back transformation of log values to ‘real’ values by taking the antilogarithm of the predicted values introduces a bias in the estimates. This occurs because the geometric mean is obtained, not the arithmetic mean (Munro, 1974, Parresol, 1999), and a correction factor needs to be applied to correct this bias (see Baskerville, 1972 and Beauchamp and Olson, 1973). The method outlined by

Beauchamp and Olson (1973) was adopted to correct the bias. A database was developed to estimate biomass and apply the correction factor and was tested using the data set in the Beauchamp and Olson (1973) study.

A linear regression was established between stand basal area at 30cm from ground level and stand aboveground biomass utilising the data for the 30 censused sites. This leads to the development of standing aboveground biomass to stand basal area ratios. [Stand biomass (t/ha): stand basal area (m^2/ha)]. These ratios can then be applied to any *C. glaucophylla* stand of known basal area to estimate its standing biomass.

3.3 Results

3.3.1 Site Characteristics

Density of the censused sites ranged from 192 to 5688 plants/ha, with an average density of 2104 plants/ha (Table 3-1). Basal area of the censused sites ranged from 4.755 to 44.340 m^2/ha, with an average basal area of 19.630 m^2/ha. Density is presented as number of stems per site and number of stems per hectare. Basal area (m^2) measured at 30 cm from ground level is expressed in a similar manner. Population stand data for the 30 censused plots in the Mitchell district are presented in Appendix 8-1. Stand dominance was determined using the population stand data.

A *C. glaucophylla* dominant site was defined by *C. glaucophylla* contributing the largest percentage of total site basal area by at least 25% (e.g. 60% *C. glaucophylla* and 30% eucalypt). However, if there was less than 25% difference (such as 60% and 38%, or 45% and 30% *C. glaucophylla* and eucalypt respectively) the site was classified as co-dominant. A *C. glaucophylla* sub-dominant site was defined by a contribution by *C. glaucophylla* of less than 25% site basal area while still providing the second largest percentage (e.g. 70% eucalypt and 20% *C. glaucophylla*).

Table 3-1 Density and basal area of censused sites. Sites C01-C25 had an area of 0.25ha (50m × 50m), and sites C26-C30 had an area of 0.2ha (5 × 100 m × 4 m).

Censused Site	Plot plant numbers	Density (plants/ha)	Plot Basal Area(m^2)	Basal Area (m^2/ha)
C01	1020	4080	5.414	21.656
C02	459	1836	2.257	9.028
C03	1116	4464	5.065	20.260
C04	1422	5688	7.293	29.172
C05	277	1108	3.257	13.028
C06	1296	5184	8.728	34.912
C07	213	852	3.303	13.212
C08	854	3416	5.005	20.020
C09	148	592	4.822	19.288
C10	443	1772	5.187	20.748
C11	156	624	5.334	21.336
C12	1088	4352	2.611	10.444
C13	202	808	4.250	17.000
C14	818	3272	11.085	44.340
C15	322	1288	7.202	28.808
C16	820	3280	3.342	25.368
C17	603	2412	5.831	23.324
C18	639	2556	4.823	19.292
C19	205	820	4.584	18.336
C20	48	192	4.156	16.624
C21	90	360	2.725	10.900
C22	68	272	3.562	14.248
C23	259	1036	3.677	14.708
C24	99	396	5.445	21.780
C25	493	1972	5.158	20.632
C26	241	2410	3.605	36.05
C27	581	1935	5.707	19.004
C28	303	2524	1.773	14.769
C29	373	1865	0.951	4.755
C30	350	1750	1.171	5.855
Average	500	2104	4.577	19.630
Minimum	48	192	0.951	4.755
Maximum	1422	5688	11.085	44.340

3.3.2 Individual tree relationships

Data from tree harvesting were used to develop regressions to estimate individual tree biomass. The exponential relationship between tree circumference at 30 cm aboveground level (circ30)(cm) and total aboveground biomass (kg) Figure 3-3) represented 97.7% of the harvested sample of 20 individuals, however the residuals for this relationship were not randomly distributed (Figure 3-4), especially for the smaller tree sizes. The residual distribution pattern indicated that a logarithmic transformation should be performed (Zar, 1984). A very strong relationship accounting for 99.3% of variation in total aboveground biomass of the sampled tree population compared with 97.7% was achieved by performing a natural logarithmic transformation (Figure 3-5). The residuals for this relationship were randomly distributed (Figure 3-6), allowing least square analysis to be performed.

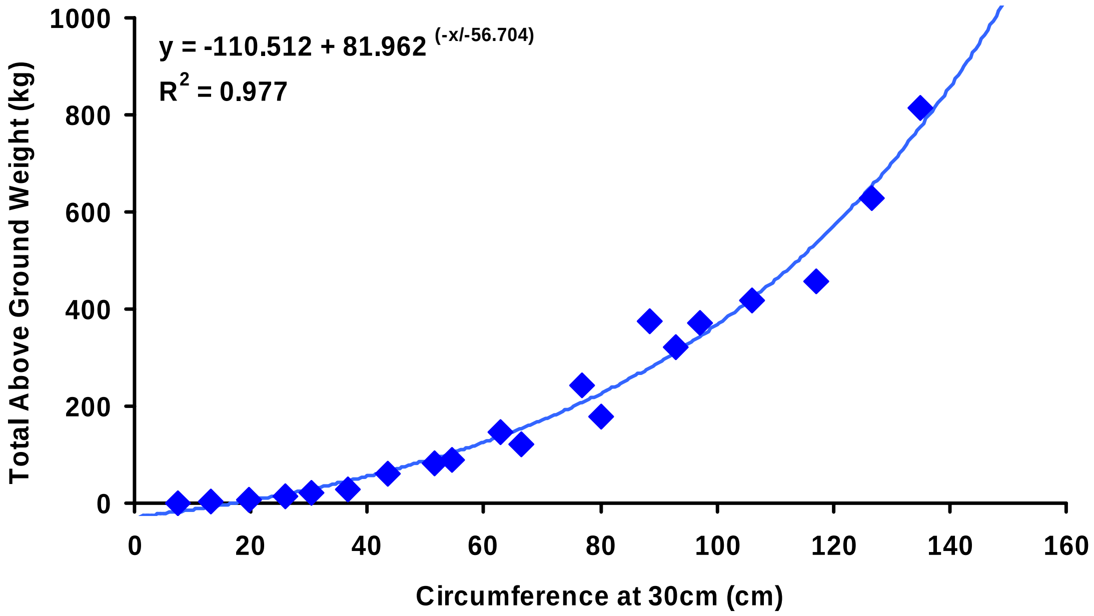

Figure 3-3 Relationship between trunk circumference (cm) measured at 30 cm aboveground and the total aboveground biomass (kg) of *C. glaucophylla*.

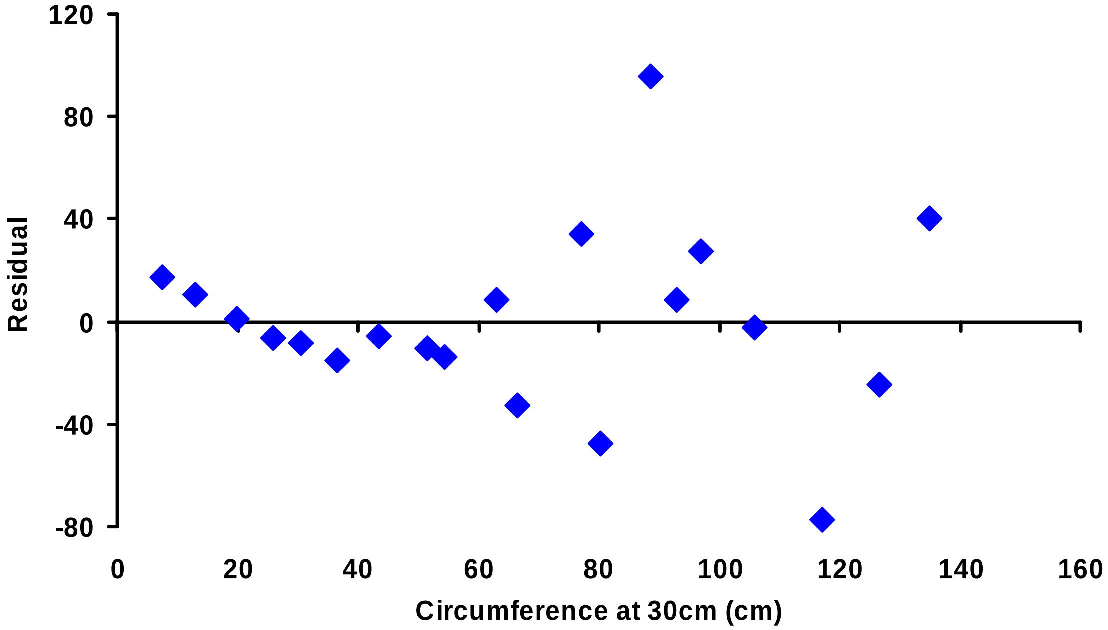

Figure 3-4 Plot of residuals for Figure 3-3 Relationship between trunk circumference (cm) measured at 30 cm aboveground and the total aboveground biomass (kg) of C. *glaucophylla.* of relationship between trunk circumference (cm) measured at 30 cm above-ground and the total aboveground biomass (kg)

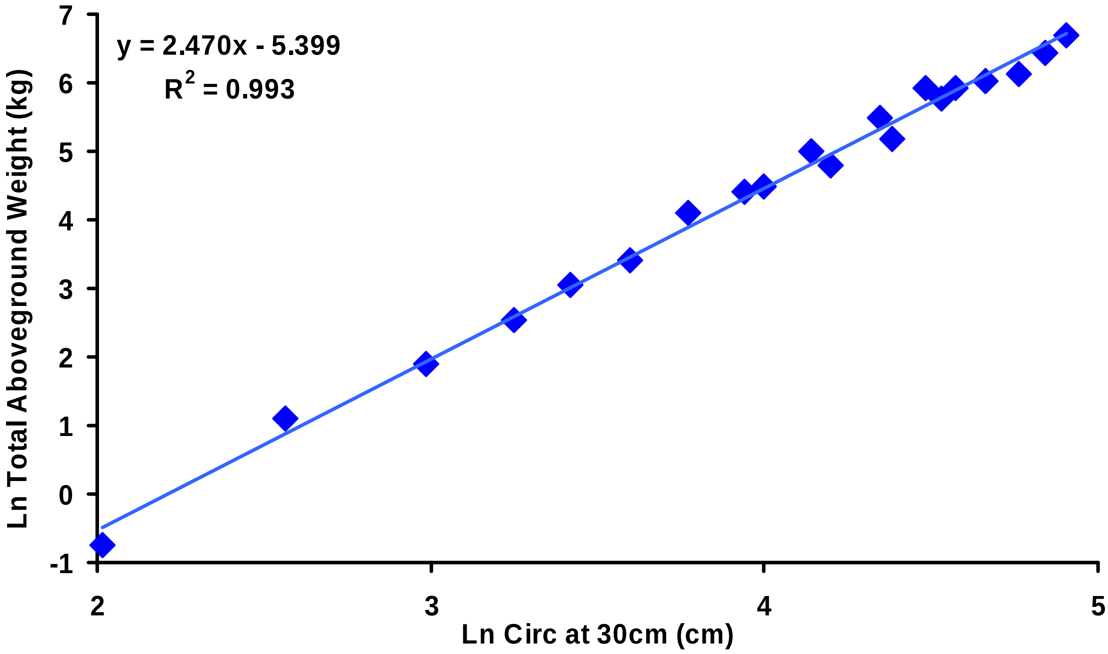

Figure 3-5 Relationship between trunk circumference (cm) measured at 30 cm above ground level and total aboveground biomass (kg) for *C. glaucophylla* after logarithmic transformation.

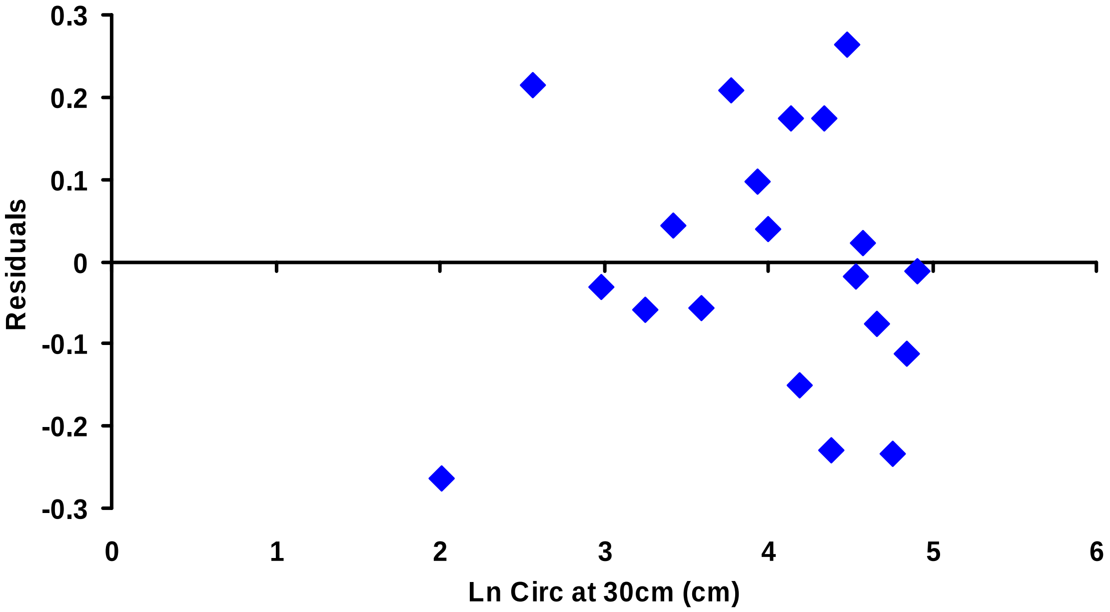

Figure 3-6 Plot of residuals for Figure 3-5 of relationship between trunk circumference (cm) measured at 30 cm aboveground and the total aboveground biomass (kg).

Relationships between other independent variables and total aboveground biomass were also investigated. Similarly, strong exponential regressions were found with non-random residual distribution and again natural logarithmic transformations were performed. These relationships between the transformed independent variables, trunk circumference at 130 cm (circ 130), tree height (H) and canopy area (CA), and total aboveground biomass accounting for 99.5%, 95.8% and 85.8% respectively, of the variation in total aboveground biomass of the sampled population (Figure 3-7 - Figure 3-9).

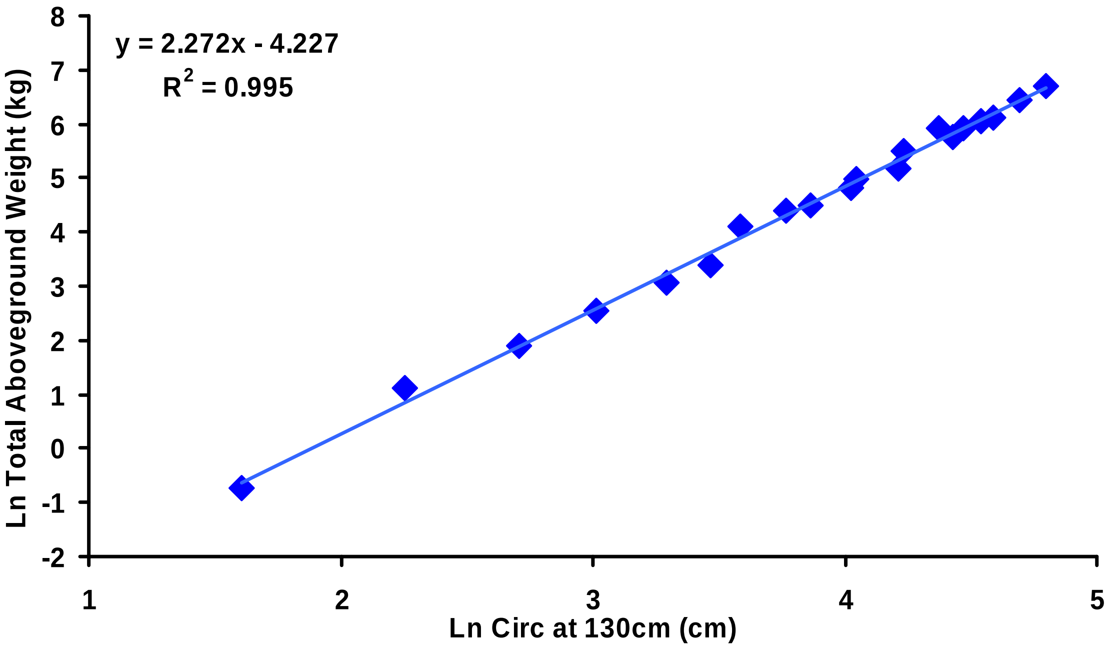

Figure 3-7 Relationship between trunk circumference (cm) measured at 130 cm above ground level (breast height) and total aboveground biomass (kg) for *C. glaucophylla* after logarithmic transformation.

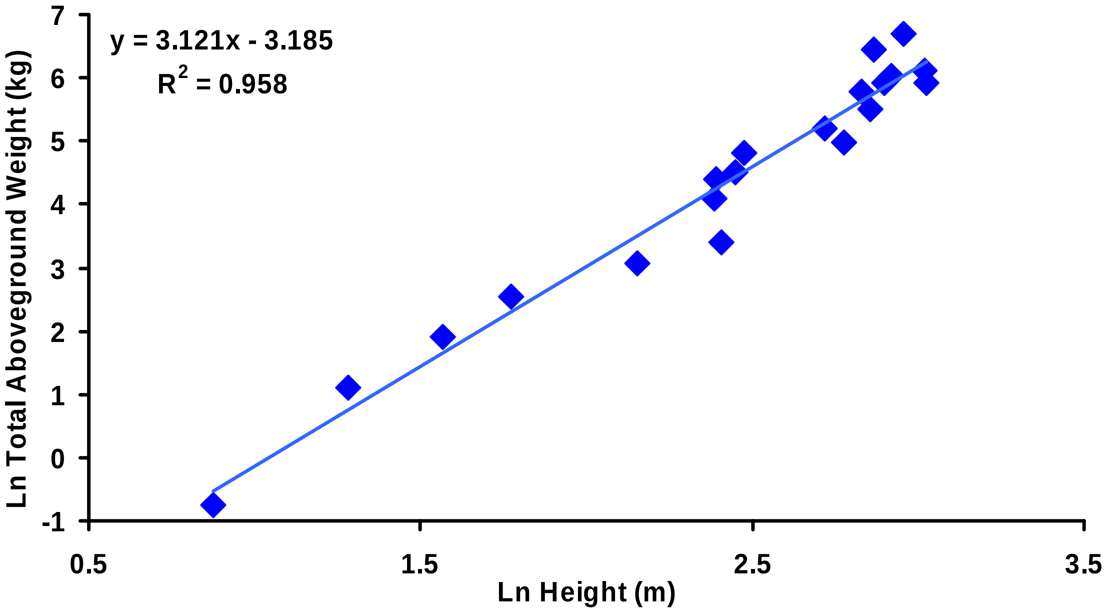

Figure 3-8 Aboveground biomass of *C. glaucophylla*: Relationship between height (m) and total aboveground biomass (kg) of *C. glaucophylla* after logarithmic transformation.

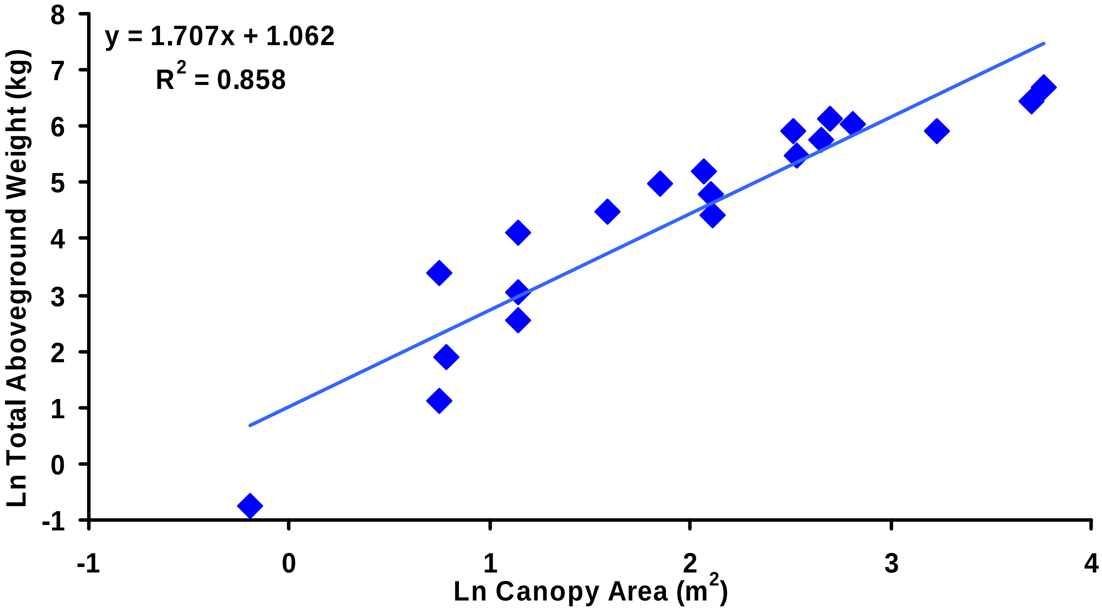

Figure 3-9 Relationship between canopy area (m²) and total aboveground biomass (kg) after logarithmic transformation.

Similarly, the relationships between the independent variables and tree component biomass were determined after a natural logarithmic transformation and R^2 values ranged from 0.783 to 0.994 (Table 3-2). Regression parameters presented include R^2, Residual Standard Deviation (RSD) and Sum of Squares of the Deviations of x (SSDx). The latter two being required to calculate the correction factor when back transforming data using Baskerville's (1972) bias correction method. Although not used in this study, it is included for readers wishing to use the regressions.

Table 3-2 Biomass relationships for *C. glaucophylla* trees with different independent variables. Equations are in the form y = a + bx. (y = dependent variable, x = independent variable, n = number of samples for regression, a = intercept, b = regression coefficient, R^2 = Coefficient of Determination, RSD = Residual Standard Deviation, SSDx = Sum of squares of deviations of x).

y	n	a	b	R^2	RSD	SSDx
Callitris glaucophylla (x = circumference at 30 cm) range = 7.5 cm – 135 cm						
Total Aboveground biomass (kg)	20	-5.399	2.470	0.993	0.171	11.847
Leaf biomass (kg)	20	-5.150	1.698	0.948	0.323	11.847
Branch biomass (kg)	20	-6.585	2.305	0.948	0.440	11.847
Trunk biomass (kg)	20	-6.685	2.694	0.990	0.224	11.846
Stem* biomass (kg)	20	-5.950	2.571	0.992	0.183	11.847
Live Stem* biomass (kg)	20	-6.716	2.250	0.896	0.624	11.847
Callitris glaucophylla (x = circumference at 130 cm) range = 5 cm – 121 cm						
Total Aboveground biomass (kg)	20	-4.227	2.272	0.995	0.148	14.026
Leaf biomass (kg)	20	-4.325	1.557	0.944	0.336	14.026
Branch biomass (kg)	20	-5.462	2.113	0.943	0.460	14.026
Trunk biomass (kg)	20	-5.413	2.480	0.993	0.187	14.026
Stem* biomass (kg)	20	-4.833	2.386	0.994	0.162	14.026
Live Stem* biomass (kg)	20	-5.594	2.056	0.885	0.654	14.026
Callitris glaucophylla (x = height) range = 2.4 m – 20.6 m						
Total aboveground biomass (kg)	20	-3.185	3.121	0.958	0.412	7.159
Leaf biomass (kg)	20	-3.509	2.097	0.874	0.503	7.159
Branches biomass (kg)	20	-4.305	2.826	0.860	0.719	7.159
Trunk biomass (kg)	20	-4.320	3.425	0.966	0.402	7.156
Stem* biomass (kg)	20	-3.631	3.243	0.954	0.449	7.159
Live Stem* biomass (kg)	20	-4.365	2.706	0.783	0.899	7.159
Callitris glaucophylla (x = canopy area) range = 0.821 m2 – 43.001 m^2						
Total aboveground biomass (kg)	20	1.062	1.707	0.858	0.758	21.429
Leaf biomass (kg)	20	-0.790	1.215	0.878	0.493	21.429
Branches biomass (kg)	20	-0.766	1.702	0.934	0.493	21.429
Trunk biomass (kg)	20	0.421	1.833	0.828	0.911	21.429
Stem* biomass (kg)	20	0.761	1.784	0.864	0.771	21.429
Live Stem* biomass (kg)	20	-1.137	1.713	0.939	0.477	21.429

*Stem represents both live and dead, branch and trunk components. Live stem represents only branches and trunk components that were live at harvest.

3.3.3 Total Harvest Biomass

The estimated biomass of the harvest plots (C01, C02) was calculated using the regressions developed from harvested trees with an independent variable of circ 30. The estimated biomass of the harvest plots were very different, as was the basal area for these sites (Table 3-3 and Table 3-1). Estimated total aboveground biomass at harvest site C01 was almost triple the total aboveground biomass at harvest site C02, within a range of 59.8 t/ha and 20.7 t/ha and 20700kg/ha). Similarly the component biomass for harvest site C01 was greater compared with harvest site C02 (Table 3-3).

Site C02 was a less developed site in terms of biomass and basal area than plot C01. Hence biomass is predictably reflected in the plot basal area. Not surprisingly, despite differences in stand biomass, both sites had similar component proportions (leaf 5.1%, branch 16.3% and trunk 72.6%) Although component portions were similar when summed they are slightly less (approx. 6%) than the biomass calculated by the total aboveground regression. Additionally, the standing aboveground biomass to basal area ratio (kg/m^2) for the two sites was approximately 4.1 t/m^2.

Table 3-3 Distribution of above ground biomass components in a *C. glaucophylla* community north of Mitchell, Queensland. Data for harvest sites (50 m × 50 m) given. Plot characteristics presented in Table 3-1.

Biomass components	Biomass** (t/ha)
Site no: C01	
Aboveground (from regression)	59.8
Trunk	43.4
Branches (live + dead)	9.8
Leaf	3.0
Stems*	53.1
Live Branches	7.5
Total (sum of components; trunk, branch, and leaf)	56.2
Site no: C02	
Aboveground (from regression)	20.6
Trunk	14.9
Branches (live + dead)	3.4
Leaf	1.1
Stems*	18.3
Live Branches	2.6
Total (sum of components; trunk, branch, and leaf)	19.4

*Stem consists of both branches and trunk components. **Rounded values

3.3.4 Site biomass

Total *C. glaucophylla* aboveground biomass was estimated for all censused sites. Stand aboveground biomass for *C. glaucophylla* ranged from 1.0 to 128.9 t/ha (Table 3-4), with approximate basal areas of 0.2 to 34.7 m^2/ha respectively. Some mature stands contained trees that exceeded the range of the circumferences of the regressions. This was due to these sites not being subjected to forestry management techniques, such as culling, therefore these trees have had their circumferences limited to the largest tree in the set harvested for regression analysis (Burrows *et al.,* 2000). Biomass estimates for these trees will actually be underestimated. However this is considered to be a more appropriate approach than applying the regression to independent variables with larger values than those employed in establishing the relationship (Burrows *et al.,* 2000).

Biomass to basal area ratios for the *C. glaucophylla* communities ranged from 2.5 to 5.1 with an average value of 3.9 t/m^2 (Table 3-4). The basal area in this table will alter from the basal area in Appendix 8-1 due to large individual trees being limited to the circumference range sampled in deriving the biomass regressions.

A strong linear relationship exists between stand basal area (m^2) and stand biomass (t) (R^2 = 0.949) (Figure **3-10**), reiterating that stand biomass can readily be predicted from the stand basal area. However, the sites consisted of three different *C. glaucophylla* stand types, dominant, co-dominant and sub-dominant. Upon further investigation it was determined that the dominant and sub-dominant communities had very strong linear trends, with R^2 greater than 90%. The co-dominant sites had a reasonable linear relationship with an R^2 of approximately 75%. See (Figure 3-11).

The co-dominant and sub-dominant sites regressions have similar slopes but different intercepts reflecting increased inter-species competition. The dominant community regression was significantly different from the other two community groups.

Table 3-4 *C. glaucophylla* Site Biomass and Site Basal Area for 30 sites. (25 sites from Mitchell study area and 5 from TRAPS data set). Biomass determined from allometrics developed from Mitchell area (Table 3-1). Note: Basal area differences from Table 3-1, large trees have been constrained to 135 mm circumference at 30 cm (the largest circumference of trees utilised in establishing the allometrics).

Site	Biomass t/ha*	Basal Area (m^2/ha)*	Biomass: Basal Area Ratio (t/m^2)*
C01	59.8	14.2	4.2
C02	20.6	5.0	4.2
C03	12.0	4.7	2.6
C04	35.4	13.2	2.7
C05	5.3	1.1	4.7
C06	38.4	15.5	2.5
C07	1.1	0.3	3.9
C08	4.1	0.9	4.7
C09	45.1	8.9	5.1
C10	32.0	7.4	4.3
C11	25.2	5.5	4.6
C12	5.6	1.2	4.8
C13	29.7	6.9	4.3
C14	128.9	34.7	3.7
C15	122.8	26.9	4.6
C16	59.3	16.0	3.7
C17	27.2	7.8	3.5
C18	8.4	3.3	2.5
C19	8.2	1.8	4.6
C20	1.2	0.3	3.5
C21	3.4	0.7	4.7
C22	1.0	0.2	4.4
C23	42.9	10.3	4.2
C24	35.8	8.1	4.4
C25	13.1	5.0	2.6
C26	54.1	15.5	3.9
C27	57.1	12.7	4.5
C28	47.9	12.8	3.8
C29	13.0	4.0	3.2
C30	8.9	2.7	3.3

*Rounded values

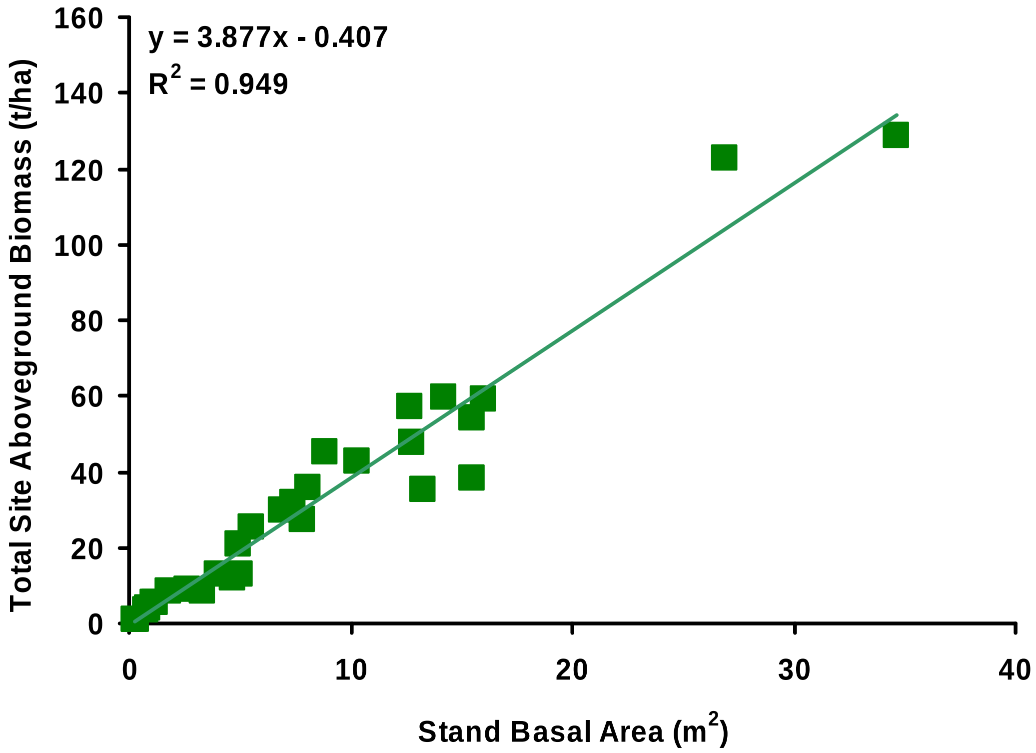

Figure 3-10 Relationship between total aboveground standing biomass (t/ha) and stand basal area (m^2/ha) for 30 *C. glaucophylla* communities in north-east Australia.

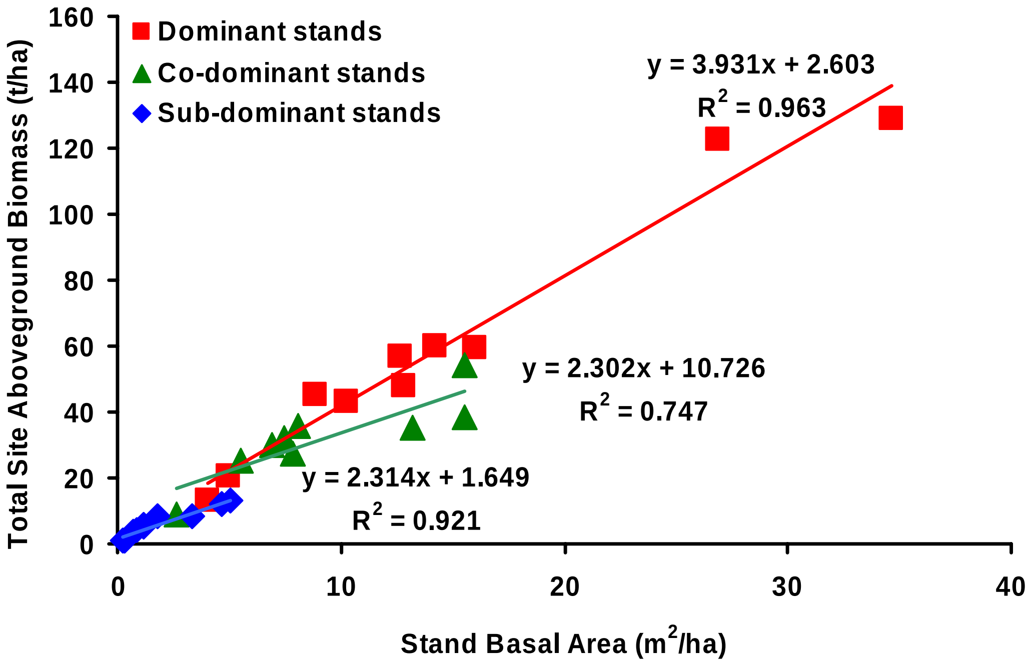

Figure 3-11 Stand basal area of aboveground biomass relationships for *C. glaucophylla* for stands where the species are dominant, co-dominant or sub-dominant (see text for description of these structural classes).

3.4 Discussion

Aboveground biomass can be estimated using whole tree regressions with stem circumference as the independent variable and stand allometrics were effective at estimating stand biomass of *C. glaucophylla*.

3.4.1 Site Characteristics

C. glaucophylla was dominant at both harvest sites (Appendix 8-1) because it contributed over 55% of the total site basal area (m^2/ha) at both sites, although site C01 had greater basal area and density than C02. Harvesting to develop the allometric relationships at both sites incorporated trees from two different communities and enabled the derived relationship to be used at 'light' and 'heavy' density sites. The harvest site summaries noted that smoothed-bark apple (*Angophora leiocarpa*), budgeroo (*Lysicarpus angustifolius*) and woody pear (*Xylomelum cunninghamianum*) comprised most of the remaining species (over 25%) while the minority were wattles and other native shrubs. Other species present within the study area, but not within the harvest sites, include *Eucalyptus crebra* and *Eucalyptus populnea*.

The 30 censused plots within the Mitchell study area highlighted the variation in dominant species and plant density (Appendix 8-1). This was due to the large study area (2400 km^2). Sites were dominated by *C. glaucophylla*, *Eucalyptus* spp., *Acacia harpophylla* or *Angophora* spp., or a combination of two species (co-dominant). *C. glaucophylla* dominant plots were also represented by regenerating communities and 'mature' communities (in most cases the mature plots had been managed by normal forestry practices).

3.4.2 Individual Tree Relationships

Tree biomass can be estimated with varying levels of predictability from four measurable variables (Table 3-2). However some variables were statistically better, having higher R^2 values, than others, although the differences were small. The canopy area and height variables were the least efficient at predicting biomass. Of these two, canopy area was statistically poorer, with the lowest R^2 and the highest residual standard deviation. However, both variables, although still statistically sound for estimating biomass, increase the probability of operator error. Height is usually estimated visually, or by using a clinometer, so that the accuracy of both methods are operator dependent. Canopy area is calculated from an estimation of two perpendicular canopy diameters, projected to ground level (chance of inaccuracy increased with each measurement). Canopy shape of individual trees is often irregular, further increasing the chance of inaccuracy. Height requires only one estimation, but is considered time consuming (Rayachhetry *et al.,* 2001), while canopy area is considered the most error prone predictor variable.

The use of stem circumference as an independent variable decreases the incidence of operator errors. Only one variable is required for each tree and it is physically measured using a flexible steel tape. Furthermore, circumference is less time consuming to measure and record than height or canopy, allowing more trees in a stand, or sites to be recorded. Foresters often use a combination of trunk diameter at 130 cm (D) and tree height (H) as a predictor value for aboveground biomass (e.g. Madgwick and Satoo, 1975; Feller, 1980). However, this method results in a small increase in the accuracy of the equation (Eamus *et al.,* 2000). As previously stated, height can be difficult to measure, thus the small gain in accuracy would be out weighed by the expense of obtaining it. For biomass estimation, height and canopy area are less efficient predictors compared with relationships that use circumference as a variable. Therefore relationships that used circumference as the independent variable were favoured.

The variable circ130 gave better results statistically than the circ30 predictor, although statistical differences between the two relationships were very small. The difference between the coefficient of determination (R^2) was 0.2% (99.5% and 99.3%) and the residual standard deviation increases slightly for circ30 when compared to circ130 (0.171 vs. 0.148). For that reason there was little statistical gain favouring the use of one of these two variables over the other in *C. glaucophylla* communities of southern Queensland.

Foresters traditionally use circ130 to record tree basal area, to avoid effects of buttressing when calculating merchantable wood volumes. However, measurements at this height can lead to some saplings being excluded. Exclusion would occur when saplings or seedlings do not reach 1.3 m in height. Inaccurate measurements of saplings at circ 130 can occur as a result of difficulties in measuring a small diameter at this height. Biased diameter measurements of trees less than 5 m in height can occur when measured at DBH (Snowdon *et al.*, 2000). This bias occurs because the point where diameter is being measured relative to the tree height differs markedly, (ie. DBH on a tree 1.4 m tall is in the “tree canopy” whereas DBH for a tree 14 m tall is on the trunk). In some communities, small trees, seedlings and saplings contribute significantly to biomass, especially if their density is large (Brown, 1997). Therefore, if total biomass at a site is required, especially if trees are small and dense, there is an advantage in using the circ30 relationship.

The relationships for tree components and the independent variable explained between 99.4% and 78.3% of the variation within the sample population (Table 3-2). Again the circumference variables provided the best predictor statistics for components. Leaf and branch biomass estimates usually were the least accurate for each independent variable (94.8% - 86.0% of variation explained) an exception to this was when canopy area was the independent variable. In latter case, leaf and branch biomass were two of the most reliable components (87.8% and 93.4% variation explained) and trunk component was the least reliable (82.8%), yet the circumference variables were again statistically better (94.3% – 94.8% variation explained) at predicting component biomass.

The relationships developed here were used to determine total aboveground biomass (total aboveground regressions and summation of predictions from component aboveground biomass regressions). Total aboveground biomass relationships were the best fitting when compared with the component regressions (with the same independent variable). Thus, it was resolved that total aboveground biomass regressions with circumference measured at 30 cm should be utilised to determine standing aboveground biomass in *C. glaucophylla* communities.

3.4.3 Harvest Site Biomass

Biomass calculated using the suite of component regressions should ideally sum to approximately the same amount (Parresol, 1999). Data presented here support reports by Parresol (1999) that total biomass may be more accurately estimated using a single regression. Although, component relationships would be sufficient in circumstances where there is no total aboveground biomass relationship, provided that it is acknowledged that the final figure may not be as accurate as a total tree regression.

Despite having different aboveground biomass and basal areas, the two harvest sites had similar biomass: basal area ratios. This fact indicates that stand basal area may be a useful variable to predict aboveground stand biomass of *C. glaucophylla* in these communities, however further investigation to confirm this would be required.

3.4.4 Stand Biomass

The greater variation within the co-dominant communities may result from increased but variable, competition for available nutrients and water in the poorer soils on which they occur. The biomass/basal area ratio varies according to which tree species tends to dominate. Thus, in some communities this would be *C. glaucophylla* and in others *Eucalyptus* spp. or *Angophora* spp.. The dominant community regression was significantly different ($P < 0.001$) from the other two community groups. Here there is very little inter-species effect, and individual trees would have similar intra-species competition. The co-dominant and sub-dominant sites regressions have similar slopes but different intercepts, as a result of different inter-species competition. However, this trend should be treated cautiously as the differences in intercepts may also reflect the differing basal area ranges of the communities (i.e. there is no overlap of basal area range between the two communities). Therefore, if a stand type is known, it would be more accurate to use the appropriate regression developed for that stand type to predict its total biomass. However, if unsure of stand composition the combined regression can still be utilised to predict stand biomass in *C. glaucophylla*.

C. glaucophylla has a smaller biomass: basal area ratio than other native woodland species. The overall biomass ratio for *C. glaucophylla* (3.90 ± 0.14 t/m^2) is lower than the ratio for eucalypt species (*E. melanophloia* (5.11 t/m^2), *E. crebra* (6.74 t/m^2), and *E. populnea* (5.81 t/m^2)) (Burrows *et al.*, 2000), *Acacia aneura* (4.79 t/m^2)

(Burrows 1976) and *A. harpophylla* (4.97 t/m^2) (Scanlan, 1991). This was expected, since *C. glaucophylla* has a much lower wood density (675 kg/m^3) compared with eucalypts (1090 kg/m^3), mulga (1089 kg/m^3) and brigalow (1000 kg/m^3) (Cause *et al.* 1989). Similarly, these latter species have a more apparent branching pattern and different tree-form to *C. glaucophylla*. *Melaleuca viridiflora* also has a greater wood density (730 kg/m^3, Cause (1989)) than *C. glaucophylla*, but it is much lower than that of the eucalypts. This reflects the high water content of the *M. viridiflora* wood. Surprisingly, *M. viridiflora* has a lower biomass: basal area ratio (3.18 t/m^2) than *C. glaucophylla* (3.90 t/m^2). *M. viridiflora* tends to have a straight slender trunk with a thin canopy (Anderson, 1993) whereas *C. glaucophylla* has a similar slender trunk but has a denser canopy. It is also found in a more xeric environment than *M. viridiflora*. Thus branching patterns, wood density and tree-form can influence biomass: basal area ratios.

Biomass: basal area ratio for *E. melanophloia* regrowth was much smaller than that for the intact *E. melanophloia* woodland. The regrowth biomass: basal area ratio was 1.94 t/m^2, which was also smaller than *C. glaucophylla* and *M. viridiflora* biomass: basal area ratio. The eucalypt regrowth has a very different form when compared with these species. The *E. melanophloia* regrowth is multi-stemmed, and originates from the original tree lignotuber. Regrowth sites initially have very dense numbers of small size saplings. Therefore, even for a reasonable basal area, the relative biomass is low.

Thus, stand biomass: basal area ratios appear to be specific to the community type, as well as the genus. When applying relationships developed within different communities it is advisable to compare ‘like with like’. Hence, in those cases where no such relationship is available, caution should be exercised in extrapolating existing relationships. These relationships need to be applied with care and acknowledgment of the likely differences between communities and possible effects this may have on the accuracy of the biomass estimates.

3.4.5 Expansion Factors

A common method for estimating the total above-ground biomass of a tree is by the use of expansion factors in conjunction with trunk volume regressions. The need to use expansion factors when estimating biomass came about because foresters traditionally measured merchantable timber volumes of trees. Burrows *et al.* (2000) identified the total aboveground biomass: trunk biomass relationship within three eucalypt species as ranging from 1.85 – 2.39 and that the *M. viridiflora* value was 2.04. The present default value adopted by the Intergovernmental Program on Climate Control (IPCC) is on expansion factor of 2.0 (IPCC, 1997). The *C. glaucophylla* expansion factor for the harvested trunk biomass at the present study sites was 1.38, much lower than that calculated for the above species and defaults. Hence using expansion factors to estimate aboveground biomass of *C. glaucophylla* would result in an overestimation of biomass and any resultant carbon sink.

3.5 Conclusion

This study has shown that the independent (predictor) variable, stem circumference (cm) was statistically the most robust variable for determining the above ground biomass of *C. glaucophylla* trees. Stem circumference was also far more efficient and accurate to use than alternative predictors, such as height or canopy area. Further, it was concluded that total aboveground biomass estimations were more accurate if a whole tree regression was used rather than summing estimates made from a suite of component relationships.

Stand allometrics with very good statistical fits ($R^2 > 0.75$) were developed for *C. glaucophylla*. These relationships can be applied quickly and easily to stand basal area of woodland communities measured with common forestry mensuration instruments (e.g relaskop, prism or Bitterlick sticks).

At the present time the best method to estimate tree community biomass in *C. glaucophylla* would appear to be by the use of allometric regressions based on individual trees and/or stand relationships. However, limited knowledge exists on the effect that different environments have on biomass allometric regressions.

Chapter 4. Tree-form and allometric regressions for estimating biomass in northeast Australian woodlands.

4.1 Introduction

Tree-form can be mathematically described by the relationship between stem circumference and tree height of individual trees and is represented by a linear regression plotted on a logarithmic scale. Limited knowledge exists on how environmental factors influence tree-form regressions, and the effect that tree-form has on allometric biomass regressions. Few allometric biomass regressions have been developed for Australia's woodlands because developing these regressions is expensive and time consuming. Therefore, the ability to use a common regression to predict biomass across both species and geographic regions would be useful. Brown *et al*. (1989), Senelwa and Sims (1998) and Montague (2005) suggest that the same regression can be applied accurately to similar species, yet both Feller (1980) and Keith *et al.* (2000) suggest that a regression is specific to both site and species.

This chapter reports two studies. The first study examines tree-form regressions in woodland communities with contrasting rainfall and soil characteristics. The second study examines an array of sites to investigate the relationship between tree-form and biomass[5], and the application of a common biomass regression.

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[5] The harvest data in Chapter 4, with the exception of the *E. populnea* regrowth data, was published in Burrows *et al*. (2000), Burrows *et al*. (2001), Williams *et al*. (2005) and Zerihun *et al*. (in press).

4.2 Study 1: Tree-form regression and the effect of soil and rainfall

4.2.1 Material and Methods

4.2.1.1 Site selection

Thirty-two sites from the monitoring data set were selected, which have a range of over-storey vegetation including:

- Dominated or co-dominated by *C. glaucophylla*; or
- Dominated by *E. populnea* - sites have a basal area of *E. populnea* greater than 60% of total site basal area. However, two sites (T2 and T15) dominated by another species have been included, as these two sites contribute significantly to the *E. populnea* distribution range; or
- Dominated by *E. melanophloia* where the sites have a basal area of E. *melanophloia* that is greater than 75% of the total site basal area.

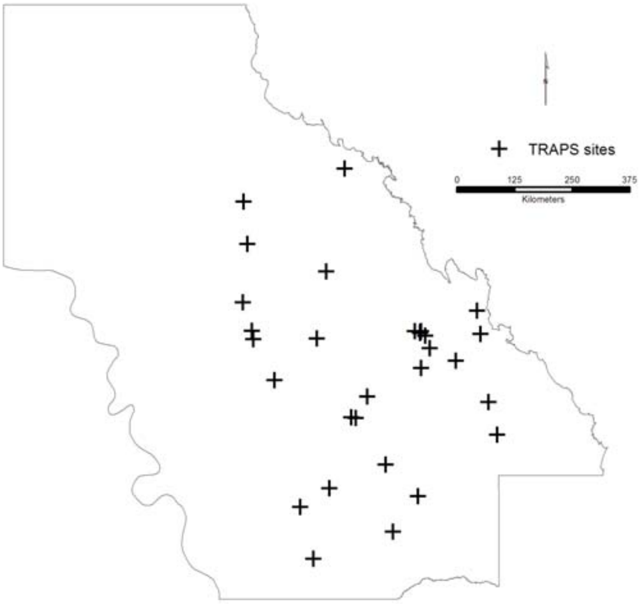

Figure 4-1. Distribution map of sites throughout north-east Australia.

4.2.1.2 Soils

The dominant soil types were established from the Atlas of Australian Soils (Northcote *et al*., 1975) by overlaying site positions onto a digital form of the Atlas of Australian Soils. However, in this overlay most soil classes contained very few sites (two or less). Therefore, these soil classes were broadened until most contained at least two monitoring sites, and this resulted in three broad soil groups for each of the two eucalypt species and two broad soil groups for the *C. glaucophylla* (**Table 4-1**).

4.2.1.3 Mean annual rainfall

Mean annual rainfall (MAR) for each site was extracted from the Queensland Department of Natural Resources and Mines' (QNR&M) Data Drill database (www.nrm.qld.gov.au/silo/datadrill). These data were interpolated daily rainfall from 1889 to 2001, which were derived from real data recorded by the Bureau of Meteorology (BOM) from Australia Post Offices and weather stations. Mean annual rainfall (mm/yr) was calculated and rainfall classes (in 50 mm increments from 400 mm/yr) were determined for each of the monitoring sites (**Table 4-1**). Due to limited numbers of monitoring sites within some 50 mm/yr rainfall classes, larger rainfall classes were also determined to investigate the potential rainfall effect on tree-form.

Table 4-1. Locations, dominant overstorey vegetation, mean annual rainfall class (mm/yr) and soil group for the sites selected to investigate tree-form.

Site number	Latitude and longitude	Dominant species	MAR class (mm/yr)	Soil group
T01	-27° 8.37 N 146° 59.54 W	*E. populnea*	450-500	Gradational
T02	-28° 11.19 N 147° 15.62 W	*E. populnea*	400-450	Gradational
T03	-21° 49.76 N 145° 55.55 W	*E. melanophloia*	500-550	Duplex
T04	-23° 11.07 N 146° 34.18 W	*E. melanophloia*	500-550	Duplex
T05	-24° 11.00 N 150° 8.00 W	*E. melanophloia*	650-700	Duplex
T06	-23° 35.39 N 149° 18.28 W	*E. populnea*	650-700	Duplex
T07	-25° 40.82 N 150° 58.30 W	*E. melanophloia*	650-700	Duplex
T08	-24° 20.00 N 149° 26.00 W	*E. melanophloia*	650-700	Gradational
T09	-23° 10.43 N 150° 33.58 W	*E. populnea*	1100-1150	Duplex
T10	-23° 34.84 N 146° 0.97 W	*E. melanophloia*	500-550	Gradational
T11	-26° 45.51 N 147° 35.05 W	*E. melanophloia*	450-500	Gradational
T12	-23° 56.02 N 149° 36.53 W	*E. populnea*	700-750	Uniform
T13	-20° 58.58 N 145° 51.10 W	*E. melanophloia*	550-600	Gradational
T14	-23° 38.73 N 150° 37.91 W	*E. populnea*	800-850	Uniform
T15	-23° 44.38 N 147° 19.80 W	*E. populnea*	600-650	Duplex
T16	-22° 23.00 N 147° 31.00 W	*E. melanophloia*	550-600	Uniform
T17	-24° 36.86 N 148° 36.09 W	*E. melanophloia*	650-700	Duplex
T18	-24° 54.44 N 148° 20.93 W	*E. melanophloia*	800-850	Duplex
T19	-25° 1.42 N 150° 47.78 W	*E. melanophloia*	600-650	Duplex
T20	-23° 45.00 N 146° 2.00 W	*E. melanophloia*	500-550	Gradational
T21	-25° 19.69 N 148° 1.23 W	*E. melanophloia*	600-650	Duplex
T22	-23° 0.47 N 145° 50.11 W	*E. populnea*	450-500	Gradational
T23	-23° 40.66 N 149° 30.88 W	*E. melanophloia*	700-750	Gradational
T24	-23° 37.56 N 149° 24.84 W	*E. populnea*	650-700	Gradational
T25	-23° 36.42 N 149° 25.20 W	*E. populnea*	650-700	Gradational
T26	-23° 35.52 N 149° 26.04 W	*E. populnea*	650-700	Gradational
T27	-27° 38.28 N 148° 52.05 W	*E. populnea*	500-550	Gradational
T28	-26° 16.89 N 148° 43.17 W	*C. glaucophylla*	550-600	Duplex
T29	-22° 51.45 N 147° 18.91 W	*C. glaucophylla*	550-600	Duplex
T30	-25° 19.69 N 148° 1.23 W	*C. glaucophylla*	600-650	Duplex
T31	-25° 20.40 N 148° 6.92 W	*C. glaucophylla*	600-650	Duplex
T32	-24° 34.69 N 146° 28.29 W	*C. glaucophylla*	500-550	Uniform

4.2.1.4 Limitations imposed on the site data

Not all sites had been recorded at the same frequency after establishment. Therefore, for the purpose of this study, the analysis of each site's data set was constrained to the most recent recording. However, where the landholder had imposed mechanical or chemical timber control treatments, the data extracted were that from the recording prior to the landholder's management practice.

The data were influenced by the TRAPS methodology. For example, TRAPS records a single height value for each tree, therefore when a tree is multi-stemmed the tallest stem is recorded as the tree height. Consequently, for multi-stemmed trees, the largest stem circumference was extracted with the recorded height. Additionally, TRAPS records the height of seedlings and young saplings. However, if the stem circumference of such plants is smaller than 50 mm, a 0 value is allocated for the stem circumference. This 0 value ensures that operators can efficiently record the presence of all seedlings, yet this may introduce bias in any tree-form regressions for sites and these data were not included in the present analysis.

The data set was further limited for each site by extracting the data for only the required species. For example, *E. populnea* information only was extracted for *E. populnea* dominated communities. However, one monitoring site was an exception because that site was dominated by *E. melanophloia* with sub-dominant *C. glaucophylla.* In this case data for both species were extracted (labelled as T21 and T30, respectively) and analysed separately.

4.2.1.5 Analysis of tree form regressions

Regressions between stem circumference[6] (cm) and height (m) were determined for each monitoring site. Regression residuals increased as circumference increased and a logarithmic transformation was necessary. Tree-form regressions (ln stem

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[6] Stem circumference in Chapter 4 and Chapter 5 were measured at 30cm above ground level unless otherwise stated.

circumference × ln height) were calculated using least-squares analysis for both, pooled site data grouped by environmental classes and for each of the monitoring sites. Analysis of covariance was performed on these linear regressions to determine if slopes were significantly different, and then, if found to be similar, it was determined if y intercept of the regressions were significantly different.

4.2.2 Results

4.2.2.1 Tree-form regressions of monitoring sites.

The tree-form regressions (ln stem circumference × ln plant height) for each of the monitoring sites, representing the three study species, generally have strong variable correlations with R^2 ranging from 0.968 to 0.489 (Table 4-2). One exception to this occurred with a *E. melanophloia* site (T10), where $R^2 = 0.053$. However, this may be due to limited range of circumference/height data. Regressions were in the form of $y = a + bx$, where y = ln height (m), x = ln circumference at 30 cm from ground level (cm), n = number of trees, R^2 = coefficient of determination. The tree-form regressions for the pooled data for each species was pooled were also strong (R^2= 0.85, 0.79, 0.84 for *C. glaucophylla*, *E. populnea* and *E. melanophloia*, respectively) (Figure 4-2.). However, despite these strong pooled data correlations, the tree-form regressions for individual sites for a given species were significantly different ($P < 0.001$) (Figure 4-3).

Table 4-2 Lognormal regressions for tree-form at selected sites in Queensland woodlands. Regressions are in the form of ln y = a + b lnx, where y = tree height (m), x = stem circumference (cm) measured 30 cm above ground level.

Site	a	b	R^2	n
T01	-0.355	0.616	0.789	31
T02	-0.161	0.556	0.790	32
T03	-0.737	0.694	0.862	30
T04	-1.227	0.868	0.712	105
T05	-0.705	0.644	0.941	36
T06	0.276	0.554	0.620	85
T07	-0.864	0.775	0.918	150
T08	0.128	0.582	0.576	69
T09	-0.518	0.715	0.596	27
T10	1.799	0.136	0.053	20
T11	-0.278	0.644	0.821	73
T12	0.747	0.767	0.827	11
T13	-1.005	0.762	0.489	22
T14	0.009	0.593	0.749	30
T15	-0.227	0.625	0.963	16
T16	-0.831	0.735	0.919	40
T17	0.570	0.459	0.712	31
T18	-0.584	0.707	0.871	40
T19	0.095	0.551	0.690	46
T20	-1.102	0.901	0.845	235
T21	-1.026	0.818	0.716	65
T22	0.598	0.397	0.626	40
T23	0.316	0.635	0.675	72
T24	-0.458	0.741	0.964	22
T25	-0.187	0.703	0.743	91
T26	-0.361	0.713	0.947	73
T27	-0.568	0.683	0.950	61
T28	-0.326	0.676	0.961	158
T29	-0.446	0.756	0.678	127
T30	-0.191	0.608	0.786	144
T31	-0.477	0.747	0.861	139
T32	-1.336	0.859	0.901	39

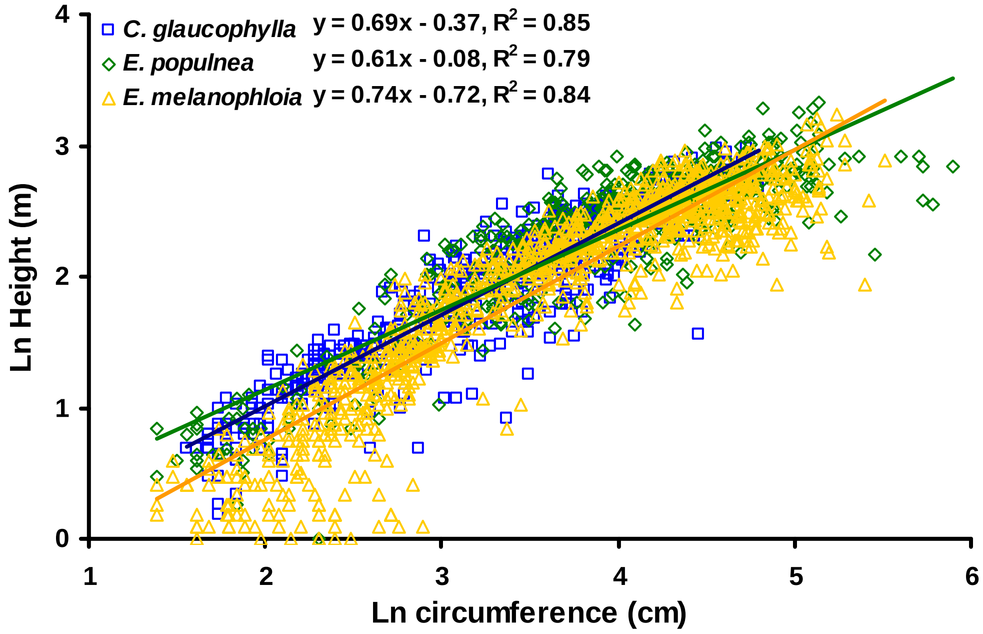

Figure 4-2. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for pooled data of *Callitris glaucophylla, Eucalyptus populnea* and *E. melanophloia*.

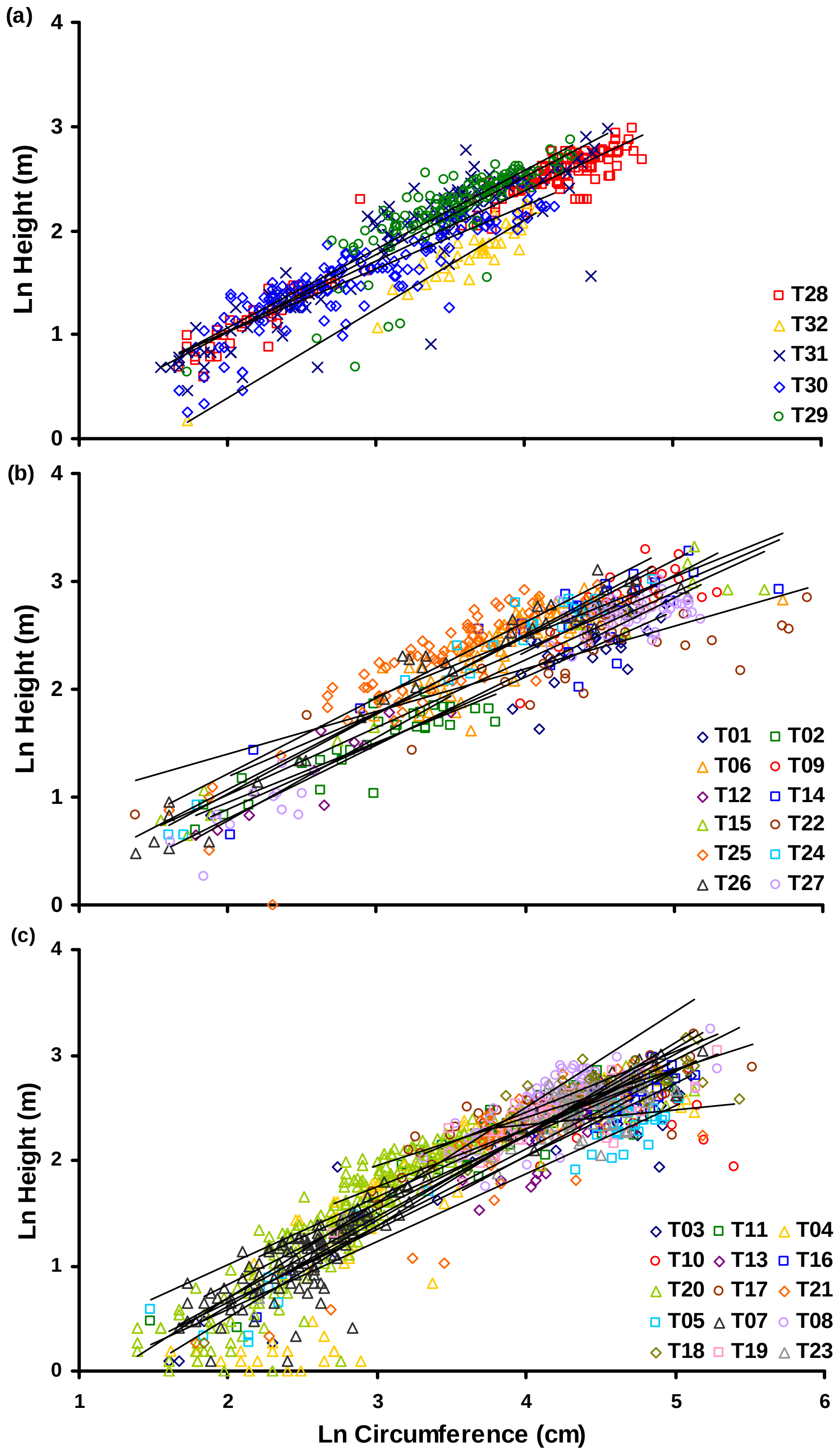

Figure 4-3. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites dominated by (a) *C. glaucophylla,* (b) *E. populnea,* and (c) *E. melanophloia*. (Independent regressions were significantly different, P<0.001). Regression details in Table 4-2.

4.2.2.2 Tree-form regressions of monitoring sites with differing environmental factors.

Tree-form regressions at each site representing the three woodland species were investigated for the effect the environmental factors of soil and rainfall. When sites were grouped by either soil group or rainfall class, trends appear within each species.

Tree-form regressions for monitoring sites grouped by soil type, for each woodland species, were not significantly different (P>0.001) (Figure 4-4).

Three major soil types (duplex, gradational and uniform) were further investigated. Monitoring sites dominated by either *E. populnea* or *E. melanophloia* represented gradational soils. Within these two species groups, tree-form regressions for monitoring sites were found to be significantly different from one another (P <0.001) (Figure 4-5). The only species representing the uniform soil group was *E. populnea*, and tree-form regressions for monitoring sites within this group were not significantly different (P>0.001) (Figure 4-6). The duplex soil group was represented by all three woodland species and tree-form regressions for these monitoring sites were significantly different for the *C. glaucophylla* and *E. melanophloia* dominated sites (P<0.001). However, tree-form regressions for the *E. populnea* monitoring sites were parallel (Figure 4-7).

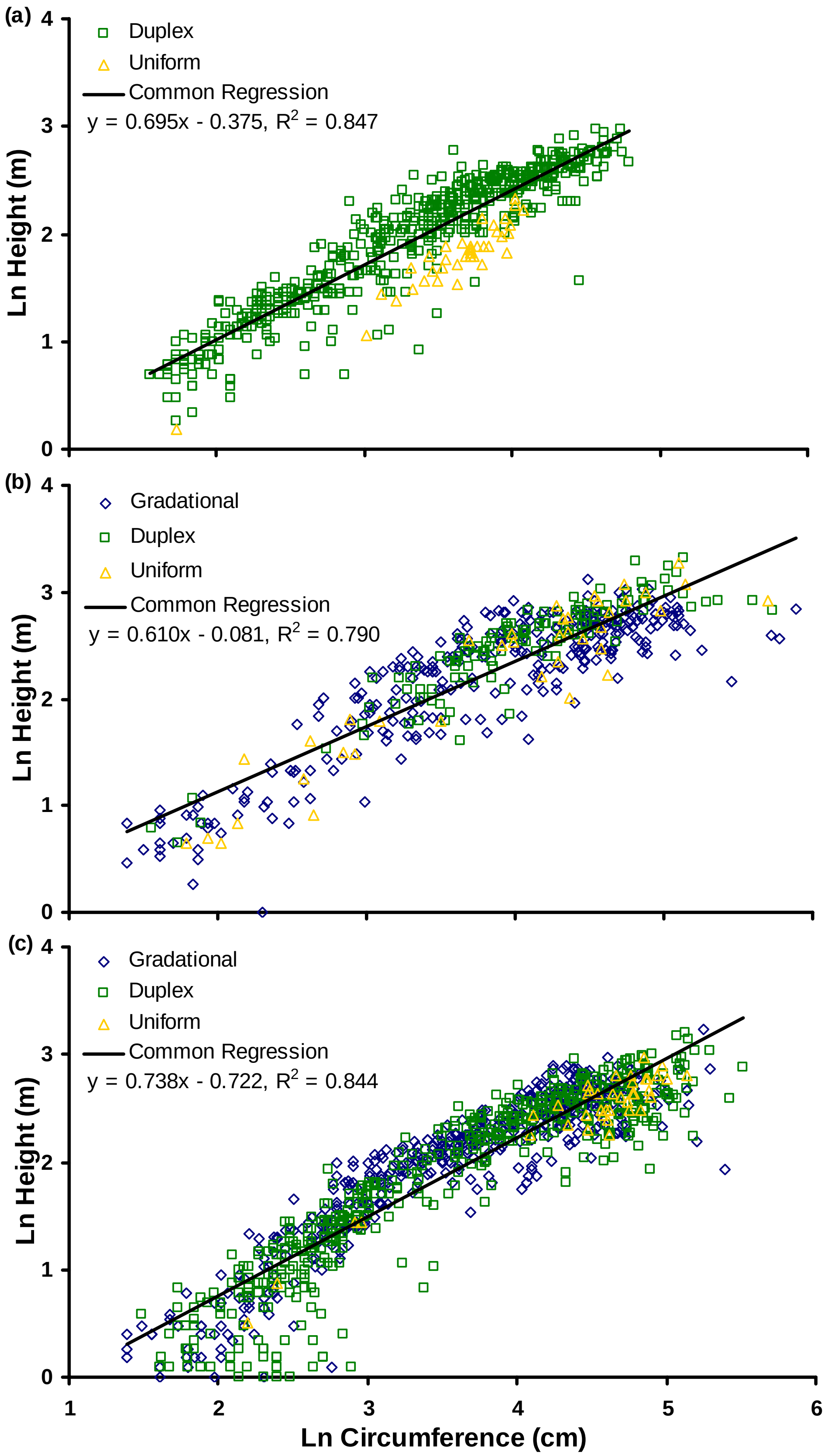

Figure 4-4. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for each soil group in (a) *C. glaucophylla,* (b) *E. populnea,* and (c) *E. melanophloia* communities. (Independent regressions were not significantly different, P>0.001).

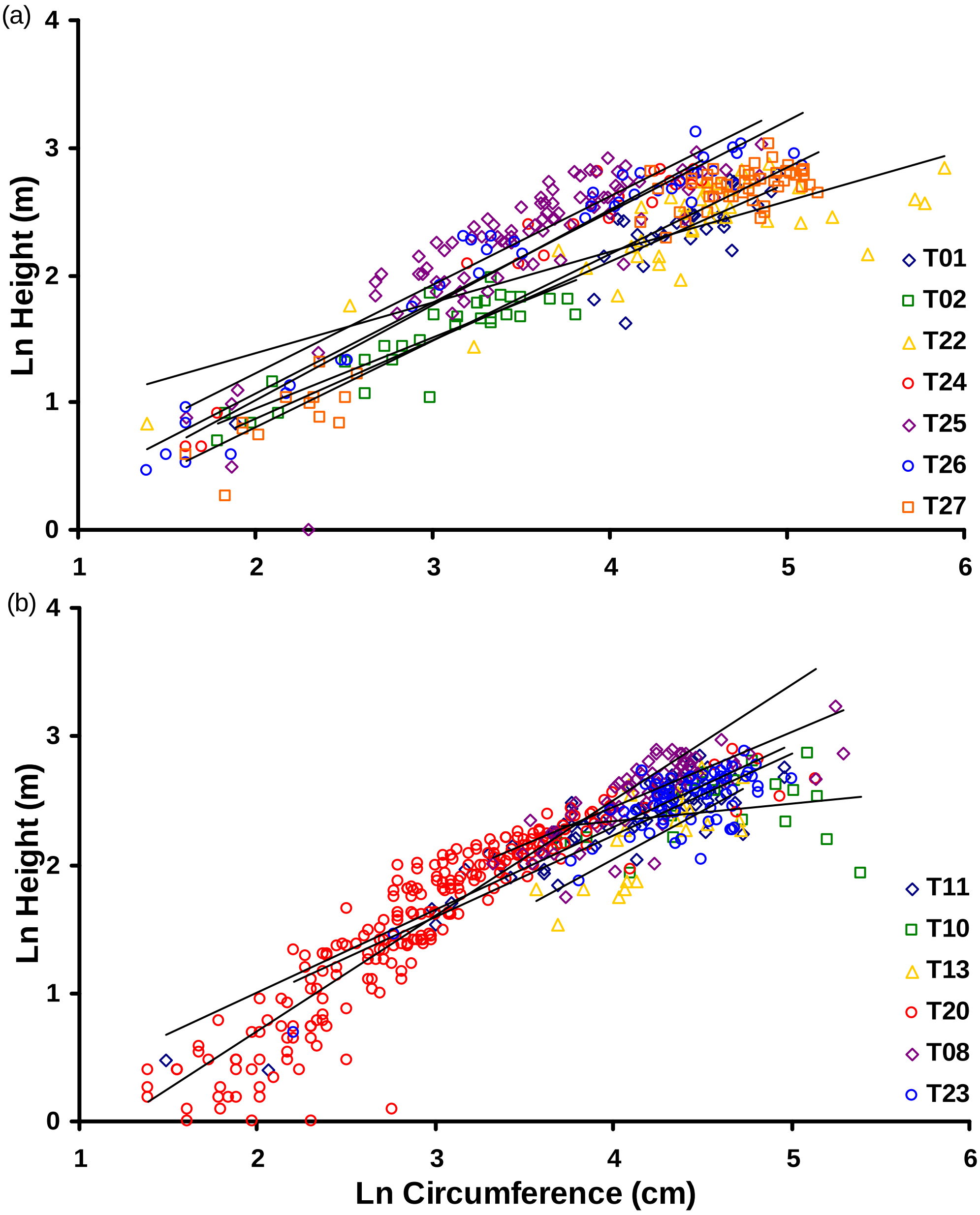

Figure 4-5. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites with gradational soil type, for each of the woodland communities dominated by (a) *E. populnea,* and (b) *E. melanophloia*. (Independent regressions were significantly different, P<0.001)

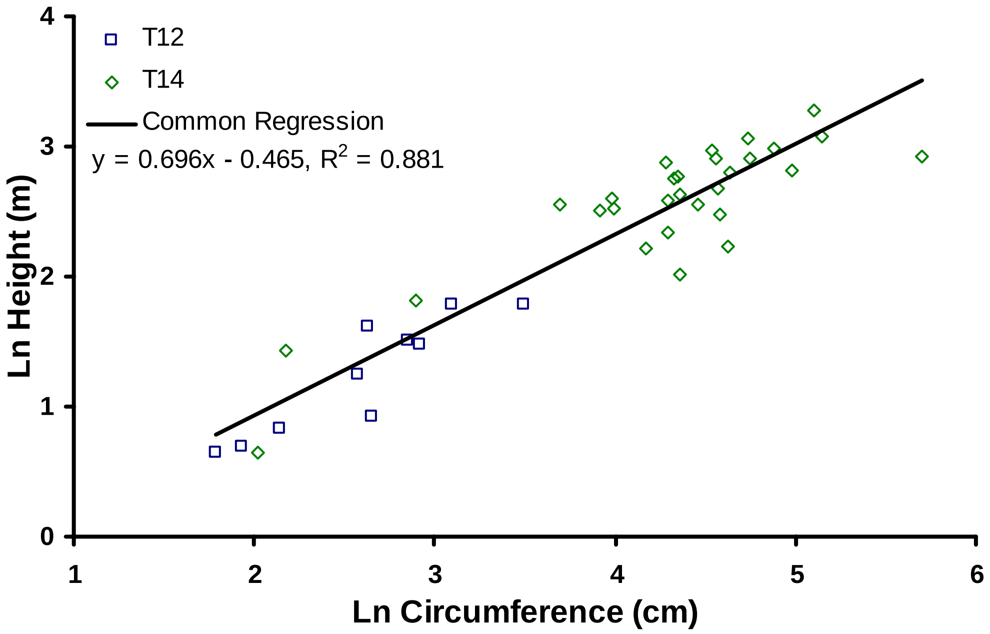

Figure 4-6. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites with uniform soil type, for each of the woodland communities dominated by E. *populnea*. (Independent regressions were not significantly different, P>0.001).

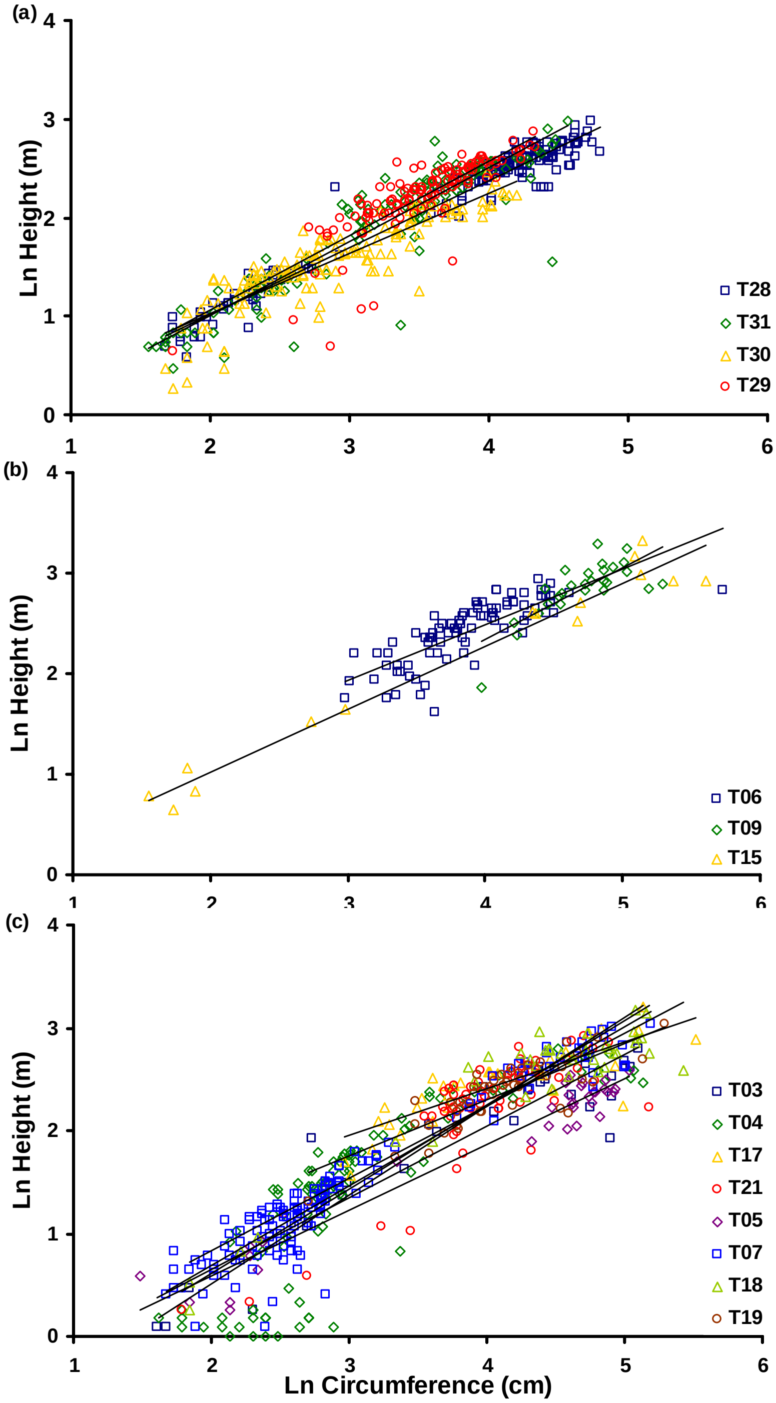

Figure 4-7. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites with duplex soil type, for each of the woodland communities dominated by (a) *C. glaucophylla*, (b) *E. populnea*, and (c) *E. melanophloia* communities. (Independent regressions in figures (a) and (c) were significantly different, P<0.001, and in figure (b) independent regressions were parallel, ie, slopes were not significantly different, P>0.001, and intercepts were significantly different, P<0.001).

When monitoring sites were grouped by mean annual rainfall increments, the trends of tree-form regressions were varied and trends within each increment class were also inconsistent across the three species. Tree-form regressions, when grouped by rainfall, for *C. glaucophylla* were parallel, for *E. populnea* were significantly different (P<0.001) and for *E. melanophloia* were not significantly different (P> 0.001) (Figure 4-8). *C. glaucophylla* tree-form regressions were significantly different (P<0.001) for monitoring sites within the 600-650 mm/yr increment class, however for sites at a lower rainfall increment (550-600 mm/yr) the tree-form regression slopes were parallel (Figure 4-9).

E. populnea tree-form regressions were not significantly different (P> 0.001) for sites in each of the 450-500 mm/yr and 650-700mm/yr rainfall classes (only these two classes had > 1 site). Tree-form regressions for rainfall classes in the more arid region (450-750 mm /yr) were significantly different (P<0.001). Yet, when this rainfall range was reduced to 500-700 mm/yr the slopes of the tree-form regressions of rainfall classes were parallel. Further investigation into the rainfall gradient effect showed that tree-form regressions for low, mid-range and high rainfall classes (400-450, 700-750 and 1100-1150 mm/yr) were not significantly different (P> 0.001)(Figure 4-10).

E. melanophloia site tree-form regressions were not significantly different in the 550-600 mm/yr and 600-650 mm/yr rainfall classes. However, site tree-form regressions in the other two rainfall classes with >1 site (500-550 and 650-700 mm/yr) were significantly different (P<0.001). Yet, tree-form regressions for the lowest, mid-range and highest rainfall classes (450-500, 700-750 and 800-850 mm/yr) were significantly different (Figure 4-11).

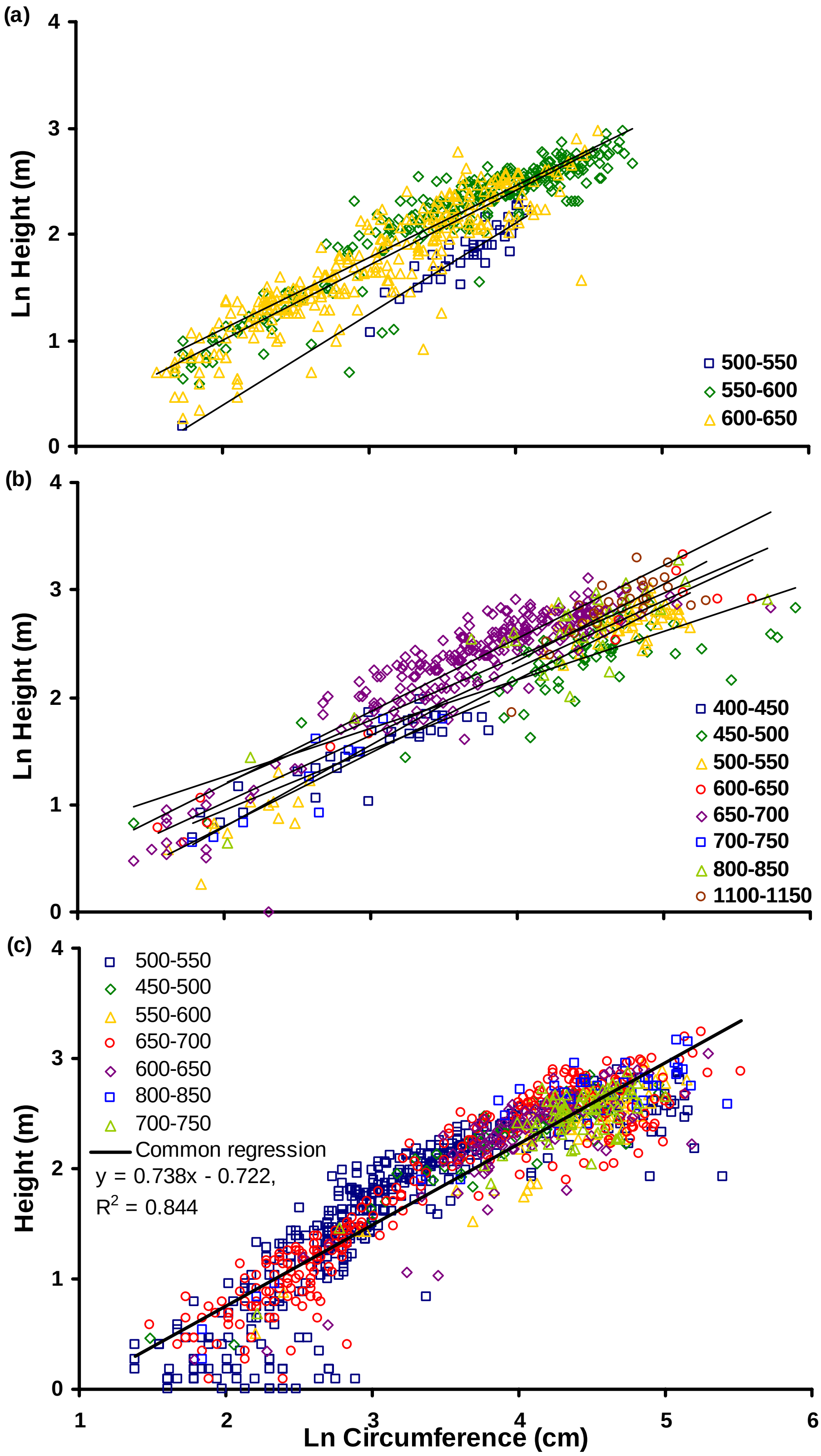

Figure 4-8. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for mean annual rainfall increments in (a) *C. glaucophylla*, (b) *E. populnea*, and (c) *E. melanophloia* communities. (Independent regressions in figure (a) and (b) were significantly different, P<0.001, and figure (c) were not significantly different, P >0.001).

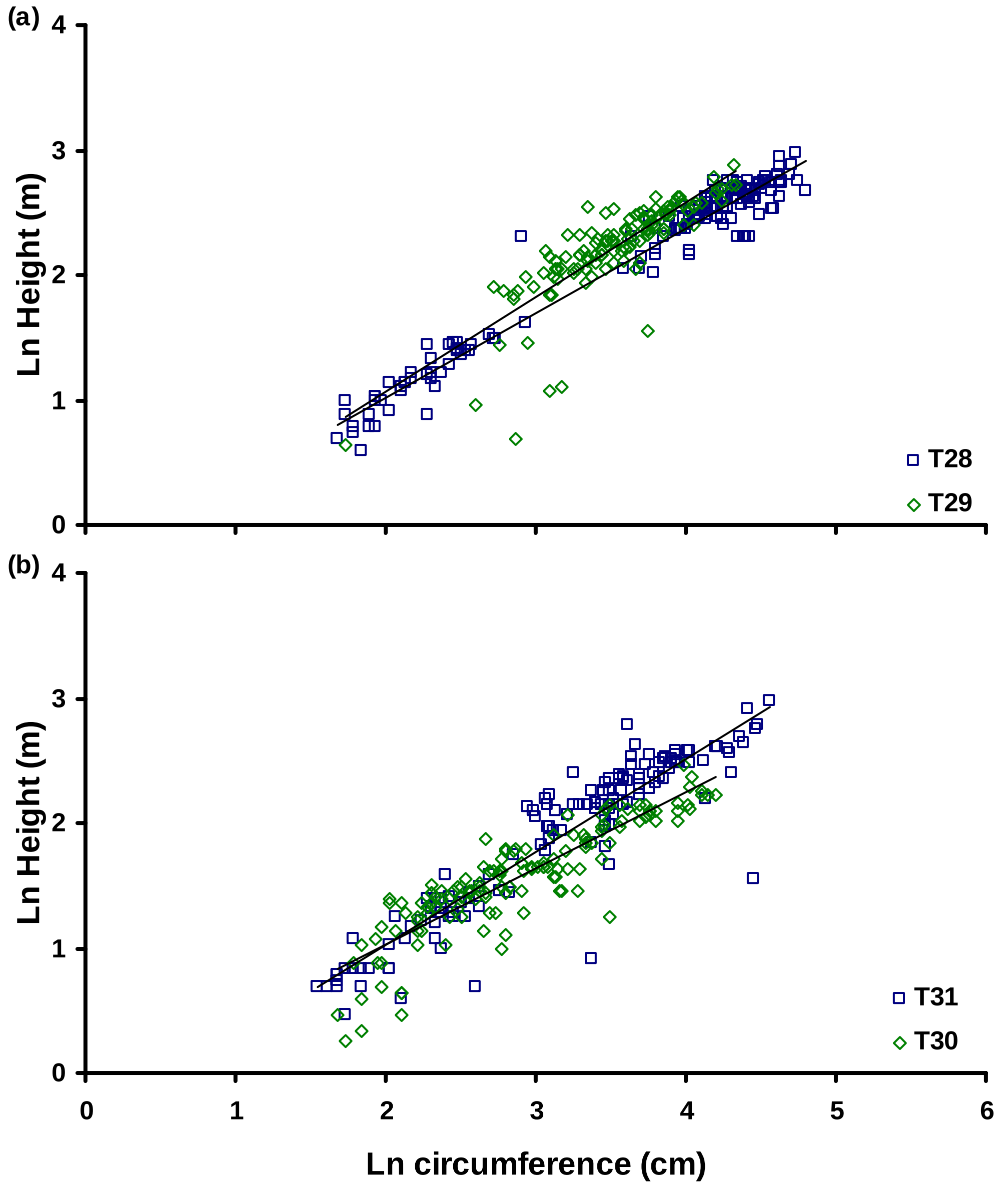

Figure 4-9. Lognormal regressions of stem circumference (cm) measured 30 cm above ground level and tree height (m) for (a) 550-600mm/yr and (b) 600-650mm/yr mean annual rainfall increments in *C. glaucophylla* communities. (Independent regressions in figure (a) were parallel, ie, slopes were not significantly different, P>0.001 and intercepts were significantly different, P<0.001 and in figure (b) independent regressions were significantly different, P<0.001).

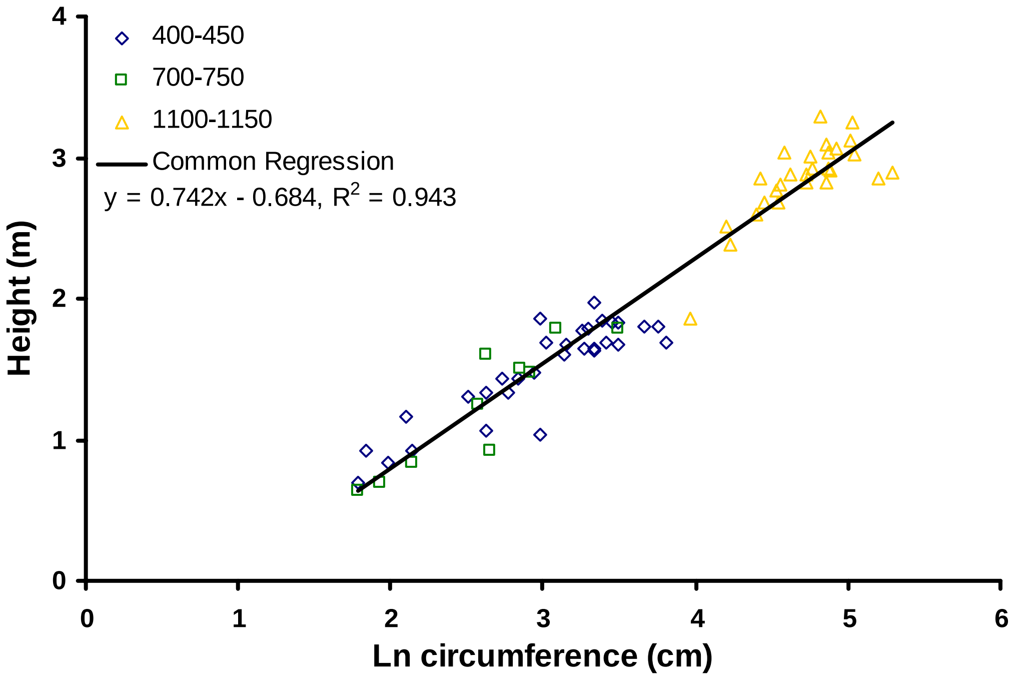

Figure 4-10. Lognormal regression of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites with mean annual rainfall increments of low, mid-range and high rainfall classes (400-450 mm/yr, 700-750 mm/yr and 1100-1150 mm/yr) in *E. populnea* communities. (Independent regressions were not significantly different, P>0.001).

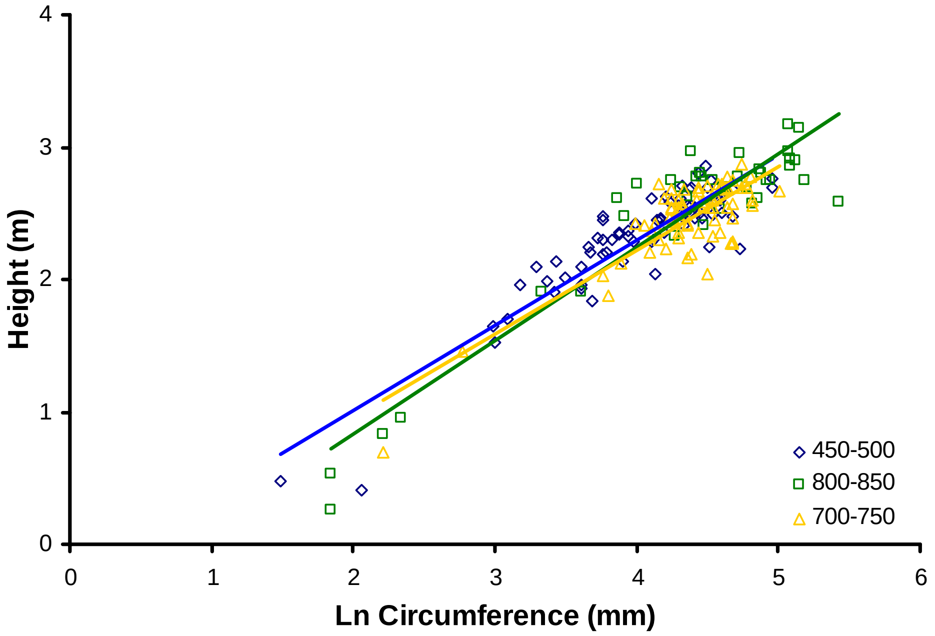

Figure 4-11. Lognormal regression of stem circumference (cm) measured 30 cm above ground level and tree height (m) for individual sites with mean annual rainfall increments of low, mid-range and high rainfall classes (450-500 mm/yr, 700-750 mm/yr and 800-850 mm/yr) in *E. melanophloia* communities. (Independent regressions were significantly different, P<0.001).

4.2.3 Discussion

Neither soil type nor rainfall appears to have influence on tree-form relationships for the tree species *E. populnea*, *E. melanophloia* and *C. glaucophylla*.

4.2.3.1 Tree form regressions

The correlation between ln stem circumference (cm) and ln tree height (m) for individual sites suggest that tree-form may be used to investigate the 'form' trends in north-eastern Australian woodlands. Furthermore, the correlations combining monitoring sites for each woodland species suggest that a single tree-form regression could represent each species. However, the sites within the woodland species groups do have significantly different tree-form regressions, initially suggesting that environmental characteristics of individual sites may influence the regressions. This observation is consistent with (Keith *et al.*, 2000; Nelson *et al.*, 1999). Further investigation was required to determine the relationships of regressions at sites with different environmental characteristics.

4.2.3.2 The effects of soil type on tree-form of some woodland species

Soil type may influence tree-form regressions. Initially, similar tree circumferences at sites with different soils appeared to have different heights (Figure 4-7b). However, if soil type alone influences tree-form regressions then it would be a reasonable expectation that the sites representing each of the soil groups would have similar trends to each other. *E. populnea* sites on uniform soils had similar tree-form regressions. However, only one site occurred on uniform soil types for the other two woodland species. Apparent similarities of the *E. populnea* sites may be due to either limited site numbers or the limited height overlap of the sites (Figure 4-6). Therefore, similarities should be treated cautiously, as comparing tree-form regressions that have different height ranges is not valid (Henry and Aarssen, 1999) and the resultant trend may merely reflect differences in stand development, age or other environmental factors.

Larger numbers of monitoring sites were available for comparison of duplex soils for all three woodland species studied. Regressions for *E. populnea* monitoring sites on duplex soils had similar slopes, although the number of these sites was still limited. Regressions for *C. glaucophylla* and *E. melanophloia* sites on duplex soils were significantly different. Similarly, regressions for *E. melanophloia* or *E. populnea* sites on gradational soil types were significantly different. These results indicated that, despite initial trends, soil type appears to have little influence on tree-form.

The trends of regressions, with respect to soil type, within the three woodland communities were inconsistent and indicate that other factors may have a greater influence on tree-form. Thus, soil type has very little effect on tree-form regressions of these species in the north-eastern Australian woodlands.

4.2.3.3 The effects of mean annual rainfall on the tree-form of woodland species

Similarities of slopes of *C. glaucophylla* tree-form regressions indicated that rainfall may have an effect on tree-form. Low rainfall sites (500-550 mm/yr) had the lowest intercept indicating that the trees have a smaller height than trees with a similar stem circumference at a higher rainfall. In contrast, the highest rainfall group (600-650 mm/yr) has the second lowest intercept. However, if rainfall influenced tree-form regressions then this intercept would be expected to be one of the highest intercepts. Thus, rainfall appears have some influence on the tree-form of *C. glaucophylla* trees however, another factor, or combination of factors, appear to have a far stronger influence.

Regressions for *C. glaucophylla* sites, with MAR of 600-650 mm/yr, had similar slopes. These two sites were geographically close (i.e. 10 km apart on the same property) and, in terms of basal area, one site was co-dominant for *C. glaucophylla* and the other was sub-dominant to *E. populnea*. Hence, differences in tree-form regressions appear to reflect an effect of stand structure rather than an effect of environmental factors. There was no effect on *C. glaucophylla* tree-form within a MAR of 550-650 mm/yr, as tree-form regressions were not significantly different. Additionally, *C. glaucophylla* was represented by a limited number of sites in a limited rainfall range (500-650 mm/yr) and trends in regressions with regards to rainfall classes were inconsistent.

The *E. populnea* sites had the largest rainfall range (400 – 1150mm/yr) and when grouped into the eight rainfall classes, regressions were not similar, indicating that rainfall may effect regressions of *E. populnea*. Additionally, if MAR alone influenced tree-form regressions it would be expected that monitoring sites within each rainfall class would have similar regressions. This holds true for *E. populnea* sites. However, these results need to be treated cautiously as all sites in the 650-700 mm/yr rainfall class are located geographically within a 20 km radius and it is likely that they would have similar forms. The numbers of sites in each of the remaining rainfall classes were limited and these trends may not be representative of the communities within this rainfall class.

The trends in *E. populnea* were also unexpected. A low rainfall class had the highest intercept and conversely, the highest rainfall class had one of the lowest intercepts.

This is indicative of trees growing taller at the drier site compared with a tree of similar circumference at the wetter site, which is contrary to results reported for eucalypts in southern Australia (Hassal and Associates, 1999), India (Tewari *et al*., 2002) and trees in tropical forests (Brown *et al*., 1989). Furthermore, the low rainfall sites have, on average, greater tree densities than the high rainfall sites, and this supports reports of high density stands having greater heights for the same circumference compared with trees in low density stands (Niklas, 1995; Henry and Aarssen, 1999). Hence, stand density appeared to have a greater influence on tree-form than rainfall.

Regressions for *E. populnea* were increasingly inconsistent when rainfall classes were compared. *E. populnea* regressions with low, medium and high rainfall classes (400-450, 700-750, 1100-1150 mm/yr) have similar regressions despite very different MAR (Figure 4-10). These similarities should be treated cautiously, as comparing tree-form regressions that have different height ranges is not valid (Henry and Aarssen, 1999) and the resultant trend may merely reflect differences in stand development, age or other environmental factors. Conversely, *E. populnea* regressions of MAR classes with a range 450-750 mm/yr were different. Trends within the *E. populnea* communities in regards to rainfall classes were also inconsistent.

Regressions for *E. melanophloia* sites were similar, and hence not influenced by rainfall. Initially, sites in the 500-550 mm/yr class were not represented by similar regressions, yet when one of these sites was removed from the analysis the regressions were similar. This site was omitted because a storm had removed the tops of most of the mature trees, hence leaving a 'flat' regression. However, once again caution should be applied when considering these trends as some rainfall classes consisted of sites with a small overlapping range of trees, which may result in misleading trends (Henry and Aarssen, 1999).

Different trends within rainfall classes for each of the woodland species suggest these species may respond differently to increasing MAR. Initially, rainfall seemed to have some influence on tree-form regressions; however this influence was the opposite of the trends reported for eucalypts (Hassal and Associate, 1999; Henry and Aarssen, 1999; Tewari *et al*., 2002). Inconsistency of trends within, and between, the species studied suggest that another factor such as stand structure, or a combination of factors, have a far greater influence than any single factor. Mean annual rainfall, on its own, appeared to have very little effect on the tree-form regressions of these species in the north-eastern Australian woodlands.

4.2.3.4 Rainfall and Soil Combinations and locations

No single factor has been identified as the cause of differing regressions. However, if both rainfall and soil type influence tree-form, as stated by Brown *et al*. (1989), Feller (1980) and Ter-Midaelian and Korzukhin (1997), it would be expected that regressions for the same species, with similar rainfall classes and different soil types (or vice-versa, same soil types and different rainfall classes) would have different trends, or at most, have similar slopes.

This appeared to be true for the *C. glaucophylla* sites, which had a limited rainfall range (500-650 mm/yr). Sites with higher rainfall and the same soil type had similar regressions and the pooled regression was significantly different to a regression with lower rainfall and different soil type. This supports the hypothesis that soil and rainfall have a synergistic effect on tree-form. Conversely, previous results have highlighted that in some cases *C. glaucophylla* sites with the same rainfall class and soil type have significantly different tree-form regressions. However, this may be a result of limited site numbers being compared rather than reflecting environmental effects on tree-form relationships.

The *E. melanophloia* sites in the 650-700 mm/yr class had two different soil types and different tree-form regressions. This finding supports the hypothesis that soil type and rainfall collectively may affect tree-form. Conversely, some sites have similar tree-form regressions while environmental factors were different. The *E. populnea* sites in the 650-700 mm/yr class follow this trend, where sites have different soil types yet the regressions were similar. This trend was repeated in *E. melanophloia* sites, which had similar regressions for MAR classes, yet soil types were different. The *E. populnea* sites with MAR classes of 400-450, 700-750 and 1100-1150 mm/yr had similar regressions despite having different soil types while representing the range of mean annual rainfall experience by this species. Therefore, it is concluded that soil type and rainfall appear to have little effect on tree-form regressions.

It is apparent from this study that the influences of mean annual rainfall and dominant soil type on regressions were inconsistent. There was no persistent evidence that regressions reflected the selected environmental factors, or combinations of factors, investigated in this study. Regressions are probably a consequence of a combination of environmental factors, stand density and management regimes, which resulted in the variety of tree-form regressions encountered in this study and reflect the range of communities in north-east Australian woodlands. However, the influence of tree-form on allometric regressions used to estimate biomass remains unknown.

4.3 Study 2: The relationship between tree-form and biomass allometric regressions

4.3.1 Material and Methods

4.3.1.1 Biomass regressions

Aboveground biomass regressions were derived from nine data sets which represented four native woodland species, *C. glaucophylla*, *E. populnea*, *E. melanophloia* and *E. crebra* from six districts. Four of these districts (Duaringa, Jericho, Dingo and Rockhampton) were located in central Queensland while the other two (Injune and Mitchell) were located in south-west Queensland.

For *E. populnea* five data sets represent a range of communities, including:

- mature stand (Dingo)
- regrowth stand (adjacent to mature site above)
- a coastal mature stand (Rockhampton),
- an inland mature stand (Roma),
- a *C. glaucophylla* invaded stand (Mitchell).

For *E. melanophloia* three data sets represent communities including:

- adjacent mature and regrowth stands (Jericho)
- regrowth stand (adjacent to mature site above)
- a *C. glaucophylla* invaded stand (Mitchell).

For *E. crebra* one data set represents a mature stand at Duaringa (Table 4-3).

Table 4-3. Harvest site details and stand types for nine data sets used to develop biomass regressions.

No.	Species	n	Circumference Range (cm)	Stand type and Region
H1	E. crebra	20	8.0 – 202.0	Mature - Duaringa
H2	E. melanophloia	20	4.0 – 165.0	Mature - Jericho
H3	E. populnea	20	14.6 – 240.0	Mature - Dingo
H5	E. melanophloia	31	4.0 – 43.0	Regrowth - Jericho
H6	E. populnea	7	13.5 – 194.0	Mature - Mitchell
H7	E. melanophloia	5	25.5 – 161.7	Mature - Mitchell
H8	E. populnea	24	6.8 – 56.2	Regrowth - Dingo
H9	E. populnea	10	10.1 – 223.5	Mature - Rockhampton
H10	E. populnea	11	5.0 – 161.8	Mature - Injune

Mature communities had no known clearing in the previous 30 years, whereas regrowth communities were those developed after mature woodlands were pulled with a ball and chain between two bulldozers. Additionally, three of the *E. populnea* sites represented stands with contrasting MAR. The Rockhampton site receives a MAR of 1098 mm/yr; 60% higher than the Dingo site (677 mm/yr) while the Injune site receives a MAR of 602 mm/yr and, although described as a mature site, has had some past clearing prior to 30 years previous.

4.3.2 Analysis of tree form and biomass regressions

Aboveground biomass of individual trees was estimated at the harvest sites using destructive harvesting techniques, outlined in chapter three of this study, and in Burrows *et al.* (2000). Tree-form regressions and biomass allometric regressions (ln tree biomass (kg) × ln stem circumference (cm)) were calculated using least-squares analysis for the harvest data sets.

Analysis of covariance was performed on linear regressions to determine if slopes were different, and then if found to be similar it was determined if y intercept of the regressions were significantly different. Differences in tree-form were compared to differences in biomass.

The trends of significant differences between biomass regressions for the harvest sites were investigated. Trends within the same species at different locations, between different species and different stand types were compared.

Common allometric biomass regressions were calculated using least-squares analysis. Regression between actual biomass and predicted biomass (1:1 regression) was calculated using least-squares analysis to further determine the strengths of a common regression. The residual errors from the common biomass regression and the site-specific biomass regressions were compared by F-test.

4.3.3 Results

4.3.3.1 Tree-form regressions.

Strong correlations were demonstrated between ln stem circumference (cm) and ln height (m) for each of the harvested sites, with R^2 ranging from 0.846 to 0.977 (Table 4-4).

Table 4-4. Lognormal regressions of circumference (cm) at 30cm from ground level and tree height (m) (tree-form) at nine sites in Queensland's woodlands. Regressions are in the form y = ax + b, where y = ln tree height (m), x = ln circumference (cm), n = sample number, R^2 = coefficient of determination.

Site	a	b	R^2	n
H1	0.756	-0.696	0.932	20
H2	0.707	-0.605	0.902	20
H3	0.521	0.390	0.944	20
H5	0.643	-0.652	0.919	31
H6	0.576	0.021	0.973	7
H7	0.688	-0.562	0.846	5
H8	0.800	-0.708	0.879	24
H9	0.775	-0.835	0.929	10
H10	0.664	-0.346	0.977	11

Regressions for harvest sites that represented mature *E. populnea* and *E. melanophloia* communities were not significantly different (P>0.001)(Figure **4-12**a and Figure **4-12**b, respectively). Regressions for harvest sites that represent mature and regrowth *E. populnea* communities at the Dingo site were significantly different (P<0.001)(Figure **4-12**c). Regressions for harvest sites that represent mature and regrowth *E. melanophloia* communities at the Jericho site had similar slopes (P>0.001)(Figure **4-12**d).

Regressions that represent mature *Eucalyptus* spp. communities at sites (Dingo, Duaringa, Jericho, Mitchell, Rockhampton and Injune) were not significantly different (P>0.001)(Figure **4-12**e). Regressions that represent regrowth *Eucalyptus* spp. communities at sites (Dingo and Jericho) had similar slopes (P>0.001)(Figure **4-12**f).

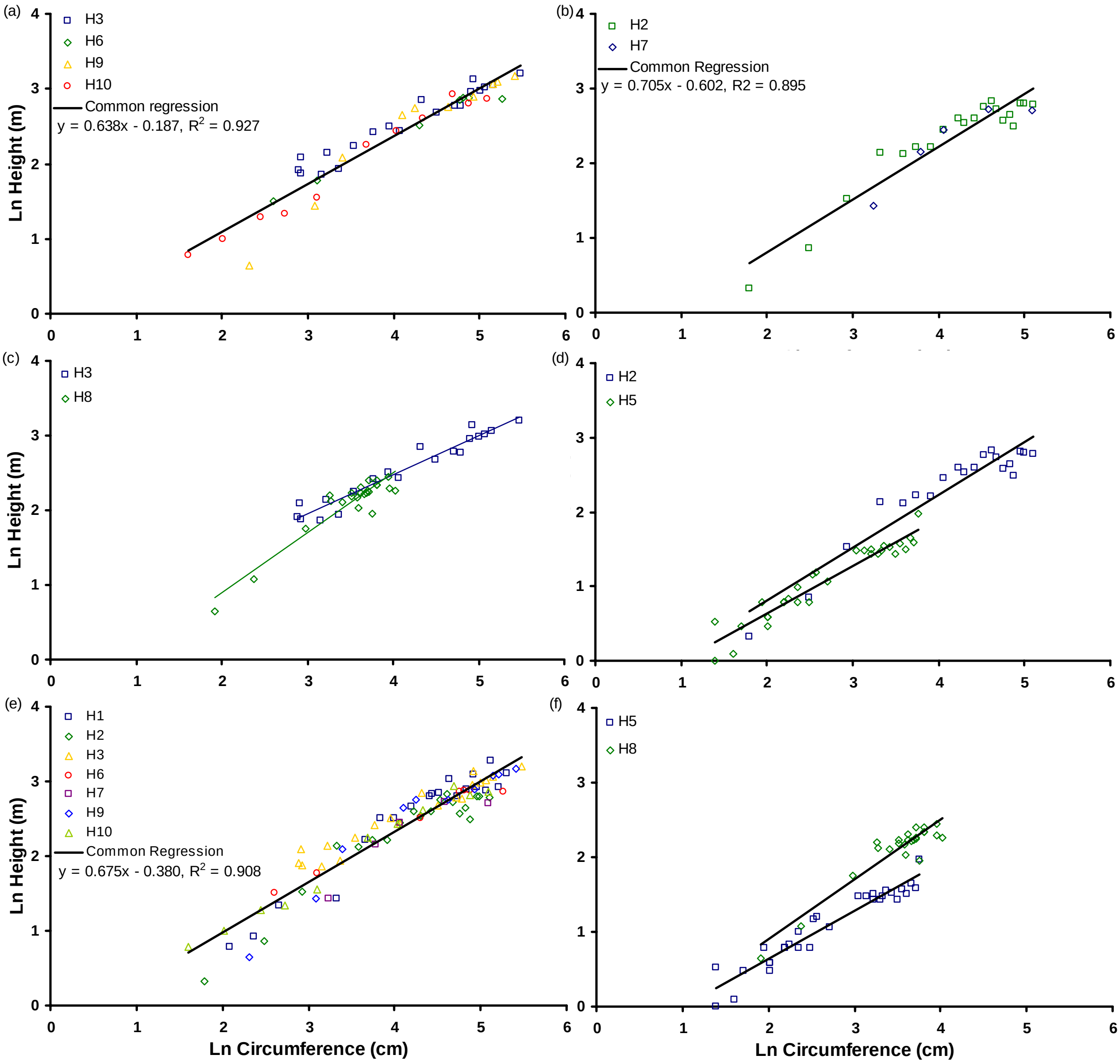

Figure 4-12. Lognormal regression of stem circumference (cm) at 30 cm from ground level and tree height (m) for nine sites representing (a) mature *E. populnea*, (b) mature *E. melanophloia*, (c) adjacent regrowth and mature *E. populnea*, (d) adjacent regrowth and mature *E. melanophloia*, (e) all mature *Eucalyptus* spp. and (f) all regrowth *Eucalyptus* spp. communities. (Independent regressions in figures (a), (b) and (e) were not significantly different, P>0.001, independent regressions in figures (d) and (f) were not significantly different in slope, P>0.001, but were significantly different intercepts, P<0.001, and independent regressions in figure (c) were significantly different, P<0.001). Independent regression descriptions are in Table 4-4.

4.3.3.2 Biomass regressions

Strong correlations (R^2 ranging from 0.963 to 0.997) were demonstrated between ln stem circumference (cm) and ln biomass (kg) for each site (Table 4-5).

Table 4-5. Lognormal regressions of circumference (cm) and biomass (kg) at nine sites in Queensland's woodlands. Regressions are in the form y = a + bx, where y = ln dry weight (kg), x = ln circumference (cm), n = sample number. R^2 = coefficient of determination. The residual standard deviation (RSD) and sum of squares of the deviations in x (SSDx) values are utilised in applying the antilogarithm correction factor (Baskerville, 1972) for estimating community biomass by using these regressions (Chapter 2 and 3).

Site	a	b	R^2	n	RSD	SSDx
H1	-6.505	2.756	0.987	20	0.309	17.456
H2	-6.553	2.726	0.991	20	0.236	15.385
H3	-4.907	2.382	0.994	20	0.164	14.001
H5	-5.209	2.325	0.967	31	0.326	16.516
H6	-5.376	2.524	0.997	7	0.164	6.034
H7	-5.547	2.499	0.963	5	0.403	2.037
H8	-5.892	2.588	0.988	24	0.144	5.438
H9	-5.160	2.414	0.986	10	0.309	9.550
H10	-5.746	2.568	0.996	11	0.196	14.471

Biomass regressions for sites representing mature *E. populnea* and *E. melanophloia* communities were not significantly different ($P>0.001$)(Figure 4-13a and Figure 4-13b). Regressions for sites representing mature and regrowth *E. populnea* communities at Dingo had similar slopes ($P>0.001$)(Figure 4-13c). Regressions for sites representing mature and regrowth *E. melanophloia* communities at Jericho were significantly different ($P<0.001$)(Figure 4-13d). Regressions that represent mature *Eucalyptus* spp. communities at all sites (Dingo, Duaringa, Jericho, Mitchell, Rockhampton and Injune) were significantly different ($P<0.001$) (Figure 4-13e). Regressions that represent regrowth *Eucalyptus* spp. communities at Dingo and Jericho were not significantly different ($P>0.001$)(Figure 4-13f).

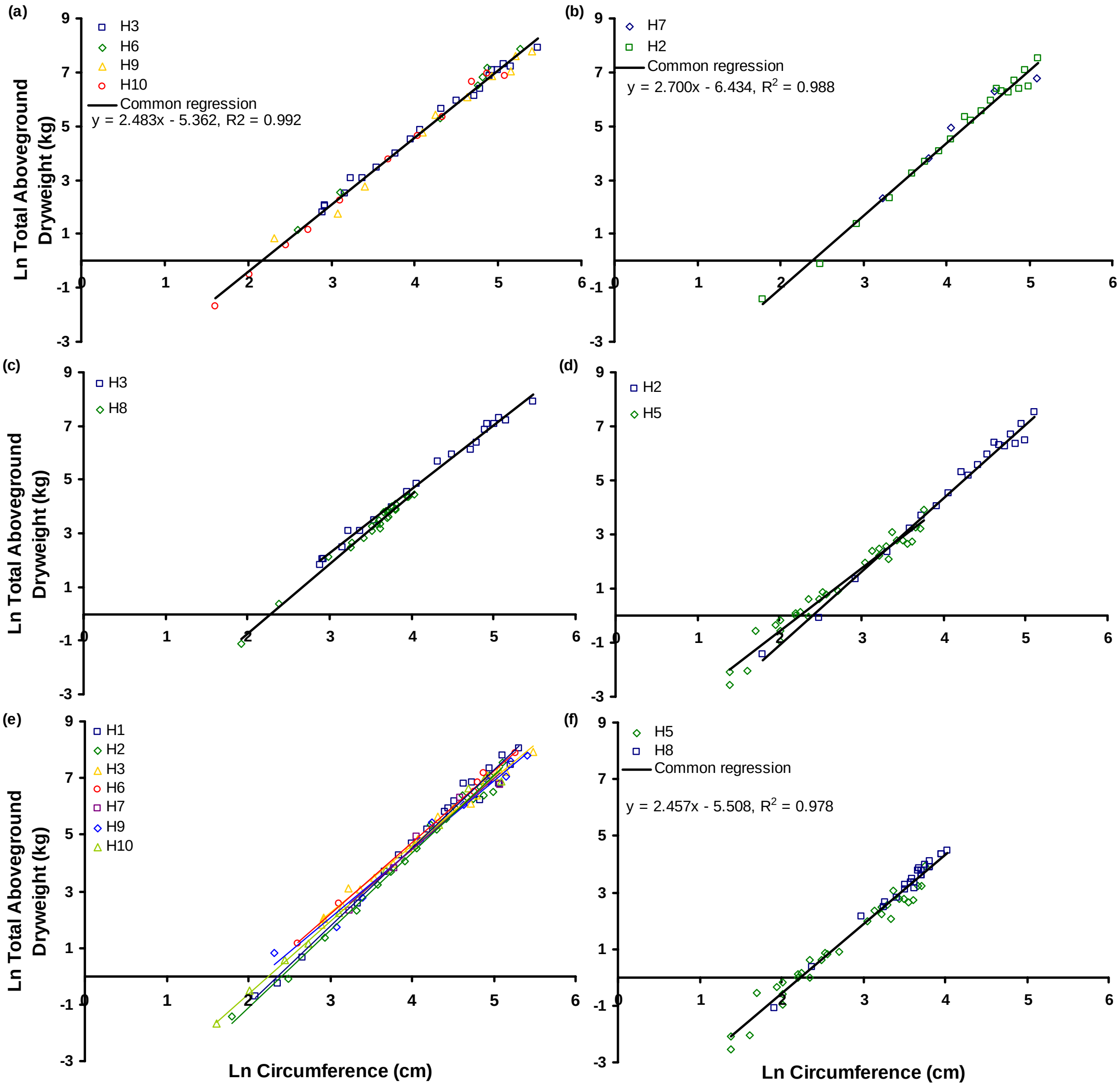

Figure 4-13. Lognormal regression of stem circumference (cm) measured 30 cm above ground level and biomass (kg) for harvest sites representing (a) mature *E. populnea*, (b) mature *E. melanophloia*, (c) adjacent regrowth and mature *E. populnea*, (d) adjacent regrowth and mature *E. melanophloia*, (e) all mature *Eucalyptus* spp. and (f) all regrowth *Eucalyptus* spp. communities. (Independent regressions in figures (a), (b) and (f) were not significantly different, P>0.001, independent regressions in figures (c) were not significantly different in slope, P>0.001, but were significantly different intercepts, P<0.001, and independent regressions in figure (d) and (e) were significantly different, P<0.001).

Biomass regressions that represent mature (pooled data from *E. populnea*, *E. melanophloia* and *E. crebra* sites) and regrowth (pooled data from both *E. populnea* and *E. melanophloia* regrowth sites) *Eucalyptus* spp. communities at sites were not significantly different (P>0.001) (Figure 4-14).

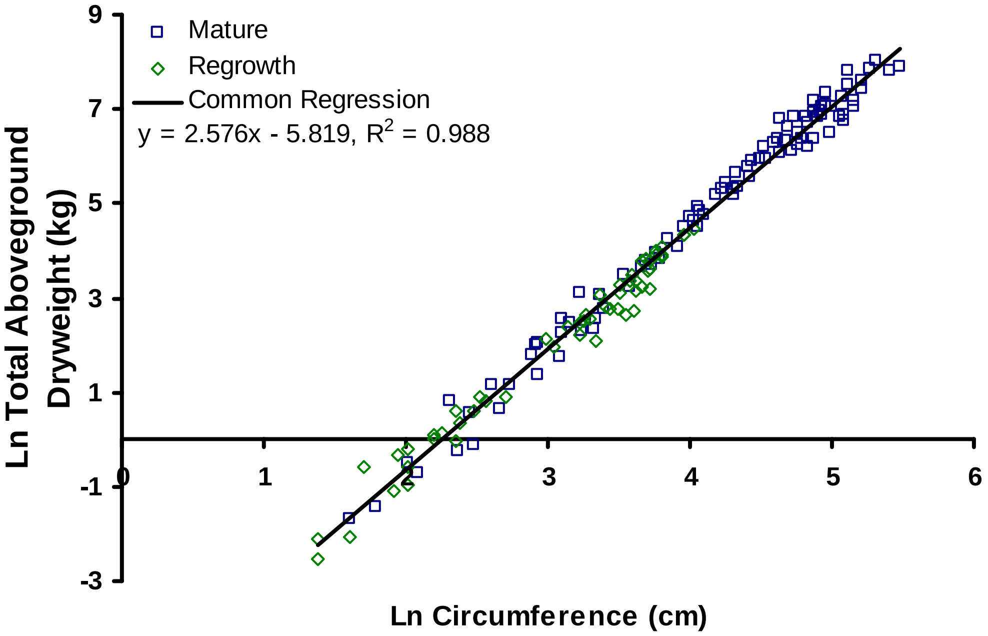

Figure 4-14. Lognormal regression of stem circumference (cm) measured at 30 cm from ground level and biomass (kg) for pooled sites representing mature and regrowth *Eucalyptus* spp. communities. (Independent regressions were not significantly different, P>0.001).

4.3.3.3 Comparison of tree-form regression trends and biomass trends.

The trends of ANOVA for tree-form and biomass regressions were compared to determine if tree-form reflected similarities or differences in biomass allometric regressions. The relationship between circumference and height was quite variable compared with the tight relationship between circumference and aboveground biomass (Table 4-4 and 4-5). Implying that statistical differences would require large differences between tree-form regressions compared with the small differences required between biomass regressions. It was expected that if tree-form had some influence on biomass regressions then similarities in trends would exist. Trends were inconsistent; for example, for all *E. populnea* sites, tree-form regressions were significantly different, however biomass regressions were not significantly different. Conversely, both tree-form and biomass regressions for mature *E. populnea* harvest sites were not significantly different (Table 4-6).

Table 4-6. Analysis of covariance of tree-from and biomass regressions for Eucalyptus spp. at the nine harvest sites (significance levels of 0.001).

Regression Grouping	Tree-form Slope	Intercepts	Biomass/Harvest Slope	Intercepts
All Harvest Sites	Common	Different	Different	N/A
All Mature Sites	Common	Common	Different	N/A
Regrowth	Common	Different	Common	Common
Dingo	Different	N/A	Common	Different
Jericho	Common	Different	Different	N/A
E. populnea Mature sites	Common	Common	Common	Common
E. melanophloia Mature sites	Common	Common	Common	Common
All E. melanophloia sites	Common	Different	Common	Common
All E. populnea sites	Different	N/A	Common	Common
Mature and Regrowth	Different	N/A	Common	Common

N/A = not applicable

4.3.3.4 Common biomass allometric regressions

Strong correlations (R^2 ranging from 0.978 to 0.988) were demonstrated between ln circumference and biomass for the pooled harvest data for mature, regrowth and all harvested sites (Table 4-7).

Table 4-7 Common lognormal regressions of circumference (cm) and biomass (kg) for *Eucalyptus* spp. communities in Queensland's woodlands.

Description	a	b	R^2	n	RSD	SSDx
Mature	-5.824	2.585	0.987	93	0.289	83.415
Mature (without H6 & H7)	-5.854	2.589	0.987	81	0.290	75.139
Regrowth	-5.508	2.457	0.978	55	0.281	30.956
All sites	-5.819	2.576	0.988	148	0.292	151.834

Strong correlations were demonstrated between ln actual biomass and ln predicted biomass for common regressions, including the mature regression in which sites were statistically different, with R^2 ranging from 0.978 to 0.988 (Table 4-8). Mitchell sites, H6 and H7, were significantly different in this suite of regressions. The relationship between predicted biomass values from common mature regression and actual values from site-specific regressions (1:1 regression) accounted for 98.7% of variation for all mature harvest sites (Figure 4-15). The 1:1 regression was calculated for harvest sites H6 and H7 and this correlation was also strong, with R^2 of 0.984 and a slope and intercept of 1.008 and 0.130, respectively. The slope was not significantly different ($P>0.001$) from one and the intercept was not significantly different ($P>0.001$) from zero.

The common regression, incorporating all harvest sites (both mature and regrowth) accounted for 98.8% of variation of biomass data, slope and intercept were 0.988 and 0.045, respectively (Figure 4-16). The slope was not significantly different ($P>0.001$) from one and the intercept was not significantly different ($P>0.001$) from zero. The variance of the residual errors of the common biomass regression and site-specific biomass regressions were not significantly different ($P>0.001$).

Table 4-8. Lognormal regressions of actual biomass (kg) and predicted biomass (kg) for groupings of *Eucalyptus* spp communities. Regressions are in the form y = a + bx, where y = ln predicted biomass (kg), x = actual biomass (kg), n = sample number. R^2 = coefficient of determination.

Description	a	b	R^2	n
Mature	0.049	0.983	0.987	93
H6 & H7	0.130	1.008	0.984	12
Regrowth	0.001	1.025	0.978	55
All sites	0.045	0.988	0.988	148

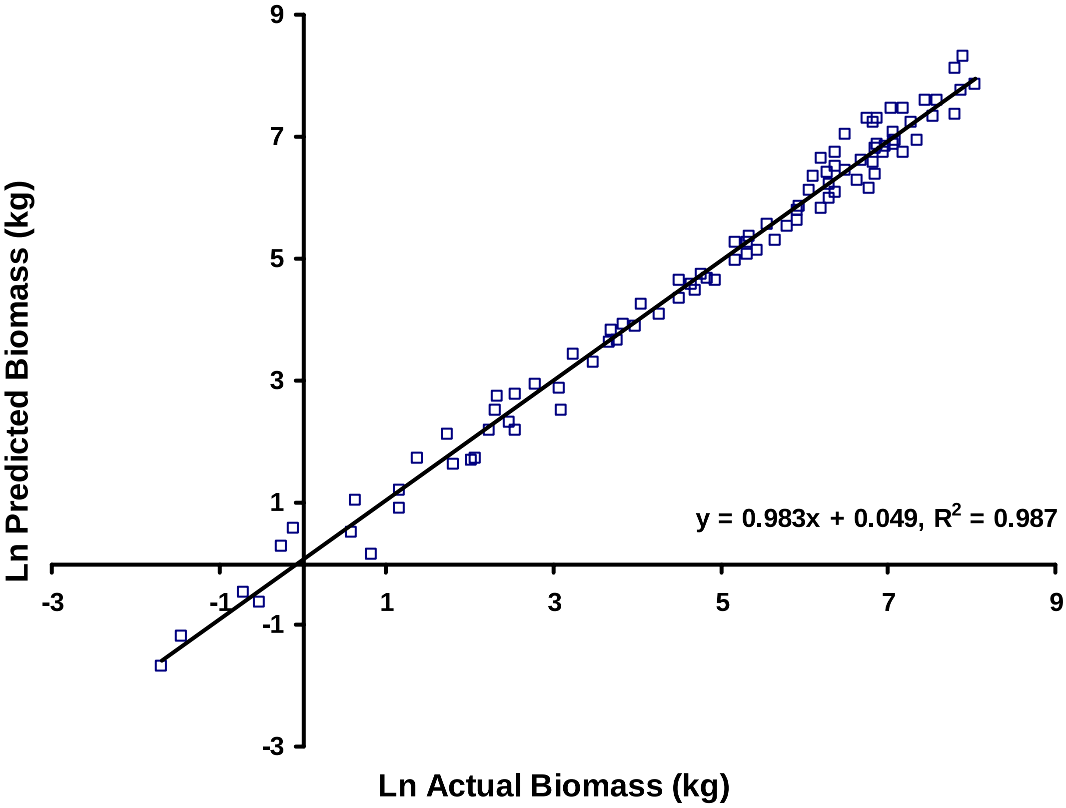

Figure 4-15. Lognormal regression of actual biomass (kg) and predicted biomass (kg) for mature *Eucalyptus* spp. communities

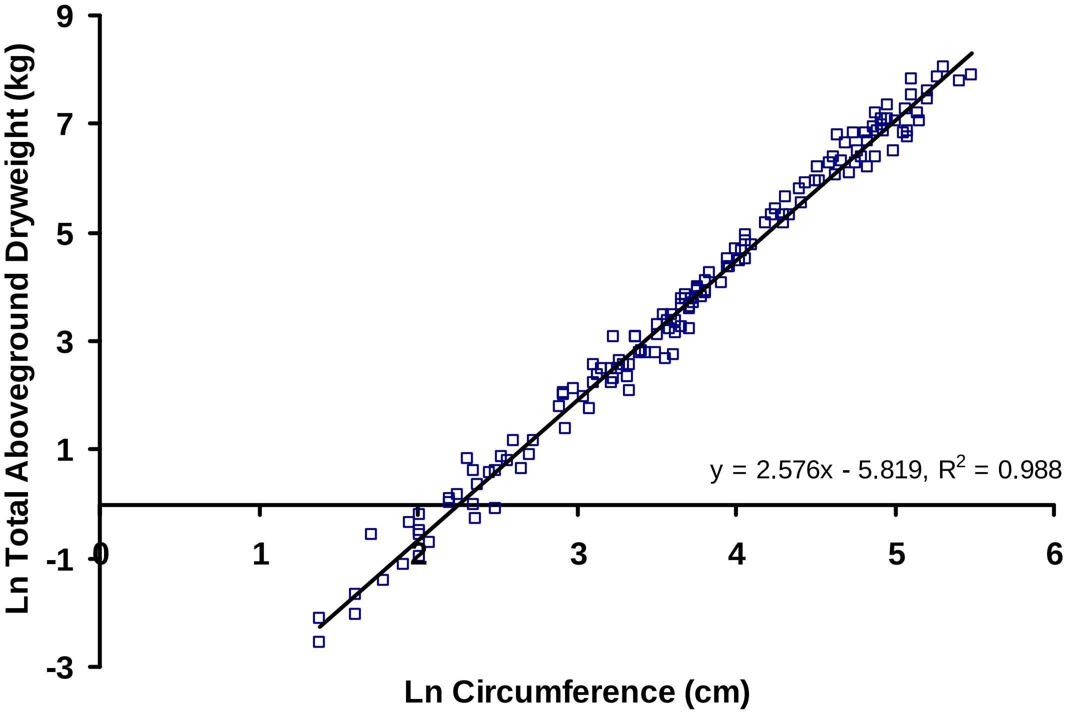

Figure 4-16. Lognormal regression of stem circumference (cm) measured 30 cm above ground level and biomass (kg) for pooled harvest sites representing *Eucalyptus* spp. communities.

4.3.4 Discussion

Rainfall, soil type and stand structure appear to have little influence on tree-form relationships, and subsequently little effect on the biomass relationships.

4.3.4.1 Relationship between tree-form and biomass

Researchers place much importance on the relationship between tree-form and biomass regressions (Feller, 1980; Brown, 1997; Keith *et al.*, 2000; Nelson *et al.*, 1999) and it is generally accepted that changes in tree-form regressions reflect similar changes in biomass regressions. For example, it is assumed that two communities with different tree-form relationships will also have different biomass regressions, and likewise, communities with similar tree-form regressions would be expected to have similar biomass regressions (Medhurst *et al*., 1999). This study investigated whether this holds true for eucalypt species in different environments of North-east Australia.

For the mature *E. melanophloia* sites both tree-form and biomass regressions were similar. The same trend occurred in the *E. populnea* sites and these results support the theory that changes in tree-form will impact biomass relationships. Conversely, the *E. populnea* sites (mature and regrowth) had different tree-form regressions and yet biomass regressions were similar. Large differences in tree-form relationships were required to produce a statistical difference due to larger variation within relationships when compared to the biomass relationships, these results do not support the theory that tree-form reflects differences in biomass regressions. Similar findings of conflicting trends between tree-form regressions and biomass regressions occurred in the *E. melanophloia* sites (mature and regrowth), regrowth stands (*E. populnea* and *E. melanophloia*), and mature eucalypt sites (*E. populnea*, *E. melanophloia* and *E. crebra*).

Trends from data suggest that similarities or differences in tree-form regressions are not a reliable indicator of potential differences in biomass regressions. Height and circumference growth increments respond differently to stand density and management (Cancino *et al*., 1999), which results in changes to tree-form regressions as stands develop. Additionally, these regressions change with increasing stand age, with younger stands having steeper curves than older stands (Eerikainen, 2002). Different site characteristics may influence the growth rates of species (West and Mattay, 1993), however as tree mass increases through increased height, stem girth also increases to maintain the mechanical strength required to support the additional mass (Niklas, 1994; Enquist, 2002). Many environmental factors may combine with stand characteristics, such as density, management, age and history, to influence tree-form relationships. Additionally, tree-form regressions do not reflect similarities of biomass allometric regressions in grazed eucalypt woodlands of North-east Australia.

4.3.4.2 Common Biomass relationships

Burrows *et al*. (2000) reported that a common biomass regression can be used with confidence for three mature stands of *E. populnea*, *E. melanophloia* and *E. crebra* in Central Queensland. However, the validity of applying a common regression across a wider range of Queensland's woodlands was not investigated by Burrows *et al*. (2000). The current study investigates the validity of applying a common regression across grazed woodlands in north-east Australia.

Biomass regressions for four mature *E. populnea* sites were not significantly different ($P>0.001$), despite having different site characteristics and stand histories. A similar finding was reported for the mature *E. melanophloia* sites. Conversely, the mature eucalypt site biomass regressions were significantly different, with the Mitchell sites (H7 and H6) being different from the remaining regressions. Despite being significantly different, the actual vs. predicted biomass regression had a slope of 0.98 and an intercept of 0.13, closely approaching a 1:1 line (slope not significantly different from one, $P>0.001$ and intercept not significantly different from zero, $P>0.001$). When the circumference range was reduced, as recommended by Henry and Aarssen (1999), the regressions were not significantly different, despite having different site characteristics, stand histories and species. Hence, a common biomass regression was adequate for estimating aboveground biomass for mature eucalypt stands across a wider range of Queensland's woodlands than investigated by Burrows *et al*. (2000).

Two *Eucalyptus* spp. regrowth sites were located in two different regions. Despite regional, environmental and age differences, biomass regressions were similar at these two sites. Hence, a common biomass regression may also be used to estimate aboveground biomass in *Eucalyptus* spp. regrowth communities.

Results from this study are consistent with similar findings in biomass regression studies of plantations, tropical forests and eucalypt forests (Lott *et al*., 2000; Senelwa and Sims, 1998; Hingston *et al*., 1981). Williams *et al*. (2005) demonstrated that a general allometric regression was applicable to a number of woodland and savanna species (including the mature species in this study) across a wide geographical range. Brown *et al*. (1989) successfully developed regional biomass regressions for tropical forests from several aboveground biomass regressions.

However, Nelson *et al*. (1999) reported that biomass estimates of secondary forests were overestimated by up to 60% when regressions with a single variable (diameter) were used that were derived from primary forests data in central Amazon. The *E. populnea* biomass regressions for adjacent mature and regrowth stands had similar slopes despite having different stand histories. In contrast, the *E. melanophloia* biomass regressions for adjacent mature and regrowth stands were different, supporting the findings of Nelson *et al*. (1999). However, the age of the *E. melanophloia* regrowth (5 years old) is younger than the *E. populnea* regrowth (14 years old). Therefore, the differences in the ranges of the measured variables for the regrowth compared to the mature trees contributing to their respective regressions were greater in the *E. melanophloia* stand than the *E. populnea* stand. The comparison of regressions with different height ranges si not valid (Henry and Aarssen, 1999). When the ranges of both *E. melanophloia* data sets were restricted to trees of approximately similar stem circumference ranges, the ensuing regressions were similar. Similarly, biomass regressions for *E. populnea* were similar when stem circumference ranges were restricted; hence biomass regressions were similar despite very different management regimes.

This is similar to the results of Montagu *et al*. (2005) where a common biomass regression for *Eucalyptus pilularis* stands in eastern Australia, inclusive of natural forests and plantations, was developed. Montagu *et al*. (2005) report that biomass regressions of young plantations of *E. pilularis* appeared to be different to biomass regressions of mature *E. pilularis* stands and these differences were due to differences in size distributions of stands and not to changes in biomass regressions. This general regression for *E. pilularis* was as accurate as the site-specific regressions for estimating aboveground biomass in either stand type. This emphasises the influence that stand age or sampling range can have when comparing biomass regressions of two different communities. In the current study, all *E. populnea* and *E. melanophloia* biomass regressions were similar despite having different stand structures and geographical ranges. Similarly, all mature sites were similar when size ranges were restricted to trees of approximately similar stem circumferences.

Conversely, regressions for all sites (both mature and regrowth for all species) were statistically different, yet circumference ranges of these independent data sets varied, with some sites having very limited circumference range. The R^2 values for independent biomass regressions were high and thus a small variance in the slope of any single regression may be determined as statistically different, when in reality the differences are more than likely very small (D. Reid, pers. comm. 2005). Thus, despite statistical differences, a common biomass regression for eucalypts appears robust enough to predict biomass for all of the harvest sites. In other words, the combined regression should provide a satisfactory estimate of community biomass for all practical purposes.

Further analysis established that the common biomass regression for regrowth eucalypt communities was not significantly different to the common biomass regression for mature eucalypt communities. Actual vs. predicted biomass regression had a slope of 0.99 and an intercept of 0.06, closely approaching a 1:1 line and hence a robust predictor of biomass for grazed woodlands. The variance of the residual errors of the common biomass regression and site-specific biomass regressions were similar. This infers that a common biomass regression can be used for regrowth and mature sites, provided that the regression is based on both stand types.

4.4 Conclusion

The influence of environmental factors on tree-form regressions was inconsistent for a range of eucalypt trees in north-east Australia. Despite initial findings suggesting that tree-form regressions (ln height × ln circumference) were dependent on environmental factors, the research reported here indicate that tree-form regressions were not an indicator of the effect of environmental factors on tree-form. The relationship between tree-form and biomass regressions was also inconsistent, suggesting that it is not essential to consider the effects of tree-form when estimating biomass of eucalypts in this study area.

This study has indicated that a common biomass regression based on the tree data from several sites provided a robust estimate of tree biomass. Thus, it is not necessary to establish new allometric regressions for estimating tree biomass on ‘new’ sites where the targeted woodland community is located within the broad species and environmental ranges examined in this study. These findings will contribute to increasing the efficiency and ease of estimating tree biomass in the study area and aid in the development of stand biomass regressions. However, there is limited knowledge of the effect of stand dominance, or stand structure on such relationships and this is investigated in the next chapter.

Chapter 5. Stand allometrics for Estimating Above-ground Biomass in North-eastern Australian Woodlands.

5.1 Introduction

Common regressions can be used to estimate the biomass of trees across a wide range of eucalypt communities to aid in the development of stand biomass regressions. However, the effects of differing stand dominance and structure on total aboveground biomass are unknown and hence, a reliable method of determining stand structure across the north-eastern woodlands of Australia is required.

Circumference distribution classes appear to be an unreliable tool to interpret stand structure or development. Mature forests possess circumference distribution curves that are typically a reverse-J curve (Leak, 1964; Moser, 1972). However, Goodburn and Lorimer (1999) reported that circumference distributions curves of uneven- and even-aged stands were also fitted with reverse-J curves. Rouviven and Kuuluvainen (2005) reported that natural stands could also be fitted with other curves. The validity of using this method of determining stand structure was further compounded by using size classes that were too large or uneven (Niklas *et al*., 2003; Leak, 1965). Due to the nature of woodlands, in some cases, the diameter distribution frequencies would be lacking due to the small numbers of individuals at some sites.

Brown and Bredencamp (2004) used height classes to investigate stand structure in the African savanna, while QDPI&F's Woodland Monitoring Group also use height classes to describe stand structure in Queensland's grazed woodlands (Burrows *et al.,* 2000).

This chapter reports the results of three studies. The first study examines the contribution of *Eucalyptus* spp. and *Corymbia* spp. to stand basal area (dominance) and the effect of dominance on stand biomass regressions. The second study develops a methodology to allocate a ranking that describes the stand structure of sites based on height distribution. Finally, the third study examines the effect of stand structure on stand biomass regressions of *Eucalyptus* spp. and *Corymbia* spp.

5.2 Study One: Basal area, aboveground biomass estimations and stand dominance

5.2.1 Materials and Methods

5.2.1.1 Selection of Sites and Species

Ninety-five woodland sites were selected across the grazed eucalypt woodlands of north east Australia and included both intact and disturbed sites (27 of these sites were included in Chapter 4). The sites represent three soil groups; duplex, gradational and uniform with rainfall varying from 1100 mm to less than 500 mm mean annual rainfall (MAR).

The analysis of all species in the communities may confound and/or conceal trends in stand analysis (Condit *et al*., 1998; Hitimana *et al*., 2004), hence only *Eucalyptus* spp. and *Corymbia* spp.[7] were investigated. Individual tree data at the final recording for all eucalypts were extracted from the TRAPS data set for each site.

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[7] For this study all *Eucalyptus* spp. and *Corymbia* spp. are collectively known as eucalypts.

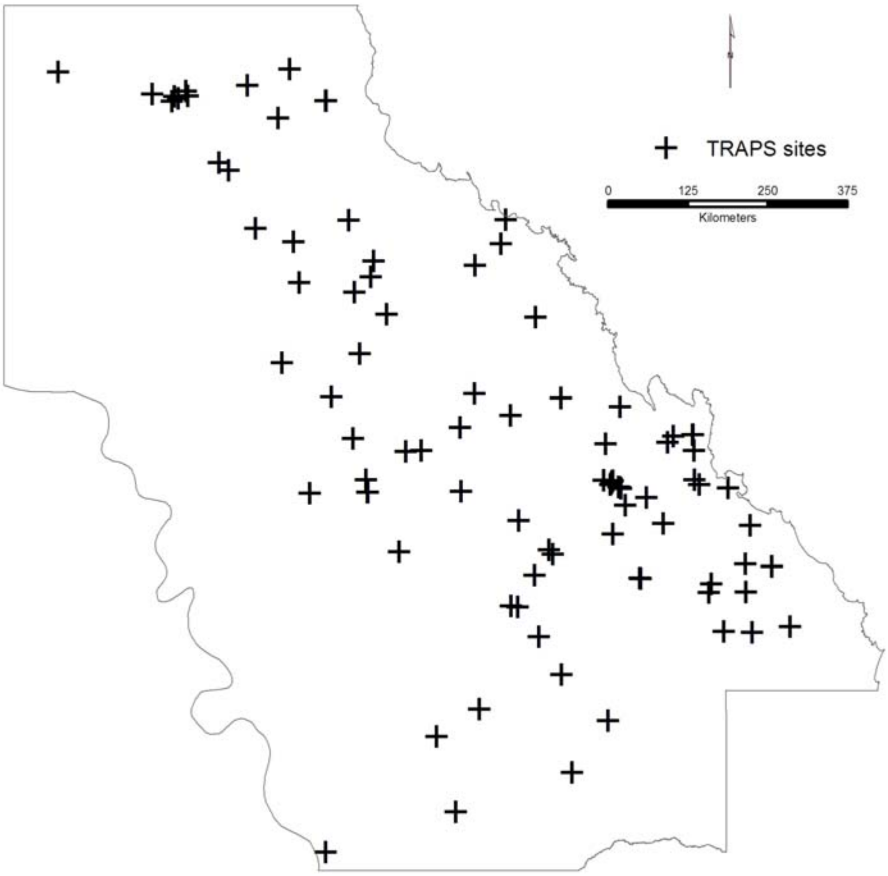

Figure 5-1. Distribution map of 95 sites throughout north-east Australia.

5.2.1.2 Basal area and aboveground biomass

Total site and total eucalypt basal areas were determined for each site and the contribution of eucalypt to total stand basal area was determined. Three different groupings of dominance were determined to investigate a range of dominance values (Table 5-1).

Table 5-1. Description of three groupings of dominance.

Grouping	Dominant	Co-dominant	Sub-dominant
Grouping 1	>70%	70-30%	<30%
Grouping 2	>65%	65-35%	<35%
Grouping 3	>60%	60-40%	<40%

Aboveground biomass was estimated by the application of a common allometric regression (chapter four) to individual eucalypt trees at each site. The methodology described in chapter three was used to estimate biomass. The bias caused by back-transformation of natural logarithmic values to biomass values was corrected using the methods outlined by Beauchamp and Olson (1973).

5.2.1.3 Eucalypt stand basal area and biomass regressions

A regression between eucalypt basal area and eucalypt biomass was determined using least squares analysis. However, regression residuals increased as basal area increased and a natural logarithmic transformation was necessary. Eucalypt stand biomass regressions (ln eucalypt basal area × ln eucalypt biomass) were determined using least squares analysis.

Similarly, stand biomass regressions were calculated for each eucalypt dominance type in each of the three dominance groupings of 70%, 65% and 60%, using least squares analysis. Analysis of covariance was performed on regressions of each dominance grouping to determine if slopes were significantly different, and then, if found to be similar, it was determined if y intercept of the regressions were significantly different.

5.2.2 Results

5.2.2.1 Basal area and aboveground biomass

Total site basal area ranged widely from 0.080 m^2/ha to 36.050 m^2/ha, with an average basal area of 11.226 m^2/ha. Eucalypt basal area ranged from 0.011 m^2/ha to 25.565 m^2/ha, with an average basal area of 9.302 m^2/ha. Eucalypt aboveground biomass estimates ranged from 0.010 to 193.317 t/ha, with an average biomass of 56.939 t/ha (Table 5-2).

In terms of basal area, 81-84% of sites were graded as eucalypt dominant at all three selected dominance groupings. The remaining sites were dominated by non-eucalypt species such as *Callitris* spp., *Melaleuca* spp. or *Acacia* spp.. Of these, 6.3 – 8.4% of sites were graded eucalypt co-dominant and 8.4 – 10.5% were graded as eucalypt sub-dominant sites for the three selected dominance groupings (Table 5-3).

Table 5-2. Summary of basal areas and biomass for 95 sites in north-east Australian woodlands. Bracketed values are ranges for data.

Site Dominant species	**n**	**Average Site Basal area (m^2/ha)***	**Dominant Eucalypt Species**	**Average Basal Area (m2/ha)***	**Average Biomass (t/ha)***
Acacia aneura	1	7.40	*E. populnea*	0.79	5.47
A. argyrodendron	1	7.67	*E. cambageana*	2.07	13.31
A. bidwillii	1	0.08	*E. populnea*	0.01	0.01
Atalaya hemiglauca	1	0.49	*E. populnea*	0.11	0.40
C. glaucophylla	1	36.05	*E. populnea*	17.67	137.77
C. glaucophylla	2	14.46 (14.15-14.78)	*E. chlorophylla*	1.55 (1.63-1.47)	9.67 (8.15-11.19)
Corymbia papuana	1	0.303	*C. papuana*	0.294	1.28
E. acmenoides	1	17.15	*E. acmenoides*	14.27	90.78
E. brownii	4	8.57 (5.19-10.24)	*E. brownii*	8.17 (5.19-9.96)	50.81 (38.94-40.63)
C. citriodora	3	12.70 (6.90-20.63)	*C. citriodora*	11.39 (5.62-18.73)	66.87 (30.99-99.98)
E. coolabah	2	9.33 (16.72-17.93)	*E. coolabah*	17.01 (16.68-17.34)	123.69 (107.72-140.16)
E. crebra	19	12.53 (2.18-31.05)	*E. crebra*	10.92 (2.17-23.48)	68.77 (16.07-193.32)
E. exserta	1	21.59	*E. exserta*	21.53	130.86

Site Dominant species	**n**	**Average Site Basal area (m^2/ha)***	**Dominant Eucalypt Species**	**Average Basal Area (m2/ha)***	**Average Biomass (t/ha)***
E. fibrosa	1	23.75	*E. fibrosa*	23.75	142.20
E. melanophloia	18	10.78 (0.76-21.72)	*E. melanophloia*	9.93 (0.66-21.19)	59.39 (1.39-115.65)
E. microneura	6	8.26 (3.13-15.94)	*E. microneura*	7.028 (2.11-14.57	38.09 (11.78-76.90)
E. moluccana	2	18.541 (12.52-24.57)	*E. moluccana*	17.52 (10.48-24.57)	120.83 (69.26-172.40)
E. orgadophila	2	5.87 (2.79-8.98)	*E. orgadophila*	4.09 (2.65-5.54)	26.98 (17.20-36.77)
E. platyphylla	1	4.88	*E. platyphylla*	4.75	30.99
C. plena	1	4.77	*C. plena*	4.10	25.53
E. populnea	14	11.22 (0.94-21.58)	*E. populnea*	9.38 (0.56-17.00)	54.56 (3.51-113.19)
E. quadricostata	1	11.228	*E. quadricostata*	10.94	62.39
E. similis	2	7.884 (7.60-8.17)	*E. similis*	7.16 (6.77-7.55)	26.49 (25.67-27.31)
E. tereticornis	3	11.03 (6.80-16.19)	*E. tereticornis*	10.960 (6.60-16.19)	80.49 (36.44-117.52)
E. whitei	1	6.75	*E. whitei*	6.37	20.80
Eremophila mitchellii	1	5.21	*E. populnea*	1.25	3.14
M. quiquinervia	1	0.53	*E. intermedia*	0.08	0.18
Melaleuca viridiflora	1	10.49	*E. tectifica*	5.71	22.23
Melaleuca viridiflora	1	10.43	*E. exserta*	3.91	8.76
Melaleuca viridiflora	1	25.19	*E. tereticornis*	11.97	84.00
Site Average		**11.23**	**Eucalypt Average**	**9.30**	**56.94**

Table 5-3. Distribution of 95 sites (number in each class) in three groupings of dominance.

	Dominant sites	Co-dominant sites	Sub-dominant sites
70% dominant group	77	8	10
65% dominant group	79	8	8
60% dominant group	80	6	9

5.2.2.2 Stand basal area and biomass regressions

A strong relationship between eucalypt stand basal area and total aboveground biomass represented 96.7% of the monitoring sites (**Figure5-2**). Stand regressions, when sites were grouped by dominance type, were also strong (R^2 ranging from 0.883-0.977) (Table 5-4). Furthermore, stand regressions grouped by dominance type, for each of the dominance groupings, were not significantly different ($P>0.001$) (Figure 5-3).

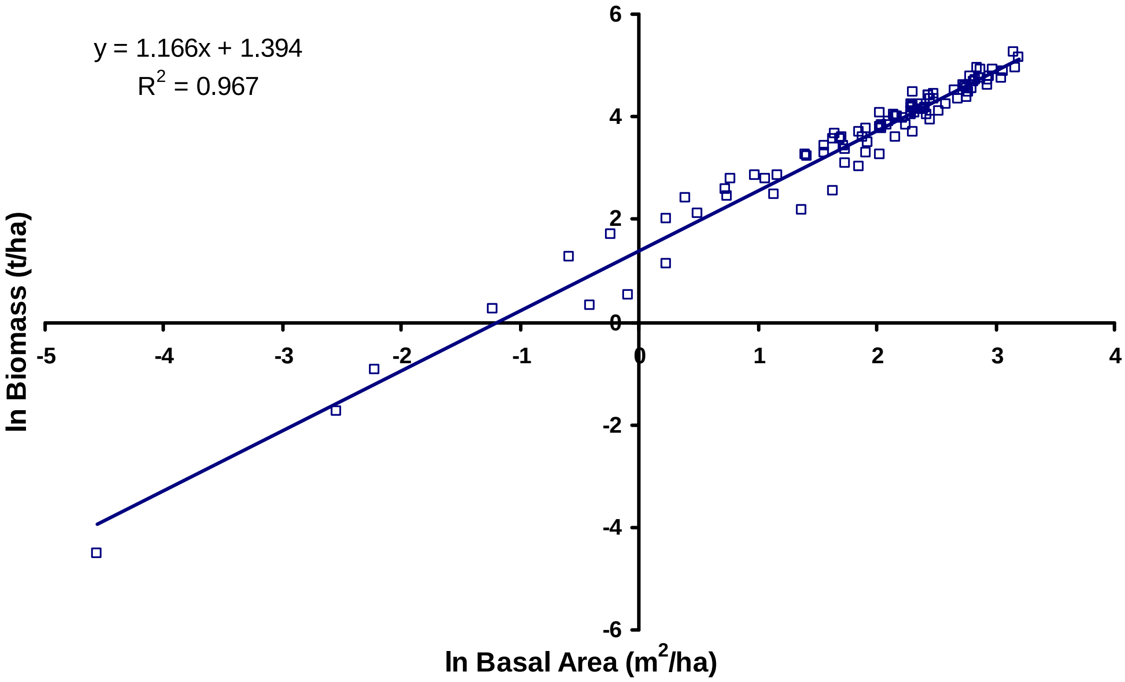

Figure5-2. Lognormal regression of eucalypt basal area (m^2/ha) and eucalypt biomass (t/ha) for 95 sites in north-east Australia.

Table 5-4. Lognormal regressions for stand regressions at 95 sites in Queensland woodlands. Regressions are in the form of lny = a + b lnx, where y = eucalypt biomass (t/ha), x = eucalypt basal area (m^2/ha).

		a	b	n	R^2	RSD	SSDx
70%	Dominant	1.407	1.156	77	0.938	0.240	48.302
	Co-dominant	1.537	1.071	10	0.894	0.388	8.849
	Sub-dominant	1.603	1.297	8	0.977	0.412	25.766
65%	Dominant	1.420	1.151	79	0.939	0.237	50.256
	Co-dominant	1.504	1.080	8	0.883	0.444	7.650
	Sub-dominant	1.603	1.297	8	0.977	0.412	25.766
60%	Dominant	1.427	1.149	80	0.938	0.237	50.428
	Co-dominant	1.622	1.062	6	0.954	0.321	7.577
	Sub-dominant	1.411	1.216	9	0.956	0.548	30.600
Common regression		1.394	1.166	95	0.967	0.276	153.733

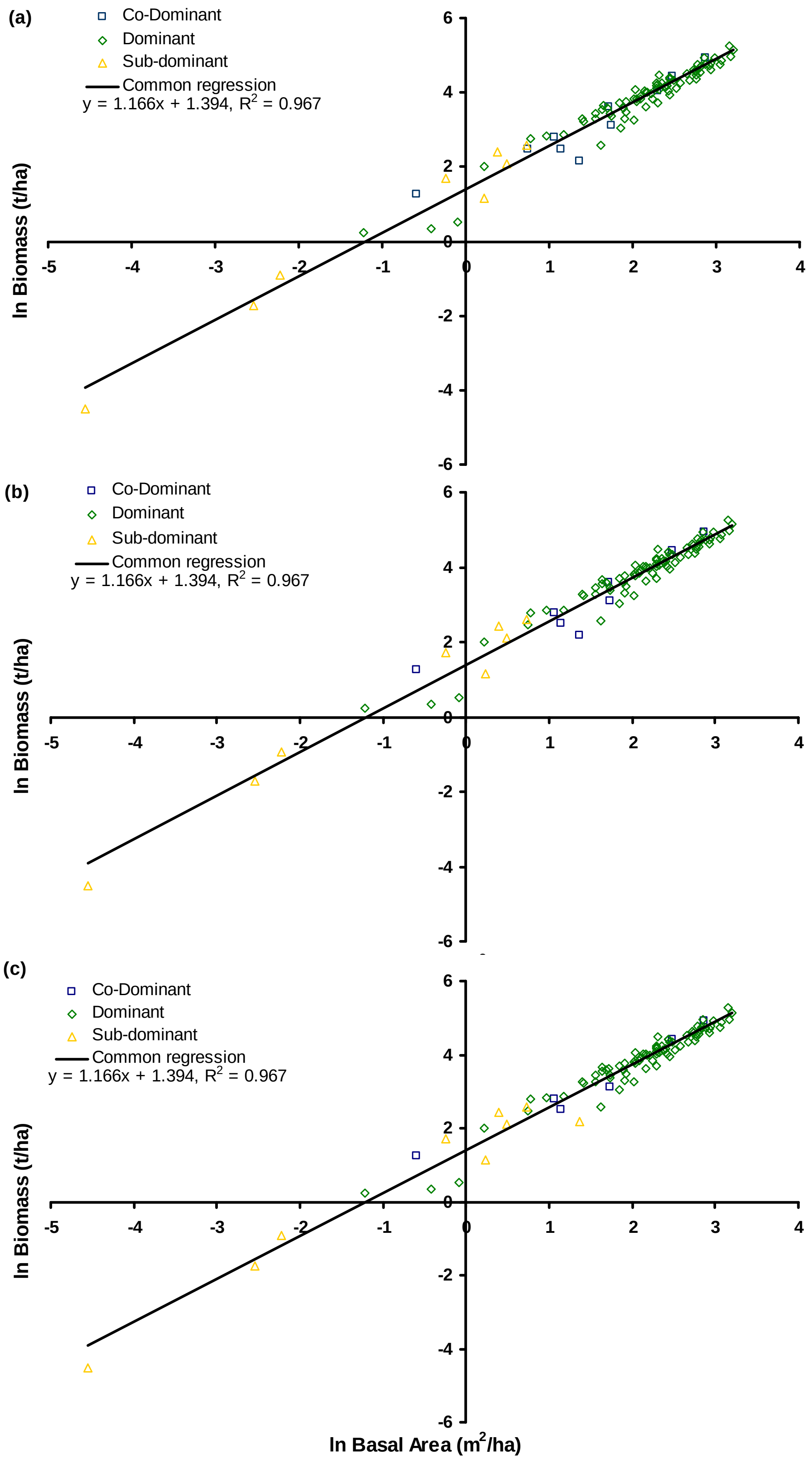

Figure 5-3. Lognormal regressions of eucalypt basal area (m^2/ha) and eucalypt biomass (t/ha) for sites at dominance levels (a) 70%, (b) 65% and (c) 60%. (Independent regressions were not significantly different, P>0.001).

5.2.3 Discussion

5.2.3.1 Basal Area and Aboveground Biomass

The 95 sites within the study area highlighted the variation of species and plant density in Queensland's grazed eucalypt woodlands. The greater percentage of eucalypt dominant sites reflects the study area selected, i.e. the grazed eucalypt woodlands. The smaller percentage of eucalypt co- and sub- dominant sites may indicate either community structural changes, usually due to human intervention, such as clearing, changing fire regimes and varying stocking rates (Archer *et al*., 1988; Landis and Bailey, 2005) or differences in community types. Changing the grouping of dominance types had little influence on the distribution of sites between each dominance grouping.

5.2.3.2 Stand Basal Area and Biomass Regressions

Unlike the *C. glaucophylla* stand biomass (Chapter Three), eucalypt stand regressions for stand dominance type were not significantly different for any of the three dominance groupings. Hence stand regression for eucalypt species does not vary according to which species dominate the stand. This is similar to findings reported by Burrows *et al*. (2000) and used by Burrows *et al.* (2002). However, the latter study was limited to mature stands across a smaller geographic range in Queensland. Findings from the current study indicate that a single stand regression can be applied at a site level to estimate site biomass of eucalypts in a variety of communities in north-east Australia.

5.3 Study Two: Application of height distribution to determine stand structure

5.3.1 Materials and Methods

The purpose of this study was to develop a consistent and repeatable methodology using height distributions to describe the stand structure of a variety of eucalypt woodland communities. This was achieved by the development of the following five stages:

- Determine height class frequency distributions for each site.
- Develop a tallest tree grade to each site that describes the mature height of the stand using the height frequency distributions and determine the actual terminal height class for each site.
- Describe typical tree categories based on tree growth stages within a stand, e.g. seedlings, saplings, pole and mature.
- Combine descriptive tree categories and height class distributions using the tallest tree grade to account for different maximum tree heights at different sites and determine the frequencies of trees in each tree category.
- Assign a descriptive rank to individual sites based on the proportion of trees in each tree category, and then combine ranks into broader groupings.

Data were evaluated at all stages by a series of queries in an MSAccess database.

5.3.1.1 Sites and Data

The same ninety-five sites from study one (5.2) were used. For each site, tree height data for all eucalypts at the final recording, were extracted.

5.3.1.2 Height class frequency distributions

Height class frequency distributions were determined for each site, and were based on traditional height classes used by Burrows *et al*. (2000) which consisted of seven classes (<0.5 m, 0.5-1.5 m, 1.5-4 m, 4-7 m, 7-10 m, 10-15 m and >15 m). However, this often resulted in a large terminal height class due to the maximum tree height at some sites being >30 m. Hence, height class frequency distributions were altered to the following; <0.5m, 0.5-1.5 m, 1.5-4 m, 4-8 m, 8-12 m, 12-16 m, 16-20 m, 20-25 m and >25 m.

5.3.1.3 Grading sites by height – Tallest Tree Grading

Height class frequency distributions were used to allocate a height grading to each site. The height grading, Tallest Tree Grading (TTG), consisted of four rankings: A, B, C or D and was determined by the frequency of trees in the two highest height classes. For example, if there were more individuals in the >25 m height class compared with the 20-25 m height class, then the site was graded as ‘A’. However, if there were less individuals in the >25 m height class compared with the 20-25 m height class, then the site was graded as ‘B’. In this way, this TTG method was used to allocate a new terminal height to each site (Table 5-5).

Table 5-5. Tallest Tree Grade (TTG) for 95 sites, where n, m, o, and p are frequencies in each height class. Presented in decreasing height classes where n is highest and p is lowest.

	TTG	>25 m	25-20 m	20-16 m	16-12 m	New terminal height class
if n ≥ m	A	n	m	o	p	>25 m
if n < m	B	n	m	o	p	>20 m
if m ≥ o	B	nil	m	o	p	>20 m
if m < o	C	nil	m	o	p	>16 m
if o ≥ p	C	nil	nil	o	p	>16 m
if o < p	D	nil	nil	o	p	>12 m
	D	nil	nil	nil	p	>12m

5.3.1.4 Tree categories

Tree categories were based on tree classification and growth stages described in Florence (1996) (Table 5-6). The percentage of individuals in each tree category was calculated for each site.

Table 5-6. Descriptions of tree categories based on tree growth stages.

Tree category	Description
Category One	Seedling, multi-stemmed, transient population
Category Two	Established seedlings and young saplings, multi-stemmed or single stemmed, stem not readily measured with a tape
Category Three	Sapling or young trees, single stemmed, young pole growth stage, fast growing and immature
Category Four	Young trees, flowering, middle pole stage
Category Five	Trees, flowering, late pole stage, approaching stand height
Category Six	Mature trees, very little height growth, continued circumference increases

5.3.1.5 Ranks and Groupings

Six Site Ranking descriptions were determined for sites based on the proportions of individuals in each of the Tree Categories. Ranks were then grouped into three groupings.

Table 5-7. Tree category values for Ranks and Groupings. (C1, C2, C3, C4 and C5 represent tree categories described in 5.3.1.4)

Rank	Group	Categories
Mature	Mature	C1 + C2 > C6 > C3 + C4 + C5
Early Mature	Mature	C1 + C2 > C5 ≥ C6 > C3 + C4
Mature without regeneration	Mature	C5 ≥ C6 > C3 + C4 > C1 + C2
Growth	Growth	C1 + C2 > C3 + C4 > C6 & C5 or C3 + C4 > C1 + C2 > C6 & C5
Early Growth	Seedling	C1 + C2 + C3 > 90%
Seedling	Seedling	C1 + C2 + C3 = 100%

5.3.2 Results

5.3.2.1 Height class frequency distributions

At some sites individual trees were distinctly shorter at maturity compared with sites where trees were taller, with similar circumferences, while the majority of sites contained mature trees between these extremes (Table 5-8). Despite increasing the number of height classes to nine, trees with different growth descriptions remained in the same classes, e.g. height class 12-16 m was described as young trees, early mature trees or mature trees (Table 5-9).

Table 5-8. Height data summaries of individual trees of 95 sites.

Dominant Eucalypt Species	n	Average Height (m)	Average Maximum Height (m)
C. papuana	1	0.6	10.0
C. plena	1	9.7	15.9
E. acmenoides	1	2.3	31.0
E. brownii	4	5.2 (3.9-8.6)	20.4 (18.3-23.0)
E. cambageana	1	7.2	16.5
E. chlorophylla	2	6.5 (2.2-4.3)	14.6 (14.5-14.7)
E. citriodora	3	4.4 (2.0-8.5)	31.1 (26.5-33.9)
E. coolabah	2	6.7 (3.5-9.8)	26.6 (23.6-29.5)
E. crebra	19	4.0 (1.1-8.8)	23.4 (14.8-33.7)
E. exserta	2	7.1 (5.4-8.8)	20.9 (17.8-23.9)
E. fibrosa	1	3.3	22.0
E. intermedia	1	0.6	4.0
E. melanophloia	18	4.4 (0.9-8.2)	10.5 (3.3-23.0)
E. microneura	6	7.3 (6.5-9.1)	13.3 (11.8-15.8)
E. moluccana	2	6.9 (5.4-8.4)	32.5 (31.5-33.5)
E. orgadophila	2	3.1 (2.2-3.9)	16.3 (16.1-16.5)
E. platyphylla	1	4.4	19.3
E. populnea	19	4.5 (0.1-9.3)	15.0 (0.7-29.7)
E. quadricostata	1	4.0	15.9
E. similis	2	4.7 (4.4-5.1)	12.9 (10.1-15.7)
E. tectifica	1	6.1	13.3
E. tereticornis	4	4.8 (1.0-9.3)	28.5 (21.7-31.3)
E. whitei	1	3.5	7.9
Average*		4.7 (0.1-9.9)	18.9 (0.7-33.9)

* Average of all sites, not the average of species.

Table 5-9. Nine height classes determined for permanent monitoring sites. Maximum height at the sites influences the description of individuals at differing sites.

Height Class	Description of individuals
< 50 cm	Multi-stemmed seedlings that are below grass height, usually not established (transient population) and controlled by fire
0.50 - 1.5 m	Established seedlings that have no measurable circumference (without breaking tapes), usually controlled by fire
1.5 - 4 m	Saplings with measurable circumference, sometimes controlled by with fire
4 – 8 m	Young trees with rapid growth, usually not setting seed
8 – 12 m	Young trees with rapid growth or early mature trees with medium growth, setting seed.
12 – 16 m	Young trees with rapid growth, or early mature trees with medium growth, or mature trees with slow growth
16 - 20m	Early mature trees with medium growth, or mature trees with slow growth
20 - 25m	Mature trees with slow growth
> 25m	Mature trees with slow growth (have reached full potential)

5.3.2.2 Tallest tree grading and tree categories

Within the 95 sites there were four different open-ended terminal height classes ranging from >12 m to > 25 m, depending on the TTG allocated to the site. Height classes allocated to tree categories One, Two and Three were consistent for every site and those allocated to categories Four, Five and Six varied, depending on the terminal class for each site. Category Four had the largest height range for the first three grades (Table 5-10).

Table 5-10. Height class values for tree categories determined by Tallest Tree Grade.

Tall Tree Grading	**Tree Categories** One	**Two**	**Three**	**Four**	**Five**	**Six**
A	0 – 0.5m	0.5 – 1.5m	1.5 – 4m	4 – 20m	20 - 25m	>25m
B	0 – 0.5m	0.5 – 1.5m	1.5 – 4m	4 – 16m	16-20m	>20m
C	0 – 0.5m	0.5 – 1.5m	1.5 – 4m	4 – 12m	12 -16m	>16m
D	0 – 0.5m	0.5 – 1.5m	1.5 – 4m	4 – 8m	8 – 12m	>12m

5.3.2.3 Site ranks and groupings

In terms of site ranks, 54% were ranked as Growth and sites ranked as Mature and Early Mature were the next largest ranks, contributing to 14% and 19% of total sites. Sites grouped as Mature contributed to 37% of total sites, whilst sites grouped as Seedlings contributed to 9% of total sites (Table 5-11).

Table 5-11. Ranking and Groupings of 95 sites in north-east Australia.

Site Number	**Site Grouping**	**Number of sites**
Early Growth	Seedling	5
Young Growth	Seedling	4
Growth	Growth	51
Early Mature	Mature	13
Mature	Mature	18
Mature without regeneration	Mature	4

5.3.3 Discussion

Height definitions of structural components of tree stands vary with different researchers defining similar stand components by differing height increments. For example, in South Africa, Skarpe (1990) define low shrub as 0.5 -1.5m tall while

Brown and Bredenkamp (2004) define low shrub as 0-1 m tall. Furthermore, Golser and Hasenauser (1997) define forest overstorey in Austria as greater than 1.3 m tall.

Height will asymptote when trees reach maturity, while circumference will continue to increase after maximum height is obtained (Shibuya *et al.,* 2004) and height curves of trees (height × age) vary from site to site (Rivas *et al*., 2004). Additionally, circumference has substantial phenotypic plasticity, resulting in increased tree circumferences compared with tree circumferences of a stand with an increased density (Knowles and Grant, 1983), which contributes to the difficulty of determining a circumference class value to mature trees at unique sites.

Height classes have been used by researchers to describe community structure (Burrows *et al*., 2000). Similarly, Brown and Bredenkamp (2004) used height classes to describe structural differences of savanna communities in South Africa. Height classes are often based on constant increments and the problem of comparing mature trees at sites having a shorter maximum tree height with mature trees at sites having a taller maximum tree height have been encountered (P.V. Back, pers. comm. 2004). Similarly, in the current study, mature trees at sites with lower maximum height classes were being compared with young trees at 'taller' sites, as the maximum height of the 'shorter' sites, 12-16 m, was approximately half that of the 'taller' sites. Furthermore, constant height class distribution increments caused the height class distribution of tree in less productive sites ('shorter' sites) to be compressed into a smaller number of classes so that changes from one class to another were less noticeable.

The process to determine the tallest tree grade (see earlier) prevented a minority of exceptionally tall trees within the population influencing the maximum height class.

The method to determine tree category size classes at individual sites (Table 5-6) was also successful and ensured that trees of similar descriptive stages within communities of different maximum heights are compared similarly. Tree category ranges were based on a reasonable evaluation of heights observed in the field and occasionally an individual tree may vary in life stage and be allocated in one category when it may belong to another. However, one advantage of this methodology was that the same judgements were applied to each tree; it was repeatable and accounted for differences in mature heights across a wide range of sites.

The current study provided a repeatable process to determine stand structures of eucalypt woodland communities. The process remained automated and consistent, as a result of the database queries that were used to complete each stage. This prevented operator bias or a knowledge of any particular site from influencing any stage of the methodology. However, the height class distributions and tree categories used were inappropriate to fit regressions to, as the class sizes are uneven and large (Leak, 1965; Condit *et al.*, 1998). The methodology successfully allocated one of six ranks and one of three groupings to each site, based on the proportion of trees in each category instead of a fitting regression to the categories.

At most sites the proportion of individuals in the tree categories indicated a change in the stand structure from a past occurrence as the number of survivors had changed (cf. Landis and Bailey, 2005). Growth, Early Growth and Seedling ranks all indicated such a change to different extents. The Growth rank indicated an increase in saplings and young trees which resulted from a past recruitment event and/or reduced competition within the stand. These stands will more than likely have a positive effect on future carbon fluxes. The continued development of seedlings in the

Seedling rank will also have a positive effect on the carbon fluxes as the stand structure develops into a Growth ranking. Sites that were ranked as Mature show size hierarchy, that is, there were many seedlings and relatively few large trees (see Weiner and Solbrig, 1984). As noted by Florence (1996), the replacement of mature trees was not obvious (ie, there were no large seedling recruitment or development events in these stands). Stands identified as ‘Early Mature’ were approaching stability, usually from an event in their long-term history that had triggered a release of seedlings.

Thus, it is the contribution of the changing proportions of large and small individuals to stand structure that may affect the relationship between stand basal areas and stand biomass (study one) in north-east Australian woodlands. This suggests that stand structure could also be an important consideration when applying basal-biomass ratios to the determination of community biomass.

5.4 Study Three: Stand biomass relationships, effect of stand structure and validity of using common stand regression to estimate biomass.

The purpose of this study was to investigate the effect of stand structure on stand biomass regressions across a range of eucalypt woodland sites.

5.4.1 Materials and Methods

Data from studies one (5.2) and two (5.3) were combined for each site in the current study. Sites were grouped according to the site ranking or site grouping.

5.4.1.1 Stand biomass regressions and site ranks

Eucalypt stand biomass regression for each site rank and grouping were determined using least squares analysis. Analysis of covariance was performed on linear regressions to determine if slopes were different, and then if found to be similar, it was determined if y intercept of the regressions were significantly different.

The common stand regression determined in study one was investigated further. Regression between actual biomass and predicted values (1:1 relationship) was calculated using least-squares analysis to determine the strengths of a common stand regression.

5.4.2 Results

5.4.2.1 Stand biomass regressions and site ranks

Eucalypt stand regressions for each of the site ranks and groupings were strongly correlated with R^2 ranging from 0.909 to 0.994 (Table 5-12).

Stand regressions of sites representing each of the six site ranks were not significantly different (P>0.001) (Figure5-4). Similarly stand regressions representing each of the three site groupings were not significantly different (P>0.001) (Figure5-5). The common stand regression was the same for both site rank and site grouping and was initially presented in study one (**Figure5-2**) as the same data points were used.

A strong correlation was demonstrated between ln actual biomass and ln predicted biomass for the common stand regression ($R^2 = 0.967$) (Figure5-6). The variance of the residual errors of the common stand regression and species rank stand regressions were not significantly different (P>0.001).

Table 5-12. Lognormal regressions for stand basal area (m^2/ha) and stand biomass (t/ha) at 95 sites in north-east Australian woodlands.

Description	a	b	n	R^2	RSD	SSDx
Ranks						
Early Growth	1.311	1.301	5	0.909	0.606	6.480
Young Growth	1.442	1.216	4	0.976	0.621	21.193
Growth	1.464	1.120	51	0.925	0.268	34.421
Early Mature	1.500	1.137	13	0.951	0.185	5.701
Mature	1.906	0.976	18	0.965	0.141	9.110
Mature no regeneration	1.967	0.885	4	0.998	0.039	2.457
Groups						
Seedling	1.355	1.222	9	0.958	0.532	29.882
Growth	1.464	1.120	51	0.925	0.268	34.421
Mature	1.740	1.032	35	0.958	0.164	19.097
Common	1.394	1.166	95	0.967	0.276	153.733

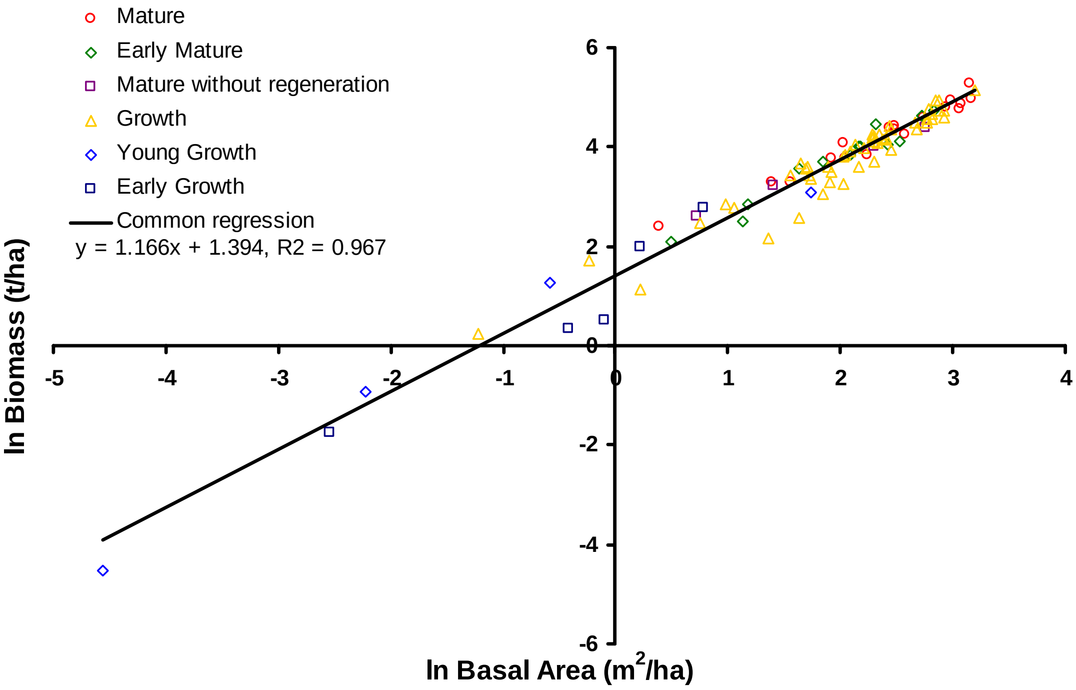

Figure5-4. Lognormal regressions of eucalypt basal area (m^2/ha) and eucalypt biomass (t/ha) for sites grouped by rankings. (Independent regressions were not significantly different, P>0.001).

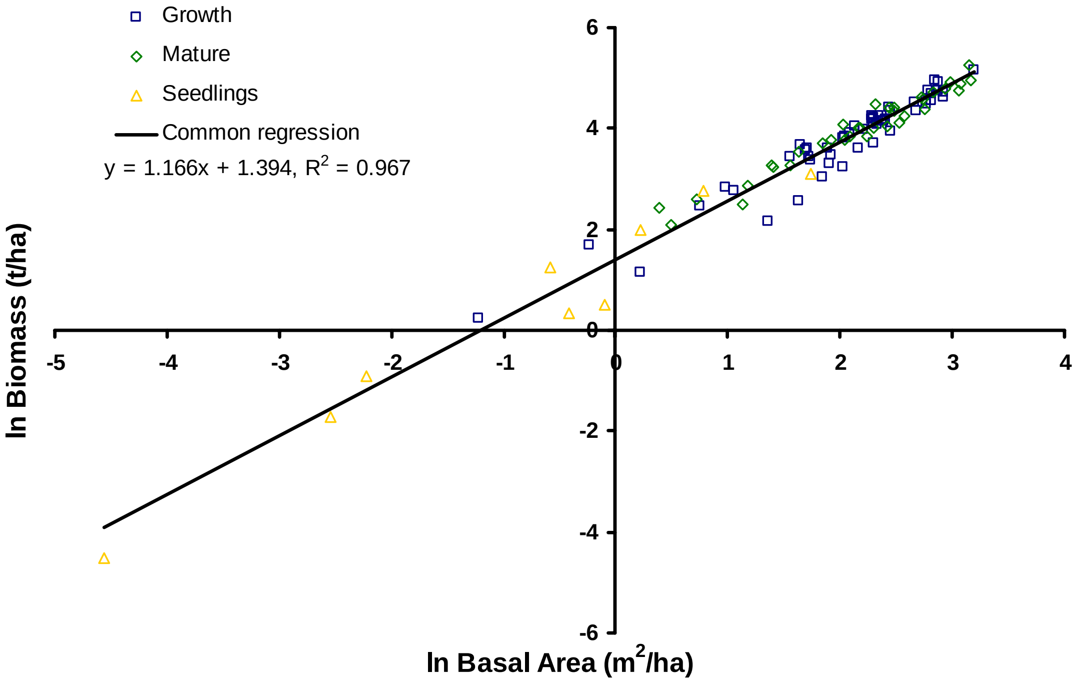

Figure5-5. Lognormal regressions of eucalypt basal area (m^2/ha) and eucalypt biomass (t/ha) for sites grouped by Site Grouping. (Independent regressions were not significantly different, P>0.001).

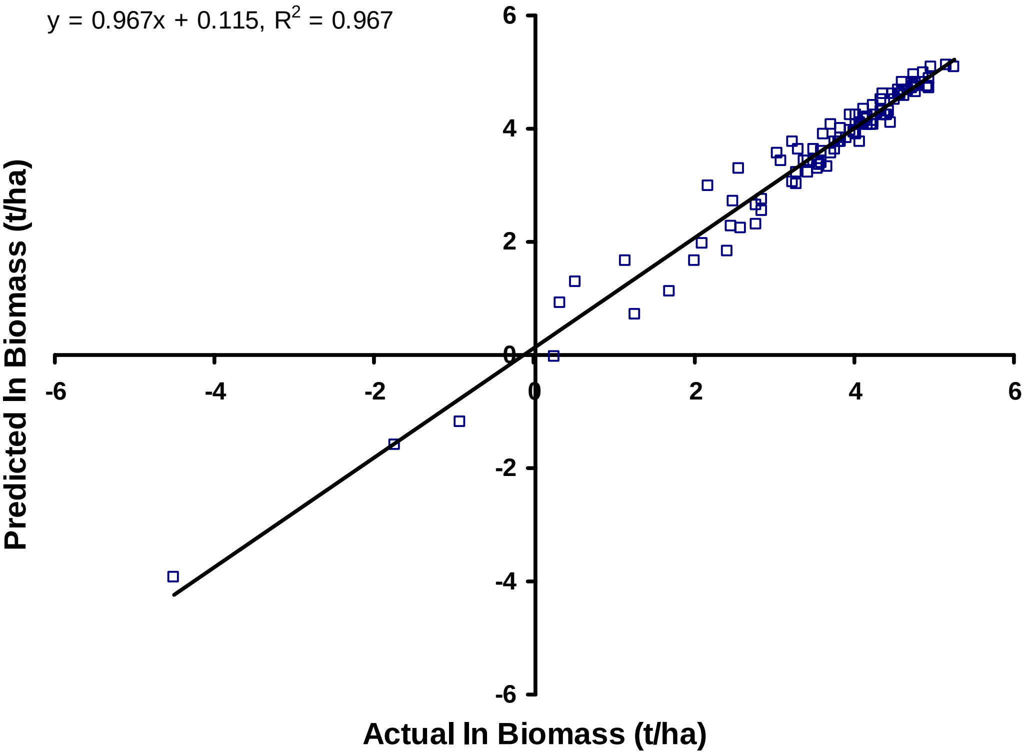

Figure5-6. Lognormal regression of actual biomass (t/ha) and predicted biomass (t/ha) for *Eucalyptus* spp. communities.

5.4.3 Discussion

Stand structure appears to have little influence on stand biomass regressions.

5.4.3.1 Site ranking and grouping

Researchers have placed importance on applying allometric regressions to communities of similar types for which they were developed (Feller, 1980; Nelson *et al*., 1999; Keith *et al*., 2000). However, there has been sparse research into the application of a stand biomass regression. It is assumed that in such cases a stand regression should similarly be applied to stands of similar dominance, and structure to which it was developed (K. Montagu, pers. comm. 2003). Yet there is very little research available that supports or opposes this argument. Burrows *et al*. (2000) reported a stand relationship for 33 mature eucalypt dominant stands in Queensland. The relationship was further expanded by increasing the number of sites to 57 (Burrows *et al*., 2002). In study one it was reported that for 95 sites the regression residuals increased with increasing basal area, thus requiring a transformation.

Stand structure, based on site rank, had no significant influence on stand biomass regressions. All site ranks had similar stand regressions despite having very different stand structures. Similarly, when sites were grouped into three stand groups (seedling, growth and mature) the stand regressions were similar. Stand structure appears to have little influence on stand biomass regressions developed in this study.

Ln predicted biomass × ln actual biomass regression had parameters that closely approached a 1:1 line, indicating that the regression was a robust predictor of stand biomass. Furthermore, the variance of the residual errors of the common stand regression and rank specific regressions were similar. Thus, a common stand biomass regression can be used to estimate stand biomass of eucalypts in North-east Australian grazed woodlands.

5.5 Conclusion

Stand dominance had no significant influence on stand biomass regressions for a range of eucalypt sites in north-east Australia. The research reported here successfully developed a methodology to determine stand structure by allocating a descriptor to woodland sites based on height class distributions. Furthermore, the effect of this stand rank was demonstrated to have little influence on stand biomass regressions.

This study has also demonstrated that a common stand biomass regression based on data from 95 sites provided a robust estimate of eucalypt stand biomass. Thus, the estimation of biomass at a site level can be achieved efficiently across a range of eucalypt communities in north-east Australia. These findings will further contribute to the ease of estimating eucalypt biomass in the study area, and thus aid in estimating the carbon flux within the eucalypt community of north-east Australia's grazed woodlands.

Chapter 6. Conclusion

Vegetation thickening and woody plant encroachment have been reported in savannas and woodlands worldwide. In the grazed woodlands of north-east Australia, the increase in biomass represents a significant carbon sink that can potentially offset Australia's reported carbon emissions (Burrows *et al*. 2002). However, an accurate measure of this sink is essential for any calculation of carbon fluxes. Allometric regressions have been used worldwide for more than 50 years to estimate the aboveground components of tree biomass, using an easily measured tree component (e.g. trunk diameter at breast height) as the predictor variable. Biomass regressions are available for a limited suite of species in north-east Australian woodlands. The research outlined in this thesis has increased the availability of suitable regressions by

i determining biomass regressions for *C. glaucophylla* and
ii expanding a site specific regression to the whole of north-east Australia.

C. glaucophylla tree biomass was estimated by the development of allometric regressions using different independent variables. Stem circumference was demonstrated to be the preferred independent variable, as it was statistically superior. Ease of measuring stem circumference compared with measuring canopy area or height would also minimise the level of operator error when using this variable. Whole tree regressions were more robust for estimating total tree biomass compared with component regressions.

The application of biomass regressions to a number of stands can be tedious and require individual tree data to be measured. Few stand biomass regressions are available that can be applied quickly and easily to stand basal area of woodland communities measured with common forestry mensuration instruments (eg. relaskop, prism etc.). Although this thesis determined stand biomass regressions for *C. glaucophylla* communities, it was demonstrated that the regression for *C. glaucophylla* dominant stands differed for co-dominant and sub-dominant stands. This may reflect structural differences of trees within these stands.

Tree-form regressions reflect the relationships between tree heights and stem circumferences. It has been reported that biomass regressions should be applied only to stands of trees for which they were developed, as the relationship between biomass and predictor variables may vary with different rainfall and soil types. Tree-form regressions were developed for a range of eucalypt communities across the study area and a comparison of regressions determined that tree-form did not reflect mean annual rainfall or soil types. Furthermore, differences or similarities in tree-form regressions did not reflect differences or similarities of biomass regressions for a range of eucalypt sites. Tree-form may be a result of community stand structure, management or a combination of numerous factors encountered within communities.

An increasing suite of species-specific biomass regressions is currently available for estimating tree biomass however, few common regressions are available for predicting biomass of eucalypts that can be applied across a wider range of eucalypt woodlands. This study has demonstrated that a common regression provides a robust estimate of biomass for both eucalypt mature communities and eucalypt regrowth communities. In eucalypt woodlands this common regression can be used to estimate biomass of individual trees that don’t have existing species or site specific allometrics within the study area. The ability to estimate eucalypt biomass for a range of sites facilitates the development of stand biomass regressions.

Stand regressions were determined from a range of sites across the grazed eucalypt woodlands and, unlike the *C. glaucophylla* communities, dominance appears to have little influence on these stand regressions. Three different definitions of stand dominance were investigated because definitions in the literature were varied. It was demonstrated that a common stand biomass regression may be used to estimate stand biomass in eucalypt woodlands. However, mature trees contain more biomass and it is assumed that sites with similar basal areas that consist of very different tree sizes (stand structure) will have different stand biomass estimates.

A reliable method to determine stand structure, free from of operator bias, was lacking in Australian woodlands. Circumference class distributions appeared to be unreliable for describing woodland stand structure. Height classes have been used successfully in South African savannas and Australian woodlands to describe communities. This study has demonstrated a methodology that successfully describes stand structure based on height distribution and tree growth descriptions of a range of eucalypt woodland communities. Furthermore, it has demonstrated that stand structure had little effect on eucalypt stand regressions, despite sites having a range of different stand structures in north-east Australia.

The findings outlined in this thesis will contribute to increasing the ease of estimating tree biomass in the study area without any further development of allometric regressions. To utilise these methods, an accurate measure of individual tree circumference or community basal area is essential. To accurately estimate carbon flux in vegetation long-term monitoring sites are necessary, and these are lacking in lacking in some forest communities (Binkley, 2004). The data at these monitoring sites should be collected using a consistent and documented methodology, which increases the estimation accuracy for landscape-scale variation in forest structure and biomass (Clark and Clark, 2000).

The robustness of common relationships for estimating aboveground biomass in eucalypt communities was demonstrated and as a result effort and resources can be concentrated on developing non-destructive processes, such as monitoring systems. QDPI&F's permanent monitoring network, based on the TRAPS methodology, have proved ideal for investigating trends at a landscape scale as they are representative of Queensland's grazed woodlands (Burrows *et al.,* 2002) and are suitable for assessing variations in basal area and biomass across landscapes (Clark and Clark, 2000). The findings from this study, combined with the long-term data in permanent monitoring networks, will enhance the estimation of carbon flux within the eucalypt communities of north-east Australia's grazed woodlands.

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Chapter 8. Appendices

Appendix 8-1 Site summaries for secondary sites of the Mitchell study area. Plot area = 0.25 ha (50m x 50m) and site summaries for five monitoring sites in containing *Callitris. glaucophylla* north east Australia. Sample areas C26 – 500 × 2.0 m, C27 – 500 × 6.0 m, C28 – 300 × 4.0 m, C29 and C30 – 500 × 4.0m.

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Site no: C01				
Acacia ixiophylla	159	636	0.175	0.700
Angophora leiocarpa	50	200	0.997	3.988
Callitris glaucophylla	128	512	3.552	14.208
Cassinia laevis	28	112	0.013	0.052
Calytrix longiflora	383	1532	0.161	0.644
Daviesia ulicifolium sub spp. stenophylla	7	28	0.004	0.016
Dodonaea boroniifolia	43	172	0.035	0.140
Leucopogon biflorus	19	76	0.008	0.032
Leucopogon mitchellii	113	452		
Lysicarpus angustifolius	88	352	0.364	1.456
Xylomelum cunninghamianum	2	8	0.058	0.232
Total	**1020**	**4080**	**5.414**	**21.656**
Site no: C02				
Acacia ixiophylla	1	4	0.005	0.020
Acacia leiocalyx	28	112	0.015	0.060
Acacia neriifolia	37	148	0.041	0.164
Angophora leiocarpa	45	180	0.515	2.060
Bossiaea rhombifolia sub spp. concolor	28	112	0.015	0.060
Callitris glaucophylla	70	280	1.242	4.968
Cassinia laevis	27	108	0.022	0.088
Calytrix longiflora	41	164	0.033	0.132
Dodonaea boroniifolia	32	128	0.019	0.076
Dodonaea peduncularis	27	108	0.015	0.060
Leucopogon biflorus	26	104	0.014	0.056
Leucopogon mitchellii	55	220	0.044	0.176
Lysicarpus angustifolius	10	40	0.142	0.568
Xylomelum cunninghamianum	32	128	0.133	0.532
Total	**459**	**1836**	**2.257**	**9.028**

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Site no: C03				
Acacia excelsa	5	20	0.288	1.152
Alphitonia excelsa	12	48	0.010	0.040
Brachychiton populneus spp. populneus	3	12	0.317	1.268
Callitris glaucophylla	729	2916	1.166	4.664
Casuarina spp.	6	24	0.046	0.184
Eucalyptus melanophloia	13	52	1.646	6.584
Eucalyptus microcarpa	1	4	0.166	0.664
Eucalyptus populnea	10	40	0.522	2.088
Hakea fraseri	1	4	0.017	0.068
Hovea longpipes	46	184	0.037	0.148
Notelaea microcarpa var. microcarpa	247	988	0.591	2.364
Opuntia tomentosa	35	140	0.161	0.644
Santalum lanceolatum	6	24	0.017	0.068
Unknown spp.	2	8	0.082	0.328
Total	**1116**	**4464**	**5.065**	**20.260**
Site no: C04				
Acacia leiocalyx	12	48	0.010	0.020
Allocasuarina luehmannii	1	4	0.005	0.020
Angophora leiocarpa	2	8	0.123	0.492
Callitris glaucophylla	1039	4156	3.309	13.236
Dodonaea viscosa sub spp. spatulata	162	648	0.131	0.524
Eucalyptus chloroclada	8	32	0.054	0.216
Eucalyptus melanophloia	178	712	3.412	13.648
Eucalyptus populnea	1	4	0.199	0.796
Hovea longpipes	18	72	0.015	0.060
Unknown spp.	1	4	0.035	0.140
Total	**1422**	**5688**	**7.293**	**29.172**
Site no: C05				
Acacia leiocalyx	15	60	0.016	0.064
Acacia leptostachya	31	124	0.025	0.100
Angophora leiocarpa	106	424	2.128	8.512
Callitris glaucophylla	24	96	0.285	1.140
Eucalyptus melanophloia	11	44	0.284	1.136
Hovea longpipes	56	224	0.045	0.180
Lysicarpus angustifolius	33	132	0.456	1.824
Unknown spp.	1	4	0.019	0.076
Total	**277**	**1108**	**3.257**	**13.028**

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Site no: C06				
Allocasuarina luehmannii	5	20	0.035	0.140
Angophora leiocarpa	33	132	2.857	11.428
Callitris glaucophylla	1109	4436	3.877	15.508
Eucalyptus chloroclada	2	8	0.109	0.436
Eucalyptus dolichocarpa	1	4	0.091	0.364
Eucalyptus melanophloia	6	24	0.098	0.392
Lysicarpus angustifolius	140	560	1.662	6.648
Total	**1296**	**5184**	**8.728**	**34.912**
Site no: C07				
Acacia decora	25	100	0.058	0.232
Callitris glaucophylla	3	12	0.068	0.272
Cassinia laevis	51	204	0.061	0.244
Casuarina cristata	12	48	0.052	0.208
Dodonaea viscosa sub spp. spatulata	6	24	0.009	0.036
Eremophila mitchellii	31	124	0.045	0.180
Eucalyptus populnea	85	340	3.010	12.040
Total	**213**	**852**	**3.303**	**13.212**
Site no: C08				
Acacia decora	18	72	0.042	0.168
Callitris glaucophylla	24	96	0.219	0.876
Cassinia laevis	74	196	0.082	0.328
Casuarina cristata	6	24	0.009	0.036
Dodonaea viscosa sub spp. spatulata	637	2548	0.934	3.736
Eucalyptus melanophloia	33	132	1.190	4.760
Eucalyptus populnea	55	220	2.509	10.036
Geijera parviflora	6	24	0.005	0.020
Lophostemon suaveolens	1	4	0.016	0.064
Total	**854**	**3416**	**5.005**	**20.020**
Site no: C09				
Acacia ixiophylla	2	8	0.010	0.040
Acacia leiocalyx	7	28	0.063	0.252
Acacia neriifolia	2	8	0.015	0.060
Angophora leiocarpa	9	36	0.777	3.108
Callitris glaucophylla	35	140	2.283	9.132
Eucalyptus chloroclada	18	72	0.485	1.940
Eucalyptus melanophloia	17	68	0.356	1.424

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Lysicarpus angustifolius	42	168	0.823	3.292
Philotheca difformis sub spp. difformis	12	48	0.007	0.028
Total	**148**	**592**	**4.822**	**19.288**
Site no: C10				
Acacia ixiophylla	1	4	0.005	0.020
Acacia neriifolia	30	120	0.189	0.756
Angophora leiocarpa	7	28	2.429	9.716
Callitris glaucophylla	36	144	1.891	7.564
Callitris spp.	268	1072	0.393	1.572
Cassinia laevis	6	24	0.011	0.044
Unknown spp.	95	380	0.268	1.072
Total	**443**	**1772**	**5.187**	**20.748**
Site no: C11				
Acacia decora	16	64	0.054	0.216
Acacia leiocalyx	41	164	0.345	1.380
Acacia neriifolia	11	44	0.057	0.228
Acacia spp.	1	4	0.007	0.028
Angophora leiocarpa	5	20	1.303	5.212
Callitris glaucophylla	41	164	1.405	5.620
Eucalyptus chloroclada	13	52	1.043	4.172
Eucalyptus melanophloia	8	32	0.338	1.352
Lysicarpus angustifolius	20	80	0.782	3.132
Total	**15**	**624**	**5.334**	**21.336**
Site no: C12				
Acacia leptostachya	62	248	0.091	0.364
Acacia macradenia	232	928	0.258	1.032
Callitris glaucophylla	6	24	0.289	1.156
Cassinia laevis	142	568	0.060	0.240
Eucalyptus chloroclada	83	332	0.863	3.452
Eucalyptus melanophloia	9	36	0.475	1.900
Hakea fraseri	6	24	0.002	0.008
Total	**1088**	**4352**	**2.611**	**10.444**
Site no: C13				
Angophora leiocarpa	2	8	0.009	0.036
Brachychiton populneus spp. populneus	1	4	0.086	0.356
Callitris glaucophylla	53	212	1.774	7.096
Eucalyptus chloroclada	87	348	1.098	4.392
Eucalyptus melanophloia	40	160	0.822	3.288

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Eucalyptus populnea	5	20	0.361	1.44
Syzygium cormiflorum 'ramiflorus form'	14	56	0.096	0.384
Total	**202**	**808**	**4.250**	**17.000**
Site no: C14				
Acacia leiocalyx	4	16	0.003	0.012
Callitris glaucophylla	765	3060	8.664	34.656
Eucalyptus dolichocarpa	3	12	0.373	1.492
Eucalyptus melanophloia	36	144	1.954	7.816
Eucalyptus spp.	1	4	0.009	0.036
Geijera parviflora	9	36	0.083	0.332
Total	**818**	**3272**	**11.085**	**44.340**
Site no: C15				
Angophora leiocarpa	10	40	0.178	0.712
Brachychiton populneus spp. populneus	1	4	0.017	0.068
Callitris glaucophylla	277	1108	6.787	27.148
Corymbia tessellaris	1	4	0.032	0.128
Eremophila deserti	6	24	0.005	0.020
Eucalyptus dolichocarpa	9	36	0.080	0.360
Eucalyptus melanophloia	8	32	0.049	0.196
Eucalyptus spp.	4	16	0.035	0.140
Leucopogon mitchellii	6	24	0.009	0.036
Total	**322**	**1288**	**7.202**	**28.808**
Site no: C16				
Acacia leptostachya	1	4	0.006	0.024
Angophora leiocarpa	1	4	0.007	0.028
Callitris glaucophylla	536	2144	3.996	15.984
Cassinia laevis	152	608	0.123	0.492
Eucalyptus chloroclada	51	204	0.649	2.596
Eucalyptus melanophloia	53	212	1.541	6.168
Leucopogon mitchellii	26	104	0.021	0.084
Total	**820**	**3280**	**6.342**	**25.368**
Site no: C17				
Acacia leptostachya	13	52	0.079	0.316
Acacia macradenia	28	112	0.065	0.260
Acacia neriifolia	13	52	0.030	0.120
Angophora leiocarpa	31	124	3.057	12.228
Bossiaea rhombifolia sub spp. concolor	78	312	0.063	0.252
Callitris glaucophylla	168	672	1.955	7.820

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Cassinia laevis	7	28	0.006	0.024
Eucalyptus chloroclada	3	12	0.050	0.200
Leucopogon mitchellii	59	236	0.033	0.132
Lomandra multiflora sub spp. multiflora	182	728	0.100	0.400
Xylomelum cunninghamianum	21	84	0.395	1.580
Total	**603**	**2412**	**5.831**	**23.324**
Site no: C18				
Brachychiton populneus spp. populneus	2	8	0.093	0.372
Callitris glaucophylla	262	1048	0.827	3.308
Cassinia laevis	19	76	0.015	0.060
Eremophila mitchellii	13	52	0.037	0.148
Eucalyptus chloroclada	109	436	0.841	3.364
Eucalyptus melanophloia	101	404	1.852	7.408
Eucalyptus microcarpa	99	396	0.988	3.952
Eucalyptus spp.	3	12	0.090	0.360
Myoporium deserti	12	48	0.028	0.080
Opuntia tomentosa	12	48	0.028	0.080
Unknown spp.	1	4	0.024	0.060
Total	**639**	**2556**	**4.823**	**19.292**
Site no: C19				
Brachychiton populneus spp. populneus	4	16	0.197	0.788
Callitris glaucophylla	68	272	0.861	3.444
Eucalyptus chloroclada	129	516	3.203	12.812
Eucalyptus melanophloia	4	16	0.323	1.292
Total	**205**	**820**	**4.584**	**18.336**
Site no: C20				
Alectryon diversifolium	2	8	0.002	0.008
Callitris glaucophylla	5	20	0.086	0.344
Cassinia laevis	1	4	0.001	0.004
Casuarina spp.	1	4	0.011	0.044
Eucalyptus chloroclada	27	108	1.675	6.700
Eucalyptus melanophloia	7	28	2.292	9.168
Geijera parviflora	2	8	0.055	0.220
Opuntia tomentosa	1	4	0.005	0.020
Unknown	2	8	0.030	0.120
Total	**48**	**192**	**4.156**	**16.624**
Site no: C21				
Callitris glaucophylla	4	16	0.183	0.732

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Canthium spp.	12	48	0.028	0.112
Eucalyptus populnea	74	296	2.514	10.056
Total	**90**	**360**	**2.725**	**10.900**
Site no: C22				
Callitris glaucophylla	2	8	0.058	0.232
Eucalyptus populnea	65	260	3.503	14.012
Hakea fraseri	1	4	0.001	0.004
Total	**68**	**272**	**3.562**	**14.248**
Site no: C23				
Bursaria spinosa	6	24	0.004	0.016
Callitris glaucophylla	223	892	2.588	10.352
Callitris glaucophylla xC. preissii ssp. verrucosa	9	36	0.251	1.004
Eucalyptus chloroclada	2	8	0.061	0.244
Eucalyptus melanophloia	6	24	0.580	2.320
Eucalyptus populnea	1	4	0.145	0.580
Opuntia tomentosa	12	48	0.048	0.192
Total	**259**	**1036**	**3.677**	**14.708**
Site no: C24				
Callitris glaucophylla	67	268	2.234	8.936
Callitris glaucophylla xC. preissii ssp. verrucosa	17	68	1.032	4.128
Eucalyptus chloroclada	6	24	1.721	6.884
Eucalyptus melanophloia	5	20	0.190	0.760
Eucalyptus populnea	2	8	0.026	0.104
Eucalyptus tereticornis	2	8	0.241	0.964
Total	**99**	**396**	**5.445**	**21.780**
Site no: C25				
Acacia harpophylla	318	1272	3.432	13.728
Eremophila mitchellii	11	44	0.158	0.632
Eucalyptus chloroclada	1	4	0.008	0.032
Eucalyptus melanophloia	196	784	1.114	4.456
Eucalyptus microcarpa	14	56	0.721	2.884
Geijera parviflora	77	308	0.384	1.536
Notelaea microcarpa var. microcarpa	7	28	0.044	0.176
Opuntia tomentosa	18	72	0.083	0.332
Unknown spp.	1	4	0.036	0.144
Total	**644**	**2576**	**5.982**	**23.928**

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Site no: C25				
Acacia excelsa	4	16	0.108	0.432
Acacia harpophylla	27	108	0.244	0.976
Brachychiton populneus spp. populneus	4	16	0.116	0.464
Callitris glaucophylla	325	1300	1.260	5.040
Casuarina cristata	1	4	0.018	0.072
Casuarina spp.	1	4	0.009	0.036
Eremophila mitchellii	10	40	0.041	0.164
Eucalyptus chloroclada	9	36	0.706	2.824
Eucalyptus melanophloia	1	4	0.008	0.032
Eucalyptus microcarpa	64	256	2.129	8.516
Eucalyptus populnea	4	16	0.246	0.984
Notelaea microcarpa var. microcarpa	25	100	0.124	0.496
Opuntia tomentosa	13	52	0.030	0.120
Santalum lanceolatum	5	20	0.120	0.480
Total	**493**	**1972**	**5.158**	**20.632**
Site no: C26 (26° 55' 25.03S, 149° 22' 08.34E)				
Acacia bidwillii	1	10	-	-
Acacia excelsa	8	80	0.052	0.52
Alectryon oleifolius	1	10	-	-
Allocasuarina luehmannii	3	30	0.013	0.13
Alphitonia excelsa	1	10	0.076	0.76
Callitris glaucophylla	127	1270	1.551	15.51
Canthium oleifolium	3	30	0.091	0.91
Corymbia clarksoniana	2	20	0.006	0.06
Corymbia tessellaris	10	100	0.058	0.58
Eremophila mitchellii	1	10	0.003	0.03
Eriocerius martinii	30	300	-	-
Eucalyptus chloroclada	2	20	0.22	2.20
Eucalyptus populnea	6	60	1.484	14.84
Geijera parviflora	7	70	0.052	0.52
Maytenus cunninghamii	6	60	-	-
Opuntia stricta	1	10	-	-
Opuntia tomentosa	32	320	-	-
Total	**241**	**2410**	**3.605**	**36.05**
Site no: C27 (26° 16' 59.60S, 148° 43' 09.77 E)				

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Acacia macradenia	14	47	5.116	17.036
Callitris glaucophylla	545	1815	0.003	0.010
Canthium oleifolium	3	10	0.341	1.136
Eucalyptus chloroclada	1	3	0.248	0.826
Eucalyptus melanophloia	6	20	-	-
Maytenus cunninghamii	1	3	-	-
Opuntia tomentosa	11	37	-	-
Total	**581**	**1934.73**	**5.707**	**19.004**
Site no: C28 (25° 20' 29.81S, 148° 6' 51.20E)				
Acacia longispicata	6	50	0.002	0.017
Angophora costata	24	200	-	-
Brachychiton populneus	4	33	-	-
Callitris glaucophylla	191	1591	1.53	12.745
Eucalyptus melanophloia	3	25	0.091	0.758
Eucalyptus chloroclada	9	745	0.104	0.866
Jacksonia scoporia	51	425	0.045	0.375
Maytenus cunninghamii	8	67	0.001	0.008
Melichrus spp.	3	25	-	-
Opuntia stricta	2	17	-	-
Opuntia tomentosa	2	17	-	-
Total	**303**	**2524**	**1.773**	**14.769**
Site no: C29 (25° 19' 47.18S, 148° 1' 9.85E)				
Acacia excelsa	4	20	0.003	0.015
Acacia leiocalyx	6	30	0.001	0.005
Callitris glaucophylla	262	1310	0.806	4.03
Eucalyptus clarksoniana	1	5	-	-
Eucalyptus melanophloia	98	490	-	-
Opuntia tomentosa	2	10	0.141	0.705
Total	**373**	**1865**	**0.951**	**4.755**
Site no: C30 (24° 34' 37.45S, 146° 28' 13.77E)				
Acacia excelsa	8	40	0.008	0.04
Archidendropsis basaltica	95	475	-	-
Brachychiton populneus	1	5	-	-
Callitris glaucophylla	38	190	0.533	2.665
Canthium oleifolium	131	655	0.002	0.01
Dodonaea viscosa	1	5	-	-

Species	No. of plants	No. per hectare	Plot Basal Area (m^2)	Basal Area (m^2/ha)
Eremophila mitchellii	2	10	0.001	0.005
Eucalyptus hybrid	2	10	-	-
Eucalyptus melanophloia	48	240	-	-
Eucalyptus populnea	19	95	0.624	3.12
Lysiphyllum hookeri	1	5	0.004	0.02
Opuntia tomentosa	4	20	-	-
Total	**350**	**1750**	**1.171**	**5.855**

Appendix 8-2 Lognormal relationships for components of Callitris glaucophylla, Hillside, south-west Queensland.

	Range	n	A	b	R2	RSD	SSDx
Callitris glaucophylla							
Equations in the form: ln y = a + b ln x.							
x = circ at 30 cm (cm), y = ln weight (kg)							
Trunk <1 cm*	13.0 - 216.5 cm	11	-2.9694	-0.4435	0.2621	0.5765	5.4010
Trunk 1-4 cm*	7.5 - 135.0 cm	20	-2.5650	0.4573	0.2297	0.6794	11.8468
Trunk 4-10 cm*	7.5 - 135.0 cm	20	-1.9496	0.8901	0.4501	0.7981	11.8468
Trunk 10-20 cm*	36.5 – 135.0 cm	15	-0.3037	0.9778	0.322	0.5806	2.1768
Trunk 20-30 cm*	63.0 – 135.0 cm	11	-15.6206	4.3986	0.6523	0.8492	0.6293
Trunk 30-40 cm*	88.5 – 135.0 cm	7	-42.0121	9.7774	0.9585	0.3583	0.1551
Branches <1 cm *	7.5 - 135.0 cm	20	-5.1834	1.5985	0.6188	1.0178	11.8468
Branches 1-4 cm*	19.8 – 135.0 cm	17	-10.2257	2.8784	0.9635	0.3356	5.3844
Branches 4-10 cm*	63.0 – 135.0 cm	11	-21.3995	5.1704	0.7192	0.8543	0.6293
x = height (m), y = weight (kg)							
Trunk <1 cm*	13.0 - 216.5 cm	11	-3.1475	-0.6499	0.3504	0.5409	3.3632
Trunk 1-4 cm*	7.5 - 135.0 cm	20	-2.2150	0.6026	0.2410	0.6744	7.1588
Trunk 4-10 cm*	7.5 - 135.0 cm	20	-1.3691	1.2142	0.5062	0.7563	7.1588
Trunk 10-20 cm*	36.5 – 135.0 cm	15	-0.4543	1.6097	0.3252	0.5792	0.8111
Trunk 20-30 cm*	63.0 – 135.0 cm	11	-15.1556	6.8312	0.5988	0.9122	0.2395
Trunk 30-40 cm*	88.5 – 135.0 cm	7	4.6046	-0.2924	0.0002	1.7586	0.0332
Branches <1 cm *	7.5 - 135.0 cm	20	-3.3329	1.8483	0.4999	1.1658	7.1588

	Range	n	A	b	R2	RSD	SSDx
Branches 1-4 cm*	19.8 – 135.0 cm	17	-7.7500	3.6721	0.8649	0.6458	2.9698
Branches 4-10 cm*	63.0 – 135.0 cm	11	-10.0476	4.2369	0.1838	1.4565	0.2395
x = canopy area (m^2), y = weight (kg)							
Trunk <1 cm*	2.117-40.432	11	-4.2466	0.2492	0.1436	0.6211	9.3743
Trunk 1-4 cm*	0.821-43.001	20	-1.2719	0.2667	0.1413	0.7173	21.4287
Trunk 4-10 cm*	0.821-43.001	20	0.6078	0.4982	0.2551	0.9289	21.4296
Trunk 10-20 cm*	2.121-43.001	15	2.9933	0.4009	0.2511	0.6102	10.0989
Trunk 20-30 cm*	6.362 – 43.001	11	0.3491	1.4547	0.4484	1.0695	3.9551
Trunk 30-40 cm*	12.370 – 43.001	7	-3.7158	2.4457	0.6139	1.0929	1.5874
Branches <1 cm *	2.117 – 43.001	20	-1.4039	1.3106	0.7524	0.8202	21.4295
Branches 1-4 cm*	2.121 – 43.001	17	1.7834	1.6570	0.8719	0.6288	14.7031
Branches 4-10 cm*	6.362 – 43.001	11	-3.9744	2.2052	0.8223	0.6797	3.9551
x = circumference at 130 cm, y = weight (kg)							
Trunk <1 cm*	9.5 – 98.7	11	-3.1449	-0.418	0.273	0.572	6.322
Trunk 1-4*	5.0 – 121.0	20	-2.399	0.434	0.245	0.673	14.026
Trunk 4-10*	5.0 - 121.0	20	-1.612	0.841	0.476	0.779	14.026
Trunk 10-20*	32.0 – 121	15	-0.078	0.956	0.325	0.579	2.297
Trunk 20-30*	55.8 - 121	11	-15.244	4.442	0.671	0.826	0.635
Trunk 30-40*	79.3 – 121	7	-43.644	10.402	0.937	0.441	0.134
Branches <1 cm*	5 – 121	20	-4.348	1.450	0.603	1.039	14.026
Branches 1-4 cm*	15 – 121	17	-9.208	2.731	0.967	0.318	6.005
Branches 4-10*	55.8 – 121	11	-20.936	5.217	0.739	0.824	0.635

* Components measured as diameter.

Appendix 8-3. Full species list of plants in the text (Sourced from Australian National Herbarium).

Acacia aneura F.Muell. ex Benth.

Acacia argyrodendron Domin

Acacia bidwillii Benth.

Acacia decora Rchb.

Acacia excelsa Benth.

Acacia harpophylla F.Muell. ex Benth.

Acacia ixiophylla Benth.

Acacia leiocalyx (Domin) Pedley

Acacia leptostachya Benth.

Acacia longispicata Benth.

Acacia macradenia Benth.

Acacia neriifolia A.Cunn. ex Benth.

Acacia spp. Mill.

Alectryon diversifolium (F.Muell.) S.T.Reynolds

Alectryon oleifolius (Desf.) S.T.Reynolds

Allocasuarina luehmannii (Miq.) L.A.S.Johnson

Alphitonia excelsa (Fenzl) Benth.

Angophora costata (Gaertn.) Britten

Angophora leiocarpa (L.A.S.Johnson ex G.J.Leach) K.R.Thiele & Ladiges

Angophora spp. Cav

Archidendropsis basaltica (F.Muell.) I.C.Nielsen

Atalaya hemiglauca (F.Muell.) F.Muell. ex Benth.

Bossiaea rhombifolia sub spp. concolor (Maiden & Betche) A.T.Lee

Brachychiton populneus (Schott & Endl.) R.Br.

Bursaria spinosa Cav.

Callitris glaucophylla Joy Thomps. & L.A.S.Johnson

Callitris glaucophylla xC. *preissii* ssp. *verrucosa* (Endl.) J.Garden

Callitris spp. Vent

Calytrix longiflora (F.Muell.)

Canthium oleifolium Hook.

Canthium spp. now *Pysdrax* spp. S.T.Reynolds & R.J.F.Hend.

Cassia spp. L.

Cassinia laevis R.Br.

Casuarina cristata Miq.

Casuarina spp. L.

Corymbia citriodora (Hook.) K.D.Hill & L.A.S.Johnson

Corymbia clarksoniana (D.J.Carr & S.G.M.Carr) K.D.Hill & L.A.S.Johnson

Corymbia papuana (F.Muell.) K.D.Hill & L.A.S.Johnson

Corymbia plena K.D.Hills & L.A.S.Johnson

Corymbia tessellaris (F.Muell.) K.D.Hill & L.A.S.Johnson

Daviesia ulicifolium sub spp. stenophylla G.Chander & Crisp

Dodonaea boroniifolia G.Don

Dodonaea peduncularis Lindl.

Dodonaea viscosa (l.) Jacq.

Dodonaea viscosa sub spp spatulate (Sm.) J.G.west

Eremophila deserti (A.Cunn. ex Benth.) Chinnock

Eremophila mitchellii Benth.

Eremophila spp. R.Br.

Eriocerius martini (Labour.) Riccob.

Eucalyptus acmenoides Schauer

Eucalyptus brownii Maiden & Cambage

Eucalyptus cambageana Maiden

Eucalyptus chloroclada (Blakely) L.A.S.Johnson & K.D.Hill

Eucalyptus chlorophylla Brooker & Done

Eucalyptus coolabah Blakely & Jacobs

Eucalyptus crebra F.Muell.

Eucalyptus dolichocarpa now *Corymbia clarksoniana* (D.J.Carr & S.G.M.Carr) K.D.Hill & L.A.S.Johnson

Eucalyptus exserta F.Muell.

Eucalyptus fibrosa F.Muell.

Eucalyptus intermedia R.T.Baker

Eucalyptus macrocarpa (Maiden) Maiden

Eucalyptus melanophloia F.Muell.

Eucalyptus microneura Maiden & Blakely

Eucalyptus moluccana Roxb.

Eucalyptus orgadophila Maiden & Blakely

Eucalyptus platyphylla F.Muell.

Eucalyptus populnea F.Muell.

Eucalyptus quadricostata Brooker

Eucalyptus signata F.Muell.

Eucalyptus similis Maiden

Eucalyptus tectifica F.Muell.

Eucalyptus tereticornis Sm.

Eucalyptus whitei Maiden & Blakely

Geijera parviflora Lindl.

Grevillea robusta A.Cunn. ex R.Br.

Hakea fraseri R.Br.

Hovea longipes Benth.

Jacksonia scoporia Sm.

Leucopogon biflorus R.Br.

Leucopogon mitchellii Benth.

Lomandra multiflora sub spp. multiflora (R.Br.) A.T.Lee

Lophostemon suaveolens (Sol. Ex Gaertn.) Peter G.Wilson & J.T.Waterh.

Lysicarpus angustifolius (Hook.) Druce

Lysiphyllum hookeri (F.Muell.) Pedley

Maytenus cunninghamii (Hook.) Loes.

Melaleuca quiquinervia (Cav.) S.T. Blake

Melaleuca viridiflora Sol. Ex Gaertn.

Melichrus spp. R.Br

Myoporum deserti Benth.

Notelaea microcarpa var. macrocarpa R.Br.

Opuntia stricta (Haw.) Haw.

Opuntia tomentosa Salm-Dyck

Philotheca difformis sub spp. difformis (Endl.) Paul G.Wilson

Santalum lanceolatum R.Br.

Syzygium cormiflorum (F.Muell.) B.Hyland

Xylomelum cunninghamianum Foreman