Phytocapping of Municipal Landfills: Evaluating the Performance of 21 Tree Species and Two Soil Depths

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A thesis submitted in partial fulfilment for the requirement of a Doctor of Philosophy

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August 2013

Declaration

To the best of my knowledge and belief, the work presented in this thesis is my own, except where specific acknowledgement has been made. Soil hydraulic, physical and chemical properties were determined by Dr Ian Philips and Associate Professor Nanjappa Ashwath and have only been used as reference in this thesis. This field trial was established in 2003 by Associate Professor Nanjappa Ashwath and data presented in this thesis is a result of series of investigations and experiments conducted since January 2005.

I hereby declare that I have not submitted this material either in whole or in part for a degree at this or any other institution.

I acknowledge the following contribution to the content of this thesis.

Kartik Venkatraman
Date: 23/08/2013

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Date: 23/08/2013

Abstract

Modern living inevitably results in waste generation. Such wastes must be managed effectively to minimise adverse environmental impacts, such as groundwater contamination due to leachate and methane emission. Local Governments in Australia are using various landfill remediation technologies such as landfill capping, leachate collection and methane flaring to address the issues outlined above.

Capping is a mandatory landfill closure procedure to isolate the waste from outside environment, mainly rain water. Clay caps are mostly expensive and often fail to limit entry of water into the waste by developing cracks due to desiccation. To reduce capping costs and to increase environmental benefits, an alternate capping system called 'Phytocapping' was trialled at Lakes Creek Road Landfill, Rockhampton, Australia. This system consists of a soil cover and vegetation. Soil cover stores water during rainfall events and the vegetation removes the stored water via transpiration. Trees also act as 'rain interceptors' by trapping certain proportion of the rainfall in their canopy. Soil and plants also contribute to reduced methane emission by supporting methanotrophs in their root zone.

In the past, phytocapping studies have used grasses, shrubs, herbaceous species and a few tree species. This thesis has examined the role of 21 tree species grown on two soil covers (Thick cover; 1400 mm and Thin cover; 700 mm) and evaluated their characteristics such as transpiration and rainfall interception in reducing percolation of water into the waste. The thesis has also assesed the importance of soil depth for tree growth and maintenance of hydrological balance of the phytocapping system. The role played by the trees and the soil cover in reducing methane emission, as well as contribution of trees to canopy interception has also been studied.

Studies in the past have associated tree mortality and low growth rates on landfills to shallow soil depth, landfill gases, soil compaction and high soil temperature. However, in this study, majority of the established species survived well and accumulated biomass at a rate comparable to those growing on natural soils. Fast growing species produced 100 to 125 t ha^{-1} shoot biomass in 3.5 years. Some species

such as *Hibiscus tiliaceus* grew well, transported a considerable amount of water and were also resilient to drought. All species except *Dendrocalamus latiflorus* exhibited a well developed tap root system with lateral root growth. The majority of species had a rooting depth of 60 cm, with the most of their fibrous roots concentrated in the top 40 cm. Root length density was highest in the top 20 to 30 cm of soil for most species.

Foliar chemical analysis showed adequate amounts of nutrients for all elements except for phosphorus. A large proportion of these nutrients were retained in leaf litter. Analysis of the foliage and the leaf litter for heavy metals revealed no elevated levels of heavy metals (except in one species that accumulated elevated levels of cobalt). The leaf litter contained slightly higher levels of heavy metals than live foliage. Comparison of mineral compositions of these plants with those published, indicated that none of the elements (except cobalt in one species) showed any concern in terms of environmental pollution or on tree growth.

The long-term transpiration monitoring data show that these trees can remove up to 6.25 mm d^{-1} of water and some of these can also survive at transpiration rates as low as 0.1 mm d^{-1}. The average transpiration rates of the 15 tested tree species ranged between 0.9 to 2.1 mm d^{-1}, with an overall mean of 1.4 mm d^{-1}. The selected species also showed rapid response to irrigation and rainfall events, and were able to reduce water uptake during dry periods.

Some species were able to intercept up to 50% of the rainfall on a per storm basis, with an overall average of 30%. This is a significant contribution towards the hydrological balance of the phytocapping system. Stemflow was estimated at 4.5% of the total rainfall. Overall, the results clearly demonstrate the need to consider plant canopy interception as an essential parameter in modelling site water balance.

HYDRUS 1D model predicted percolation rates of 16.7 to 23.8 mm yr^{-1} in Thick and Thin phytocaps respectively. This equates to 2.1% to 3.1% of the average rainfall at Rockhampton. Removal of all vegetation components in the water balance model suggested percolation rates of 17% to 19% of rainfall. This emphasised the importance of vegetation in maintaining site hydrological balance.

Methane flux in the Thick phytocap was >0.0009 g m^{-2} d^{-1} and in the Thin phytocap was <0.0007 g m^{-2} d^{-1}. These values were up to four times lower than those measured in the adjacent non-vegetated landfill that had only the day cover (>0.0036 g m^{-2} d^{-1}). Although the measured flux was less than that reported in literature, there is a clear indication that the phytocaps can make a significant contribution to reduced methane emission from landfills.

Literature review and published reports on economics of landfill capping suggest that phytocaps are less expensive (up to 50% cheaper) to construct than clay caps in many instances, particularly to councils that do not have local supplies of clay. Overall cost of construction and maintenance of clay cap (for at least 30 years) is costly. However this may vary with specific site conditions; for example landfills situated in high rainfall regions which may require thicker layers of soil cover. Most of the cost analysis conducted for landfill remediation/capping till date includes only cost of construction of the capping system and not the costs of maintenance. Phytocaps cost much less to maintain than conventional covers, and hence long-term maintenance cost of phytocaps may be more important than construction cost itself. If long term maintenance costs are factored in the analysis, then phytocaps would have a better advantage.

Overall, 1000 to 1500 mm of soil cover and establishment up to 10 selected tree species should be able to reduce percolation of water into the waste at the Rockhampton landfill site. Data from this research has been instrumental in undertaking nation wide studies on phytocapping. This research has also contributed to a relaxation in the landfill license condition and assisted in gaining government approval to use phytocap as the final cap at Lakes Creek Road Landfill, Rockhampton, Australia.

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Abbreviations

AACAP	Australian Alternative Cover Assessment Program
ACAP	Alternative Cover Assessment Program
ACT	Australian Capital Territory
AEC	Alternative Earthen Final Caps
AEST	Australian Eastern Standard Time
ALCD	Alternative Landfill Cover Demonstration Project
ANOVA	Analysis of Variance
AWT	Alternative/Advanced Waste Technology
BCC	Brisbane City Council
BOD	Biochemical Oxygen Demand
BOM	Bureau of Meteorology
C&D	Construction and Demolition
CFI	Carbon Farming Initiative
COD	Chemical Oxygen Demand
CPRS	Carbon Pollution Reduction Scheme
CQLGA	Central Queensland Local Government Association
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DBH	Diameter at Breast Height
DEHP	Department of Environment and Heritage Protection
DERM	Department of Environment and Resource Management
DoCC	Department of Climate Change
DPI	Department of Primary Industries
EfW	Energy from Waste
EPA	Environmental Protection Authority/Agency
EPIC	Erosion-Productivity Impact Calculator
ET	Evapotranspiration
ETC/ET	Evapotranspiration Caps (ETC),
GCL	Geosynthetic Clay Liner
HBM	Heat Balance Method
HDPE	High Density Polyethylene
HELP	Hydrologic Evaluation of Landfill Performance
HRM	Heat Ratio Method
INEEL	Idaho National Engineering and Environmental Laboratory"
LAI	Leaf Area Index
LEACHM	Leaching Estimation and Chemistry Model
LI	Light Interception
LSD	Least Significant Difference
MEDLI	Model for Effluent Disposal using Land Irrigation
MRET	Mandatory Renewable Energy Targets
MRF	Materials Recovery Facility
MSW	Municipal Solid Waste
NSW	New South Wales
NYSDEC	New York State Department of Environmental Conservation
OECD	Organisation for Economic Co-operation and Development
PET	Potential Evapotranspiration
PVC	Poly Vinyl Chloride
QMS	Queensland Maritime Safety
RCRA	Resource Conservation and Recovery Act

RH	Relative Humidity
RLD	Root Length Density
RRC	Rockhampton Regional Council
SHAW	Simulation of Heat and Water
SPC	Soil-Plant Caps
SRC	Store-and-Release Caps
STOMP	Subsurface Transport over Multiple Phases
SWIM	Soil Water balance and Infiltration Model
TDP	Thermal Dissipation Probes
TPI	Transpacific Industries
UK	United Kingdom
US	United States
USDoE	United States Department of Energy
VLC	Vegetative Landfill Caps
VPD	Vapour Pressure deficit
WBC	Water Balance Caps
WCS	Wright Corporate Strategy
WMAA	Waste Management Association of Australia
WTS	Waste Transfer Station

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Awards and Recognition

1. Growing the Smart State PhD Award, Queensland Government (2005)

2. Growing the Smart State PhD Award, Queensland Government (2006)

3. Best Project Award (Regional), Institute of Sustainable Resources (ISR), The Society for Sustainability and Environmental Engineering (SSEE) and Waste Management Association of Australia (WMAA) (2007)

4. Sustainable Project Award, Institute of Sustainable Resources (ISR), The Society for Sustainability and Environmental Engineering (SSEE) and Waste Management Association of Australia (WMAA) (2007)

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Acknowledgments

A PhD is a colossal task which cannot be achieved without guidance and help from numerous people, to whom I am very grateful. The scholarship from the Centre for Plant and Water Sciences (CPWS) and CQUniversity has been extremely supportive in helping me reach completion.

To begin with, I express my gratitude to my supervisors Assoc. Prof. Nanjappa Ashwath and Dr. Ninghu Su (James Cook University) for their constant support, understanding, guidance, direction and sharing their experiences during this research. I would also like to thank Richard Yeates (Phytolink Pty Ltd), Craig Dunglison (Rockhampton Regional Council) and Rockhampton Regional Council for providing me with on-site technical assistance, labour and on-ground support as and when required. I would like to thank Prof. David Midmore for giving me the opportunity to work with CPWS and also for his guidance throughout my research.

I would like to make a special mention of Roshan Subedi and Alison Clarke and thank both of them for their tireless effort and assistance in my field work and experiments. I would also like to thank the two international exchange program students; Alma Garcia from Mexico and Valeria Venturie from Italy for their assistance in field work and data collection.

I offer my sincere gratitude to Linda Ahern, Dr Ajay Sharma, Dr Surya Bhattarai, Dr Ajay Krishnamurthy, Ben Kele and Dr Phul Subedi for their continuous guidance and assistance during my research. I am thankful to Centre for Plant and Water Science (CPWS) and staff for their timely support during my research. I am grateful to Jenny Moore (Central Queensland Local Government Association) for her support to my thesis and her help in propagating the technology throughout Central Queensland and Michael Asnicar (Manager – Logan Waste Services) for granting me the flexibility and time to work on my thesis during work hours. I am also grateful to Mr Chris Waites (Director – Operations, East Gippsland Shire Council) for his support.

I was fortunate to be awarded the CPWS scholarship, CQU Scholarship (UPRS) in 2005, which allowed me to pursue my PhD without any financial stress. I was also fortunate receive the Smart State Award in 2006 and 2007, which allowed me to pursue my PhD without any financial stress and incorporate extensive research and additionally present a series of papers at various National and International Conferences. This Award also presented me with an opportunity to work under the mentorship of Dr Bodapati Naidu and Dr Ram Dalal from the Department of Environment and Resource Management.

Further, I would like to place on record the following people without whom the completion of this research wouldn't have been possible.

Kerrie Thompson (Office of Research) and her team for administering my research progress and catering to my research needs as and when I required

Dr Jirka Simunek (University of California) for assisting me with HYDRUS 1D Modelling

Assoc. Prof. Yuyue Yang (Xiangtan University) for assisting me with STOMP Modelling

Alec Downy (ICT international) for assisting me with the setup of sapflow sensor technology

Nigel Tuckwood, Lindsay Best and John Russell (Rockhampton Regional Council) for their on-site assistance

James Fox (Brisbane City Council) for providing information on the economics of capping

Queensland Maritime Service for providing tidal information

Bureau of Meteorology for providing Rockhampton meteorological data

Department of Primary Industries for giving me access to their equipment as and when required

Prof. Mark Westoby & Dr Ian Wright (Macquarie University) for giving me the opportunity to use their leaf toughness tester

Dr Natalia Deeva (PELM, CQ University) for assisting me with an electron microscope to determine leaf hair density

Lee Barnett (CQUniversity), Andrew Hammond (CQUniversity), Trudy Baker (Capricorn Pest Management), Jennifer Smith (Australian Red Cross), David

Moy (Techsearch Pty Ltd) and Lakshmi Nathan (HSBC Bank Middle East Ltd) in proof reading my thesis

I arrived in Rockhampton with all sorts of different feelings; however the friendship and support from my friends here have made this place a “home” for me. Last and certainly not the least, I thank Swathi, my wife and Tvisha, my daughter, for their patience, encouragement and motivation throughout my research. My deepest respect to my parents, brother, my in-laws and the extended family for constantly supporting me and boosting my morale. Finally, I thank GOD for His blessings which helped me reach research completion.

1

General Introduction*

1.1 Introduction

Modern living inevitably results in waste generation. This waste needs to be managed appropriately, as it causes environmental problems. Local Government "Councils" in Australia strive hard to minimise these problems through various technologies, predominantly landfilling of Municipal Solid Waste (MSW). The term "Landfill" is used to describe a physical facility that contains solid wastes and waste residue on the earth's surface (Tchobanoglous and O'Leary 1994). Councils use landfills, as land disposal of waste has been found to be the most economic means of dealing with MSW (Scott *et al.* 2005, Izzo *et al.* 2009, and Tonini *et al.* 2009).

There are two types of MSW landfills: - (i) open dumps, which are covered with a layer of soil (Themelis and Ulloa 2007); and (ii) engineered or regulated landfills that have liners, gas and leachate collection systems and clay caps at the top (Tchobanoglous and O'Leary 1994, Themelis and Ulloa 2007). Both types of landfills currently exist in Australia.

1.1.1 The Predicament

Australia is one amongst the ten highest solid waste generators in the Organisation for Economic Cooperation and Development (OECD) (Scott *et al.* 2005, Weikhardt 2006), as 50% to 60% of the total MSW generated is placed in landfills (Weikhardt 2006, EPA 2007). In Queensland alone, around 1.7 million tonnes of domestic waste was deposited in landfills in 2006, of which 60,000 tonnes of waste was generated in Central Queensland (CQ) alone (EPA 2006). Due to increasing quantities of wastes being produced, landfill owners and operators have great concerns about

* Some information from this chapter has been accepted for publication in *Critical Reviews in Environmental Science and Technology* [ISSN: 1064-3389]

environmental problems such as gas emissions and leachate generation from these landfills.

In Australia, more than 70% of the population live in coastal areas (EPA 2006). Thus construction of landfills has predominantly occurred in low lying areas and closer to urban developments. This inadvertent practice of siting landfills in the past combined with no or minimum environmental considerations has introduced numerous environmental problems such as groundwater contamination and methane generation which contributes towards global warming and affects urban populations that dwell around landfills (CSIRO 2001, EPA 2002).

Municipal solid waste landfills predominantly contain putrescible wastes which produce leachate and methane gas (CSIRO 2001) when these come in contact with water. "Leachate is the medium by which soluble materials contained in the landfill are transported to the environment" (Christensen *et al.* 1994). Leachate generation occurs due to water entering into the landfill. Rain is the major source of water which moves through the landfill system through condensation, evaporation, precipitation, runoff, infiltration and transpiration (Fig. 1.1).

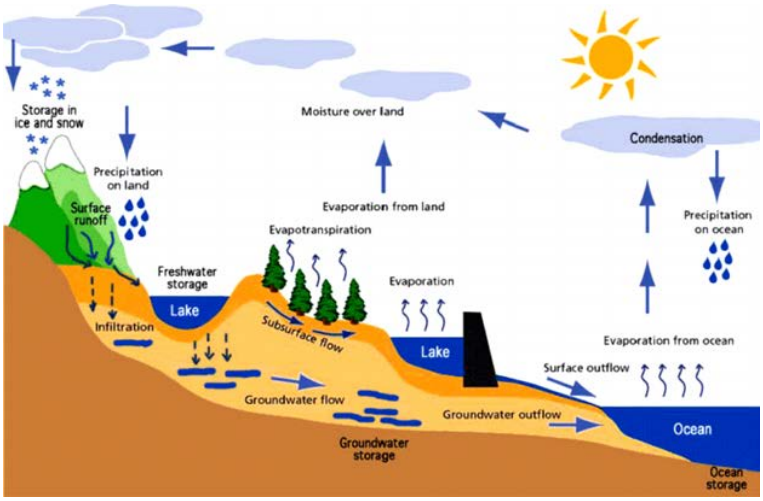


Figure 1.1: Hydrological Cycle
(Source: http://dardel.info/images/hydrologic_cycle.gif)

Today several policies and regulations are in place to reduce leachate generation and release of landfill gases into the environment. Waste management practices such as landfill capping, lining, waste segregation, leachate recycling and gas recovery

systems have been employed in Australia to reduce adverse environmental impacts of landfills on the surrounding environment (Scott *et al.* 2005). A report by the Environmental Protection Agency (EPA) reveals that only 2% of the landfills in Queensland have gas collection systems and only 28% have leachate collection systems (EPA 2006). In CQ, only 12% of the landfills have leachate collection systems, but none of the landfills have been installed with a gas collection system (EPA 2006).

1.1.2 State of Landfills in Australia

Landfilling practice in Australia has undergone significant changes over the last few decades (Scott *et al.* 2005). In the early 19th century, wastes were collected and disposed into open dumps. Concerns over public health and other environmental issues arising due to open dumping led to the introduction of sanitary/engineered landfills by the 20th century in Australia (Scott *et al.* 2005), and US and Canada (US EPA 2002). This also contributed to significant changes to requirements of siting, management and operation of landfills (Scott *et al.* 2005).

Many of the MSW landfills that were built in Australia prior to 1993 have liner systems (Friends of the Earth 2000) and have become major sources of local groundwater contamination (Janechek and Prosser 1995, Clister *et al.*1997) and gas emission (Bogner *et al.* 1995). The Resource Conservation and Recovery Act (RCRA) instituted in 1976 in the US made it mandatory to cap the landfills as closure remediation to prevent water from percolating into the waste (Licht *et al.* 2004).

In the past, landfills in Australia were operated with minimal consideration of long term environmental impacts (Thornton 2002). A survey conducted by Xu *et al.* (1999) revealed that only one-third of landfills in Australia were lined, just over half had gas collection systems and only two-third had installed leachate treatment systems. Landfill lining is generally accomplished using a low hydraulic conductivity material such as compacted clay, Geosynthetic Clay Liners (GCLs) and/or artificial liners such as High Density Polyethylene (HDPE) (Benson 2000). Capping of landfills has also been undertaken in addition to placing liners at the

bottom of the landfill. Leachate recirculation systems have been extensively used in the US and in other parts of the world (Reinhart 1996). Research into potential application of leachate recirculation systems have also been conducted in Australia (Yuen 1999).

There are more than 600 landfills in Australia (Bateman 2005, Johnston 2009) and these include MSW landfill, Co-disposed landfills (MSW and Industrial); and Monofills/Monocells (industrial and hazardous wastes) (Scott *et al*. 2005).

1.1.3 Environmental Impacts of Landfills

1.1.3.1 Waste Collection and Separation

Waste is separated at source through kerbside recycling and/or via self-haulage of green waste, recyclables such as cardboards, glass and plastics and regulated waste such as tyres, oil, gas bottles and batteries to increase landfill capacity and reduce the impact of landfills on the environment. Although several efforts have been made to reduce waste going to landfill through awareness programs, waste reduction programs and introduction of State and Federal policies and strategies; a significant amount of waste still generated and landfilled every month via kerbside collection, special burial, commercial skip bins and self-hauled household and commercial waste to transfer stations. These wastes attract vermin, birds, generate odour, and produce leachate and gases that have an adverse impact both on the environment and on the community.

1.1.3.2 Vermin, Birds and Odour

Wastes deposited in landfills attract vermin, rodents and birds which in turn create nuisance to landfill operators and the local community. Ibis, an Australian native bird is a great nuisance in and around landfills in Queensland. These birds are a threat to the surrounding flora and fauna. Odour is another issue faced by landfill operators. Several landfill operation technologies such as deodorising units, landfill lids and waste compaction are being adopted by landfill operators to reduce the impact of vermin, birds and odour on the community and environment. Many Councils in Queensland have adopted the Ibis management program to control these birds within and around landfill sites.

1.1.3.3 Leachate

Landfills have adverse environmental impacts as these produce landfill leachate and generate landfill gases; predominantly methane (Scott *et al.* 2005). These impacts may give rise to explosions, vegetation damage, odour, groundwater pollution, contamination of water bodies and air pollution (El-Fadel *et al.* 1997). Landfill leachate not only contaminates groundwater but also deposits organic solvents, salts and heavy metals in the soil and groundwater (Al-Yaquot 2003). Leachate composition and concentration varies between landfills depending on the type of waste deposited (Kalyuzhnyi *et al.* 2003) and the age of the landfill (Ragle *et al.* 1995). Leachates have high Chemical Oxygen Demand (COD) (700 mg COD L^{-1}) (Kalyuzhnyi *et al.* 2003). High levels of COD are also due to slow degradation of organic waste (Fatta *et al.* 1999). Leachates from Australian landfills could have a Biological Oxygen Demand (BOD_5) of 11,400 mg L^{-1}; conductivities of 240 to 22,000 µS cm^{-1} and lead concentrations up to 0.1 mg L^{-1} (Scott *et al.* 2005). Scott *et al.* (2005) reviewed environmental impacts of various landfills in Australia, and a summary of constituents of leachate are presented in Table 1.1.

Table 1.1: Leachate composition in Australian landfills

Landfill leachate composition	Values in mg L^{-1} except where indicated	Landfill leachate composition	Values in mg L^{-1} except where indicated
pH	5.3 - 8.9	K	62 - 1200
COD	50 - 14000	Na	71 - 3900
BOD_5	6.8 - 11400	As	0.008 - 0.07
DO	0-4	Ba	< 1 - 22.5
EC (µS/cm)	240 - 22000	B	1.3 - 2.2
Temp (°C)	15 – 38	Cu	< 1
SS	8.6 - 2600	Cd	< 0.05
TDS	270 - 14000	Cr	0.13
TOC	59 - 6200	Fe	0.56 – 235
TKN	0.14 - 42.7	Pb	< 0.1
NO_3^-	< 0.05 -2.3	Mn	< 0.05 - 0.95
NO_2^-	< 0.05 - 0.21	Hg	< 0.0002
N (organic)	< 2 – 70	Ni	0.07 - 0.08
PO_4 (Inorganic)	31.5 µg/L	Se	< 0.01
P (total)	80.6 µg/L	Ag	< 0.05
Cl^-	40 - 6400	Zn	0 – 12
SO_3^{2-}	0.2 - 1.03	Phenols	0 – 244
SO_4^{2-}	1.4 - 295	Toluene	0 – 1
$CaCO_3$	490 - 4500	Xylene	0 - 0.5
Ca	70 – 350	Benzene	0 - 0.04
Mg	0.74 - 540	Ethyl Benzene	0 - 0.12

COD: Chemical Oxygen Demand, BOD_5: Biological Oxygen Demand, DO: Dissolved Oxygen, EC: Electrical Conductivity, SS: Suspended Solids, TDS: Total Dissolved Solids, TOC: Total Organic Content, TKN: Total Kjeldahl Nitrogen. Source: (Scott *et al.* 2005)

Many cases of groundwater pollution have been reported globally due to poorly-designed landfill sites and the malfunctioning of landfill capping. For example, Assmuth and Strandber (1993) in Finland, Badv (2000) in Argentina, Neufeld (2000) in Canada, Bocanegra *et al.* (2001) in Iran, Van Nooten *et al.* (2009) in Belgium, Modin *et al.* (2009) in Sweden, Haarstad *et al.* (2009) in Norway, Aivalioti *et al.* (2009) in Greece and Jeevanrao and Shantaram (2003) in India reported groundwater contamination from landfills. Similar problems were cited at the South Fremantle landfill in Western Australia (Dunnet 2004). Adverse impacts of landfill leachate on groundwater have prompted numerous studies since 1980, which in turn has led to the evolution of new technologies (Fatta *et al.* 1999). Flyhammer (1995) examined leachate quality in several Swedish landfills and Sanchez *et al.* (1993) in Madrid.

Similar studies were undertaken on leachate quality by Zheng *et al.* (1991) in New Jersey, Gailey and Gorelick (1993) in Canada and Blight (1995) in South Africa.

1.1.3.4 Methane Production in Landfills

In 2002, the Australian waste sector (MSW disposal on land, waste water disposal and waste incineration) contributed to approximately 3.2% of the total (total = 550.1 Mt) carbon dioxide equivalent emissions, of which 87.6% was derived from solid waste disposal sites (Scott *et al.* 2005). Problems associated with landfill gases are:

- Accumulation of gas to explosive quantities (Scott *et al.* 2005)
- Odour (Hudgins *et al.* 2002, Sullivan *et al.* 2004)
- Groundwater acidification (Park and Lee 2002)
- Reduced plant growth after site restoration (Department of the Environment 1986).

Reports of accidents involving explosions and migration of methane from landfills have been reported in recent years (Gendebien *et al.* 1992, Christensen *et al.* 1996, Cooper 2008).

Methane, a major component of landfill gas currently counts for 15% to 20% of the greenhouse budget (EPA 2003). Landfill gas is produced from the biological degradation of wastes (Jones and Nedwell, 1993, Giani *et al.* 2002) and comprises methane (45% to 60% v/v) and carbon dioxide (40% to 60% v/v) (Swarbrick and Dever, 1999). In Australia, methane concentrations range from 50% to 60% (v/v) of the total landfill gas produced (Duffy *et al.* 1996, Hansen *et al.* 1998, Yuen 1999) and is comparable with the global emissions (Bogner *et al.* 1996). However a study conducted by the Department of Climate Change (2006) showed that methane composition of Australian landfill gases was ca. 75% of the total gas released; which may possibly be due to presence of hot tropical conditions in many parts of the country.

Methane production in landfills normally takes place in two stages – the non-methanogenic stage and the methanogenic stage (Farquhar and Rovers 1973, Whalen *et al.* 1990, Czepiel *et al.* 1995, Borjesson and Svensson 1997). The non-methanogenic stage is initiated by a hydrolytic process which reduces complex organic matter to smaller soluble compounds such as fatty acids, simple sugars,

amino acids and other low molecular weight organic compounds (Imshenetsky 1968). This phase accomplishes further modification of the organic material by capturing energy and forming organic acids, ammonia, water, hydrogen and carbon dioxide (Farquhar and Rovers 1973). The methanogenic stage is initiated by microorganisms under aerobic conditions (Urmann *et al.* 2007). The microorganisms active in the methanogenic stage are generally bacteria of the genus *Methanobacterium*; common inhabitants of soil, sewage (Alexander 1971) and marine sediments where both oxygen and methane are present (Hanson and Hanson 1996). Odorous gases such as nitrogen and hydrogen sulphide are also produced during this stage (Alexander 1971).

Normally, a steady rate of methane production is reached after 80 to 500 days of waste deposition, and this state is maintained for 10 to 20 years (Moore *et al.* 1998). The time required for degradation of waste in landfills and the amount of gas formed depends upon a number of factors. These include type and amount of buried waste, water content of the waste, compaction and leachate recycling procedures (Farquhar and Rovers 1973).

1.1.3.5 Methane Emissions from Landfills

The average annual methane emissions from a municipal solid waste landfill in Canberra were estimated to be 10 kg methane per tonne of waste (Denmead 1995, unpublished data).

Soils are also the only known biological sinks ((Mosier 1998, Hutsch 2001, Reay *et al.* 2004). Reay *et al.* (2001) demonstrated the effect of five species (alder, oak, Norway spruce, Scot pine and grassland) on methane oxidation in temperate soils in an experimental plot in Northwest England. The trial plot consisted of (0.2 ha) four pure stands of Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* L.), alder (*Alnus glutinosa* L.), oak (*Quercus petraea*), together with a grass control. Each of the five treatments was planted in triplicate blocks, with treatment plots being located randomly within each block (Fig. 1). No fertilizers were applied either at planting or subsequently. The trial plot showed low methane oxidation rates compared to the Norway spruce, Scots pine and grassland. European beech (*Fagus sylvatica*) exhibited oxidation rates three times higher than that of Norway spruce on

the same soil type in Germany (Borken *et al.* 2003). This difference was due to the variation in monoterpenes in different species (Maurer *et al.* 2008). Monoterpenes are naturally occurring compounds produced by plants (Amaral *et al.* 1997). European beech produces and releases these compounds into the soil in very low concentrations (Holzke *et al.* 2006), whereas Norway spruce produces high concentrations of monoterpenes in needles, twigs, bark and buds (Bufler *et al.* 1990). Monoterpenes have a tendency to inhibit growth and activities of methanotrophs in the soil (Amaral *et al.* 1998).

Methane is the main hydrocarbon present in the atmosphere, with an average concentration of 1.7 parts per million (ppm) (Borjesson and Svensson 1997, Humer and Lechner 1999) and "a molecule of methane in the atmosphere has a global warming potential approximately 25 times higher than that of a carbon dioxide molecule (Forster *et al.* 2007) because of methane's higher infrared activity" (Abichou *et al.* 2006, IPCC 2007). Bingemer and Crutzen (1987), Richards (1989) and Bogner (2003) estimated 9 to 70 Tg yr^{-1} of methane emission from landfills. The rate of emission of methane (methane flux) varies over orders of magnitude, from less than 0.0004 g m^{-2} d^{-1} to about 4000 g m^{-2} d^{-1} (Bogner *et al.* 1997), and in some occasions more than 10,000 g m^{-2} d^{-1} was found in the US (Spokas *et al.* 2006). For example, Jones and Nedwell (1993) and Nozhevnikova *et al.* (1993) reported 0 to 38.4 g m^{-2} d^{-1} in the UK and Moscow. De Visscher (1999) reported 0.0048 to 4000 g m^{-2} d^{-1} in a laboratory-based soil column study in Belgium, and Tohijima and Wakita (1993) found 200 g m^{-2} d^{-1} in Tokyo. Methane flux of 45 g m^{-2} d^{-1} in Germany (Jager and Peter, 1995) and 10.5 g m^{-2} d^{-1} in France (Pokryszka *et al.* 1995) have also been reported. Sengupta *et al.* (1998) found 0.3 to 0.7 g m^{-2} d^{-1} of methane flux in Indian landfills. Similarly, a laboratory test conducted by Gebert *et al.* (2009) showed an oxidation rate of 0.2 to 426 g m^{-2} h^{-1}.

1.1.3.6 Effect of Methane on Plants

Methane is generally non-toxic to plants or other organisms (Chan *et al.* 1991, Trotter and Cooke 2005). A major effect of methane in soils is methane oxidation which depletes oxygen present in the landfill, increases carbon dioxide levels (Trotter and Cooke 2005) and also raises soil temperatures (Fischer 1999). These changes in soil gas composition may contribute to plant death by enhanced

asphyxiation due to lower solubility of gases inside plant cells or by drying of soil (Fischer 1999).

1.1.3.7 Vegetation Management

Vegetation in and around landfills have to be well maintained and weeds need to be cleared regularly and the landfills need to be sprayed with herbicide as part of the regular maintenance schedule. Weeding and clearing of vegetation from buffer zone is essential to avoid landfill fires to spread into surrounding properties.

1.1.3.8 Daily Cover and Final Landfill Capping

To mitigate adverse effects of landfills, the State Government has made it mandatory to cover the waste on a daily basis using soil (daily cover, 300 mm deep). The State Governments in Australia has also made it mandatory to cap the landfill using clay once the landfill is completely filled and to maintain the integrity of the clay for at least 30 years to ensure low risk to the community and lower environmental impacts due to leachate and gas generation.

1.2 Current Landfill Technologies

Landfills pose threat to the environment due to their very presence and also due to operations involved in managing and burying waste. Landfill operations include, spreading of waste using a dozer, compacting waste and stripping the daily cover. These operations are carried out to maintain the integrity of landfills, increase landfill capacity and reduce environmental impacts. Another landfill operation technology is the bioreactor landfill, which has been successfully implemented in Australia.

Bozkurt *et al*. (2000) have developed a conceptual model of landfill wastes and their associated physico-chemical properties that lead to the production of leachate. Leachate collection systems (Rowe *et al.* 1997, Rittman *et al.* 1998) and landfill liners (Alston *et al.* 1997, Halse *et al.* 1990, Benson 2000) have been extensively used in landfill design and construction. A number of other techniques can also be used to reduce impact of leachate, and these are shown in Table 1.2.

Table 1.2: Various leachate treatment techniques

Types of Treatment	Reference
Aerated lagoon	Robinson *et al.* (1992)
Anaerobic lagoon	Blakey *et al.* (1992)
Rotating biological contactors	Knox (1992)
Physico-chemical treatment	Cossu *et al.* (1992)
Reverse osmosis	Li and Heine (2007)
Activated sludge technique	Avezzu *et al.* (1992)

Gas emission from landfills has led to significant climate change over the last few decades (El-Fadel and Massoud 2000). Significant amounts of money and time have been spent in treating, extracting and trying to reduce methane emission and leachate generation from landfills. Methane recovery, although an expensive affair, is being implemented and adopted in developed countries. Feasibility of this system greatly depends on types of gases, and the amount of methane gas emitted by the landfill (Willumsen 2002). Methane from landfills has been extracted, recovered and used for various purposes including bioenergy generation (De Walle *et al.* 1978, Borjesson *et al.* 2000), and these activities have considerably reduced the quantities of methane being released into the atmosphere. Willumsen (2002) reported approximately 25 methane gas recovery plants operational in Australia in 2001, and this number should now have increased. In 2006, 103 MW of electricity was generated from landfill gas in Australia (WMAA 2007). Landfill capping can contribute to reduction in methane emissions. A study by Grossman *et al.* (2002) examined the natural attenuation of landfill methane through aerobic oxidation in landfill soils. Different types of liners and caps are also being used to reduce water infiltration with a view to reducing methane generation. Numerous scientists, researchers and engineers have been working on technologies that are required to reduce environmental hazards caused by landfills. These technologies and their drawbacks are summarised in Table 1.3.

Table 1.3: Landfill remediation technologies and their drawbacks

Technology	Reference	Drawbacks	Reference
Leachate collection system	Ramke (1989), Rowe *et al.* (1997), Rittman *et al.* (1998)	Clogging due to microbial reaction and calcium carbonate deposition	Korfiatis and Demetracopoulos (1986), VanGulck *et al.* (2003)
Compacted clay liners (CCL)	Anderson (1982), Nirmala *et al.* (1995)	Pure organic liquid with high acidity increases the hydraulic conductivity	Nirmala *et al.* (1995), Alston *et al.* (1997)
Composite liners (High Density Poly-ethylene; HDPE)	Halse *et al.* (1990)	Sudden potential stress, cracking and unexpected rupture of plastic	Benson (2000)
Composite liners (Poly-Vinyl Chloride; PVC))	Benson (2000)	They contain plasticizers that make the polymer flexible and the plasticizers evaporate over time	Shackelford *et al.* (2000), Benson (2000)
Geosynthetic Clay Liners (GCL)	Shackelford *et al.* (2000), Melchoir (1997), Benson (2000)	Bentonite is not chemically compatible and it hydrates when it comes in contact with leachate	EPA (2001), Melchoir (1997), Benson (2000),
Compacted clay covers (CCC)	Drumm *et al.* (1997), Benson and Othman (1993), Melchoir (1997), Khire *et al.* (1997)	Desiccation and cracking of clay, short life span and does not allow optimum diffusion of oxygen required for methane oxidation	Drumm *et al.* (1997), Abichou (2004), Vasudevan *et al.* (2003)
Composite covers (HDPE)	Melchoir (1997), Floess *et al.* (1996), Khire *et al.* (1997)	Expensive. They act as protective barriers to clay caps	Melchoir (1997), Floess *et al.* (1996), Khire *et al.* (1997)
Composite covers (PVC)	Stark *et. al.* (2008)	Expensive.as protective barriers to clay caps	Benson (2000)

Despite evolution of new technologies, there remain several problems in controlling adverse environmental impacts from landfills. Moreover, most landfills in regional areas are unsupervised and charge no gate fee, which puts many of the councils under budget constraints; thereby discouraging them from implementing more advanced technologies which are costly.

1.2.1 Landfill Technologies in Australia

Lining of landfills has been practiced in Australia since early 1990s (Friends of Earth 2000) to prevent the flow of leachate from landfills into groundwater. Geosynthetic clay liners (GCLs) are commonly used in both liquid and solid waste containment in Australia (Phillips and Eberle 1999). Australia instituted the Commonwealth Environmental Protection Agency (CEPA) in 1992. Individual states had also established their own Environmental Protection Agencies/Authorities (EPA) between

1970's and 1990's. At present CEPA has classified landfills according to their types and the types of waste accepted at each site.

Currently, more than 600 operational landfills exist in Australia (Bateman 2005, Johnston 2009). A survey of 230 closed landfills revealed that 70% used clay capping systems, 20% composite and 10% unspecified alternative caps (Bateman 2005). Most sites in Australia had a clay liner and over half of the larger landfills had composite liners. Most landfills had a leachate collection system and many landfill operators treated the leachate before disposal (Bateman 2005). About 40% of the larger landfill sites (>100,000 tonnes yr^{-1}) collected and used methane gas to generate electricity and about 90% of the sites surveyed flared the gases (Bateman 2005). In 2002, there were around 25 methane gas recovery systems installed in landfills throughout Australia (Willumsen 2002), which should have increased by now due to introducing stringent legislation and Carbon Farming Initiative (CFI).

Leachate recirculation is a technique used to encourage saturation of waste, with a view to stimulating the waste degradation processes to achieve quicker stabilisation of the landfill (Scott *et al*. 2005). Rather than relying on intermittent rainfall, leachate recirculation has been employed to enhance and maintain waste saturation. Leachate recirculation also benefits in liquid volume reduction via evaporation (spray irrigation) (Percy and Truong, 2005). Concerns have been expressed about elevated pollutant levels around landfills but various studies have found no evidence of elevated pollutant levels in landfills that use leachate recirculation systems (Doedens and Cord 1989, Ehrig 1989, Reinhart and Basel 1996).

Several experiments have shown that methane generation will be enhanced if the acid generation phase is minimised (Komilis *et al.* 1999); leachate recirculation is known to minimise acid generation. Leachate recirculation has the potential to reduce quantity of leachate being produced. Research into potential application of leachate recirculation has been conducted in Australia (Waste Services 1998, Yuen 1999). Bowman *et al.* (2002) and Percy and Truong (2005) demonstrated the use of vetiver grass technology and parklands, respectively, for managing landfill leachate. Despite several new technologies such as gas recovery systems, leachate recirculation systems, liners and caps, very little is known about their long-term

benefits and sustainability (Scott *et al.* 2005). Various technologies currently used in landfills to reduce adverse environmental impacts of landfills are shown in Table 1.4.

Table 1.4: Landfill remediation technologies practiced in Australia

Technology	Reference
1. Leachate collection	Bateman (2005), EPA (2005)
2. Leachate recirculation	Scott *et al.* (2005), Yuen (1999)
3. Leachate disposal	EPA (2005), Percy and Truong (2005)
4. Leachate treatment	Bateman (2005), Scott *et al.* (2005)
5. Methane gas recovery	EPA (2005), Willumsen *et al.* (2002)
6. Gas flaring	Bateman (2005)
7. Caps	
Clay	EPA (2005), Gourc *et al.* (2009)
GCL	Bateman (2005)
Alternative caps	Ashwath and Venkatraman (2007), Travor *et al.* (2009), Yuen *et al.* (2011)
Caps made of industrial by-products	Ronkainen *et al.* (2009)
8. Liners	
Clay liners	EPA (2005)
GCL	Bateman (2005)
9. Biofiltration	Dever *et al.* (2007)
10. Bioreactor	Yuen (1999)

1.3 Landfill Capping

Landfill capping is practiced to isolate landfills from the outside environment, mainly rainwater (Vasudevan *et al.* 2003). Rain and temperature are the two key elements to be considered in designing a capping system (Berger and Melchoir 2009). Similarly, suitability and type of capping material depends on climatic conditions and topography of the site (Berger and Melchoir 2009). Capping involves placement of clay (usually with materials that are similar to the liner) over filled landfill to minimise percolation of rain or surface water into the waste (Scott *et al.* 2005). In recent years, conventional capping systems made of compacted clay (Othman *et al.* 1994), GCLs (Benson, 2000), PVC (Levin and Hammond 1990) and HDPE (Simon and Muller 2004) have been used extensively in both developed and developing countries. Landfill capping systems minimise the entry of water into waste, which is primarily responsible for generation of leachate (Bendz *et al.* 1997)

and methane gas emission (Whalen *et al.* 1990). Unlike the caps which are usually placed on completion of landfilling, placement of a layer of top soil or local soil is applied daily (daily cover) to reduce odour, stop the waste from being blown away and to prevent vermin from accessing the buried waste (Rohrs and Fourie 2002).

Once MSW landfills reach the end of their life or the permitted maximum height, current regulations require that the waste be covered with low hydraulic conductivity soil layer to minimise entry of water into the waste (Leone *et al.* 1977). Environmental regulatory authorities in Australia have prescribed the use of clay capping to minimise percolation of water into the waste (EPA 2005). A capping system recommended for landfills in Queensland is shown in Figure 1.2.

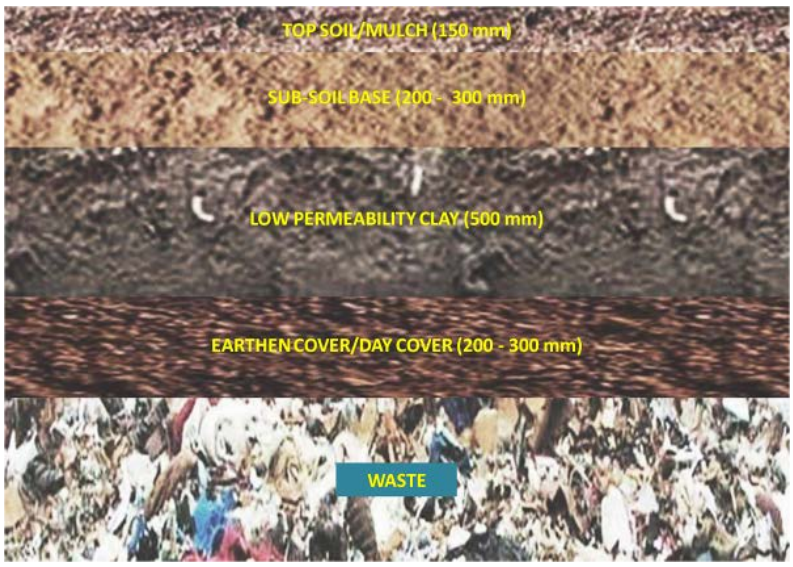


Figure 1.2: Typical clay capping system in Queensland
Source: (EPA 2005)

Landfill caps improve aesthetics; prevent waste from blowing across landscape; reduce odour; control insects, rodents and flies; reduce fire hazards and most importantly minimise infiltration of water into the waste (Leone *et al.* 1977). Although many uses have been proposed for completed landfills, experience has shown that they are generally unsuitable for excavation, crop production or for suburban development. A large number of landfills have been used as play grounds or parks (Pereboom *et al.* 2009). Recent research studies from Austria (Tintner *et al.* 2009) and Italy (Delbarba and Mazzata 2009) have shown the possibility of crop production on closed landfills.

The performance of a landfill cap depends on the interactive effects of various processes that regulate a site's water balance (Paige *et al.* 1996). Negligence in the understanding of these processes by traditional remedial engineering techniques (Nyhan *et al.* 1990) has led to failure of many remediated landfills (Jacobs *et al.* 1980, Hakonson *et al.* 1982). Various techniques such as use of clay capping, GCLs, PVC or HDPE capping (Bowers 2002, Bateman 2005) and phytocapping (Ashwath and Pangahas 2004) have been suggested as effective techniques for landfill capping. Clay caps (Fig 1.2) have been used in more than 70% of the landfill sites in Australia (Bateman 2005). Amongst the larger sites (>100,000 tonnes yr^{-1}), 25% of the landfills have used GCLs in the capping system, whereas 10% of the larger sites used alternative capping systems (not specified) (Bateman 2005).

Clay caps rely on low saturated hydraulic conductivity of the caps to minimise entry of water into buried waste. However, this is not often achieved due to the formation of micro and macro cracks as the clay caps age and due to drying and wetting cycles associated with seasonal changes in rainfall and relative humidity (Albright and Benson 2001, Albright et al. 2003, Dwyer 2001, Melchoir 1997, Simon and Muller 2004). Clay caps require heavy maintenance and regular environmental monitoring to avoid any occurrences that enhance water infiltration. Clay caps have a limited life span (Vasudevan *et al.* 2003). Recent studies have shown that clay caps, although popular, often fail over a longer term due to cracking (Albright and Benson 2001, Albright *et al.* 2004, Benson and Othman 1993, Gourc *et al.* 2009, Khire *et al.* 1997, Melchior 1997, Othman *et al.* 1994) and they do not allow optimal interaction between methane and oxygen, which is necessary for methane oxidation (Abichou 2004).

Geosynthetic Clay Liners comprise a bentonite layer between two Geotextiles layers. These are preferred due to their low cost and space saving (Simon and Muller 2004). In the past, studies on GCLs have reported failure of the capping system due to the freezing/thawing effect and damage by desiccation (Lin and Benson 2000, Melchoir 1997) in temperate regions. GCLs are also susceptible to leakages through holes left behind during construction (Board and Laine 1995, Croizer and Walker 1995). Vegetation such as trees and shrubs that establish voluntarily, or those grown intentionally, will continually compete with grasses for available space sending their

roots down to deeper soil layers, thereby both creating and leaving cracks in the capping system following root death (Stonell 1986). HDPE capping systems are very expensive (Simon and Muller 2004) compared to conventional clay caps.

Cossu *et al.* (1995) recommends the use of bio covers with high permeability to fasten waste degradation process and environmental impacts of landfills. They also reported that bio covers have a good field capacity and have the potential to reduce odour from the buried waste. According to Benson (2000), various researchers recommended the use of a PVC geomembrane capping system for sanitary landfills (New York State Department of Environmental Conservation; NYSDEC) due to its flexibility and durability. However, PVC caps often lose their plasticizer, and the corresponding elongation properties reduce by 63% (Stark *et al.* 2008).

A new technology commonly known 'Phytocapping' was introduced in 1991 by the Idaho National Engineering and Environmental Laboratory (INEEL) for the US Department of Energy (USDoE). This concept was also tested at Lakes Creek Landfill in Rockhampton, Australia. Phytocapping has been trialled throughout US (Ankeny *et al.* 1997) as the first demonstration project conducted by INEEL in 1991. The trial demonstrated that phytocaps can be as efficient as the clay capping technique in many parts of the US (US EPA 2003).

1.4 Phytocapping – *The Concept*

The phytocapping concept has been well described by Hauser *et al.* (2001). In brief, phytocaps have two major components; the plants that act as 'bio-pumps' and 'rain interceptors' and the soil that acts as 'storage' and a 'screen'. The vegetation and soil together minimise percolation of water into the waste (Fig. 1.3). In this system, the soil acts mainly as the 'storage' to absorb and store water. Soil may also act as a 'screen' to restrict methane release into the atmosphere, while also serving as the substrate for plant growth. Plants act as 'bio-pumps' thereby use up stored water and minimise movement of water into the waste (Fig. 1.3) (Venkatraman *et al.* 2006). Plants also act as 'rain interceptors' by blocking a certain proportion of the rain from reaching the ground surface (via direct evaporation from the tree canopy)

(Venkatraman and Ashwath 2006). Trees may also minimise methane emission. (Fig. 1.3) by promoting methane oxidising microorganisms in their root zone.

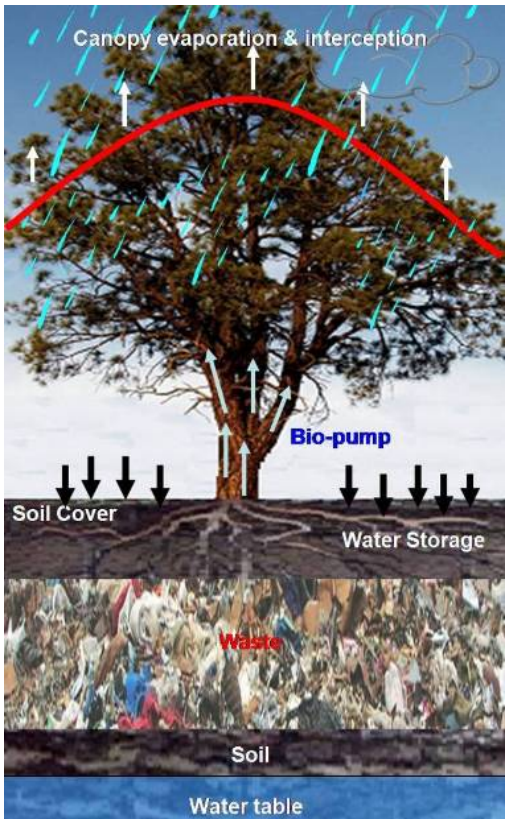


Figure 1.3: Phytocapping concept

Sustainability of the phytocap depends upon thickness and properties of the soil used and types of the trees chosen for the phytocapping system (Albright *et al.* 2003, Dwyer 2001, Hauser *et al*. 2001). EPA (2005) recommended a standard saturated conductivity of 10^{-8} m s^{-1} for clay caps to reduce the 'bath tub effect' that increases potential of the waste to generate leachate. Phytocapping systems tend to prefer medium textured soils that not only hold reasonable amounts of moisture but also allow good root activity. This system also prefers vegetation that can adapt well to local conditions such that these will remove water rapidly during the wet season and minimise their water uptake during dry periods, with the view of ensuring sustainability of the established phytocaps on landfills. The vegetation should also be tolerant of landfill gases and able to support methanogenic bacteria to enhance methane oxidation. Local soil is preferred to reduce cost of the established phytocaps

(Hakonson 1997, Dwyer 2003). Phytocaps has proven to perform better than clay caps at certain sites in the US (Albright *et al.* 2006, Benson *et al.* 2002).

Phytocaps are also referred to as phytocovers, evapotranspiration caps (ETC), water balance caps (WBC), alternative earthen final caps (AEC), vegetative landfill caps VLC), soil-plant caps (SPC) and store-and-release caps (SRC) (EPA 2003, Madalinski *et al.* 2003). The design of phytocaps is based on quantifying water balance components of a given site, which include water storage capacity of the soil, precipitation, surface runoff, evapotranspiration, infiltration (Dwyer 2001, Hauser *et al.* 2001, Albright *et al.* 2003, Madalinski *et al.* 2003) and canopy interception. Greater storage capacity of soil and evapotranspiration properties of plants will reduce potential for percolation through the phytocap (Hakonson 1997, Hauser *et al.* 2001, Dwyer 2003). Phytocaps are half as expensive as clay caps and these offer other environmental benefits whilst performing equally well as conventional caps (US EPA 2003, Scanlon *et al.* 2005).

Two types of phytocaps (monolithic barriers and capillary barriers) are used in the US (US EPA 2003). Monolithic barriers, also referred to as monofills, contain a single vegetated soil layer to retain water until it is either transpired or evaporated from the soil surface (Madalinski *et al.* 2003). Capillary barriers contain a fine-grained soil layer overlying a coarser-grained layer (Wing and Gee 1994). Studies evaluating performance of phytocaps have been conducted at USDoE sites (Nyhan *et al.*1990, Chichchester and Hauser 1991, Waugh *et al.* 1994, Anderson *et al*. 1997, Gee and Ward 1997, Dwyer 2001). More discussion of these capping systems can be found in Koerner and Daniel (1997), Krieth (1994) and Tchobanalogous *et al*. (1993) and Albright *et al.* (2010). Following success of the Alternative Capping Assessment Program (ACAP) study in the US, a similar study with a similar concept and design, is has been undertaken in Australia. This is known as the Australian Alternative Capping Assessment Program (A-ACAP) (Yeates *et al.* 2008, Salt *et al.* 2011, WMAA 2013**).**

1.4.1 Soil Cover

In phytocapping systems, soil covers are used both to retain moisture and to support plant and microbial growth. Generally, locally available soils are used in phytocapping projects. It is also possible that the soil may be synthesised using readily available materials such as clean fill, green waste, biosolids and crushed concrete. The soils used in phytocapping, however, must retain adequate quantities of moisture and be free from chemicals that may affect survival and growth of plants and microbes. The properties such as its texture (Auge *et al.* 2001), water holding capacity (Williams *et al.* 1983, Schmitz and Sourell 2000), porosity (Johnson *et al.* 2003, Bird *et al.* 2005), bulk density (Kalman *et al.* 1996), hydraulic conductivity (Hopmans and Dane 1986), chemical composition (Lee and Foster 1991) and organic matter content determine the suitability of soil to phytocapping.

Depth of the soil used will have a marked impact on phytocapping, not only from the point of view of its effectiveness, but also from cost of constructing the landfills. It is therefore important to choose the right depth considering the climate (particularly rainfall pattern), nature of the plants used and regulatory requirements.

Hauser *et al.* (2001) examined the effect on soil depth on plant growth, and concluded that the phytocaps were as effective as clay caps in many cases. A study conducted by Moffat and Houston (1991) at Pitsea Landfill site, Essex, observed that the trees grown in 1500 mm of soil cover grew better than those established in a 200 mm soil cover. Similar conclusions were drawn by Warren *et al.* (1996) from a four-year trial at a site that had a mean annual rainfall of 2000 mm. They reported a percolation of 300 mm in soil caps having 1500 mm with trees and shrubs, 240 mm in 1500 mm caps that had grass, and 400 mm in 900 mm soil caps that had grass only. The highest percolation in 900 mm cap was found to be due to insufficient depth (Warren *et al.* 1996) and high soil density (Hauser *et al.* 2001, Jones 1983). Soil density in the 900 mm cap with grass was reported to be 1.86 Mg m^{-3} (Warren *et al.* 1996) which is higher than that permissible for normal root growth (1.1 to 1.5 Mg m^{-3}) (Grossman *et al.* 1992, Hauser *et al.* 2001). Soil densities above 1.7 Mg m^{-3} restrict root growth in most soils (Timlin *et al.* 1998). Thus it is extremely important to optimise the two main factors, viz soil depth and soil density, in designing

phytocapping systems, with the view to improving water holding capacity and promoting root growth.

Inclusion of topsoils in phytocaps promotes plant growth as they contain nutrients, essential microbes and soil seed banks. Likewise, addition of green waste or organic matter will minimise soil evaporation, salinity build up around the root zone, as well as the weed competition (Ashwath 2011). The addition of green waste and top soil can also reduce soil erosion (Madalinski *et al.* 2003). McGuire *et al.* (2001) recommended a minimum depth of top soil of 150 mm to support vegetation.

Overall, based on the previous studies and research, it is certain that optimising soil depth is critical to maintain the site water balance of the phytocapping system. Results from the study conducted by Warren *et al.* (2006) also suggest that selecting the right type of vegetation is important for good performance and long term sustainability of the phytocapping system.

1.4.2 Species

Plants play a key role in maintaining the water balance of the phytocapping system (Weand *et al.* 1999). However, selecting the right species is very important for a good and sustainable phytocapping system.

In addition to good interception and transpiration properties, plants that are leachate tolerant, drought tolerant and salt tolerant should be selected. For these reasons native plants are preferred (Dwyer *et al*. 1999), as they adapt to local conditions and are less likely to disturb the ecosystem. Native plants also have the ability to regenerate in case of natural disasters such as storms and bushfires.

Some of the species used in previous studies include wheat grass, clover, rabbit brush, sagebrush, willows, hybrid poplar (Von Der Hude *et al.* 1999, US EPA 2000, Benson *et al.* 2002, Licht *et al.* 2004, Scanlon *et al.* 2005) bluegrass, alfalfa prickly rose (US EPA 2003), switch grass, bermuda grass, Italian rye grass, black mustard, prickly lettuce, sandburg bluegrass, needle grass, California brome, bluebunch,

western wheat grass, blue gramma, green needle grass, creeping wild rye and crested wheat grass (Albright *et al.* 2004).

In some of the sites, grasses are preferred due to their ability to control erosion and produce extensive fibrous roots (Hauser *et al.* 2001). Salt cedar (*Tamarix* sp.) has also been used in landfills in the US (Nicholas, 1994) due to its high water uptake capacity of 9.5 mm d^{-1} (Gay and Fritschen 1979) and high drought tolerance (Cleverly *et al.* 2007). Apart from grasses, poplars and willows have also been extensively used in phytocapping trials in the US due to their local occurrences, good performance record in phytoremediation projects in Europe (Elowson 1999) and North America (US EPA 1998, 2000, 2001) and in phytocapping (EPA 2003, Dwyer S: per. Comm. 2006), tolerance to leachate (Aitchison 2005), efficient in water uptake (up to 4.8 mm d^{-1}) (Hinckley *et al.*1994, Aaronson 1996, Bassman 2000, Aaronson and Perttu 2001, Negri *et al.* 2003), good root development (1500 mm deep) (Licht 1990, Licht *et al.* 2004) and high LAI (Heilman *et al.*1994, Hinckley *et al.*1994, Chappell 1997). Past studies have demonstrated the potential of hybrid poplars to reduce the amount of water reaching wastes (Dobson and Moffat 1993, Madison *et al.*1991) and metabolise pollutants and release these into the atmosphere (Hinchman *et al*. 1996 Kopp *et al.* 2001, Reimenschnieder *et al*. 2001).

About 450 species of *Salix* (willows) are found worldwide; distributed mostly in the northern hemisphere (Argus 1997) with 106 species found in North America (Zomleter 1994). *Salix* species also grows in subtropical and tropical regions in all continents except Antarctica and Australia (Kuzovkina and Quigley 2004). *Salix* species are best known to grow in soils with low nutrient levels (Heijden and Kuyper 2003), have high transpiration rates (Ebbs *et al.* 2003) and are able to accumulate high levels of heavy metals like cadmium (Klang-Westin and Eriksson 2003), and are tolerant drought and salinity (Kowalchik 2001) and produce large quantity of biomass (Perttu 1993).

In Australia, over 100 native species (including trees, shrubs and grasses) have been tested in various systems (Aswathappa *et al.* 1986, Ashwath *et al.* 1993, Ragupapthy *et al.* 1998) since 2003. However in this study, potential of 21 species to reduce percolation of water into the waste has been tested.

Studies in the US and Australia have tested and endorsed the potential of a wide range of plant species to grow in landfill conditions. Numerous parameters like LAI, Root Length Density (RLD), root depth, transpiration and tree growth rate have been measured (Lee and Schnabel 2000, Benson *et al.* 2002, Madalinski *et al.* 2003, Licht *et al.* 2004, Scanlon *et al.* 2005, Venkatraman and Ashwath 2007, 2010, Ashwath 2011). Various simulations of site water balance using hydrological models have also confirmed importance of plants in a phytocapping system (Lee and Schnabel 2000, Benson *et al.* 2004).

Overall, the choice of plant species usually depends on site conditions such as physical properties of the soil material placed over the waste and thickness of the soil cover (Department of the Environment 1984), degree of exposure to environmental factors and the local climatic conditions (Hibberd 1989). Other factors such as rooting depth (Kiniry *et al.* 1995) and the ability to pump large amounts of water (Robinson *et al.* 2003) also play a vital role in defining the sustainability of plant species in a phytocapping system.

1.4.3 Findings from the ALCD and ACAP

The Sandia National Laboratories at Kirtland Air Force Base near Albuquerque, New Mexico, tested four alternative capping designs including monolithic ETC, capillary barrier ETC, anisotropic barrier ETC and GCL as part of the ALCD project (Dwyer *et al.* 2000, Dwyer 2001, Scanlon *et al.* 2005). Results from a three year study (May 1997 to June 2000) with a total rainfall of 7745 mm showed a seepage flux of 0.19 mm day^{-1} for ETC, 0.16 mm day^{-1} for anisotropic barrier ETC, 0.87 mm day^{-1} for capillary barrier ETC and 1.81 mm day^{-1} for GCL (Dwyer *et al.* 2000).

The ACAP was established in 1997 by the US Environment Protection Agency (US EPA) with the view to evaluating performance of various capping systems under different agro-climatic conditions in the US (Albright *et al.* 2003). Eleven pilot scale test sections were established using conventional and alternative capping systems (Scanlon *et al.* 2005). In 2003, more than 64 landfill sites in the US were found to have incorporated the phytocapping system (Madalinski *et al.* 2003). This included 56 projects with monolithic capping and 20 projects with capillary barrier capping.

The number of landfill sites with phytocaps increased to 93 in 2007. This included 47 demonstration trials and 46 full scale applications (www.cluin.org/products/altcovers). Today there are 223 alternative cover projects (demonstration, full scale or pilot) across US, ET/phytocapping systems have been readily implemented and used as a final closure for MSW landfills in many parts the US (Aitchison 2005) and negotiations are occurring between landfill operators and the EPA in Australia (Yeates *et al.* 2008). All studies undertaken in the US have considered only the site water balance of the system and not the individual components of trees and soil cover (e.g. species, capping thickness, tree characteristics, methane reduction). A few studies have measured methane emissions.

Results of the ACAP trials suggested that the percolation rates for phytocaps in arid and semi-arid regions with mean rainfall of less than 500 mm can be less than 1 mm yr^{-1} at some sites whilst the percolation rates in humid areas where the mean rainfall ranged between 450 mm yr^{-1} to 1250 mm yr^{-1} were 12 to 128 mm yr^{-1} (Benson *et al.* 2002). In the same study, at one of the sites in Albany (1280 mm rain), Benson *et al* (2002) found that percolation rates decreased drastically with tree growth from 360 mm yr^{-1} to 14 mm yr^{-1}. A similar study on composite covers revealed a percolation rate of 1 mm yr^{-1} in arid and semi-arid regions and 5 mm yr^{-1} in humid regions (Benson 2002).

Benson *et al.* (2001) demonstrated a field evaluation of phytocaps under different agro-climatic conditions using five different methods for assessing percolation. These include water balance, trend analysis, Darcy's Law calculations, tracer methods and lysimeters. Each of the methods varied significantly in their results. The trend analysis method was found least accurate followed by the water balance method. Percolation rates computed according to Darcy's Laws were reasonably accurate but Benson *et al.* (2001) found the use of lysimeters to be the most accurate method for estimating percolation rates (200 m^2). The lysimeters gave an accurate result with a precision of 0.00004 mm yr^{-1} to 0.5 mm yr^{-1}. Benson *et al.* (2001) also suggested that lysimeters of the area >100 m^2 are required as sensitivity of the drainage measurement increases with the size of the lysimeters.

Another study conducted by the US EPA at the Polson landfill in Montana and New Mexico in 1999 (US EPA 2003), used 150 mm of top soil, 450 mm of compacted silt and 600 mm of sandy gravel. The cover was seeded with a mixture of grasses, forbs and shrubs including wheat grass, bluegrass, alfalfa and prickly rose shrubs. Percolation was measured using lysimeters and soil temperature with a heat dissipation unit. Results showed that the phytocapping system had a cumulative percolation of 0.5 mm from 1999 to 2002, in a region that received a total rainfall of 837 mm during the same period. On the other hand, HYDRUS 2D software predicted approximately 0.6 mm percolation in the first year and 0.1 mm over the next 9 years for the same site (US EPA 2003).

Madalinski *et al.* (2003) reported their findings of a demonstration phytocapping project in the US using wheat grasses, clover, rabbit brush, sagebrush, willows and hybrid poplars grown on three different capping systems. Monolithic cover contained soil cover ranging from 600 mm to 3000 mm, whilst the depth of capillary barrier (with an additional layer of gravel) covered with fine grained soil ranged between 420 mm to 1520 mm, and the coarser grained covers ranged from 152.4 mm to 600 mm. HYDRUS 2D was used to predict site water balance. Similar studies were conducted by the US EPA in landfills at Los Angeles where a 910 mm silty sand/clayey sand layer was established. This trial used UNSAT-H model and predicted a cumulative percolation of 500 mm for phytocaps and 950 mm for conventional cap over 10 years. The study also found that performance of phytocaps improved with maturity of plants grown in this system.

The study from an ALCD project found that percolation varied between 0.05 mm and 0.54 mm depending on the type of phytocaps used at a site that received 267 mm of rainfall in 1997. In 2002, the percolation was reduced to 0 mm. Similarly, the ACAP study that commenced in 2000 at Altamont, California (semi-arid region) reported percolation of 0 mm in the first year in the 225 mm rainfall zone and 1.5 mm percolation in the 300 mm rainfall zone (US EPA 2003). In 1999, at Polson, 0.05 mm of percolation was recorded 300 mm in the first year at an annual rainfall of 300 mm, and 0.45 mm at 250 mm rainfall in the second year (US EPA 2003). A study conducted by Albright *et al.* (2004) estimated a seepage flux of 60 mm yr^{-1} in the site receiving an annual rainfall of 760 mm. The ACAP and ALCD projects

tested the performance of phytocaps against composite and compacted clay caps (Dwyer *et al.* 2000, Dwyer 2001 and Scanlon *et al.* 2005). Results showed that phytocaps worked equally well or better than conventional caps in most occasions (Albright *et al.* 2004, Benson *et al.* 2004).

1.5 Modelling Site Water Balance

The ALCD and ACAP project in the US used numerical models such as Hydrologic Evaluation of Landfill Performance (HELP), Vadose/W, HYDRUS 1D/2D and UNSTAT- H. Several other modelling studies were undertaken to evaluate the hydrological performance of landfill covers (Chai and Miura 2002, Ho *et al.* 2004) to predict seepage production (Dho *et al.* 2002, Ham 2002). Studies in the past have compared the ability of hydrologic models to predict water balance of landfill final caps. Models used in previous studies include UNSAT-H, HYDRUS, Simulation of Heat and Water (SHAW), Vadose/W, Soil Water Balance and Infiltration Model (SWIM), VS2DTI, and HELP (Fayer *et al.* 1992, Fayer and Gee 1997, Khire *et al.* 1999, Scanlon *et al.* 2002, Benson *et al.* 2004). Of these models, HYDRUS, UNSAT-H, Leachate Estimation and Chemistry Model (LEACHM) and Vadose/W are used most frequently in practical situations (Benson *et al.* 2004) due to their reliability and ability to incorporate more plant, soil and climatic data and use Richards' equation.

Fayer and Gee (1997) compared water balance data from eight non-vegetated lysimeters located in the semi-arid south-eastern region of Washington State, using UNSAT-H. Soil-water storage was under-predicted during winter months and over-predicted during summer. Khire *et al.* (1997) compared water-balance predictions made with HELP and UNSAT-H models with lysimeters data for two resistive barrier covers located in Georgia (humid climate) and Washington state (semi-arid climate). The cover profile at both locations consisted of a vegetated surface layer overlying a compacted fine-grained layer. Meteorological, vegetation, and soil data were used, but predictions from UNSAT-H were in better agreement with the measured water balance than those from HELP. Percolation was found to be grossly over-predicted by HELP and slightly under-predicted by UNSAT-H. Errors in

predicting snowmelt and frozen ground affected most water-balance parameters and were found to significantly affect runoff predictions during winter months.

Khire *et al.* (1999) described a comparison between predictions made with UNSAT-H and the field data from a capillary barrier test section consisting of a 150 mm layer of silt overlying a 750 mm layer of sand. The comparison showed that UNSAT-H under-predicted runoff (within 100 mm) and over-predicted percolation (within 50 mm) and the soil-water storage was predicted accurately within 30 mm of measured soil-water storage. Scanlon *et al.* (2002) compared predictions made with HELP, HYDRUS, SHAW, Vadose/W, SWIM, UNSAT-H, and VS2DTI with data collected from landfill caps in the semi-arid states of Texas, New Mexico, and Idaho, over three years. The cover profile at the Texas site consisted of (from top to bottom) 300 mm of sandy clay blended with 15% gravel, 1700 mm of compacted sandy clay, and 1000 mm of sandy gravel. A 1070 mm thick monolithic cover of silty sand was evaluated at the New Mexico site and a 3000 mm thick monolithic cover of sandy silt was evaluated at the Idaho site. Models employing Richards' Equation predicted the water balance more accurately than the HELP model. Scanlon *et al.* (2005) suggested that the relationship between the abundance of vegetation, evapotranspiration, and water availability was an important factor affecting accuracy of water-balance predictions, and that most models used today do not account for this interaction.

Benson *et al.* (2004) compared measured water-balance data from a monolithic cap at a semi-arid site to predictions made with UNSAT-H and Vadose/W. On-site data was incorporated in the model. The study found that more accurate predictions were obtained with Vadose/W than with UNSAT-H. Surface runoff was over-predicted by UNSAT-H, which affected all sub-surface hydraulic processes. In contrast, Vadose/W accurately predicted surface runoff, evapotranspiration, and temporal variations in soil-water storage. Neither model predicted percolation accurately nor succeeded in capturing a key change in the transpiration pattern during the last winter-summer period of the monitoring program. Differences in the method used to simulate precipitation intensity were found to be partly responsible for differences in the accuracy of predicted surface runoff. Simulations conducted to evaluate the importance of the lower boundary condition showed that essentially the same

predictions were obtained when the lower boundary was assigned as a unit gradient or seepage face condition.

Chammas *et al.* (1999) compared HELP and Vadose/W model and concluded that soil cover models gave more accurate predictions and that HELP could only conservatively predict percolation in dry environments. Despite being inaccurate, HELP has been extensively used by various researchers in phytocapping. Interesting conclusions against HELP were made by Fleenor and King (1995) and Khire *et al.* (1997). Khire *et al.* (1997) compared HELP and UNSAT-H and concluded that HELP over-estimated and UNSAT-H under-estimated percolation, but that HELP performed better than UNSAT-H. HELP and the Flow Investigation of Landfill Leachate (FILL) model have been extensively used in Australia to predict site water balance of landfill caps and liner systems (Khanbilvardi *et al.* 1995). However, recent Australian Field Lysimeters and Leachate Yield Tests in Queensland and South Australia (SA) found that the HELP model overestimates infiltration rate and leachate volumes generated in Australian landfills (Bowers 2002).

In 2004, Hauser and Gimon compared Environmental Policy Integrated Climate (EPIC), HELP, HYDRUS and UNSAT-H, and found that EPIC was better than HELP for phytocaps but UNSAT-H and HYDRUS were more accurate and could be used in phytocapping research. Further details and comparison between the various models are given in Chapter 8.

Based on the literature review, it is evident that the phytocapping trials conducted in the US used very few tree species and little or no information has been provided on role of trees in maintaining a desirable site water balance. The ACAP used mixtures of grasses, shrubs, forbs and trees, predominantly willows and hybrid poplars (EPA 2003). None of the previous studies have emphasised the role of plant species in a phytocapping system. Phytocaps can also reduce methane emissions from landfills via methane oxidation. Not much research has been conducted on the role of trees in reducing methane emissions. Phytocapping studies in the past have measured various soil parameters, tree growth, LAI, soil moisture, biomass production and transpiration but have not investigated vital characteristics of plants to intercept

rainfall (Crockford and Richardson 1990). Canopy rainfall interception plays a significant role, not only in the hydrological balance of the phytocapping system, but also by helping to optimise soil thickness, which is the most expensive component of the phytocapping system.

Phytocaps can be a preferred future landfill design for many of the landfill sites to achieve stability while maximising leachate volume reduction, minimising methane emissions and encouraging native species (Licht *et al.* 2004), especially in semi-arid regions. Results from the ACAP study suggested that phytocaps performed better in semi-arid and arid regions than in tropical and temperate regions because of the dominance of summer precipitation (62% to 80%) that corresponds with the periods of high ET (EPA 2003, Dwyer *et al.* 2005).

1.6 Knowledge Gap

Phytocaps offer several advantages over conventional capping systems. They are more sustainable and provide ecological and aesthetic benefits beyond their transpiration functionality. Phytocaps also provide shelter for many native flora and fauna, thus enhancing the biodiversity value of that region. A series of studies conducted in the US (Albright *et al.* 2004) has laid a foundation to promote phytocapping as an alternative technique for landfill post closure management. There continue to be issues which need further attention and in-depth study such as plant growth and root development, transpiration, canopy rainfall interception and methane oxidation in phytocaps.

Studies in the past have demonstrated effectiveness of phytocaps in comparison with other clay capping systems using lysimeters, but different components (i.e. trees and soil) of the capping system with regards to their performance, integrity and sustainability are barely reported. Secondly, the ability of phytocaps to reduce methane emission has not been explored. For these reasons, performance of trees in reducing infiltration of water into waste, the role of soil depth, and effects of trees on methane emission, survival and long term sustainability of established trees needs to be defined so that the phytocapping technique can be further developed and used in landfill remediation.

Canopy rainfall interception (rain intercepting effect) is another component that plays a significant role in maintaining the hydrological balance of the phytocapping system. This particular feature of trees not only improves the efficiency of a phytocap but also reduces cost of phytocapping by optimising soil depth, as it reduces amount of rain reaching the soil. Finally, previous trials in the ACAP and the ALCD projects have used very few or no tree species. Thus in-depth study and further understanding of the species to be used in a phytocapping system is warranted.

1.7 Summary

Landfills cause environmental hazards due to leachate generation and methane gas emissions (EPA 2002, CSIRO 2001). Efforts have been made to mitigate these impacts through developing landfill technologies such as leachate collection systems (Rittman *et al.* 1998), landfill gas collection systems (EPA 2006) or construction of landfill caps and liners (Bowers 2002). These technologies are expensive and not practically viable for many of the small and medium sized landfills (landfills accepting <100,000 tonnes yr^{-1}). A minimum of 100,000 tonnes yr^{-1} of waste is required to successfully run a 1 MW engine for electricity conversion (Molloy 2008). Many of the small and medium sized landfills constructed prior to 1990 are not lined. Furthermore, the mandatory requirement to clay cap, revegetate and continuously monitoring the landfill for 30 years after its closure has put a lot of pressure on many of the councils in Australia.

Landfill caps are often used to isolate landfills from the outside environment; chiefly water (Vasudevan *et al.* 2003). In recent years, conventional capping systems made of compacted clay (Othman *et al.* 1994), GCLs (Benson 2000), PVC (Levin and Hammond 1990) and HDPE (Simon and Muller 2004) have been used extensively in developed and developing countries. Environmental Protection Authority in the US and Australia have prescribed use of clay capping to minimise percolation of water into buried waste. However clay caps fail over time (Albright *et al.* 2004) and do not allow for optimal interaction of methane with oxygen, which is essential for methane oxidation (Abichou *et al.* 2004). GCLs have also reported as constant failure due to the thawing effect and damage by desiccation (Melchoir 1997, Lin and

Benson 2000). GCLs are also susceptible to leakages through holes left behind during construction (Board and Laine 1995, Croizer and Walker 1995). HDPE capping systems in turn are very expensive and susceptible to degradation from sunlight as well as chemical and biological degradation (Simon and Muller 2004). Conventional caps require heavy maintenance and constant monitoring to avoid occurrences that enhance water infiltration. PVC caps often lose their plasticizer and the corresponding elongation properties were reduced by 63% (Levin and Hammond 1990).

Hence, a new technology called phytocapping' was trialled at Lakes Creek Landfill, Rockhampton, Australia from 2003 to 2007. Field studies conducted in the US have proved that phytocaps perform equally well or better than clay caps. Phytocaps have been successfully trialled and adopted in the US, but several aspects such as soil depth, plant species to be grown, their transpiration rates, rainfall interception potential, effect of methane on tree growth, effect of plants on methane emission and sustainability of the plant species in harsh landfill conditions are still to be understood to successfully implement phytocaps in Australia.

This is the first study of its kind in Australia and the first study to consider various plant related aspects such as identification of trees to be grown at each site, role of plants in reducing methane emission, sustainability of the established vegetation, their contribution to site water balance via canopy interception and transpiration, and the plant/soil combinations that offer the best site water balance. An additional feature, i.e. role of soil thickness in determining site water balance and sustainability of the established vegetation as well as mineral composition of trees established on phytocaps has also been studied.

1.7.1 Conclusions from Published Research

Some broad conclusions can be drawn from previous studies on landfills and landfill caps:

- Landfills are the easiest and most economic means of disposing waste, and this practice will continue to be used in the future.

- Despite progress made in landfill technology and management, the associated environmental impacts are still a major concern globally and in Australia.
- Landfill capping using compacted clay has been made mandatory in Australia, but clay caps are expensive and often fail to perform over a period of time.
- Properly designed phytocapping - an alternative capping technology has shown to perform equally or better than clay caps in the US and is yet to be proven in Australia.

There is a need for further research on various aspects, such as:

- The type of species to be grown on phytocaps
- Thickness of the soil required for tree growth and water retention
- The role played by canopy rainfall interception
- The effect of phytocaps on methane emission

1.8 Aim and Objectives

Aim:

The aim of this research is to demonstrate whether phytocapping technique would be effective in minimising entry of water into the buried waste while also reducing methane emission from landfills.

Objectives:

The specific objectives were to:

- Select suitable species for phytocapping based on their actual performance in the phytocapping system as assessed by their survival, growth, transpiration and canopy rainfall interception.
- Optimise depth of soil to be placed over the waste and to model the site water balance for the phytocapped landfill site using HYDRUS 1D.
- Quantify water uptake by various species.
- Determine canopy rainfall interception potential of various species and test effect of the same on site water balance.

- Investigate the effect of phytocapping on methane emission and test the role of soil thickness on methane oxidation.

1.9 Rationale and Structure of the Thesis

Phytocapping has the potential to provide equal or better performance compared to clay capping system, especially in arid and semi-arid environments and in regions where clay is not readily available. Studies in the past have not used a diverse range of tree species and the hydrological balance was determined using lysimeters. Further research into phytocapping is required with regards to type of species to be used and the role of plants in reducing entry of water into the waste and emission of methane from the buried waste.

This thesis addresses two broad aspects; firstly, the role of trees and soil in maintaining the water balance of a phytocap, and secondly, ability of the phytocapping system to reduce methane emission from landfills. Various aspects that address these issues are presented in Chapters 1 to 10.

This Chapter provides a literature review on various aspects that are largely pertinent to the work of this thesis. The key objectives and the importance of this research to the waste sector in Australia and globally are also highlighted in this Chapter.

Chapter 2 describes the site conditions and provides details of establishment and commissioning of the experimental plot. It also describes general procedures followed in various experiments. Specific details corresponding to each Chapter are described in separate Chapters.

Survival and growth patterns and their inter-relations and influence by various species and soil thickness are discussed in Chapter 3. Shoot and root growth patterns of 19 species are also included in this Chapter.

Chapter 4 examines chemical composition of leaves and leaf litter with the view to ascertaining the tree will not mobilise heavy metals buried in the waste.

Transpiration rates of various species and in factors influencing transpiration rates is discussed in Chapter 5. The importance of transpiration in a phytocapping system is also discussed.

Chapter 6 deals with canopy rainfall interception, which was studied for the first time. Canopy rainfall interception is a newly recognised parameter which makes a significant contribution to site water balance. This aspect is detailed in Chapter 6

Chapter 7 investigates effects of phytocapping on methane emissions from landfills. This chapter also highlights spatial and temporal variations and quantifies depth-wise variations of methane concentrations in phytocaps and the potential of phytocaps. The results are also discussed in view of the upcoming carbon tax.

Various parameters contribute to site water balance. Soil, plant and climatic parameters are entered into HYDRUS 1D code with the view to simulating percolation of water from Thick and Thin phytocaps. HYDRUS 1D predictions of water percolation from Thick and Thin phytocaps are discussed in Chapter 8.

Phytocapping is known to reduce the costs of landfill remediation while also providing other environmental benefits. These aspects are discussed in Chapter 9.

Finally in Chapter 10, a brief discussion, relevant conclusions and the implications of the results of the study is provided by drawing information from various Chapters and relating these to current literature and legislation.

The next Chapter will explain procedures used in site establishment and data collection.

2

Materials and Methods

2.1 Introduction

Phytocaps have been successfully trialled and adopted in the US, but several aspects such as soil depth, plant species to be grown, performance of Australian native species with regards transpiration rates, canopy rainfall interception, ability to withstand landfill gasses and low oxygen levels in landfill soils and ability to contribute to reduction in methane emission are still unknown. Each of these aspects could constitute a detailed study. However every effort has been made to address each aspect by conducting small to medium investigations. This study also required various types of equipment for different investigations and the principle behind using these to get accurate results. The principle behind using this equipment, their use and the protocols followed are described in this chapter.

In 2003, a decision was made to conduct a field trial across Queensland at landfill sites in Nudgee, Pomona and Rockhampton to prove its effectiveness in controlling site water balance. A research trial was set up in September 2003 at Lakes Creek Road Landfill, Rockhampton, Queensland, Australia, in collaboration with the Rockhampton Regional Council and Phytolink Pty Ltd, Brisbane. In this trial 21 species were grown over two soil depths (viz. 1400 mm & 700 mm). Set up of field experiments and data collection commenced in January 2005, which is part of the PhD thesis. The primary objective of this trial was to optimise the soil depth and evaluate the performance of selected species to reduce percolation of water into the waste. To achieve these objectives, a number of experiments were conducted and various soil and tree parameters were measured. This trial also included monitoring of methane emission from phytocaps and non-vegetated areas of the landfill.

Various performance indicators such as plant growth, plant water use, methane emissions and canopy rainfall interception were studied to select suitable plant

species and optimise soil thickness required for long term sustainability of phytocaps. Each performance indicator was studied separately over a period of three years. Species that were grown on phytocaps were also studied for their growth and their potential to intercept rainfall and transpire and abate methane emission. Thickness of soil cover was optimised based on the site water balance predicted by HYDRUS 1D software (Simunek *et al*. 2005). HYDRUS 1D has been extensively used for site water balance studies and is continuing to gain popularity amongst hydrologists and environmental engineers. HYDRUS 1D has a finite element solution to Richards' equation for one dimensional flow in variably saturated media (Simunek *et al.* 2005).

This chapter describes the experimental set up, soil thickness and species used. In addition, it also describes the method used in installing piezometers to monitor water levels and to record the influence of tide on ground water table. It must be noted that this Chapter includes only the general methods used in setting up of the experiment. Detailed aspects of certain measurements are provided in respective Chapters 3 to 8.

2.2 The Experimental Site

Rockhampton is situated on the Tropic of Capricorn (23.5°S). It receives rain mostly between November to March (Fig. 2.1) with a winter rainfall received during June and September. Rockhampton's average rainfall is 780 mm yr^{-1} (average of 47 years) with a Potential Evapotranspiration (PET) rate of 1632 mm yr^{-1} (BOM 2007). The daytime temperature ranges between 32°C (max) to 22°C (min) during summer (wet season) and 23°C (max) to 9°C (min) during winter (dry season) (BOM 2007).

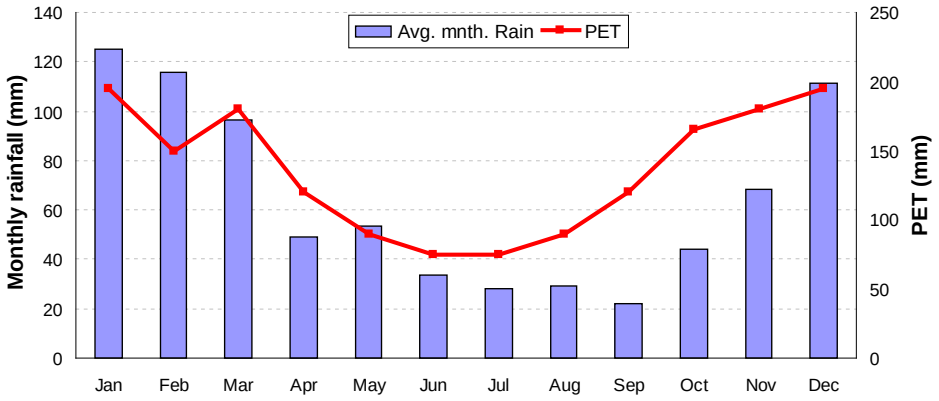


Figure 2.1: Monthly rainfall and PET in Rockhampton

Source: (BOM 2007)

Lakes Creek Road Landfill was constructed in 1981 and is the largest (45 ha) landfill currently operated and managed by the Rockhampton Regional Council (Fig 2.2). It is located within the Fitzroy River flood plain, at approximately 1 km from the Fitzroy River. The landfill site is located adjacent to residential properties to the North of Rockhampton City, a water body to the West, and a mangrove dominated creek to the South and East. Once filled, this landfill site will have to be capped as per the Department of Environment and Heritage Protection (DEHP); previously known as the Department of Environment and Resource Management (DERM) guidelines.

The site is entirely unique from other landfill sites in Queensland due to its topographic conditions, tropical climate, where PET is greater than annual rainfall by 2 fold, rainfall distribution, depth of the ground water (approx. 3.8 to 4 m from the surface of daily cover (Fig 2.8), depth of the buried waste (5 to 6 m) which is influenced by tidal fluctuations. Variation in the depth of groundwater table is greatly influenced by the distance of that site from the river (Kim *et al.* 2006).

Figure 2.2: An aerial view of Lakes Creek Landfill, Rockhampton (Photo: Google Earth, 2006)

In Australia, coastal zones are heavily urbanised. As a result many landfills in have been constructed in mangrove habitats making it even more harmful to the surrounding environment. It is therefore important to understand the complex nature of groundwater containing nutrients and heavy metals (Suh *et al.* 2003). The fluctuation in ground water levels due to tidal flux is critical due to its impact on the buried waste and methane emission into the atmosphere. Tidal fluctuations can affect groundwater behaviour up to a distance ranging from 21 m (lateral) (Suh *et al.* 2003, Cheng *et al.* 2004, Richard *et al.* 2005) up to 1 km (lateral) (Momii *et al.* 2005) with a certain time lag (Neilsen 1990, Li *et al* 1997, Baird *et al* 1998, Richard *et al.* 2005).

Tidal fluctuations are important as they may be linked to ground water discharge (Li *et al.* 2002). Groundwater may contain heavy metals and ion from landfills and reclaimed land (Suh *et al.* 2003). Piezometers (Lallahem *et al.* 2004) and level sensor (PS21000) manufactured by GreenSpan (Australia) were used in study to understand the influence of tides on the groundwater. Figure 2.4 shows the influence of tidal behaviour on the groundwater fluctuation recorded during this study.

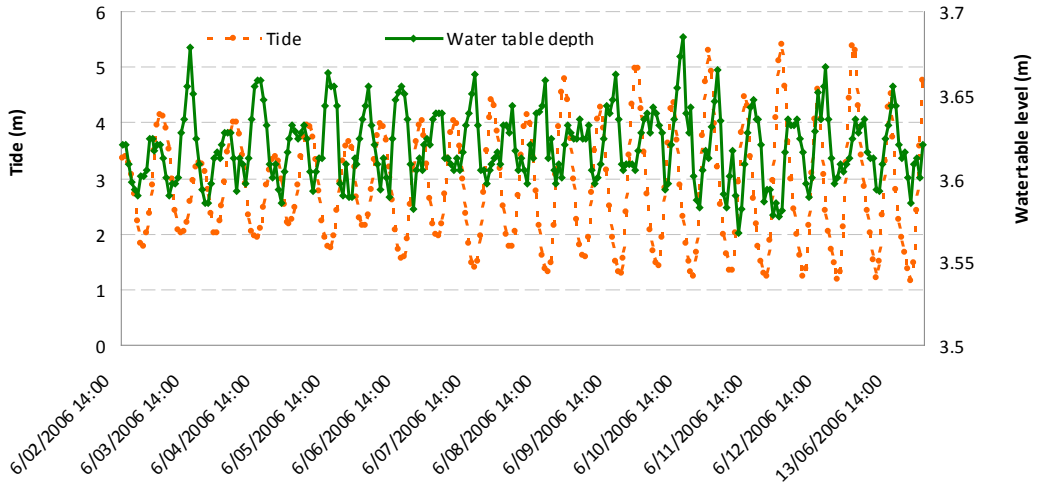


Figure 2.3: Effect of tide on the site's ground water table (June 2006)

2.2.1 Site Establishment

A large landfill site (15,000 m^2) which had reached the final height of waste buried was chosen in consultation with the council staff. The site was tested for its waste composition and most importantly, for the depth of day cover placed over the waste (Fig. 2.4). Based on the uniformity of the day cover and the composition of the buried waste, an experimental site of 5000 m^2 area was selected for this study from the 15,000 m^2 area. The selected test site predominantly composed of timber, plastics, organics, construction and demolition (C&D) wastes, tyres and other recyclables (e.g. glass, cans) (Table 2.1). Depth of the waste ranged from 5 to 6 m (personal communication: Craig Dunglison). The soil used in the construction of the phytocaps was subjected to physical, chemical and hydrological analysis/investigation. Results of the investigation/analysis are presented in Appendix A.

Table 2.1: Composition of buried waste and their depths

Test Pit	Top Soil Description	Types of Waste Buried	Depth to Waste (mm)
1	Black soil	Timber	600
2	100 mm topsoil, road base	Timber	700
3	100 mm topsoil, road base	Timber	700
4	Stoney soil	Plastic	1100
5	Stoney soil	Plastic, tyres	900
6	100mm topsoil, black soil	Tyres	900
7	Topsoil	Timber	400
8	300 topsoil, black soil	Bags	700
10	Black soil	Timber	800
11	500 Black soil, gooey clay	Plastic	1300
12	Topsoil	Plastic, bottles	800
13	Topsoil, road base, alluvial soil	Cans, wire	800
14	Clay	Timber, plastic	1000
15	Brown soil	Plastic, bottles	600
16	Gooey clay	Plastic	500
17	Dark soil and clay	Timber	700
18	Topsoil	Plastic	800
19	600 clayey soil, black soil	Timber, concrete, plastic	1100
20	Brown soil	Plastic, wire	300
21	Road base	Plastic, rubbish	100
22	Gravelly base	Sheet metal	400
23	Topsoil, gravelly base	Bags, cans	500
24	Brown soil	Timber, plastic	600
25	Brown soil	Timber, plastic	300
26	Topsoil	Bags, cans	300
27	Topsoil	Bricks, paper, timber	300
28	200 gravelly base, black soil	Timber, plastic	300
29	Brown soil	Timber	500
30	Black soil	Timber, vegetation, plastic	300
31	Black soil	Timber, plastic	200
32	Gravelly base, black soil	Plastic, bottles	600
33	Stoney black soil	Plastic	800
34	Stoney base	Plastic, paper	700
35	Stoney brown soil	Plastic, cans, timber	500
36	Black soil	Plastic	600
37	Road base, black soil	Plastic	500
38	Road base	Timber, plastic	300
39	Road base	Tyre, plastic	600
40	Brown soil	Cans, paper, plastic	500
41	Base material	Timber	600
42	Black soil	Timber	400
43	Topsoil	Glass, plastic, timber	200
46	Black soil	Tyre, car seat	400
47	Black soil	Timber	800
48	Black soil	Timber, plastics	200
51	Large stones, black soil	Timber	700
52	Stoney base material	Timber	800
53	Topsoil, black soil	Timber	600

Table 2.1 contd

56	Stoney black soil	Timber, tyres	800
57	Stoney base material	Concrete, fibro sheeting	900
58	Black soil	Timber	400
61	Black soil	Timber, concrete	400
62	Stoney black soil	Timber	700
63	Topsoil	Cans, plastic	600
66	Alluvial brown soil	Plastic, wire	500
67	Black soil	Concrete	600
68	Black soil	Timber, vegetation	200
71	Brown soil	Timber, plastics	1100
72	Black soil	Timber	700
73	Black soil, blue chip metal	Plastic, oil filter	700
76	Stoney brown soil	Glass, plastic	1100
77	Black soil	Timber	1000
78	300 black soil, base material	Plastic, bottles	700
81	Brown soil	Tyres	200
82	100 topsoil, gooey black soil	Timber	800
83	300 black soil, base material	Timber, plastics	1100
86	Brown soil	Concrete, steel	300
87	200 topsoil, gooey black soil	Concrete	500
88	Black soil, base material	Plastics	1000
91	Clay	Timber, plastics	700
92	100 topsoil, black soil	Timber, plastics, concrete, steel	1000
93	Stoney black soil	Plastics, bottles	800
101	Brown soil	Plastic	600
102	Brown soil	Timber, plastic	400
103	Brown soil	Steel, plastic	700
104	Brown soil	Paper, plastic	300
105	Brown soil	Plastics, bottles, cans	300
106	Brown soil	Timber, plastics	700
107	Black soil	Foam, plastics	1200
108	Black / grey / orange mottled clay	Chipboard, wood	1000
109	Black / grey / orange mottled clay	Plastic drums, hoses	1200
110	Black clay / grey	Plastic, cement, glass	900
111	Orange / grey clay	Plastic	1200
112	Orange / grey clay	Plastic bags	1450
113	Orange / grey clay	Plastic bags	1400
114	Grey / black clay, topsoil	Plastic, hoses	800
115	Grey / black clay, topsoil	Plastic, glass	700
116	Grey / black clay	Plastic	400

Figure 2.4: Excavated experimental site (Photo: N. Ashwath)

2.2.1.1 Site Preparation

This experimental site was covered with local clay to varying depths of 100 mm to 1200 mm; this cover was excavated and graded (Fig. 2.5) to ensure that approximately 400 mm of the interim uncompacted clay was retained on the waste. The experimental site had two soil depths treatments (Thick soil cover, 1400 mm and Thin soil cover, 700 mm; Fig. 2.6). These treatments were replicated twice. Soil depths were chosen based on the Model for Effluent Disposal using Land Irrigation (MEDLI) prediction (Fig. 2.7). In the Thin soil cover, only 300 mm of sandy loam soil and 100 mm of green waste mulch was placed over the pre-existing 400 mm un-compacted clay soil (total soil cover of 700 mm) (Fig 2.8). In the Thick soil cover, four layers of soil were placed over the pre-existing 400 mm clay soil. This consisted of 200 mm of sandy loam, 300 mm of Yaamba clay and 300 mm of Andersite clay, 200 mm of sandy loam soil and 100 mm of green waste mulch (soil cover of 1400 mm) (Fig. 2.8). Both Thick and Thin soil cover treatments were mulched with a layer of shredded green waste (100 mm). Physical, chemical and hydrological properties of these soils are presented in Appendix A,

Figure 2.5: Various steps involved in constructing the Thick and Thin phytocaps (Photo: N. Ashwath)

Figure 2.6: Thick and Thin soil covers (Photo: N. Ashwath)

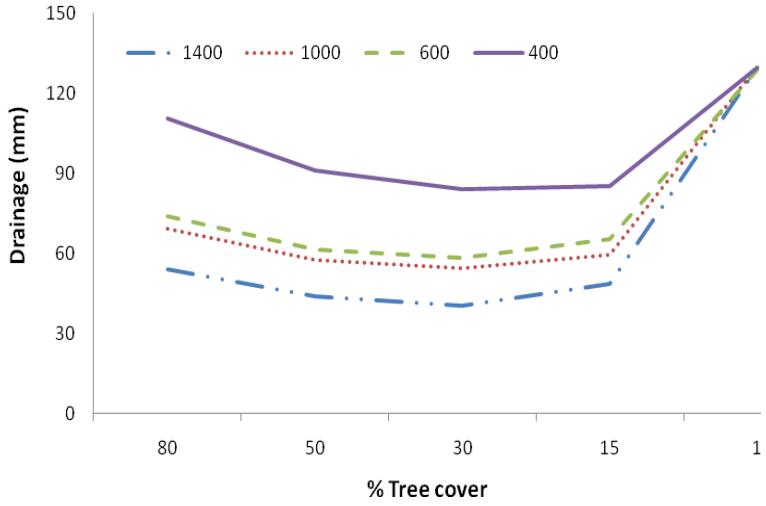


Figure 2.7: MEDLI predictions of percolation of water through various soil depths (mm) and tree density (%)

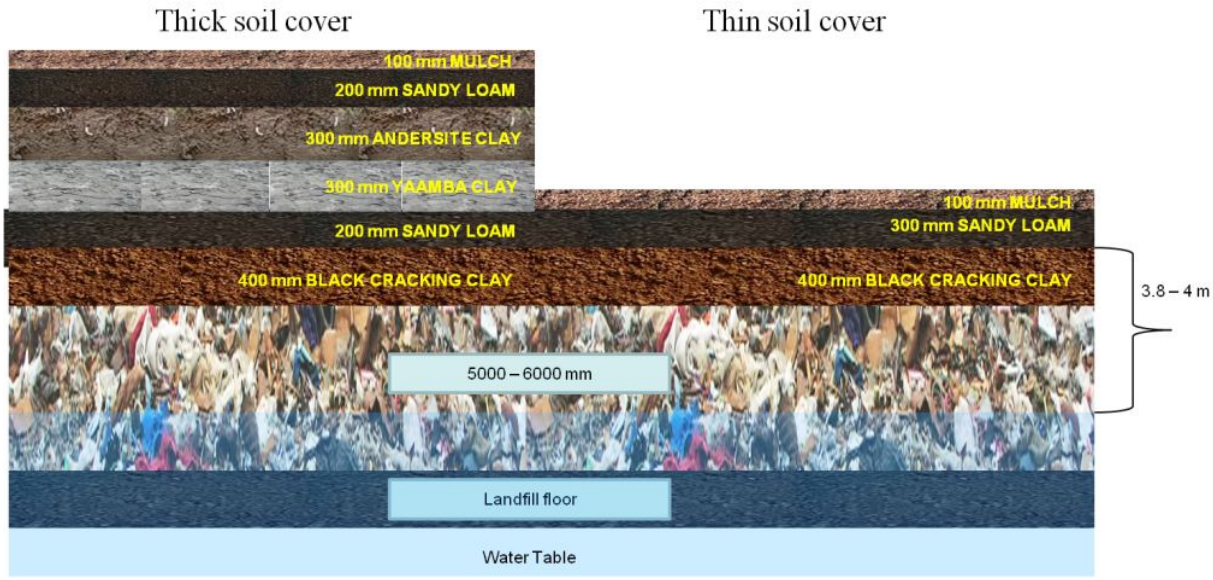


Figure 2.8: Different types of soils and their depths used in Thick (left) and Thin (right) phytocaps

The top layer of both Thick and Thin soil covers contained sandy loam soil which essentially promotes vegetation growth and reduces erosion (Madalinski *et al.* 2003). A drip irrigation system was installed at a row spacing of 2 m, with the drippers spaced at 500 mm apart with the view to providing irrigation during establishment. The dripper system in each plot was connected to a water meter to quantify the amount of water added during the study. The dripper systems were also connected to a timer (Fig. 2.9). After setting up of the drip irrigation system, 18 seedlings of each

species were planted at 2 m x 1 m spacing in each plot (Fig. 2.10). The plots were then mulched (100 mm) to restrict soil evaporation and weed infestation (Fig. 2.11).

Figure 2.9: Setting up of the drip irrigation system (Photo: N. Ashwath)

Figure 2.10: Planting tree seedlings (Photo: N. Ashwath)

Figure 2.11: Spreading mulch layer (Photo: N. Ashwath)

2.2.1.2 Experimental Layout

- A split plot design was used with two replications (Fig. 2.12).
- Each replication consisted of two capping treatments: Thick and Thin soil covers (Fig 2.13). The size of each treatment plot was 25 m wide x 50 m long.
- Within each plot/cap, 21 species were planted in individual plots in groups of 18 plants per plot (6 m x 6 m) (Fig. 2.13).
- The seedlings were planted in three rows (2 m spacing between rows), with a metre spacing between the seedlings within a row (Fig.2.14).
- An additional 9 species were established along the borders of the plots. The batter slopes of the experimental plots were hydro seeded with three native and one exotic grass species, with the view to minimising erosion from steep slopes (Fig. 2.15).
- Two years after planting, every alternate seedling was harvested with the view to determining intermediate plant growth (biomass) and optimising spacing between the trees (2 m x 2 m).

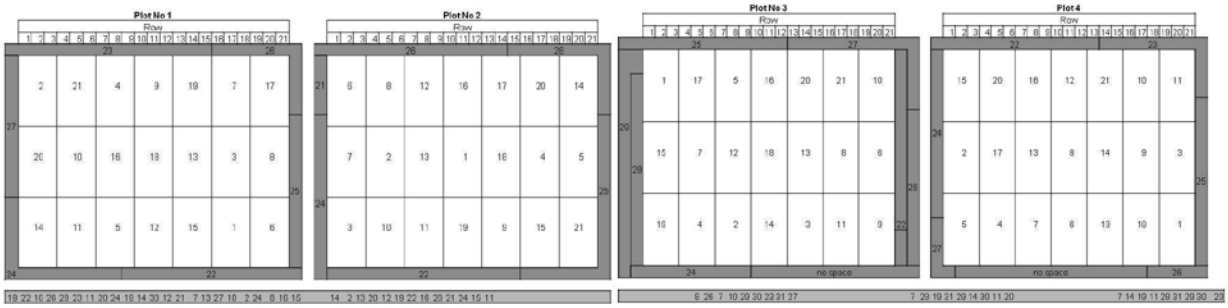


Figure 2.12: Schematic layout of the experimental site
Note: List of species and their associated numbers are given in Table 2.2
Not to scale

Figure 2.13: Top: Thick and Thin phytocap; Bottom: Different species planted with 18 seedlings/species/replication (Photos: N. Ashwath)

Figure 2.14: Photo showing individual species plot where the seedlings were planted at 1 m spacing (Photo: N. Ashwath)

Figure 2.15: Batter slopes were seeded with three native grass species and one exotic species (Photo: N. Ashwath)

2.2.1.3 Tree Species

This study used 21 species (Table 2.2). The species were chosen based on a number of criteria. These include salt tolerance (Ashwath *et al.* 1987), drought tolerance, leachate tolerance (Ashwath and Hood 2001), adaptability to local conditions, species with commercial potential, aesthetic values and their ability to support wild life (e.g. koalas, birds).

Table 2.2: List of species grown on Thick and Thin phytocaps

1 Acacia harpophylla *2 Acacia mangium* *3 Callistemon viminalis* *4 Casuarina cunninghamiana* *5 Casuarina glauca* *6 Cupaniopsis anacardioides* *7 Dendrocalamus latiflorus* *8 Eucalyptus grandis* *9 Eucalyptus raveretiana* *10 Eucalyptus tereticornis* *11 Ficus microcarpa* var. *hillii*	*12 Ficus racemosa* *13 Glochidion lobocarpum* *14 Hibiscus tiliaceus* *15 Lophostemon confertus* *16 Melaleuca leucadendra* *17 Melaleuca linariifolia* *18 Pongamia pinnata* *19 Populus nigra italica* *20 Salix* sp. *21Syzigium australis*	***22Casuarina equisetifolia*** ***23Eucalyptus robusta*** ***24Coralia brachiata*** ***20 Cynoptera iripa*** ***25Syzigium* sp. cv *Jamboo*** ***26Morus alba*** ***27Salix matsudana*** ***28Moringa* sp.** ***29Ficus elastic* (rubber plant)** ***30Acacia aulalocarpa***

*Species in **bold** were used either as border species within the experimental area or were planted directly into the waste in the trench excavated at the north of the experimental site.

2.3 Piezometer Installation

Piezometer wells were installed at the site - 3 wells x 2 replications. One set of 3 wells were installed near the trial plot at the eastern end (Plot 1) and a second set of 3 wells were installed 100 m away from the plots (towards the Fitzroy river; Plot 2). Piezometers at each location were installed at 2 m, 5 m and 9 m depths. During the installation process, the ground surface was drilled to the desired depth (e.g. 2 m, 5 m, 9 m) and piezometer wells were installed (Fig. 2.16). While drilling the 9 m piezometer well, soil samples were collected from various depths to determine moisture content and composition of soil material (Table 2.3). Samples were collected to a depth of 9 m at 0.5 m increments (Fig. 2.17). Each piezometer well was capped at the top end and was insulated with a screen or a mesh cloth at the bottom end for protection from clogging. Then, wells were insulated by sand and bentonite (Sprecher 2000). After installation, the wells were covered and protected by a metallic tube with a lid and a lock (Fig. 2.18) to protect wells from external environment. During installation, a temperature sensor was also attached to the bottom of the well to observe variations in water temperature. Temperature was recorded using the smart logger which used for sap flow measurements.

Figure 2.16: Drilling of piezometer wells

Figure 2.17: Wet waste samples from 6 m depth

Figure 2.18: Piezometer tube completely protected with metal casing

Table 2.3: Composition of the landfill material at various depths

Depth (m)	Plot 1	Plot 2
1	soil + stones	soil + stones
2	soil + dry waste + Stones	soil + dry waste + stones
3	soil + stones + dry waste	soil + stones + dry waste
4	soil + dry waste	soil + dry waste
5	wet waste	wet waste
6	wet waste	wet waste
7	wet waste	wet waste
8	wet waste	clay water (7.8 m)
9	clay water (8.5 m)	clay water

2.4 Meteorological Data

Monthly micro-climatic data was gathered from the Bureau of Meteorology (BOM) site at Rockhampton and from a weather station that was installed at the landfill site. Hourly, daily, monthly and yearly temperature, relative humidity, wind direction, radiation, evaporation wind velocity, rainfall, rainfall duration and rainfall intensity data were obtained for the duration of the study. This data was required to evaluate the performance of the phytocaps for methane emission and species transpiration rates and canopy rainfall interception potential. This data was also used in simulating the site

water balance using HYDRUS 1D. Tidal information was obtained from the Queensland Maritime Safety (QMS) Office for Fitzroy River.

2.5 Summary

This Chapter has summarised the procedures used in establishing the experimental site.

The next Chapter extends the research findings into survival and growth of the trees in a phytocapped landfill site.

3

Tree Growth and Biomass Production*

3.1 Introduction

Organic and inorganic components present in the municipal solid waste may restrain plant growth (Gendebein *et al.* 1992) due to the occurrence of toxic chemicals (Zacharias 1995). Several studies in the past have examined growth rates of trees established in landfill sites. Ettala (1988) studied survival and growth rates of plants established in six landfills in southern Finland. Of the six species studied, *Salix aquatica*, *Betula pendula* and *Populus rasumowskyana* survived and grew well. Nixon *et al.* (2000) studied the growth of poplars and willows in landfills in Southern England to determine their biomass production capacity. They found that trees could produce up to 20 t ha^{-1} biomass in a year. Gilmam *et al.* (1981) tested ten different species grown on a 10 year-old completed landfill and found that only 3 species, viz. *Nyssa sylvatica*, *Gingko biloba* and *Pinus thunbergii* grew well in landfill conditions as compared to other tree species in the first two years of planting. This may have been due to low levels of oxygen and high carbon dioxide levels in the top 40 cm of the soil (Moffat and Houston 1991, Gendebein *et al.* 1992 and Leone 1997), low moisture content and elevated soil temperature (Waisel *et al.* 1991). Several other factors such as trees and root spacing may also affect plant growth (Cresswell and Causton 1988). Trees growing nearby often compete with one another for energy and resources (Wyckoff and Clarke 2004).

*Some data from this chapter have been included in the following paper:
Venkatraman, K. and Ashwath, N. (2009) Phytocapping: importance of tree selection and soil thickness, *Journal of Water, Air and Soil Pollution*, 9: 421-430.

Presence and absence of plants, their growth rates as well as the behaviour of established trees define sustainability of phytocaps, as functioning of phytocaps relies upon plant's transpiration potential. In the past, phytocapping studies have used shrubs, grasses, willows and hybrid poplars (Von Der Hude *et al.* 1999, EPA 2000, Benson *et al.* 2002, EPA 2003, Licht *et al.* 2004, Albright *et al.* 2004, Scanlon *et al.* 2005) which have survived and grown reasonably well on phytocaps (Licht *et al.* 2004, Lee and Schnabel 2000). Poplars and willows have been used in Europe (Elowson 1999) and North America (temperate climates) (EPA, 1998; EPA, 2000 and EPA, 2001) due to their quick growth rates, deep roots and their ability to transpire large quantities of water (Hinckley *et al.*1994, Aaronson 1996, Bassman 2000, Aaronson and Perttu 2001, Dwyer S pers. Comm., EPA 2003). However, tree species are sparingly used and tested on phytocapped landfills in the tropics.

Plant growth on capped landfills has often been reported to be poor (Moffat and Houston 1991) due to limited availability of water to plants (Poore and Fries 1987) and elevated soil temperature in landfill sites (Waisel *et al.* 1991, Mackay and Barbar 1984). Hence, to test the potential of trees to survive and grow well on the phytocaps, a series of growth measurements were taken. Tree survival in its growth phase is vital for the phytocaps as this phytocapping system heavily relies upon transpiration and canopy interception potential of the plants (Venkatraman and Ashwath 2007). Both transpiration and canopy rainfall interception play a key role in hydrology of the phytocapping system. Initial survival of the 21 species was closely monitored and the mortality of the species was assessed by a monthly census counting. Several other growth parameters were also monitored.

Growth parameters including height (Downes et al. 1999, Kim and Lee 2005), stem diameter (Yokozawa and Hara 1995), canopy spread (Cole and Lorimer 1994, Sterck *et al.* 2003), Leaf Area Index (LAI) (Dewar 1996, King *et al.* 2005), shoot biomass (including stem, leaves and branches) and root biomass (Snowdon *et al.* 2002) have been determined at two time intervals to assess plant growth..

Tree height and stem diameter are both good indicators of plant growth. Tree height has been reported to change as they mature and change with seasons, climatic conditions and soil moisture levels (Ferri 1979, Lovdahl and Odin 1992). However, stem diameter increment is influenced more by plant water relation features rather than by mere presence of water in the soil (da Silva *et al.* 2002).

Canopy spread is a good indicator of plant growth and this depicts competitive effect of a tree (Cole and Lorimer 1994). Sterck *et al.* (2003) reported various effects of horizontal and vertical canopy spread on overall growth of plants. Primary functions of the canopy are light energy assimilation via photosynthesis (Herwitz *et al.* 2000), release of energy by respiration (Wang and Jarvis 1990) and transport of water to the atmosphere via transpiration (Granier 1987). These functions are performed by leaves (Wang and Jarvis 1990), which are reflected through the Leaf Area Index (LAI) (Dewar 1996).

The LAI is also a key structural characteristic of phytocapping systems, due to the role played by their leaves in controlling many biological and physical processes in plant canopies (Chen *et al.* 1997). LAI is directly proportional to the light intercepting capacity (LI) of the species (Balster and Marshall 2000). The LAI depicts leaf density in the canopy (Dewar 1996) which in turn reflects growth (Sterck *et al.* 2003). On the other hand, biomass produced by trees influences transpiration rates (Singh and Bhati 2003) and root's potential to take up water efficiently and effectively (Eamus *et al.* 2006).

Shoot biomass estimation (leaves, branches and stem) is vital to understand the relationship between growth (stem diameter; Rayachhetry *et al.* 2001) and harvest gain (Grote 2002). Biomass studies are also important for estimating nutrient cycling and heavy metal accumulation (Grote 2002) in plants. A typical above ground biomass distribution in a 9 year-old plantation of *Eucalyptus grandis* constituted 3% foliage, 9% branches, 16% bark and 72% wood (Birk and Turner 1992), and this fraction is constant in all trees (Krichbaum et al. 1992, Makela and Valentine 2004). Apart from being a

good indicator of growth, biomass production has environmental benefits such as the potential carbon dynamics in trees (Drake *et al.* 2003) and their associated sequestration (Lu *et al.* 2002).

Root biomass studies are also essential to determine distribution and flow of materials within the phytocapping system (Santantonio et al. 1977, Wu *et al.* 1985, Lynch 1995) and to examine plant water uptake (Cannell 1985). Roots are the structural base of the plant (Santantonio *et al.* 1977, Gregory 2006), and they help plants store water and nutrients (Santantonio *et al.* 1977). Root development and distribution are influenced by soil moisture availability, as the root depth increases with the lack of water availability in the top layers of the soil (Erricsson 1994). Root biomass is generally determined by destructive methods (Amato *et al.* 2008) which are time consuming, expensive and yet provide data at moderate levels of precision (Snowdon *et al.* 2002).

A recent global analysis of root depth distribution reported that at least 50% of the root biomass was concentrated in the upper 300 mm of the soil (Schenk and Jackson 2002) and this may prove to be good for carbon accounting as only the upper 300 mm depth of the soil profile is taken into consideration while accounting for carbon (IPCC 2007). The 21 species grown in the Thick and Thin phytocapping systems of this study were subjected to a detailed assessment to determine root depth, root distribution and the impact of soil thickness on root growth.

At a landfill site, presence of landfill gases, leachate, heavy metals and organic contaminants determine ability or potential growth of plants (Gendebein *et al*. 2002, Gower *et al.* 1996). Decline in the groundwater table (Horton and Clarke 2001), build-up of salts in soil (Rawat and Banerjee 1998), soil compaction (Kozlowski 1999), soil temperature (Guo and Sims 2001) and soil structure (Passioura 1991) also influence plant growth. These factors greatly affect the ability of plants to avail resources (Wyckoff and Clarke 2004). Root penetration is often restricted by compacted soil (Passioura 1991). Soil compaction limits trees from supplying adequate water to its leaves (Barraclough and Weir 1988). Fine roots (<2 mm in diameter; McKenzie *et al.*

2001) are the main corridors for water and nutrient in plants (Jackson *et al.* 1997). Fine roots control water absorption and groundwater and atmospheric fluxes (Paruelo and Lauenroth 1995).

Fatality to plant species (Moffat and Houston 1991) and poor growth rates of trees results from shallow soil depth in phytocaps (Warren *et al.* 1996). Healthy growth of plants in a phytocapping system is vital for maintaining the hydrological balance of the site via transpiration (Eamus *et al.* 2006) and canopy rainfall interception (Aboal *et al*. 1999 Witkowski and Lamont 1991). Hence the current study was undertaken to monitor survival and growth of trees established in Thick and Thin phytocaps with the view to determining the ability of phytocaps to control site water balance and to abate methane emission.

3.2 Materials and Methods

3.2.1 Survival, Growth and Biomass

Tree growth is the increase in size with time by means of cells, organs and tissue expansion (Brack 1997), which highly vary between species. Growth was monitored over a period of two and half years. Height, stem diameter, canopy spread and LAI were monitored. Above ground biomass (includes stem, leaves and branches) and root biomass (includes coarse and woody roots) were also determined (Snowdon *et al.* 2002).

3.2.1.1 Tree Survival

Survival of the species was monitored by counting the number of dead plants in each plot. Trees were declared dead when they lost their leaves and the stems dried out (Rawlinson *et al.* 2003).

3.2.1.2 Tree Height

Tree height was measured using a calibrated 8 m tall collapsible pole. During the three years of monitoring, five trees per species per plot. There were a total of 18 plants per

plot before thinning and a total of 9 plants per plot after thinning (Fig 2.14). The selected trees were tagged and the height of the same plants was measured at six monthly intervals. The relative tree height increment was calculated from height measurements.

3.2.1.3 Stem Diameter

Stem diameter was measured using a digital Vernier calliper (Cooper Hand Tools, Australia). The calliper was able to record measurements up to 200 mm. The five trees that were tagged for height measurement were used to determine stem diameter at 50 cm from the ground (D_{50}) and the Diameter at Breast Height (DBH - 1.37 m). Both D_{50} and DBH were measured twice a year as per height measurements. The relative stem diameter increment was calculated for each species from the average of tree measurements taken per species grown in the phytocapping system.

3.2.1.4 Canopy Spread

Canopy spread was calculated from an average of three radius measurements taken at right angles from the stem using a measuring tape. Canopy spread was calculated using the formula of the area of circle, assuming that the canopy will more or less take the shape of a circle. Three trees from each species from each plot (total of 4 plots) (a total of 18 plants/plot before thinning and a total of 9 plants/plot after thinning; (see Fig 2.14) were measured annually.

3.2.1.5 Leaf Area Index (LAI)

Leaf Area Index is defined as the area of leaves per unit area of soil surface. It is a ratio and hence has no unit. LAI was determined every 6 months for two consecutive years using an AccuPar ceptometer (Decagon Devices, Pullman, USA). AccuPar ceptometer calculated LAI based on the above the canopy PAR measurements along with other variables that relate to canopy architecture and position of the sun (Zenith angle). A minimum of three measurements were taken above the canopy per species per plot. Similarly, ten measurements were recorded below the canopy per species per plot, and the average was calculated.

3.2.1.6 Biomass Estimation

Above Ground Biomass

Biomass was determined by harvesting the shoots and roots (Poorter and Garnier 1996, Snowdon *et al.* 2002). Biomass was estimated on two different occasions: i) two and half years after planting, and, ii) three and a half years after planting.

In the first harvest, alternate tree from each plot (nine trees out of 18) were harvested from each Thick and Thin phytocaps and from both replications (thinning operation). Of the 9 trees harvested, three were used to determine above ground biomass, and the other six were harvested and measured for their D_{50}, DBH, sapwood depth and height. The three harvested trees were separated into stems, leaves and branches. The stem was cut into several pieces and fresh mass determined on site. Samples of these materials were drawn, weighed, placed in paper bags and labelled. The samples were then transported to the laboratory, oven dried at 70°C (Salisbury and Ross 1991) until they attained a constant weight. Once dried, samples were weighed and dry mass were recorded. Similar procedure was followed during the second harvest in 2007. Of the nine trees harvested during second harvest, three trees were selected for the above ground biomass determination and the remaining (up to six plants/plot) were measured for height, D_{50}, DBH and sapwood depth. The same three trees that were harvested for shoot biomass were also excavated to determine their root biomass.

Root Biomass

A wide range of techniques has been employed to estimate root biomass. However, excavation and core methods remain the most widely used method (Snowdon *et al.* 2002). Bengough *et al.* (2000) have provided further information on sampling strategies and statistics associated with root sampling. During the current study, the total roots that were excavated were partitioned into woody roots (>25 mm) and coarse roots (<25 mm). Soil samples were drawn from every 10 cm depth to determine fine roots (<2 mm). A back hoe (Fig. 3.1) was used to dig a 1 m deep trench beside the trees at about 1 m from the main stem to observe root distribution profile (Snowdon *et al.* 2002, Burrows *et al.* 2000) (Fig. 3.2 and Fig. 3.3). Lateral root distribution was also measured

with the help of a measuring tape. The trench method was employed only to one of the two replications (Plot 3 and Plot 4) of Thick and Thin phytocaps, as it was time consuming and laborious. This excavated trench was used to draw horizontal cores for determining fine roots.

Horizontal soil cores were drawn using PVC tubes (5 cm in diameter and 12 cm long) (Snowdon *et al.* 2002) at the depths of 10 cm, 20 cm, 30 cm, 40 cm and 50 cm. The samples were then placed in paper bags and labelled. Roots from the remaining plot 1 and plot 2 were collected by scraping off the top soil, layer by layer, using a back hoe, collecting the roots from the scraped soil, measuring lateral distribution of roots and finally lifting the root ball (>1 m depth) using the excavator bucket (Fig 3.4). Roots that were removed during scraping were also collected in separate bags and labelled. The root ball was then placed in a large trough (Fig. 3.5) and soaked in tap water for 1 to 2 hours. The soaked root ball was washed free of soil using a high pressure system. Once washed completely, the roots were placed on a tarpaulin to drain the water. Woody roots were separated from coarse roots and were weighted separately. Root samples were placed in fertilizer bags and labelled accordingly. Samples were then oven dried at 70°C for 5 to 10 days until they attained a constant weight (Snowdon *et al.* 2002). Once dried completely, the final weight of each sample was measured.

Figure 3.1: Back hoe carrying excavated roots of *Hibiscus tiliaceus* during root distribution study

Figure 3.2: Root depth and lateral root measurement (Left) One meter deep trench was dug to observe root distribution; (Right) Lateral root length measured using a measuring tape

Figure 3.3: A trench excavated next to the stem (1 m) to measure root depth

Figure 3.4: Root ball of *Dendrocalamus latiflorus* (bamboo) was excavated using a back hoe

Figure 3.5: A large trough was used to wash root ball

Root Length Density (RLD)

Soil cores were drawn by driving PVC cores (5 cm in diameter and 10 cm in length) into the soil at different depths. Core samples with PVC tubes were placed in paper bags

and labelled. Initial weight of these samples was taken on-site and then the samples were split into two equal halves. The soil core was placed on a wax paper and made into two equal halves by weighing. The first half of the sample was used to determine moisture content of the soil, and, the second half was used to determine RLD and diameter. The first half of the sample was dried in an oven at 70°C for up to 5 days and the dry weights were determined. The second half of the sample was placed in plastic bags and stored in a cool room until washed and root lengths determined (<2 weeks). These samples were carefully washed and the roots were sieved through 1 mm, 600 µm and 425 µm mesh sieves to fine roots. This ensured minimum loss of roots while washing. The fine roots were then removed from the sieves with the help of a thin brush and forceps and were placed in trays designed for root scanning. The RLD and diameter of these roots were determined using Delta- T scan (root length scanner) (Bouma *et al.* 2000). The values were expressed as cm of roots per cm^3.

Leaf Litter Biomass

Initially, a 50 cm x 50 cm quadrat was constructed using PVC tubes. The quadrats were placed randomly between tree stands within each plot (species) and all leaf litter samples present in the quadrat were collected and placed in paper bags. Three such samples were collected for each species in each plot. The samples were then washed, blotted dried and oven dried at 70°C for 3 days, before determining dry weight.

During the later part of this study, plastic trays (50 cm x 50 cm) were used to determine annual litter fall (Grigg and Milligan 1999). This experiment was performed in both Thick and Thin phytocaps. Three plastic trays were randomly placed under the canopy of each species in each plot. Leaf fall from each tray was collected every 4 months in a year. The samples were washed, dried and dry weights were determined according to Grigg and Milligan (1999) and Pauses (1997).

3.3 Statistical Analysis

The data were tested for outliers, normality and homogeneity of error variances before subjecting them to ANOVA using Genstat ver. 8.0 (Wass 2011, Payne 1997). The parameters that showed significance for the F test (P<0.05) were subjected to t test. The effects of time were also tested for some parameters that were measured repeatedly. Least significance differences (l.s.d) are presented in figures where the treatment, capping, species effect, time or their interactions were significant (P<0.05). Standard errors of differences are provided for some parameters where there were insufficient data available for ANOVA, or when the F test was found not significant (P<0.05).

3.4 Results and Discussion

There are several factors such as landfill gas, soil contamination, soil compaction, high soil temperature, drought, waterlogging, leachate, waste type, lack of oxygen in the soil layers and age of the landfill that potentially affect tree growth and survival, particularly in their early growth stage (Cureton *et al.* 1991, Chan *et al.* 1997, Chan *et al.*1999, Liang *et al.* 1999, Hilger *et al.* 2000). Hence a detailed survival and growth study was undertaken. The present study has investigated the growth of 19 of the 21 tree species established on a landfill to analyse performance of each species. Results from ANOVA are presented in Table 3.1.

Table 3.1: ANOVA for various growth parameters

Growth Parameter	ANOVA	d.f.	Significance (P)	Growth Parameter	ANOVA	d.f.	Significance (P)
Height				Shoot biomass 2007			
	Cap	1	<0.001		Cap	1	<0.001
	Date	6	<0.001		Species	18	<0.001
	Species	18	<0.001		Cap.Species	18	0.556
	Cap.Date	6	0.02	Branch biomass 2007			
	Cap.Species	18	0.155		Cap	1	0.14
	Date.Species	108	<0.001		Species	18	<0.001
	Cap.Date.Species	108	0.603		Cap.Species	18	0.257

Table 3.1 contd.

Stem diameter				Leaf biomass 2007			
	Cap	1	0.154		Cap	1	0.408
	Date	6	<0.001		Species	18	<0.001
	Species	18	<0.001		Cap.Species	18	0.823
	Cap.Date	6	0.811	Stem biomass 2007			
	Cap.Species	18	0.584		Cap	1	<0.001
	Date.Species	108	<0.001		Species	18	<0.001
	Cap.Date.Species	108	0.058		Cap.Species	18	0.556
Canopy spread				Shoot biomass 2006			
	Date	2	<0.001		Cap	1	0.551
	Species	18	<0.001		Species	18	<0.001
	Cap	1	0.012		Cap.Species	18	0.351
	Date.Species	36	<0.001	Root biomass			
	Date.Cap	2	0.022		Cap	1	0.018
	Species.Cap	18	0.162		Species	18	<0.001
	Date.Species.Cap	36	0.988		Cap.Species	18	0.139
LAI				Coarse root biomass*			
	Cap	1	0.259		Cap	1	0.207
	Date	1	0.007		Species	18	<0.001
	Species	18	<0.001		Cap.Species	18	0.444
	Cap.Date	1	0.916	Woody root biomass			
	Cap.Species	18	0.46		Cap	1	0.408
	Date.Species	18	<0.001		Species	18	<0.001
	Cap.Date.Species	18	0.621		Cap.Species	18	0.823
Shoot fraction$^{\Psi}$				Root fraction$^{\Psi}$			
	Cap	1	0.739		Cap	1	0.687
	Species	18	0.013		Species	18	0.009
	Cap.Species	18	0.782		Cap.Species	18	0.796
Root Length Density							
	Depth	4	<0.001				
	Species	18	0.081				
	Depth.Species	72	0.825				

*$\log_{10(}$Transformed, $^{\Psi}$ SQRT transformed

3.4.1 Survival

Of the 21 tree species, all species except *Populus* sp. and *Salix* sp. survived for up to 3.5 years (2004 – 2007).. Normally, on landfills more than 10% of the newly planted seedlings die in the first year (Bradshaw *et al.* 1995), which in this case was less than 9%. *Populus* sp. and *Salix* sp. grew well for the first 12 months after planting. These died during the onset of summer where the day temperature exceeded 40°C. *Populus* sp. and *Salix* sp. are known to adapt better to temperate climate (Francis *et al.* 2005). *Populus* sp. should ideally be planted when the soil temperature is at least 14°C (Zelesny *et al*. 2005) while *Salix* sp. should be planted in warm temperatures and moist soils (Abrahamson *et al*. 2002).

3.4.2 Tree Growth

3.4.2.1 Tree Height

Tree species showed an annual relative height increment of 0.63 m which is an increase of 5.2 cm per month (Fig 3.6).

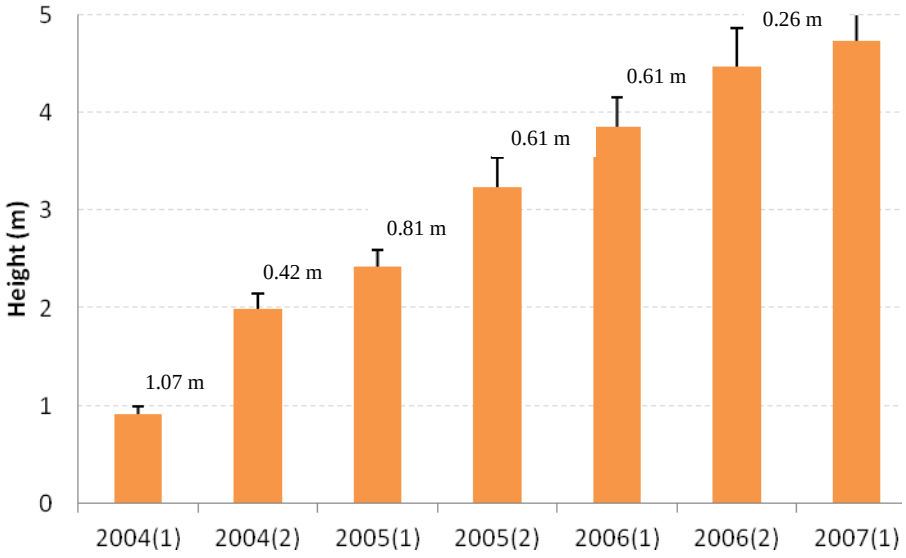


Figure 3.6: Relative height increment of trees grown on a phytocap over 3.5 years
(Bars represent standard error of means). Marked values show height increments per every 6 months.

Height growth in the established species varied significantly between species (P<0.001) (Table 3.1, Fig. 3.7) and may be attributed to genetic differences between species

(Lovdahl and Odin 1992, White 1974) and differences in their stand characteristics (Huang and Titus 1999).

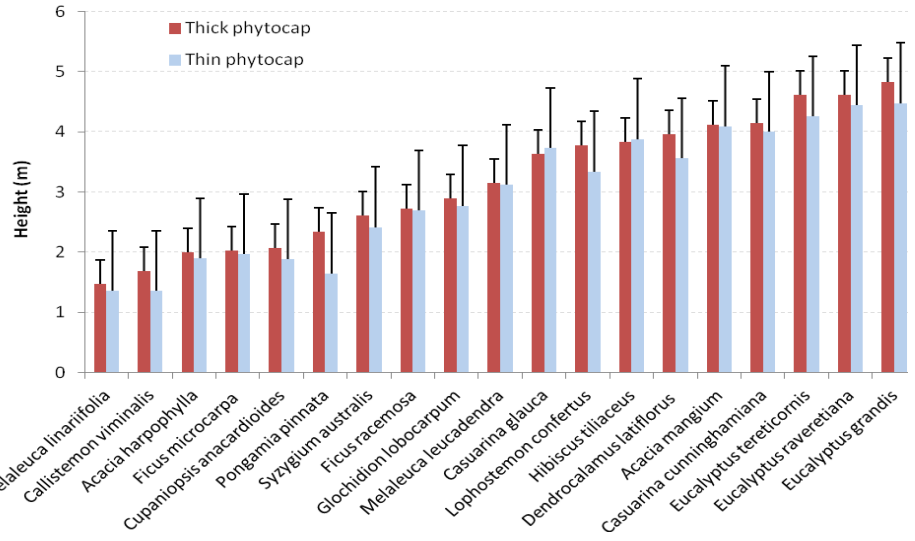


Figure 3.7: Height growth of 19 species grown on Thick and Thin phytocaps
(Bars represent l.s.d. 0.397)

Fast growing species such as *A. mangium*, *H. tiliaceus*, casuarinas, bamboo and eucalypts grew more than 6 m tall (Fig. 3.8) in 3.5 years. Most species grew over 2 m yr^{-1} (Fig. 3.8). *Calistemon viminalis* and *M. linariifolia* were the only two species that showed very slow growth rates, as they are known to grow slowly (Wright and Westoby 1999).

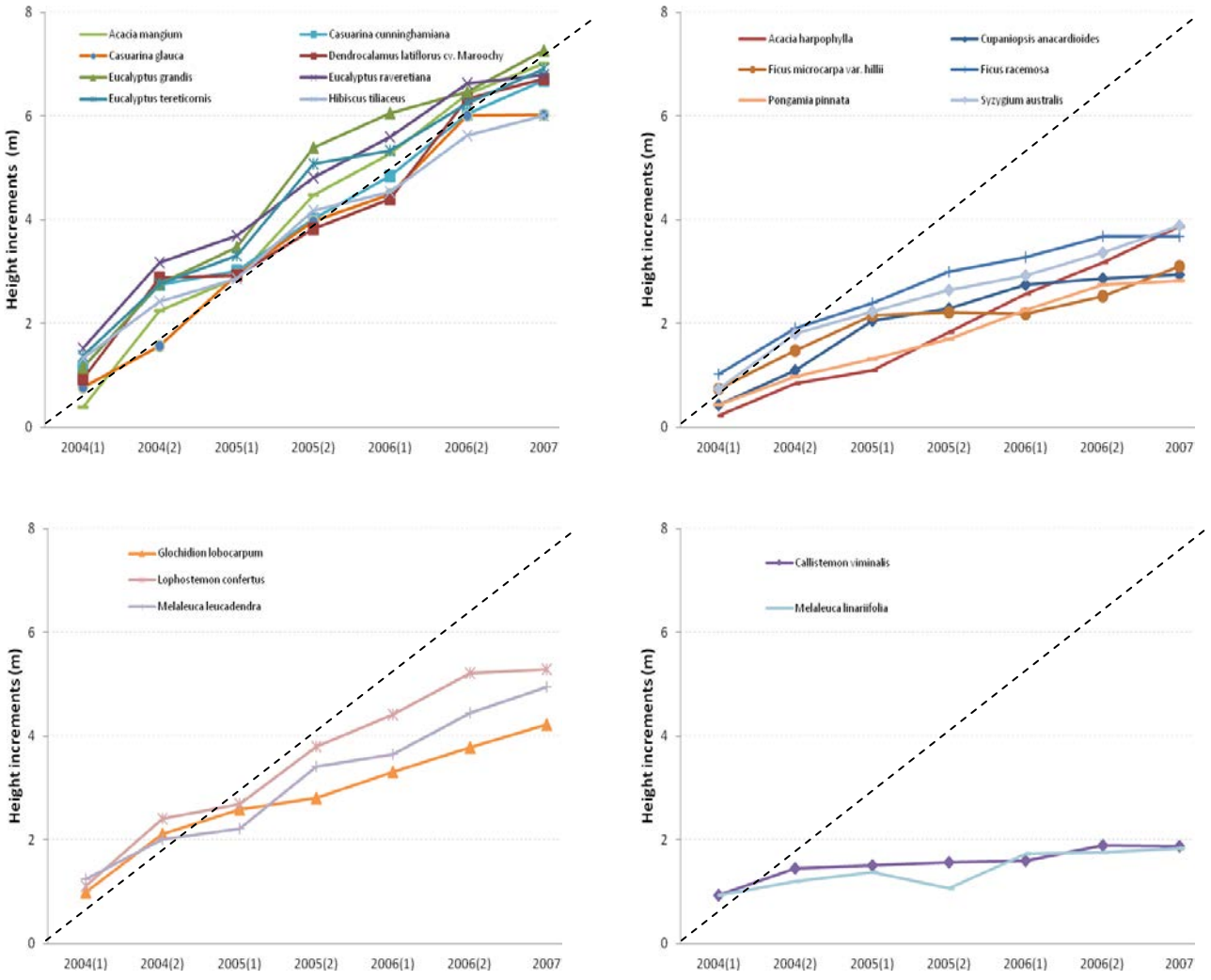


Figure 3.8: Height increment in 19 different species grown on a phytocap (l.s.d. 0.569), dotted line was drawn to show the patterns of increment by different groups of species

Tree height varied significantly (P<0.001) between Thick and Thin phytocaps (Table 3.1, Fig. 3.9). Most species grew faster in the Thick phytocap than in Thin phytocap possibly due to better nutrient availability (Maurice 2005), better root development (Moffat and Houston 1991, McGuire *et al.* 2001) and increased water availability (Fu-sheng *et al.* 2005). *Eucalyptus grandis, E. raveretiana, E. tereticornis, A. mangium* and *C. cunninghamiana* grew more rapidly than the others in the Thick soil cover than in the Thin soil covers (Fig. 3.7).

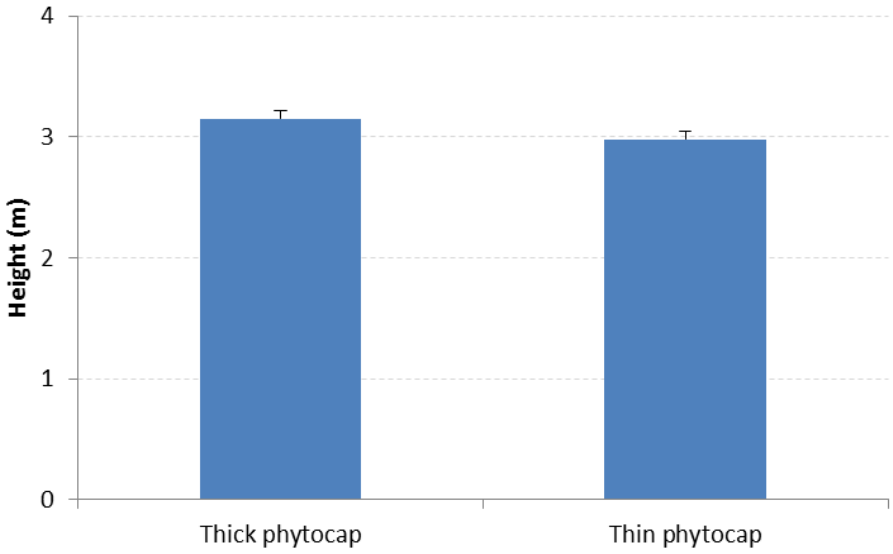


Figure 3.9: Tree height in Thick and Thin phytocaps
(l.s.d. 0.069)

3.4.2.2 Stem Diameter

The species grown in the phytocapping systems showed an annual relative stem diameter increment of 12.3 mm (Fig 3.10) which is well within the range (-0.48 to 11.4 mm yr^{-1}) reported by Manokaran and Kochummen (1993), Condit *et al.* (1995), Clark and Clark (1999) and Da Silva *et al.* (2002).

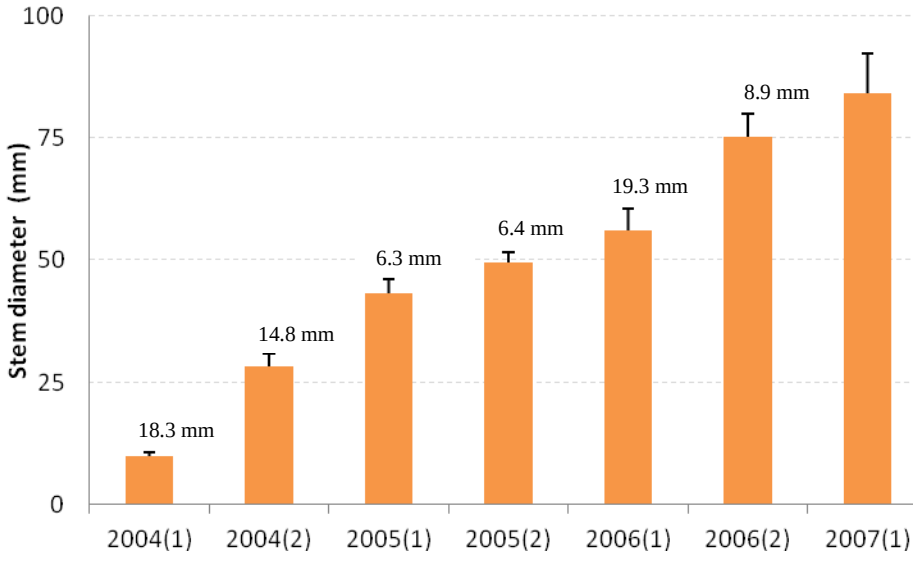


Figure 3.10: Relative stem diameter increment in trees grown on a phytocap over 3.5 years (Bars represent SE of means). Marked values show height increments per every 6 months.

Stem diameter varied significantly (P<0.001) between species (Fig. 3.11) and over time (Table 3.1) and this may be due to genetic differences, changes in the climatic conditions and internal changes in the canopy structure and hence exposure to sunlight (Kammesheidt *et al.* 2003), rainfall, air temperature, solar radiation and vapour pressure deficit (VPD) (Xiong *et al*. 2007). The sharp increase in stem diameter between 2006(1) and 2006(2) (Fig. 3.12) can be attributed to thinning (harvesting) that was done immediately after 2006(1) measurements were taken. Stem diameter increments are highly influenced by transpiration (McLaughlin *et al.* 2003) and the presence of water in the soil (da Silva *et al.* 2002). The slower increment in stem diameter in *P. pinnata*, *C. viminalis*, *F. microcarpa*, *A. harpophylla*, *C. anacardioides* and *M. linariifolia* could also be due to their inherent nature, wherein they grow slowly during initial stages and then grow fast after they have accumulated sufficient root growth. Stem diameter varied significantly (P<0.001) between species (Table 3.1), with some species attaining around 30 mm, the others 50 mm, and a few 60 mm over 3.5 years. Stem diameter increment from 2004 to 2007 is shown in Fig. 3.12. Slower increment from 2005 (1) to 2006 (1) is perhaps due to increased competition between the seedlings (18 plants/plot) and exceptionally dry season. A rapid increase in stem diameter from 2006(1) to 2006(2) can be attributed to thinning operation, wherein the number of seedlings were reduced from 18 to 9 in each plot.

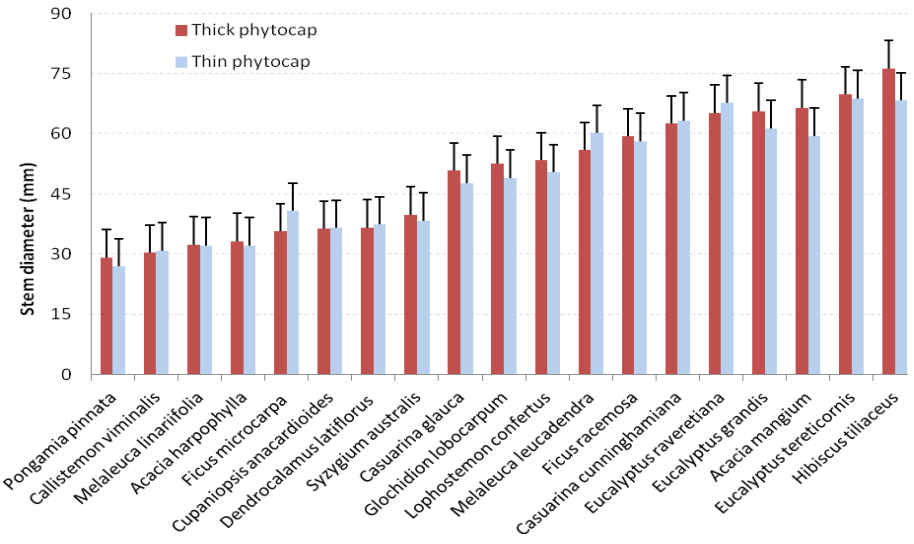


Figure 3.11: Stem diameter of 19 species grown on Thick and Thin phytocaps
(Bars represent l.s.d. 6.93)

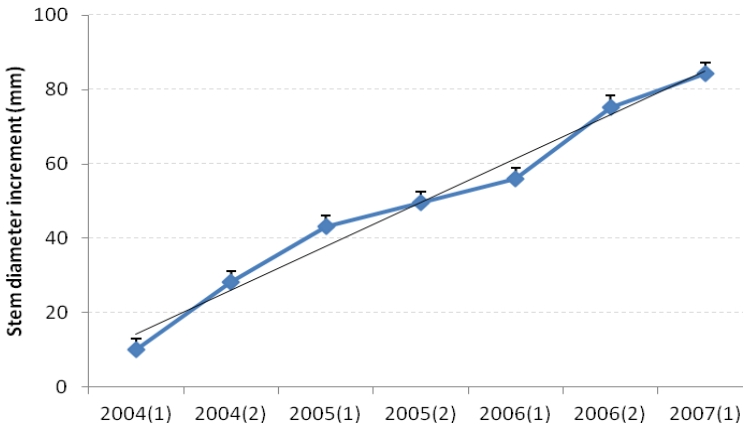


Figure 3.12: Stem diameter increments of 19 species grown on a phytocapping system.

Hibiscus tiliaceus, casuarinas and eucalypts developed a stem diameter of c.120 mm (Fig. 3.13). All species had a stem diameter of more than 30 mm in 3.5 years (Fig. 3.13). The patterns of changes in stem diameter mirrored those of stem height increments (Fig 3.8), wherein three categories of responses (fast-growing, moderately-growing and slow-growing) were noticed. Greater stem diameter coupled with quicker increments will enhance transpiration rates in trees, which is vital to maintain appropriate site water balance.

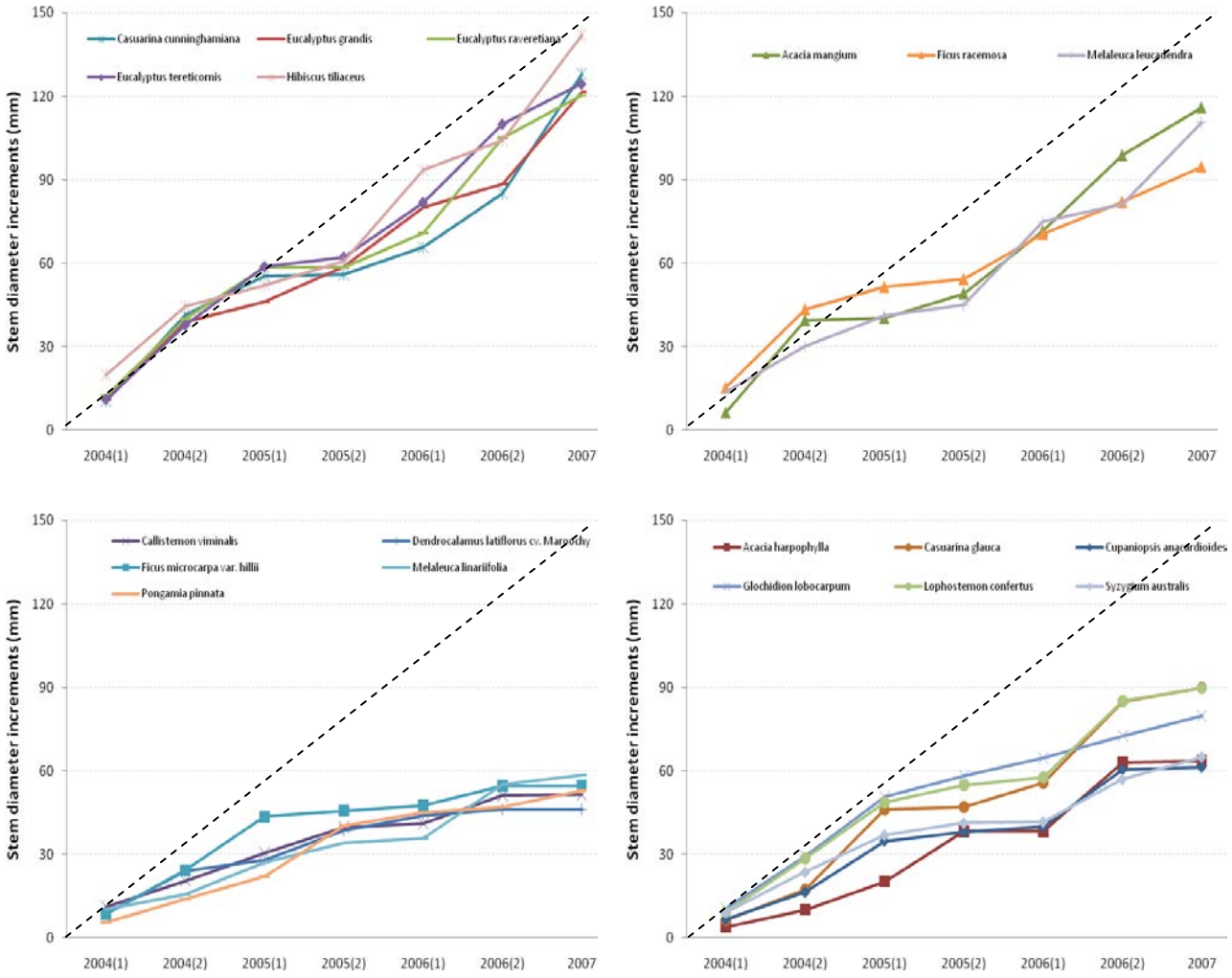


Figure 3.13: Stem diameter increments over 3.5 years in 19 species grown on a phytocap (l.s.d. 18.36). Dotted line was drawn to show the patterns of increment by different groups of species

3.4.2.3 Leaf Area Index

Leaf Area Index of any vegetation is a major determinant of its growth potential and it also contributes to the tree's water relations. LAI reflects photosynthetic and transpiration capacity of plants (Chen *et al.* 1997). LAI in this study varied significantly (P<0.001) between 2005 and 2006 (Table 3.1, Fig. 3.14) and this was associated with changes in their growth rate, wherein they had limited competition between the seedlings in 2005 (see Fig 2.14) and high competition during the first half of 2006 before they were thinned. As a result, highly significant (P<0.001) date* species interaction was found (Table 3.1). Tree Variations between species in their annual height increment (Whitford *et.al*. 1995, West *et.al*. 1988, Sterck *et al.* 1999 and King *et al.* 2005) and spacing (Foli *et al.* 2003) can also contribute to changes in LAI. Furthermore, senescing nature of plants (e.g. *Pongamia pinnata*), which show contrasting response have also contributed to significant interactions between age of the trees and height increment. Some species exhibited large variations in LAI between 2005 and 2006 (Fig. 3.14), which can be attributed to the degree of senescence in each season.

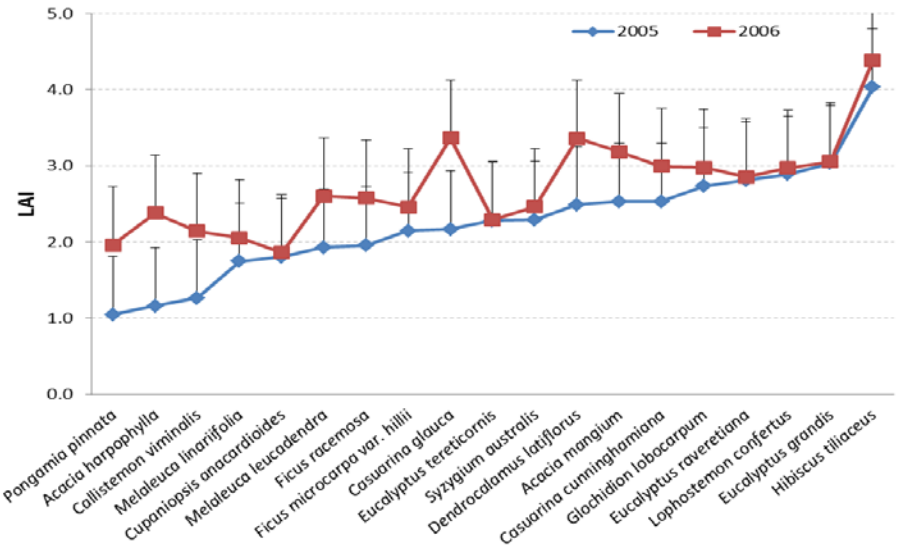


Figure 3.14: LAI of 19 species grown on Thick and Thin phytocaps at the ages of 2 and 3 years (l.s.d. 0.764 for Cap x Species interaction).

LAI also varied significantly (P<0.001) between species (Fig. 3.15, Table 3.1). The established species attained an LAI of 2.4 after 3 years. The maximum achievable is in the order of 6 to 8 for a mature forest (Beadle 1993). *Hibiscus tiliaceus* attained the maximum LAI of 3.7 (Fig. 3.15) in three years. These LAI data revealed that many of the species closed their canopies after 3 years of planting (Fig. 3.14); see Appendix C. This in turn is expected to maximise canopy rainfall interception (Sands 2004, King *et al.* 2005).

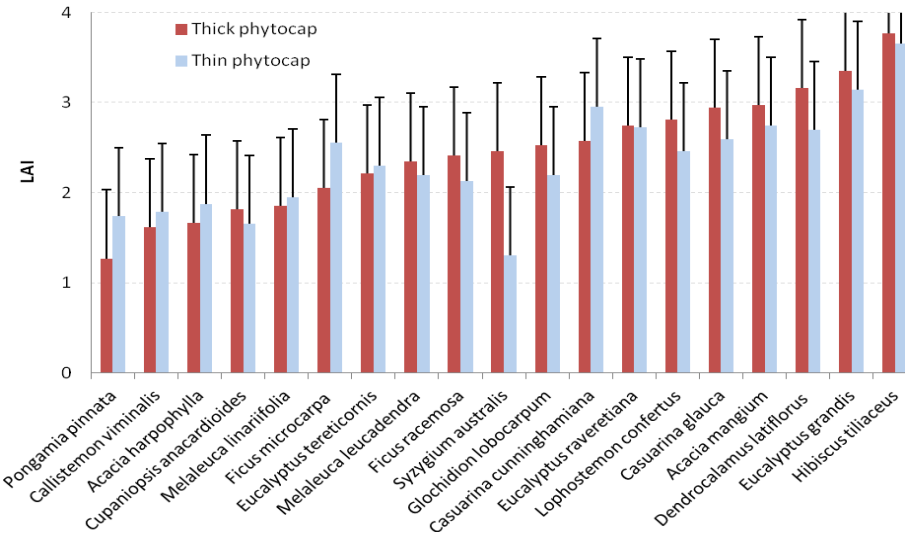


Figure 3.15: LAI in 19 different species after 3 years of planting
(l.s.d. 0.540)

LAI also depicts canopy density (Dewar 1996), which in turn reflects growth of trees (Sterck *et al.* 2003) which contributes to transpiration and canopy rainfall interception, both of which are important for maintaining site water balance. The canopy density analysed by HemiView (Dynamax, USA) in four different species (after 3 years of planting) grown in the phytocapping systems is shown in Appendix C.

3.4.2.4 Canopy Spread

Canopy spread which shows competitive effect of an individual tree (Cole and Lorimer 1994) differed significantly (P<0.001) between species and over time (Table 3.1). The steep increase in canopy spread between the year 2006 and 2007 (Fig. 3.16) was largely

due thinning. Other factors, such as variations in rainfall during 2005 and 2006, have also influenced canopy growth in many plant species. Trees with greater canopy cover will increase canopy rainfall interception which in turn will help increase water retention capacity of the soil (Joffre and Rambal 1988) and may reduce preferential flow. Trees with greater canopy spread will also increase stemflow by funnelling of rainwater (Mauchamp and Janeau 1993), thus contributing to nutrient cycling and tree growth (Potter 1992).

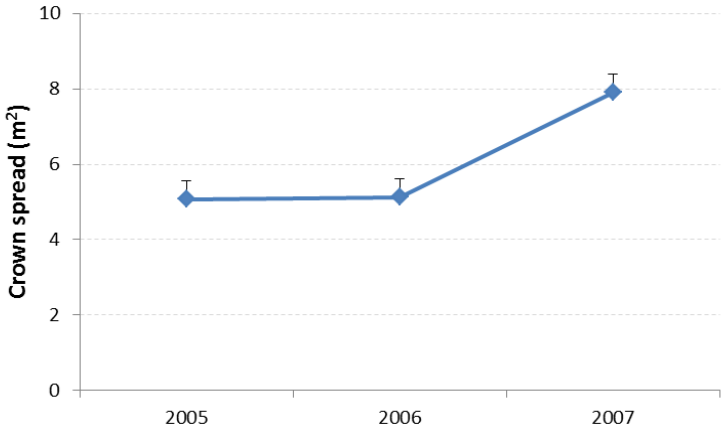


Figure 3.16: Canopy spread increment (average of 19 species) over three years (l.s.d. 0.489)

Canopy spread also varied significantly (P<0.001) between species (Fig. 3.17). Fast growing species such as bamboo, hibiscus, *Eucalyptus* sp. and *Casuarina* sp. closed their canopies well within the first three years, reflecting their dominance over the medium and slow growing species. Overall, *C. cunninghamiana* to *H. tiliaceus* (Fig. 3.17) closed canopy in 3 years with better growth rates, making them water thirsty species. These types of tree species will be suitable for phytocapping as they have the potential to transpire more water than the slow growing species. These trees are also likely to intercept more water during rain events, provided they can also tolerate drought via controlling transpiration. Canopy spread will increase over time with the height increments.

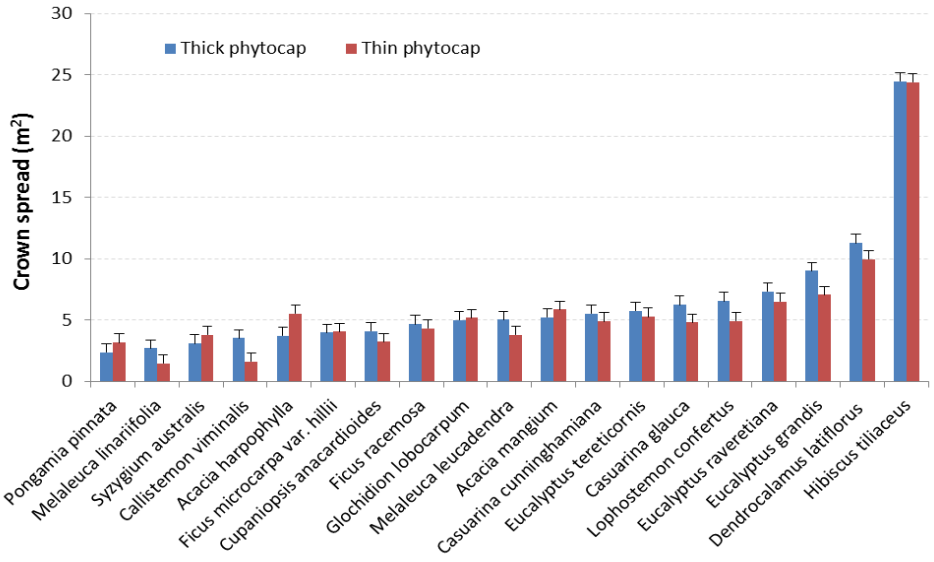


Figure 3.17: Canopy spread increments in 19 different species over 3 years (l.s.d. 2.13)

3.4.2.5 Shoot Biomass

Biomass accumulation in shoots correlates well with water uptake (Therakan *et al.* 2000, Singh and Bhati 2003). Biomass in this study was separated into shoots and roots, wherein the shoots were further separated into stems, branches, leaves and litter, and the roots into coarse and woody roots.

Shoot biomass in 2006 and 2007 significantly (P<0.001) differed between species (Table 3.1). Figure 3.18 shows the shoot biomass of 19 species grown in the phytocapping systems after 3.5 years of planting (2007). This can be attributed to variations in tree morphology and climatic conditions; growth rates (height, stem diameter and canopy spread); ability to adapt to local conditions such as drought, salinity, landfill gases and soil temperature. It is also interesting to note that the biomass produced by the trees grown in this phytocapping site comprised of 80% above ground biomass and 20% below ground biomass; similar to that reported by Swaby *et al.* (2004) (83% in trees as above ground and 17% as below ground biomass).

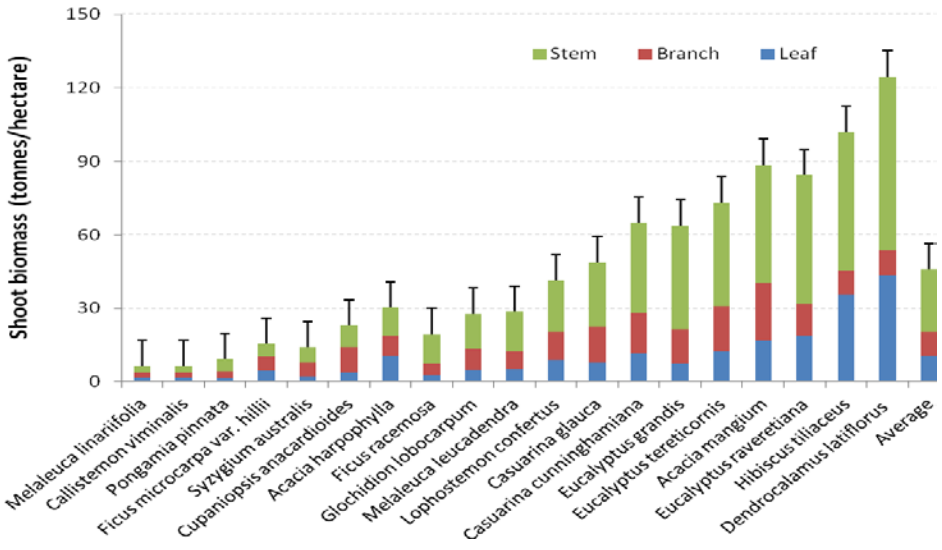


Figure 3.18: Shoot biomass of 19 species grown in Thick and Thin phytocap after 3.5 years of planting (L.s.d. for total shoot biomass 10.53). Values are means of thick and thin phytocaps.

Shoot biomass also varied significantly (P<0.001) between Thick and Thin phytocaps (Fig. 3.19, Table 3.1). Trees grown in the thick soil cover accumulated more biomass due to higher moisture availability and better rooting depth (Fig. 3.26). *Hibiscus tiliaceus* and *D. latiflorus* produced more than 100 t ha^{-1} of shoot biomass within 3.5 years, with most species accumulating 30 to 90 t ha^{-1} in 3.5 years (Fig. 3.18). *Melaleuca linariifolia*, *C. viminalis* and *P. pinnata* accumulated less than 10 t ha^{-1} biomass in 3.5 years (Fig. 3.18).

The biomass produced by *E. grandis* (63 t ha^{-1}) in this study (Fig. 3.18) corresponds well with that reported for an effluent-irrigated plantation in Yeppoon, Queensland (65 t ha^{-1}; Sharma 2008). The current results demonstrate that most species established on the Thick and Thin phytocaps were growing at their maximum potential despite the environmental conditions encountered on the landfill. Biomass accumulated by these trees can be used for carbon sequestration (Fang *et al.* 2007) and/or bioenergy production (Marland and Schlamadinger 1997), under the Mandatory Renewable Energy Targets (MRET) (RIRDC 2010) and the newly released Carbon Farming Initiative (CFI) (http://www.climatechange.gov.au/cfi).

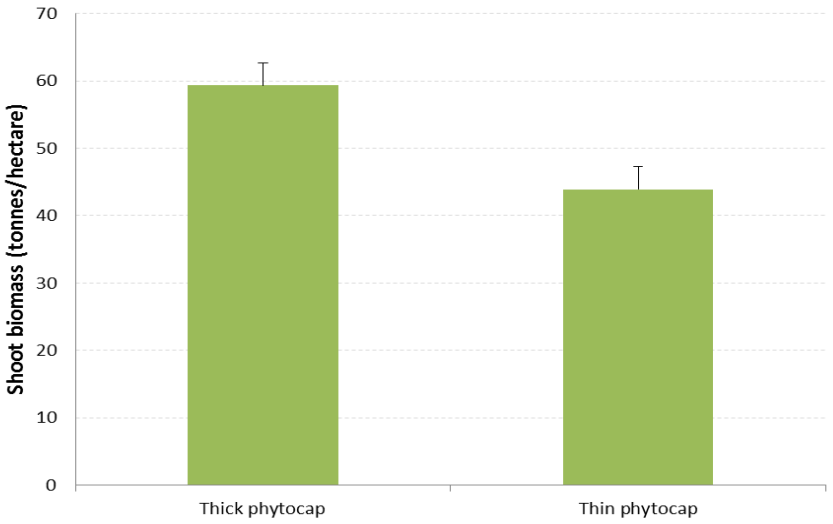


Figure 3.19: Shoot biomass of 19 species in Thick and Thin phytocaps (l.s.d. 3.417)

The biomass accumulation in stem in 2007 differed significantly (P<0.001) between Thick and Thin caps (Table 3.1), as phytocaps differed in their soil composition, depth and possibly in compaction regime (Thick cap was more compacted than Thin capping, due to the vehicular traffic in spreading the clay and top soil). Stem biomass also varied significantly (P<0.001) (Table 3.1) between species (Fig 3.21, Table 3.1). Plants may respond differently to biomass accumulation in stems, and variation between the two soil covers can also induce further changes in their stem biomass accumulation. However, the lack of cap*species interaction suggested that all species responded similarly in this case. With time, the competition for water and nutrients will increase and it is expected that each species will respond differently to soil covers and hence they are likely to show interactions with the capping.

Plants grown in the Thick phytocap accumulated significantly higher quantities of stem biomass (30 t ha^{-1}) than those growing in the Thin phytocap (22 t ha^{-1}) (Fig. 3.21) and may be attributed to increased availability of water and nutrients.

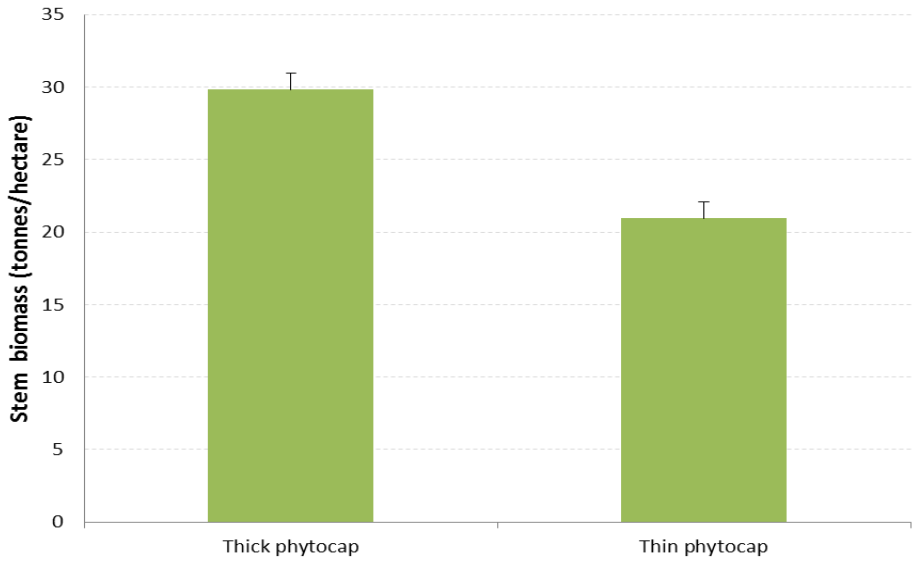


Figure 3.20: Stem biomass of 19 trees on Thick and Thin phytocaps after 3.5 years of planting (l.s.d. 1.151)

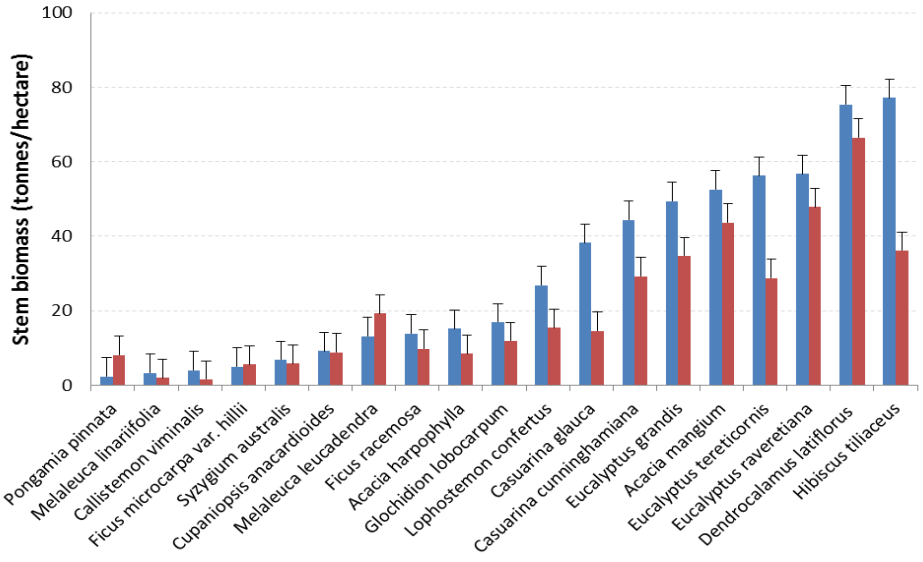


Figure 3.21: Stem biomass of 19 species in Thick and Thin phytocaps after 3.5 years of planting (L.s.d. for capping and species interaction 5.015)

Leaf biomass significantly (P<0.001) (Table 3.1) differed between species (Fig. 3.22), but not between the two capping systems. Leaf biomass was the highest in *D. latiflorus* and *H. tiliaceus*, as they grew fast (43 t ha^{-1} and 35 t ha^{-1} respectively) and produced large numbers of leaves compared to other species having the same height. *Pongamia pinnata, M. linariifolia* and *C. viminalis* had low leaf biomass (1.56 t ha^{-1}, 1.65 t ha^{-1}

and 1.8 t ha^{-1} respectively). This is due to slow growth rates and smaller leaves. In the case of *P. pinnata*, leaves senesced every year.

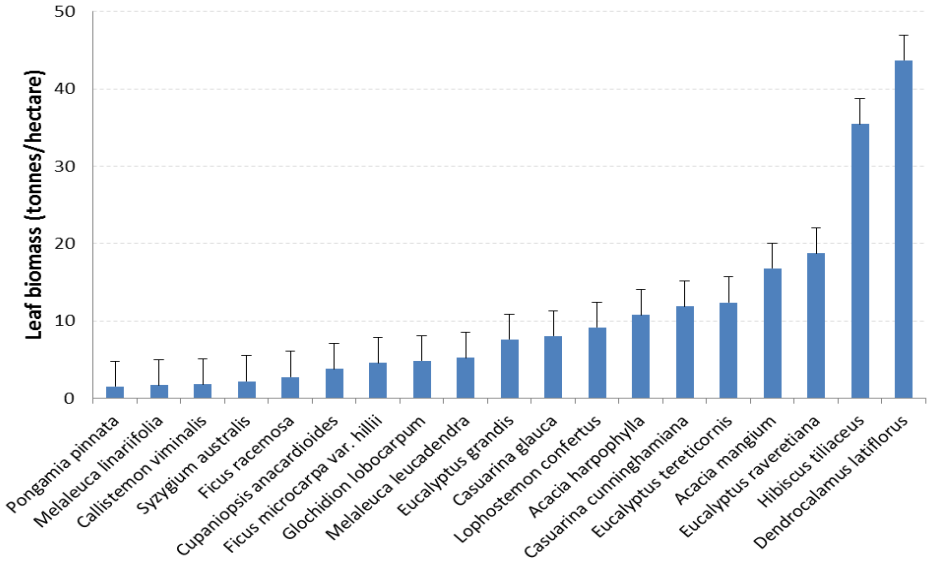


Figure 3.22: Leaf biomass of 19 species grown on a phytocap
(Bars represent l.s.d. 3.27) (Average of Thick and Thin phytocaps)

After 3.5 years (in 2007), *D latiflorus* and *H. tiliaceus* produced higher quantities of leaf litter compared to the other species (Fig. 3.23). Trees grown in the Thin phytocap produced more leaf litter than those established in the Thick phytocap (Fig. 3.23) and this was associated with lesser availability of water in the Thin phytocap during dry seasons whereas, some species hardly dropped any leaves (e.g. *M. linariifolia*). *Dendrocalamus latiflorus* and *A. mangium* senesced large number of leaves contributing to form a thick layer on the floor (up to 10 cm) (Fig. 3.23). This in turn will enhance leaf litter interception and also increase mobilisation and dispersion of nutrients into the environment.

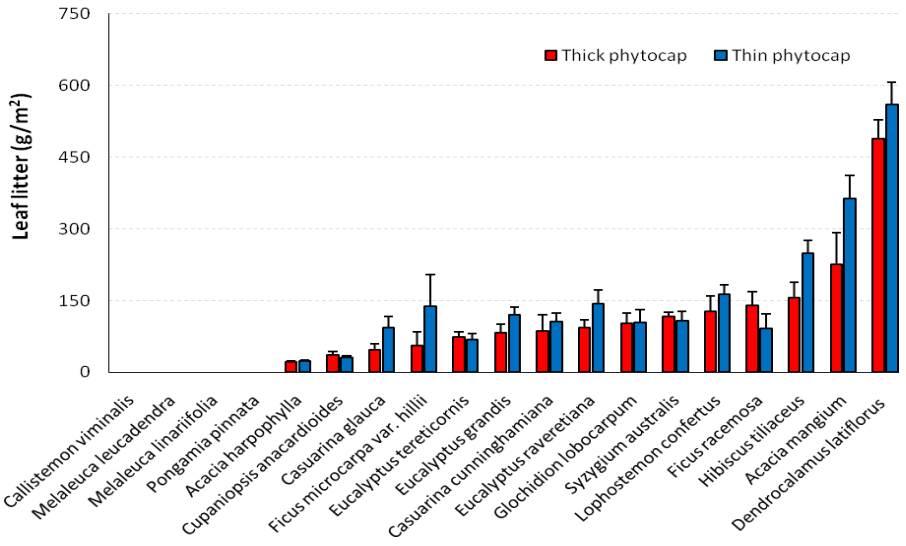


Figure 3.23: Leaf litter weight of 19 species grown in Thick and Thin phytocaps after 3.5 years of planting. Bars represent SE (n =3)

The leaf drop was moderate in *H. tiliaceus* (dropped 100 to 150 g m^{-2} per year) and in many other species. Within their growth periods, the established plants were exposed to three dry seasons and this would have induced them to respond as they will on natural circumstances. *Dendrocalamus latiflorus*, *A. mangium* and *H. tiliaceus* responded to dry conditions by dropping their leaves. *Hibiscus tiliaceus* also showed wilting symptoms to cut down water use to a bare minimum (300 ml/tree/day vs 10,000 ml/tree/day in wet season; see Fig 5.13). *Acacia mangium* and *D latiflorus* coped well with the dry conditions by dropping large numbers of leaves in comparison with *H. tiliaceus* which managed to survive drought conditions by wilting and drooping leaves.

When choosing plants for phytocaps in arid and semi-arid regions, emphasis should be placed on selecting species that can close down stomata during dry periods and grow fast during the wet season, so that they can survive during critical dry periods and take up large quantities of water during the rainy season. In this case, *H. tiliaceus* seems to meet this requirement.

3.4.2.6 Root Biomass and its Distribution

All species except bamboo exhibited a well-developed tap root system and their highest root density was concentrated in the top 40 cm of the soil. These roots formed matting

on the surface layer. Root biomass differed significantly (P<0.001) (Table 3.1) between species and caps (P<0.018) (Fig. 3.24, Table 3.1). *Dendrocalamus latiflorus* differed markedly from other species, as it accumulated a large quantity of woody roots (Fig 3.24).

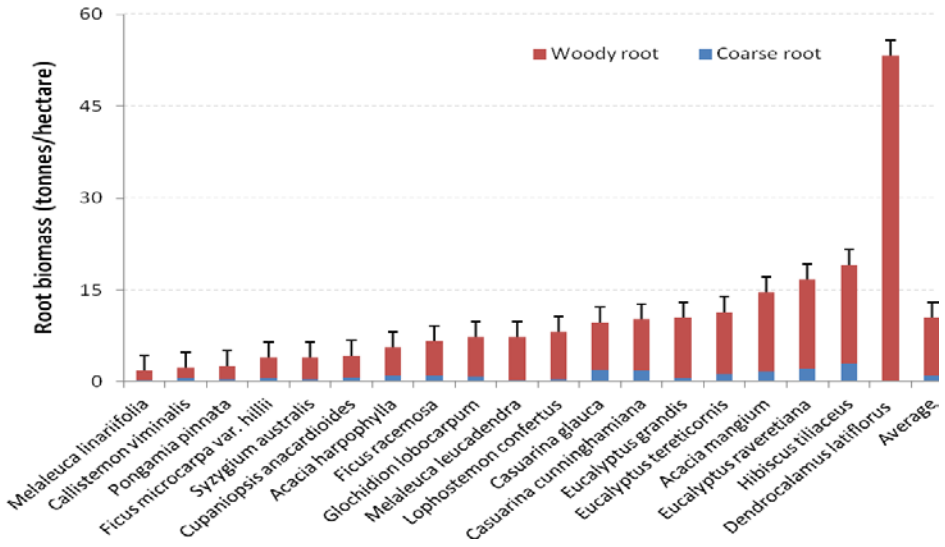


Figure 3.24: Root biomass of 19 species grown on a Thick and Thin phytocap
(Bars represent l.s.d. 2.5)

Lateral roots are the structural elements and they also provide anchorage to the tree. These roots showed a tendency to spread horizontally, perhaps to avoid high soil temperature experienced at the soil-waste junction. The lateral spread of roots exceeded the canopy spread in many species and this has been observed by various scientists in the past (Toky and Bisht 1992)

At harvest, the lateral root distribution was observed and the root distribution was quantified by measuring the lateral root length from the base of the stem. Figure 3.25 shows differences between species in the root spread, with the fast growing species showing greater spread than those of slow growing species. The exception to this was *A. harpophylla* which did not accumulate large quantities of biomass (Fig 3.18) but still had similar root spread as the fast growing species. This species is known to produce extensive root systems in clayey soils. In the current study, trees had an average lateral

root extension of 1.8 m with a root depth of 700 mm in the Thick phytocap and 500 mm in the Thin phytocap (Fig. 3.26). *Hibiscus tiliaceus* had the longest lateral root growth of 3.8 m (Fig. 3.25). This species is renowned for producing extensive root system and hence its root distribution exceeded the spacing provided (1 m on either side). As a result, this species had begun to explore soils from adjacent areas and hence competing with other species for water and nutrients.

Generally, the minimum soil temperature for root growth is about 5°C and the maximum is about 40°C (Waisel *et al.* 1991). As the landfill soil temperature could go over 40°C, this elevated temperature might have had some effect on root length and diameter. This has been ascertained by Mackay and Barbar (1984).

Irrigation in any form discourages root penetration deep into the soil (DPI & F 2011). In this study, plants established in the Thick phytocap had a tendency to have a larger root spread than those grown on the Thin phytocap, particularly for the fast growing species. By extrapolating this trend, it can be concluded that the root spread is likely to be much larger in Thick phytocaps as the trees mature, and also increasing their demand for water and nutrients.

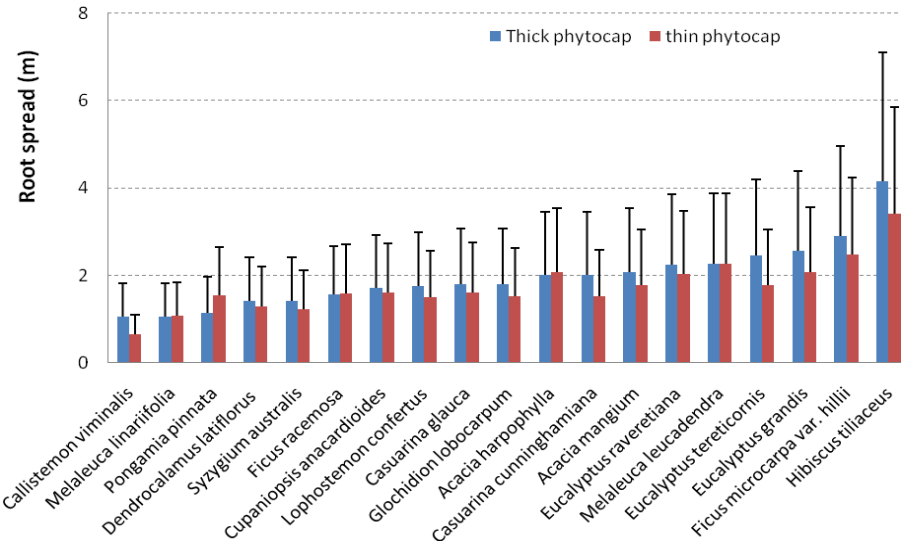


Figure 3.25: Lateral root spread in 19 species in Thick and Thin phytocaps after 3.5 years. Bars represent standard errors (n=2)

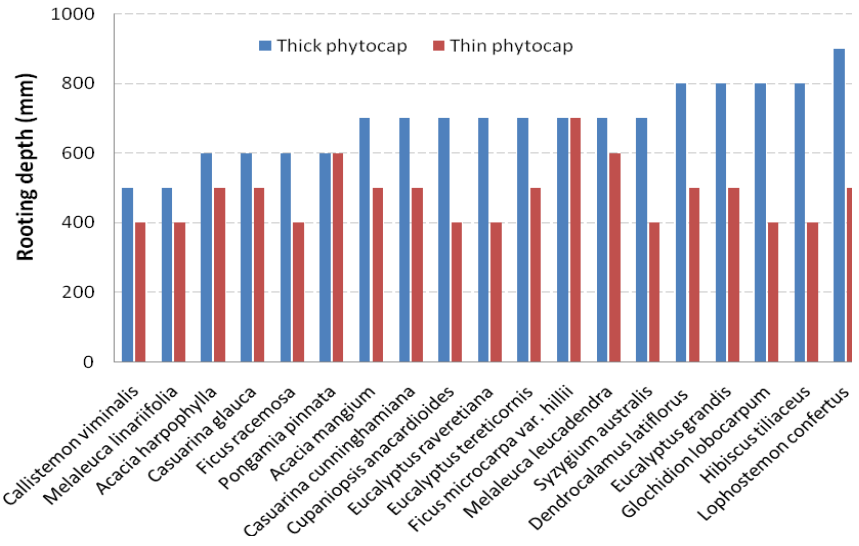


Figure 3.26: Root depth in 19 species in Thick and Thin phytocaps after 3.5 years of planting

Root distribution through space and time is usually influenced by both genetic characters of the plant species and the associated soil conditions (Huck 1983). In this study, roots of many species penetrated to a depth of 700 mm in the Thick soil cover and 500 mm in the Thin soil cover (Fig. 3.26). The deepest rooting was observed in *L. confertus* and *F. microcarpa* and the shallowest rooting was observed in *C. viminalis*, *A. harpophylla* and *M. linariifolia* (Fig. 3.26). It is expected that the species having deep roots and more spreading lateral roots will take up nutrients and water more efficiently from deeper layers, and over a wider area. The deep roots also provide firm anchorage for the tree in the soil, thereby making the tree wind-firm. Fast growing species with deep main root system and moderate lateral root length are suited for climates having wet and dry cycles.

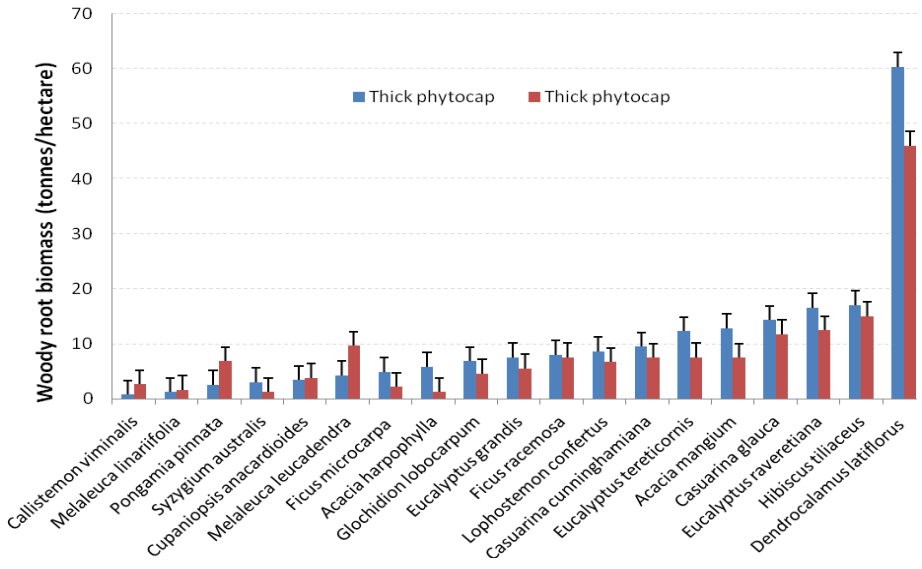


Figure 3.27: Woody root biomass of 19 species in Thick and Thin phytocaps (Bars represent l.s.d. 2.56)

The root system of a species is often guided by local climatic conditions (Das and Chaturvedi 2006). Between the species, there was a significant variation (P<0.001) in the biomass accumulated in woody and coarse roots. Woody root biomass was highest in *D. latiflorus* (Fig. 3.27). Woody roots were higher in the thick capping system (Fig. 3.27) and this was due to thicker soil profile, enabling better root development and distribution. This is also evident from the rooting depth of each species (Fig. 3.26).

The coarse root biomass varied significantly (P<0.001) between species (Table 3.1, Fig. 3.28). *Hibiscus tiliaceus* (3 t ha^{-1}) and *E. raveretiana* (2.2 t ha^{-1}) accumulated higher quantities of coarse roots (Fig. 3.28), with an overall average of 0.97 t ha^{-1}. Course roots are primarily responsible for water uptake (Poore and Fries 1987) and they explore large volumes of soil with a wide range of soil moisture content (Phillips and Riha 1994). This will also help plants to cope with various water regimes in the phytocaps.

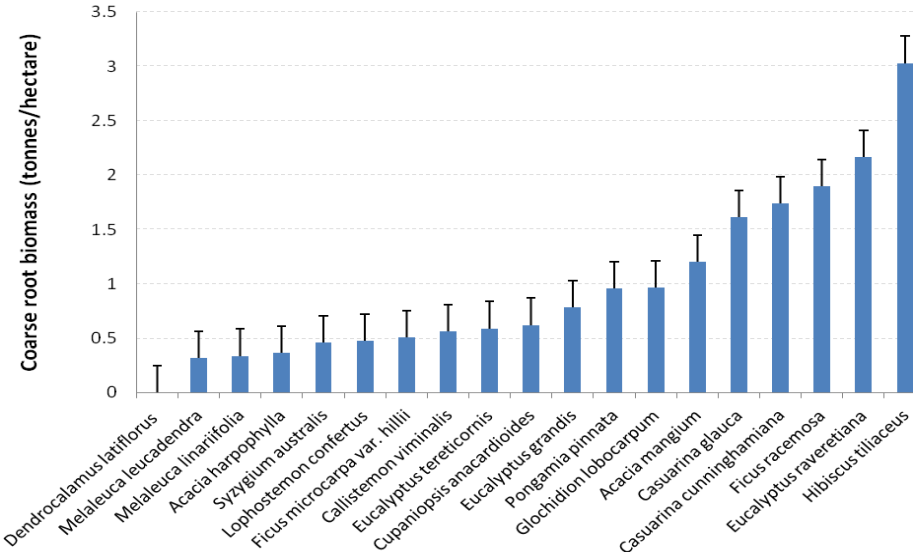


Figure 3.28: Coarse root biomass of 19 species in Thick and Thin phytocaps
(Bars represent l.s.d. 0.246)

Root and stem fractions were determined and analysed to see if the soil thickness had any influence on the root and stem growth. Root and stem fractions varied significantly P=0.009 and P=0.013 respectively, Fig. 3.29 and Table 3.1) between species. This variation may be attributed to initial growth rates in different species, which may change as the trees mature. However the root fraction did not differ between the two capping systems (P=0.687).This may change as the trees mature. After 3.5 years of growth, in the Thick and Thin phytocaps, plants showed major differences in their root and shoot fractions. It will be rather interesting to explore these criteria further, which may help optimise soil thickness required for a sustainable phytocap.

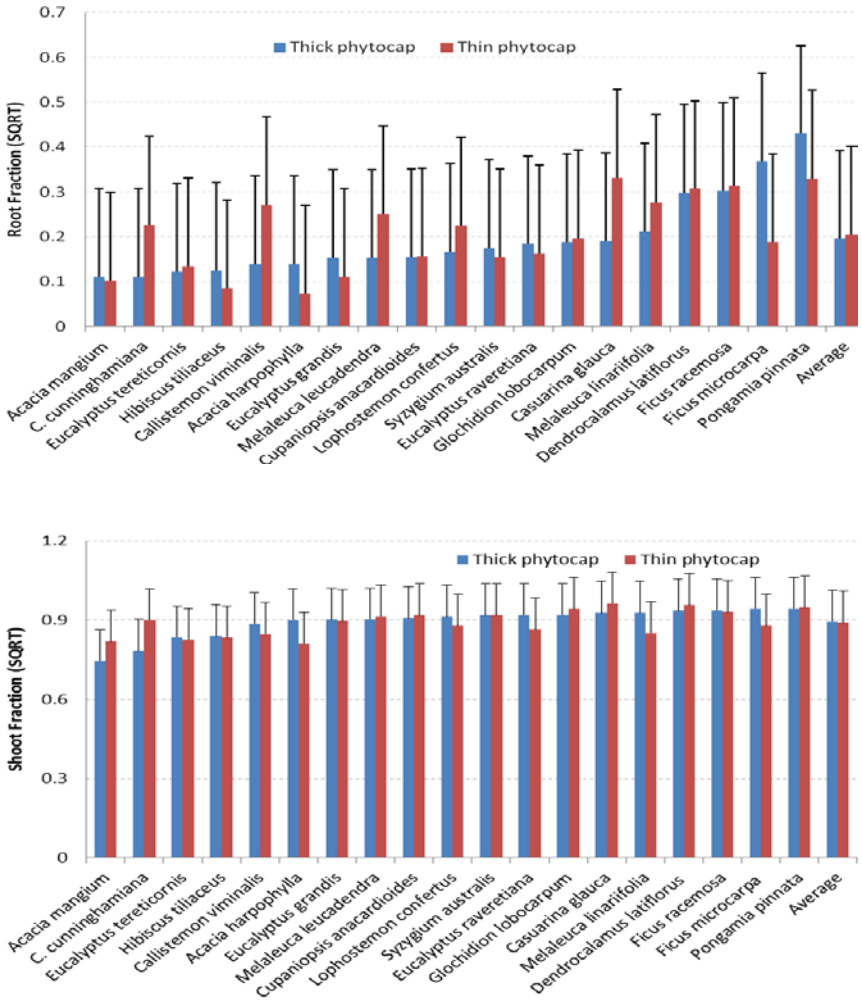


Figure 3.29: Root (top) and Shoot (bottom) fractions of 19 species in Thick and Thin phytocaps (Bars represent l.s.d. 0.197 & 0.119 for root and shoot fractions)

The root length density (RLD) is an important parameter that includes water and nutrient uptake by the plants (Cornelissen *et al.* 2003, Zuo *et al.* 2004). RLD varied significantly ($P<0.001$) between soil depths and species ($P=0.018$) (Fig. 2.30; Table 3.1). RLD was highest in the top 20 cm of the soil profile (Fig. 3.31). In most species RLD was highest in the top 30 cm of the soil profile (Fig. 3.31). Fast growing species such as *A. mangium*, *D. latiflorus*, *H. tiliaceus* and *E. grandis* had higher RLD compared to other species. This may be correlated to their increased water uptake capacity (Sharratt and Gesch 2004), row spacing (Sharratt and Gesch 2004) and faster growth due to high nutrient supply (Ryser and Lamber 1994, Ryser 1996). Studies have

shown that increased RLD can contribute to reduced nutrient losses (Lambers and Pooter 1992).

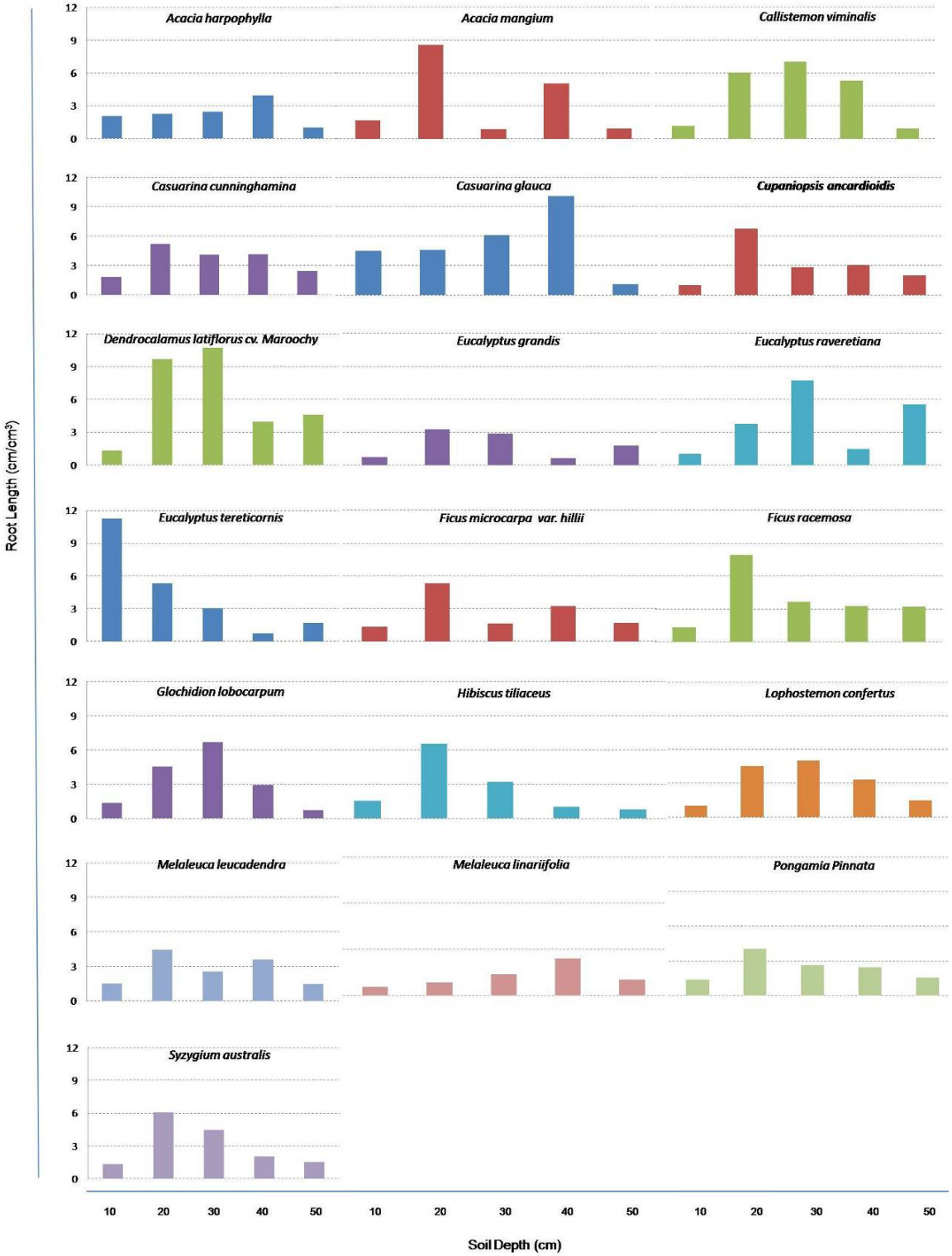


Figure 3.30: Root length density in 19 species after 3.5 years of plantation
Note: P <0.001

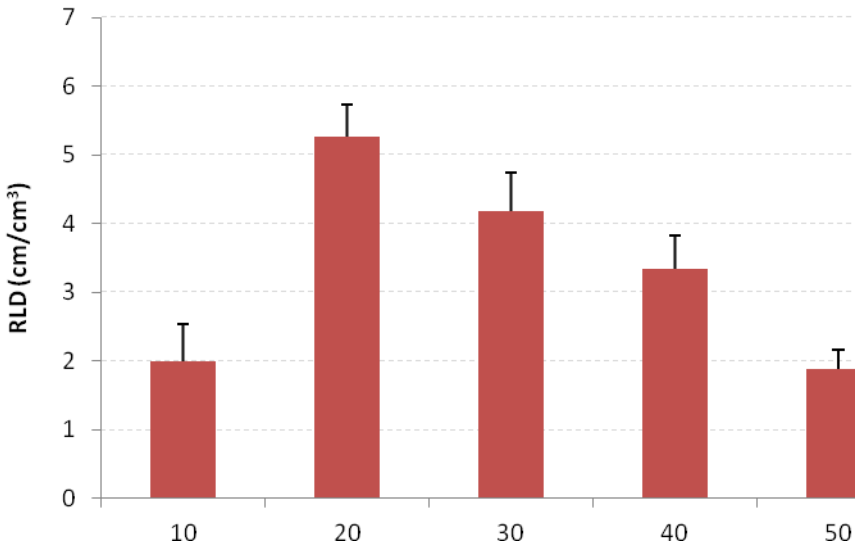


Figure 3.31: Depth at which RLD was concentrated
(Bars represent SE;n=19)

Overall, roots of eucalypts, hibiscus and casuarinas have the tendency to grow rapidly and display opportunistic growth with respect to water (Dabral *et al.* 1998). Roots of acacias can take up and exude between 75 to 225 L (Ludwig *et al.* 2004). In the dry season the roots play a vital role in sending a signal to the canopy to direct the leaves to reduce canopy conductance (Gollan *et al.* 1986, Davies and Zhang 1991). This property helps balance the hydrology of the phytocaps by sustaining survival of plants during dry seasons. Many species such as acacias have the ability to mobilise water from wet areas of the soil to dry areas of soil (Caldwell *et al.* 1998). This characteristic is extremely important for phytocaps as the distribution of water in phytocaps in not uniform.

3.4.2.7 Correlations

Height growth was correlated with various other parameters. Based on the criteria (>0.5 = strongly correlated, 0.5 – 0.2 = moderately correlated and <0.2 = weakly correlated), tree height showed a strong correlation with DBH, LAI and D_{50} and weak correlation with canopy spread (Table 3.2). Studies in the past have reported strong relationships between heights and stem diameter, and the current observations suggest that tall species have stronger and thicker stems (Rich *et al.* 1986). Tree height also correlated well with the DBH and D_{50} in both Thick and Thin phytocaps (Table 3.2). Height also strongly correlated with LAI in Thin phytocap. Biomass, root depth and root length also

influenced tree height at moderate levels due to supply of adequate nutrient levels in the soil (Witkowski and Lamont 1991). However, canopy spread, root depth, root biomass and root length did not show a high correlation with the height (Table 3.2). This may change as the trees mature and the competition for water and nutrients increase.

Table 3.2: Correlation (r^2) between tree height and various growth parameters for 19 species grown in Thick and Thin phytocaps

Parameter	Thick phytocap	Thin phytocap	Combined
DBH	0.67***	0.68***	0.65***
D_{50}	0.57***	0.65***	0.61***
Total biomass	0.36**	0.37**	0.37***
Root biomass	0.20^{ns}	0.17^{ns}	0.19**
Root length	0.15^{ns}	0.18^{ns}	0.17**
Rooting depth	0.28*	0.014^{ns}	0.10*
LAI	0.47**	0.57***	0.51***
Canopy spread	0.21*	0.22*	0.21**

Note: ***; P <0.001, **; P <0.01 and *; P <0.05 probability

Trees showed gradual increments in height since the establishment, and the height will tend to increase rapidly during their maturity (Brack 1997). Consequently, it is likely that the differences in growth between the two soil covers are likely to increase as the trees mature.

D_{50} is a new parameter used to measure tree growth (Sharma 2008). D_{50} showed strong correlation with height, AI and DBH (Table 3.3). Similarly, DBH showed a linear correlation with height and D_{50} (Table 3.3) .O'Brien *et al.* (1995) have also reported high correlation between the above parameters. Overall, from Table 3.3, height, D_{50} and LAI showed a very good correlation with DBH, while total biomass and canopy spread showed a weak correlation with DBH. Interestingly also root biomass showed very weak correlation with the DBH and D_{50}.

Table 3.3: DBH and D_{50} versus various tree components
(Thick and Thin phytocaps combined)

Parameter	DBH	D_{50}
Height	0.65***	0.61***
D_{50}	0.83***	N/A
DBH	N/A	0.83***
Total biomass	0.3**	0.30**
Root biomass	0.03^{ns}	0.08^{ns}
LAI	0.35**	0.40***
Canopy spread	0.15*	0.20**

Note: ***; P <0.001, **; P <0.01 and *; P <0.05

Trees grown on phytocaps have shown high gradual increments in stem diameter since establishment and this will increase with age, thus contributing to transpiration rates (Eamus *et al.* 2006). Consequently, it is possible that the trees in the Thick soil cover will develop thicker stem due to increased water and nutrient availability, and reduced exposure to high temperature.

3.5 Conclusions

There are several factors in a landfill that can affect tree growth and their performance. Studies in the past have shown that tree mortality and low growth rates in trees grown on landfills was due to high soil temperature, tree spacing, heavy

metals, landfill gases, soil compaction and waterlogging. Soil thickness has also been a factor for normal tree growth. However, trees in this study showed the expected growth rate (starting 2003) that is comparable with what is achievable in a normal soil, and they closed canopies within the first three years of establishment.

In this study, *Populus* sp. and *Salix* sp. did not survive due to high temperature encountered in Rockhampton. The other 19 species grew well and showed moderate to good growth rates as assessed by their height, stem diameter, canopy spread, LAI and biomass. Fast growing species grew quickly in the early stages, which is essential for the initial sustainability of the phytocapping system. However long term sustainability of these species in the landfill can only occur if those species area also able to cope with the drought and other conditions associated with the landfills. On an average the species grown in the phytocapping systems attained a LAI of 2.4 with the maximum achievable LAI of 4 to 8.

Shoot biomass varied significantly between Thick and Thin phytocaps after 3.5 years. Trees grown in the Thick phytocap accumulated higher biomass than those established in the Thin phytocap. Fast growing species produced more than 100 t h^{-1} of shoot biomass, but at the same time they could cope well during drought period by wilting or dropping leaves. Biomass produced by *E.grandis* was similar to that reported by Sharma (2008) for an effluent irrigated plantation in Yeppoon, Queensland. Leaf biomasss was high in bamboo and *H. tiliaceus* as they grew fast. Biomass accumulated by the species can be used for bioenergy production and carbon sequestration under the recently introduced Carbon Farming Initiative (Anon 2012). The biomass produced by the trees grown in this phytocapping system comprised of 80% shoot and 20% roots.

Dendrocalamus latiflorus and *A. mangium* produced high quatities of leaf litter biomass as they dropped significant amounts of leaves to survive and cope with high temperature and drought conditions in Rockhampton. Different trees possess different mechanisms to cope with drought and high temperatures. However, the leaf dropping mechanism in the larger trees such as *A. mangium* and *D. latiflorus* controls their transpiration rates by taking up maximum water during wet season and

by reducing their water uptake during drought. This characteristic has been displayed by *H. tiliaceus* and other fast growing species in this study.

All species except bamboo exhibited a well developed tap root system, with the maximum density of roots concentrated in the top 40 cm of the soil system. Greater distribution of roots in the top layer can be attributed to increased availability of oxygen in the top layers. It can also be due to higher temperatures and low oxygen experienced in the deeper layers of the phytocapping system. Roots tend to avoid high temperatures found at the soil-waste interface.

Irrigation discourages root penetration deep into the soil. However, *F. microcarpa* and *L. confertus* had the deepest roots. Species in this study had developed lateral roots and the lateral roots in some species grew longer than their canopy spread. Roots of *H. tiliaceus* grew 3.8 m laterally. It is interesting to note that *A. harpophylla* grew slowly with moderate root growth but accumulated large amounts of shoot biomass. There was no variation in root and stem fraction in both Thick and Thin phytocaps. This may change as trees mature. Root length density was higher in the top 20 cm of the soil layer and this may also change as trees mature.

Height correlated well with DBH and D_{50}, but it weakly correlated with the canopy spread. Similarly DBH and D_{50} showed good correlation with height and LAI but showed a weak correlation with canopy spread. There was no significant correlation between stem diameter and root biomass. Literature suggests that trees have grown well on landfills that had soil thickness of 600 mm. In this study, plants grown in the Thick soil cover (1400 mm) grew better than those established in Thin soil cover cover (700 mm). Based on this data, it can be concluded that a soil thickness of at least 1 m will be needed to support growth of a wide range of species in phytocaps..

Highly variable plant growth was observed in these phytocapping systems. Australian plants very considerably in their growth and adaption and hence are highly suited for phytocapping system as the conditions of phytocapping system also differ considerably. Thus success of establishing sustainable plant communities on phytocapping system relies upon careful selection of plant species, establishment of

a wider range of species and provision of suitable agronomic care during initial stages of establiment.

This Chapter discussed tree growth using various paramaters. The next Chapter presents changes in foliar chemical composition of plants established in Thick and Thin phytocaps.

4

Foliar Chemical Composition*

4.1 Introduction

All plants depend on mineral nutrients for survival, good health and growth. There are 18 essential plant nutrients of which 15 are absorbed from the soil and three, oxygen, carbon dioxide and hydrogen, are absorbed from air and water. Table 4.1 lists the essential nutrients required for plant growth, which are categorised into macronutrients and micronutrients (Barker and Pilbeam 2007). The first seven elements (Table 4.1) are classed as macronutrients. These are required in higher concentrations, in the order of >1000 mg kg^{-1} dry matter (Salisbury and Ross 1991). The last eight elements are micronutrients or trace elements that are required in lower concentrations in the order of <100 mg kg^{-1} dry matter (Salisbury and Ross 1991). All these nutrients are essential for plant growth (Hawkins and Sweet 1989) and transpiration (Tartachnyk and Blanke 2004); both macro and micro nutrients play a pivotal role in maintaining the hydrological balance of the phytocapping system.

Table 4.1: Essential mineral nutrients for plant growth

Class	Subclass	Elements
Macronutrients	*Primary nutrients*	N, P, K
	Secondary nutrients	S, Ca, Mg, Si,
Micronutrients		Fe, Mn, B, Zn, Cu, Mo, Cl, Ni

Source: Salisbury and Ross 1991

* Some data from this chapter have been included in the following papers:
Venkatraman, K. and Ashwath, N. (2009) Environmental performance of a phytocapped landfill, *The Environmental Engineer*, 10: 20-25.

Plants grown in landfills are affected by surface environmental conditions as well as the nutrient supply from the buried waste (Maurice 2005). Waste in a typical MSW constitutes more than 50% organics (ABS 2006) which are the major sources of nutrients for plants established on landfills. Organic wastes in Australian landfills predominantly contain food scraps, green waste, paper and cardboard (ABS 2006). The table below gives the nutrient composition of paper and pulp waste, which can be used as an indicator for MSW composition.

Table 4.2: Nutrient composition of paper and pulp ash

Element	mg kg^{-1}	Element	mg kg^{-1}
N	4520	B	95
P	3000	Zn	183
K	13,300	Cu	67
S	-	Mo	15
Ca	120000	Pb	72
Mg	7730	Ni	16
Si	-	Cr	75
Fe	6260	Co	14
Mn	2600	Cd	2

Not reported; Source: Muse and Mitchell 1995

Other than organic waste, landfills also contain heavy metals such as arsenic, boron, cadmium, chromium, cobalt, copper, iron, manganese, mercury, lead, nickel and zinc (Adefemi and Awokunmi 2009, Alker *et al.* 2003). Consequently, trees grown on these landfills will be exposed to the above chemicals (Gigliotti *et al.* 1996, Al-Khateeb and Leilah 2005) and may be released into the environment through the food chain (Cortet *et al.* 1999, Bruger 2002, Nahmani and Lavelle 2002, Pugh *et al.* 2002, Vandecasteele *et al.* 2003).

In general, nutrient and heavy metal uptake by plants are influenced by nutrient retention ability of the soil, nutrient demand of different species, growth rate, biomass distribution (Miller 1984), bio-availability of heavy metals (Greger 1999), organic matter content of the soil and soil temperature (Vitousek and Sanford 1986). Trees store most nutrients in the leaves (Kirschbaum *et al.* 1992). Similarly, trees take up heavy metals and store them in the leaves and branches (Fatoki 2000, Luyssaert *et al.* 2002, and Mertens *et al.* 2005) to protect themselves from insects

and fungi (Chaney *et al.* 1997). Nutrients and heavy metals that are taken up by trees are eventually distributed to the environment via litter fall (Jelaska *et al.* 2007, Friedland *et al.* 1983, Vitousek 1998). Nutrient removal through plant uptake and litter fall increases with foliar biomass production (Miller 1986, Miller 1989) and the rate of nutrient supply rate (Binkley and Vitousek 1987).

Ecosystems differ in nutrient supply rates due to variations in leaf litter decomposition rates, mineral weathering and other processes (Chapin *et al*. 1986). Studies show that the leaf litter decomposition rate is more rapid in nutrient rich sites than in nutrient poor sites (van Vuuren *et al.* 1993). A similar situation exists in landfills where the nutrient status of the soil is influenced by composition of the waste, decomposition rates of the waste and the availability of minerals. However, nutrient and heavy metal availability may vary from one landfill to another and also within landfills (Fitter 1994). Nutrient levels and heavy metal concentrations of the plants grown on phytocaps were assessed with the view to confirming if the established plants were healthy, and also to test if the same plants accumulate unusual levels of heavy metals that could adversely impact on the environment.

Foliar chemical analysis is a good method to assess plant nutritional stress (Lichtenthaler 1996) and heavy metal concentration (Pugh *et al.* 2002); both of which are indicators of processes occurring at the ecosystem level (Duquesnay *et al.* 2000). Mineral nutrients are essential for plant growth (Salisbury and Ross 1991). However, deficiencies in N, P or K mostly occur in mature leaves (Reuter and Robinson 1997) as these nutrients are translocated from old to young leaves over time (Hill *et al.* 1979). Differences in nutrient mobilisation may reflect greater internal requirements in young versus old leaves. Pastor and Post (1986) reported that over a period of time plants will affect nutrient availability by producing organic litter of varying chemical and physical properties which may have adverse impact on tree growth. Hence, considering the complex nature of the nutrients, their availability, translocation within plants and within an ecosystem, it is important to evaluate nutrient status of foliage and leaf litter on a phytocapped landfill site.

Plants require heavy metals such as zinc, copper, manganese and iron in trace amounts to grow (Marschner 1995). However, excessive uptake by plants may cause

serious health problems to plants and micro and macro fauna (Al-Khateeb and Leilah 2005, Vartanian *et al.* 1999). Most landfill soils contain elevated levels of heavy metals (Adefemi and Awokunmi 2009), which may be released into the environment via trees (Grant *et al.* 2002). Leaves are a good indicator of heavy metal concentrations in the root-zone and soil (Cox and Hutchinson 1979) and hence the foliage of species grown in the phytocapping system was assessed for their heavy metal concentrations.

Several researchers have shown great concern about the flow of heavy metals into the environment through litter fall and/or the food chain. There have been concerns about lead concentrations in landfill soil because lead is toxic even at low concentration (Haggins and Burns 1975). Scrap tyres and mechanical parts of vehicles found in many MSWs are a good source of zinc, cadmium, nickel and chromium (Evans et al. 1980, Adefemi and Awokumi 2009). Adefemi and Awokumi (2009) also reported the presence of arsenic, chromium and copper associated with waste from sludge incineration and fly ash. Heavy metals released into the environment have an adverse impact on macro-fauna such as caterpillars, earthworms, beetles, birds (Cortet *et al.* 2000, Bruger 2002, Nahmani and Lavelle 2002, Vandecasteele *et al.* 2003) and plants as they affect photosynthesis (Greger 1999) which subsequently affect growth rate of plants (Lagriffoul *et al.* 1998). This effect will vary between species (Landberg and Greger 1994) as photosynthesis reduction is dependent on canopy class, stand management, canopy dimensions, infections and seasonality (Luyssaert *et al.* 2002). However, studies in the past have reported low toxicity symptoms by trees (Riddle-Black 1993) suggesting their use of enhanced tolerance mechanisms by evolving ecotypes that help gain more tolerance to heavy metals in order to survive under harsh conditions (Kahle 1993).

The aim of this study was to assess the health of plants grown in a phytocapping system by examining heavy metal uptake and their release into the ecosystem via litter fall.

4.2 Materials and Methods

Detailed foliar chemical analysis was undertaken to determine nutrient and heavy metal composition of 19 species grown on Thick and Thin phytocapping systems. Foliar analysis was conducted twice during this study; once in 2005 and then in 2006. In the first instance, the youngest fully expanded leaves were analysed for nutrients and heavy metals. Then, in the second instance mature, young and the youngest fully expanded leaves were analysed for nutrients and heavy metals. Foliar chemical analysis was also conducted on leaf litter from the 3 year-old trees.

4.2.1 Youngest Fully Expanded Leaf (2005)

The youngest fully expanded leaves were collected from 9 plants per species per plot in the trial. Fifty to sixty such leaves were collected randomly from the 2 year-old trees and placed in labelled plastic bags which were placed in on ice in an insulated storage container. To ensure removal of dust from the leaves, the samples were washed subsequently in a series of four buckets of distilled water. Once washed, the samples were blot dried and then oven dried at 70°C for up to 96 hours until they attained a constant dry weight. Once completely dried, the leaf samples were ground to <600 µm using the Mikro-Feinmuhle-Culatti (MFC) grinder. The finely ground samples were then placed in polycarbonate tubes, labelled and sent for chemical analysis at WESFARMERs CSBP LTD, Perth Western Australia. See Appendix B for protocols used to analyse different elements. The foliage nutrient concentrations of these samples were compared with the standard nutrient concentrations reported by Herbert and Schonau (1989), Drechsel and Zech (1991) and Reuter and Robinson (1997), with the view to detecting whether the observed concentrations were low, adequate or excessive for plant growth.

4.2.2 Mature, Young and Youngest Fully Expanded Leaves (2006)

A mixture of mature, young and the youngest fully expanded leaves were sampled from 9 plants per species per plot. In addition, 50 to 60 leaves were randomly collected from the top, bottom and middle layers of the canopy of the 3 year-old trees. A similar procedure was followed as described in section 4.2.1.

4.2.3 Leaf Litter

A 50 cm x 50 cm quadrat was used for leaf litter sample collection. Senescing leaves that were about to fall from the plants were also collected during this process. Leaves were collected in the 2 & 3 year-old plantation. The quadrat was thrown randomly between stands of 9 plants in Thick and Thin phytocaps and in both replications and leaf litter samples were collected within those randomly selected quadrats. Un-decomposed leaf litter was collected from three quadrats per species in each replication. The leaf litter was washed free of dust as per live leaves (Chapter 3), dried, ground and sent to WESFARMERS CSBP LTD, Perth Western Australia for chemical analysis.

4.3 Statistical Analysis

Mineral composition data was statistically tested for outliers, normality and homogeneity of error variances before being subjected to analysis of variance (ANOVA) using Genstat ver. 13 (Payne 1997, Wass 2011). The effects of soil thickness, species and the interactions between soil thickness and species were tested. The effects of time were also tested for the leaf parameters that were measured repeatedly. Least significance differences (l.s.d) are presented where the treatment, capping, species, time or their interactions were significant ($P<0.05$). Standard errors are provided where there were insufficient data available for ANOVA or when the F test was found not significant ($P<0.05$).

4.4 Results and Discussion

Results from the nutrient analysis were compared to the data of by Herbert and Schonau (1989), Drechsel and Zech (1991) and Reuter and Robinson (1997) (Table 4.3) for optimum nutrient concentration. Similarly, results from the heavy metal analysis were compared with the heavy metal concentrations of soils/plants (Gupta and Lipsett 1981, Brady 1984, Xue *et al.* 2001, Vandecasteele *et al*. 2002, Shankar *et al.* 2004, Molina *et al.* 2006, Reuter and Robinson 1997) (Table 4.4). Foliar and leaf litter compositions were used to determine variability in the performance of each species over two soil thicknesses and over time. Results from ANOVA are presented in Table 4.5.

Table 4.3: Optimum nutrient concentrations in plants

Element	Optimum concentration	Unit	Reference
N	1.48–3.0	%	Herbert and Schonau (1989), Drechsel and Zech (1991)
P	0.1–0.5	%	Reuter and Robinson (1997)
K	0.75	%	Herbert and Schonau (1989), Drechsel and Zech (1991)
S	0.20	%	Herbert and Schonau (1989), Drechsel and Zech (1991)
Cl	0.273	%	Reuter and Robinson (1997)
Ca	1.60	%	Herbert and Schonau (1989), Drechsel and Zech (1991)
Mg	0.3	%	Herbert and Schonau (1989), Drechsel and Zech (1991)
Na	0.3–0.42	%	Reuter and Robinson (1997)
Al	160	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)
Cu	12	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)
Zn	18	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)
Mn	600*	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)
Fe	110	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)
B	17	mg kg^{-1}	Herbert and Schonau (1989), Drechsel and Zech (1991)

Note: Concentration of Mn is for tropical species with a range from 28 to 2257 mg kg^{-1}, with most species containing 30 to 500 mg kg^{-1} (Drechsel and Zech 1991)

Table 4.4: Baseline heavy metal concentrations in soils and plants

Elements	Plant/soil	mg kg^{-1}	Reference
As	Soil	7.2	Brady (1984)
Pb	Soil	19	Brady (1984)
Ni	Soil	19	Brady (1984)
Cr	Plant	18	Shankar *et al.* (2004)
Co	Plant	2.75	Reuter and Robinson (1997)
Cd	Soil/Plant	0.35–0.40	Brady (1984), Vandecasteele *et al*. (2002)
Se	Soil	1	Xue *et al.* 2001
Mo	Plant	1	Gupta and Lipsett (1981)
Hg	Plant	0.16	Molina *et al.* 2006

Table 4.5: ANOVA for leaf and litter nutrient and heavy metal compositions (2005 & 2006)

Parameter	ANOVA	d.f.	Significance (*P*)	Parameter	ANOVA	d.f.	Significance (*P*)
Foliar (nutrients)				Foliar (heavy metals)			
	Cap	1	<0.001		Cap	1	<0.001
	Species	18	<0.001		Species	18	<0.001
	Year	1	<0.001		Year	1	0.43
	Cap.Species	18	0.05		Cap.Species	18	<0.001
	Cap.Year	1	0.08		Cap.Year	1	0.54
	Species.Year	18	<0.001		Species.Year	18	1
	Cap.Species.Year	18	0.147		Cap.Species.Year	18	0.999
Litter * (nutrients)				Litter * (heavy metals)			
	Cap	1	0.256		Cap	1	0.38
	Species	13	<0.001		Species	12	<0.001
	Cap.Species	13	0.372		Year	1	1
					Cap.Species	12	0.777
					Cap.Year	1	0.21
					Species.Year	12	1
					Cap.Species.Year	12	0.136

*nutrient (N, P, K, S, Na, Ca, Mg, Cu, Zn, Mn, Fe, B) and heavy metal (Cr, Co, Ni, As, Se, Mo, Cd, Hg, Pb) analysis was conducted in species that had significant quantity of litter in all plots/replications.

4.4.1 Foliar and Leaf Litter Nutrient Composition

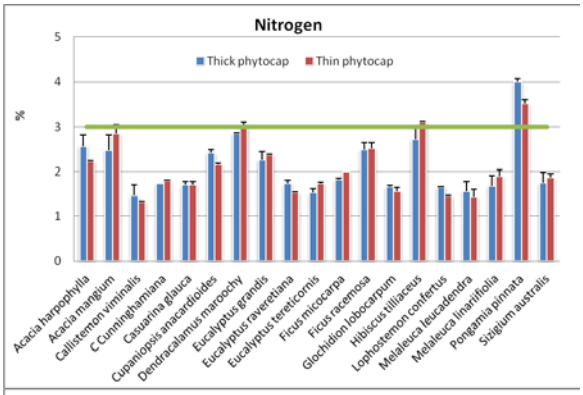
4.4.1.1 Foliar Nutrient Composition in 2 Year-Old Trees

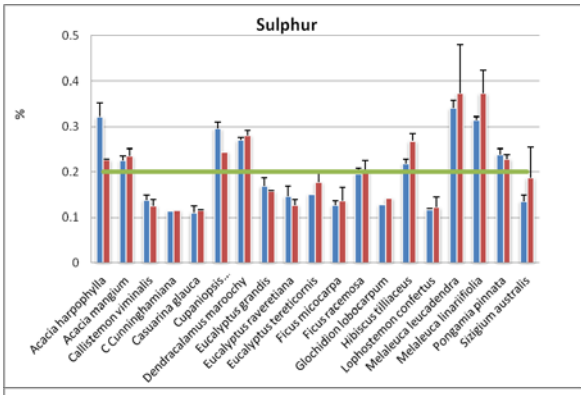
Leaves of trees at 2 years of age contained adequate nutrient levels to support growth. Trees grown in the phytocaps did not show any nutrient deficiency in their early stages of growth (Table 4.6). This may not be the case when they mature and compete with other species in the stand. The 2 year-old trees showed sufficient concentrations of nitrogen, sulphur, calcium, copper, manganese and magnesium to remain healthy and growing (Table 4.6). However, a few species contained slightly higher concentration of nitrogen, calcium and magnesium (Fig. 4.1), but these elevated levels were unlikely to have affected their growth as the Australian plants can sustain such variability (Ashwath pers. comm.). Presence of elevated concentrations of potassium, iron, zinc and boron can affect plants (Drechsel and Zech 1991). However in this study, although some plants had slightly elevated concentrations of potassium, iron, zinc and boron (Fig. 4.1), present were not at the

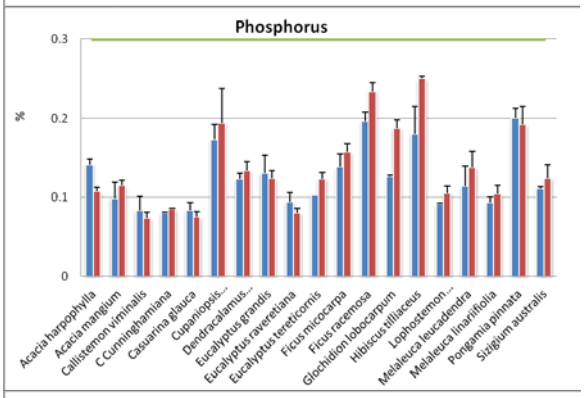
levels likely to negatively affect plant growth (Table 4.6). All plants grown in the phytocapping system showed significantly low levels of phosphorus (Fig. 4.1). Overall, in the 2 year-old trees with the exception of phosphorus, all other elements were found to be adequate for plant growth and the sodium content was lower than the threshold limit (except for *A. mangium*). A low level of phosphorus is a concern, but Australian native species have been shown to grow in low phosphorus conditions (Phillips 1994). The results also suggest that the poor growth of *Salix* and *Populus* species was not due to a lack of excess nutrients (Fig. 4.2) but possibly associated with external and agro-climatic conditions, of the region such as high temperature (>40°C) encountered during some months.

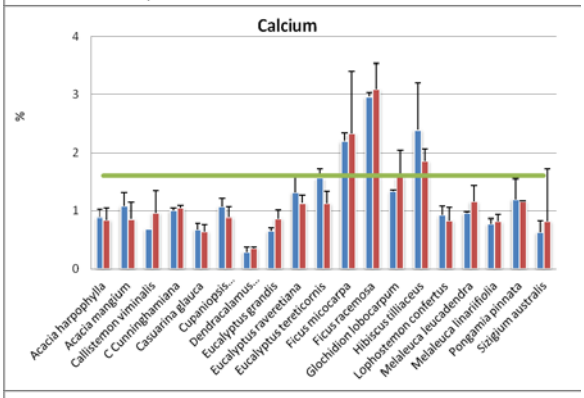
Table 4.6: The lowest, highest and mean nutrient concentrations in 2 and 3 year-old trees

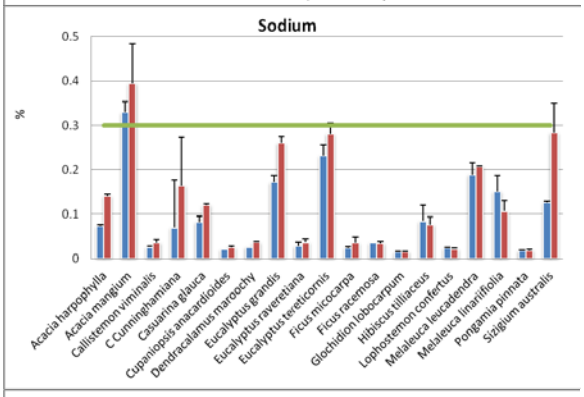
		N %	P%	K%	S%	Ca%	Na%	Mg %	Cu mg/kg	Fe mg/kg	Zn mg/kg	Mn mg/kg	B mg/kg
Leaves (2005)	Lowest	1.4	0.1	0.7	0.1	0.3	0.016	0.1	3.8	78.4	12.9	27.1	13.4
	Highest	3.8	0.2	2.0	0.4	3.0	0.4	0.7	10.9	294.3	34.4	535.2	115.5
	Mean	2.1	0.1	1.2	0.2	1.2	0.1	0.3	5.8	157.9	21.3	163.8	47.6
Leaves (2006)	Lowest	1.5	0.1	0.7	0.1	0.6	0.008	0.2	2.9	145.6	15.0	36.7	14.4
	Highest	4.3	0.2	2.1	0.4	3.3	0.5	0.6	9.6	455.7	41.4	628.3	109.0
	Mean	2.2	0.1	1.1	0.2	1.5	0.1	0.4	5.1	287.2	24.1	182.0	54.0
Leaf Litter (2006)	Lowest	0.8	0.1	0.4	0.3	1.1	0.032	0.2	2.3	316.4	15.4	66.1	26.0
	Highest	3.4	0.2	1.6	0.1	3.7	0.3	0.6	8.8	607.2	42.7	645.7	169.0
	Mean	1.4	0.1	0.7	0.2	1.7	0.1	0.3	3.8	388.5	21.1	190.6	63.3

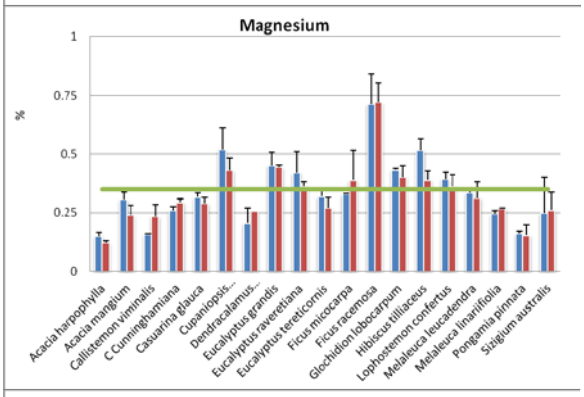

Nitrogen
Thick phytocap
Thin phytocap
%
0
1
2
3
4
5
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendracalamus maroochy
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Sulphur
%
0
0.1
0.2
0.3
0.4
0.5
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus maroochy
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Phosphorus
%
0
0.1
0.2
0.3
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus...
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon...
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Calcium
%
0
1
2
3
4
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus...
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Sodium
%
0
0.1
0.2
0.3
0.4
0.5
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendracalamus maroochy
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Magnesium
%
0
0.25
0.5
0.75
1
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus...
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
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Melaleuca leucadendra
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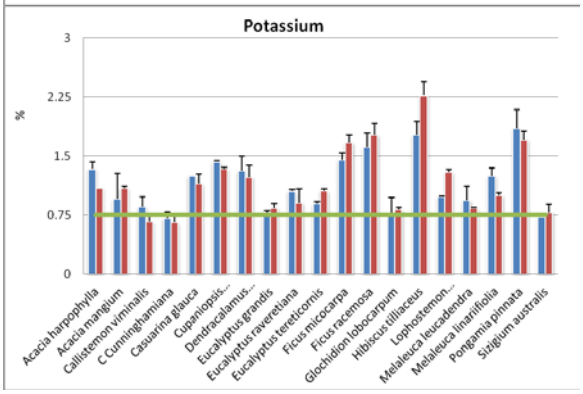
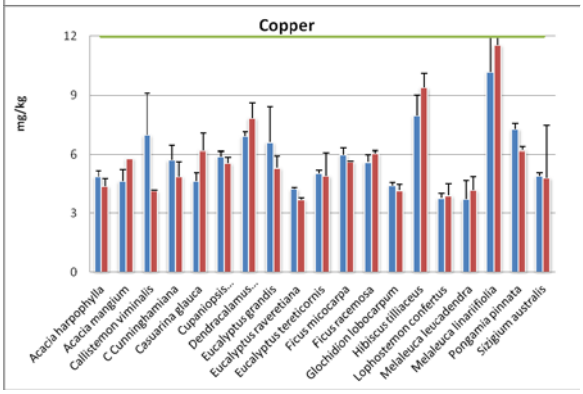
Pongamia pinnata
Syzigium australis


Potassium
%
0
0.75
1.5
2.25
3
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus...
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon...
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis


Copper
mg/kg
0
3
6
9
12
Acacia harpophylla
Acacia mangium
Callistemon viminalis
C Cunninghamiana
Casuarina glauca
Cupaniopsis...
Dendracalamus...
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus micocarpa
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzigium australis

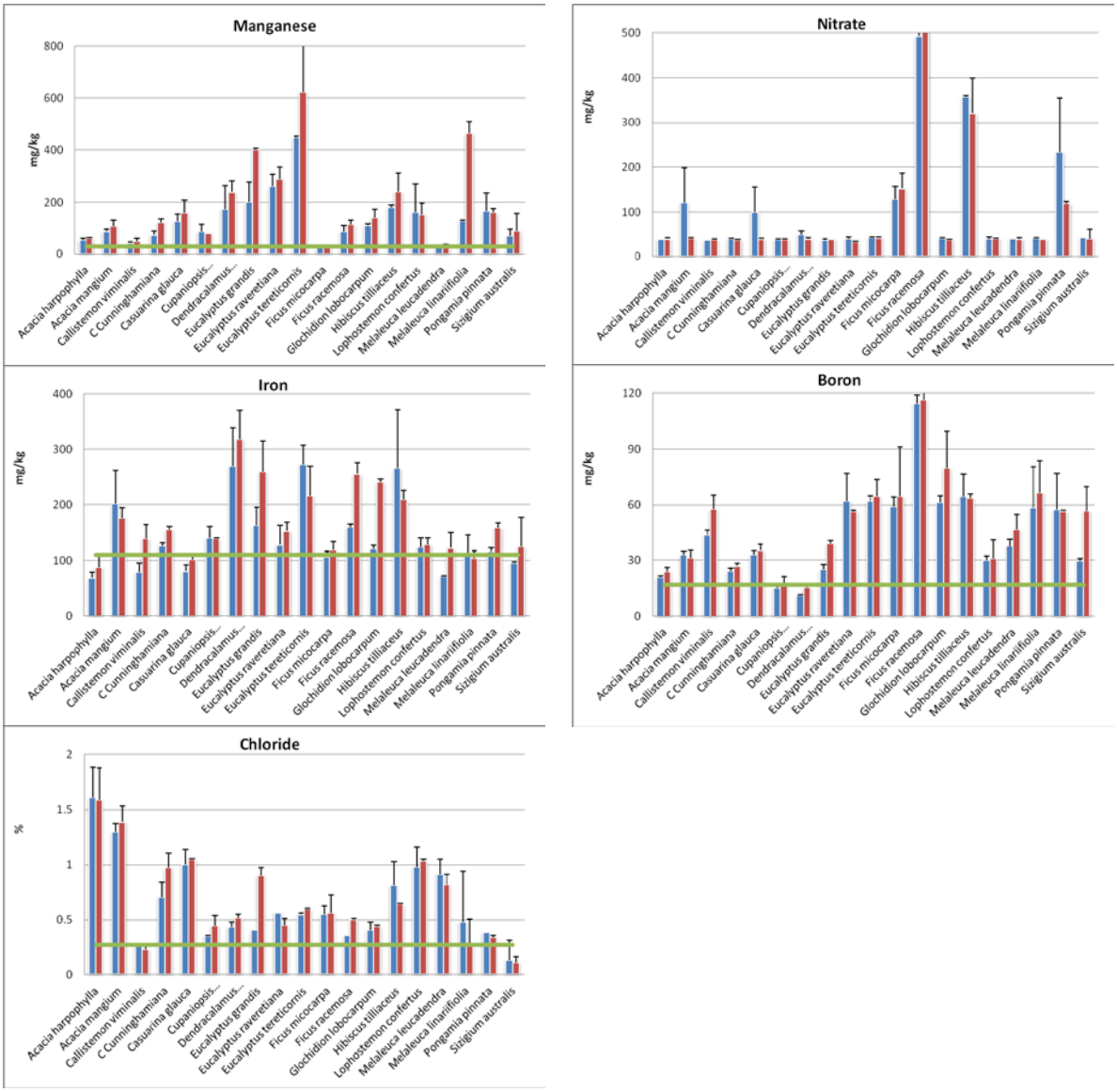


Figure 4.1: Average foliar nutrient concentrations in 2 year-old species grown in the Thick and Thin phytocapping systems.
Bars represent standard errors. The horizontal line shows the optimum levels recommended for normal growth of plants according to Table 4.3.

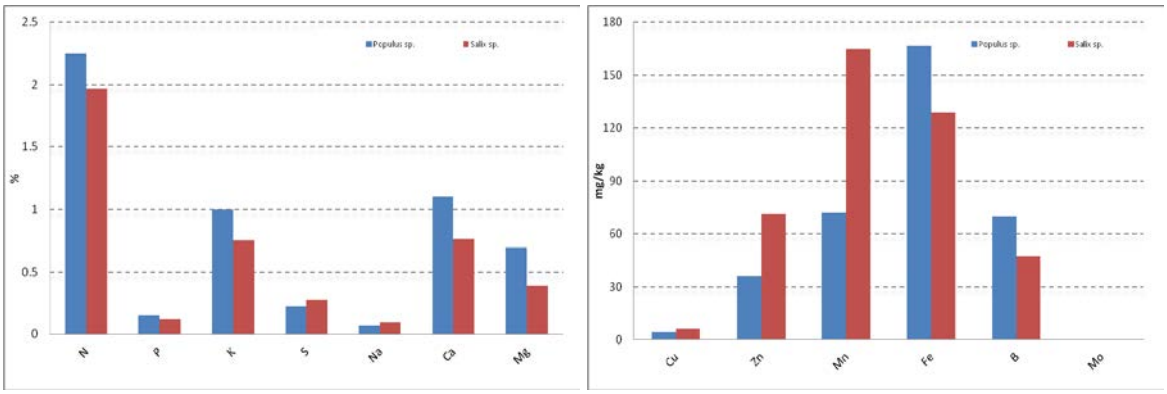
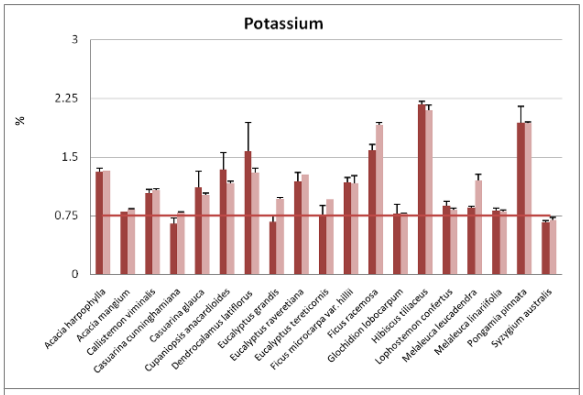


Figure 4.2: Foliar nutrient concentrations in 2 year-old *Populus* sp. and *Salix* sp. grown in the phytocapping systems at Rockhampton (Average over Thick and Thin phytocap)

4.4.1.2 Foliar Nutrient Composition in 3 Year-Old Trees

At 3 years of age, the trees showed no elevated levels of nutrients (Table 4.6). Nitrogen concentration was slightly higher in *P. pinnata* at age 3 than at age 2 (Figs. 4.1 and 4.3), and this may be associated with its nitrogen fixation potential. Sodium, sulphur, calcium, magnesium, copper and manganese concentrations were well within the optimum levels for plant growth (Fig. 4.3). The 3 year-old *A. harpophylla, C. anacardioides*, bamboo, *M. leucadendra* and *P. pinnata* had slightly higher levels of sulphur (Fig. 4.3) but these levels were unlikely to have affected plant growth. Potassium levels were high in most species (Figs 24), but the levels are not that high to affect their health. *Ficus microcarpa, F. racemosa* and *H. tiliaceus* showed higher concentrations of calcium than other species at the age of 2 and 3 (Figs. 4.1 and 4.3). Zinc concentrations were slightly higher in *F. racemosa, G. Lobocarpum* and *P. pinnata* (Fig. 4.3). Iron concentrations showed elevated levels in the 3 year-old stand compared to the 2 year-old stand (Figs. 4.1 and 4.3). Phosphorus was still below the optimum required level (Fig. 4.3). However, trees were growing well in both Thick and Thin phytocaps. It is interesting to note that phosphorus levels were similar in trees growing in Thick and Thin phytocaps (Fig. 4.3). This shows that this element was not governed by the thickness of the soil cover. Boron concentrations were higher than recommended for normal plant growth in most species, except in *C. anacardioides* and *D. latiflorus* (Figs. 4.3).


Potassium
%
3
2.25
1.5
0.75
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
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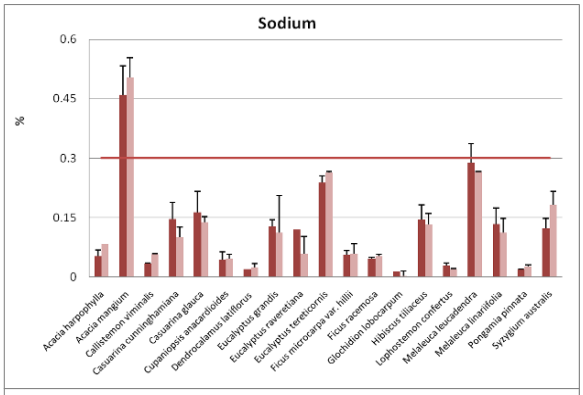
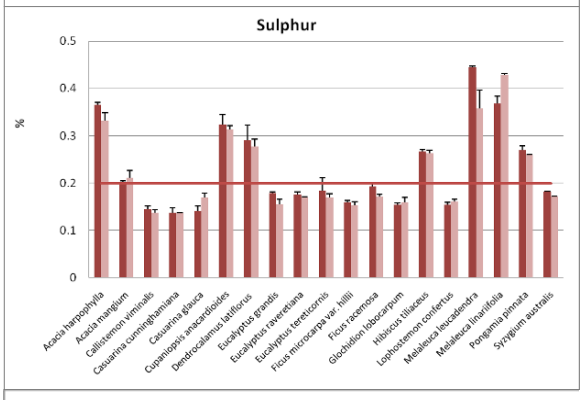
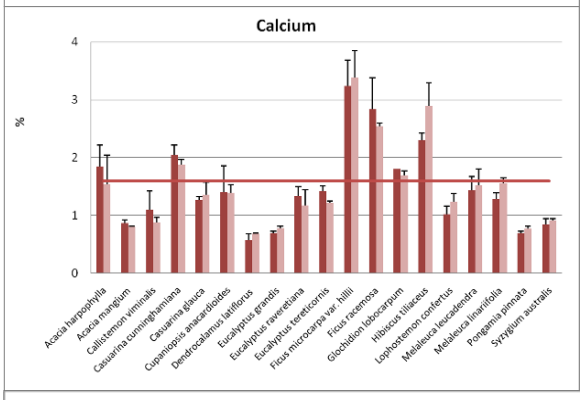
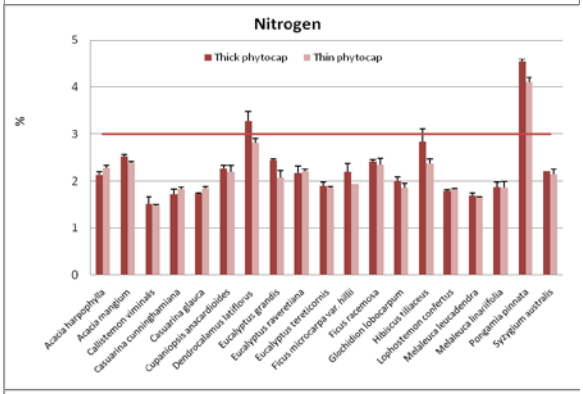
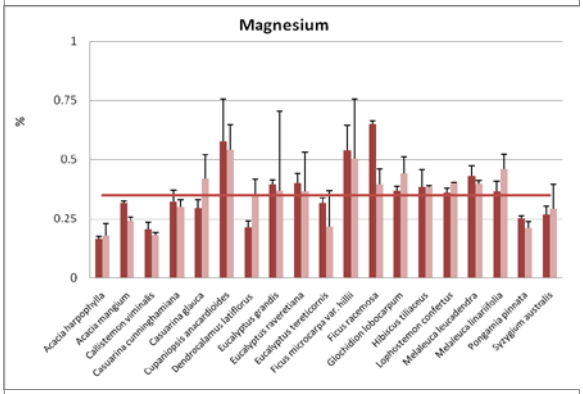
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

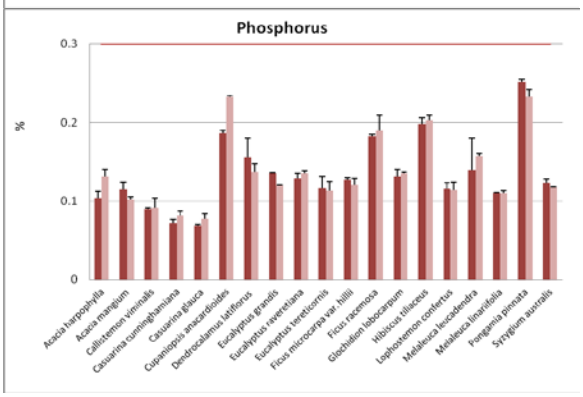

Sodium
%
0.6
0.45
0.3
0.15
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

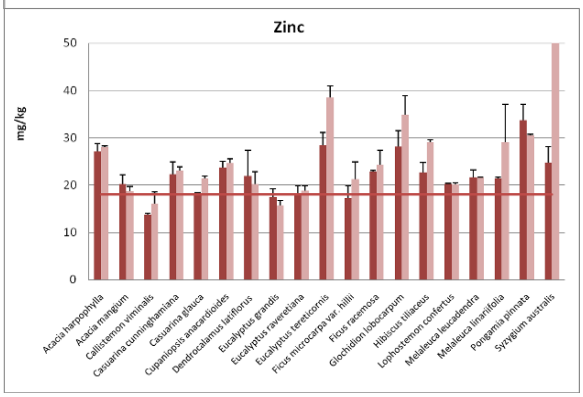

Sulphur
%
0.5
0.4
0.3
0.2
0.1
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Calcium
%
4
3
2
1
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Nitrogen
Thick phytocap
Thin phytocap
%
5
4
3
2
1
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Magnesium
%
1
0.75
0.5
0.25
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Phosphorus
%
0.3
0.2
0.1
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Zinc
mg/kg
50
40
30
20
10
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

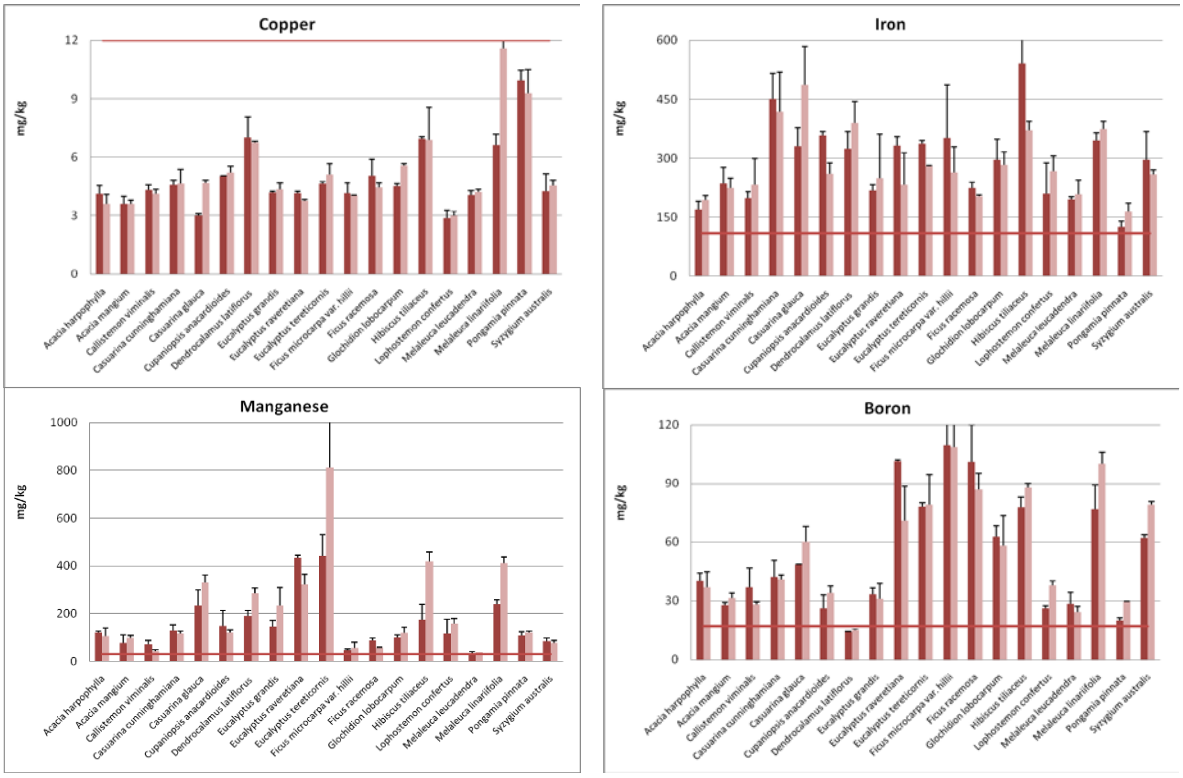


Figure 4.3: Average foliar nutrient concentrations in 3 year-old species grown in Thick and Thin phytocapping systems.
Bars represent standard errors. The horizontal line shows the optimum levels recommended for normal growth of plants according to Table 4.3.

4.4.1.3 Effect of Maturity on Foliar Nutrient Composition

Results from the analysis conducted in 2 and 3 year-old trees reveal that the foliar nutrients (N, P, K, S, Na, Ca, Mg, Cl, Cu, Zn, Mn, Fe and B) were adequate for their growth in the landfill environment, even though the nutrient content differed significantly (P<0.001) (Table 4.5) between species over a year (Figs. 4.1 and 4.3). The variation in nutrient levels among trees of the same species may be attributed to composition of the waste (Vetousek and Sanford 1986), soil composition (Drechsel and Zech 1991) and root distribution (Pregitzer and King 2005). In a fertile soil, concentrations of nutrients in leaves are found at higher levels that those in poor soils (Tanner 1985). Likewise except copper and phosphorus, all other nutrients were present in adequate levels in most species. Results from this exercise suggest that the trees grown on both Thick and Thin phytocaps had adequate nutrient levels to support their initial growth, and contribute towards the overall performance of the phytocapping system.

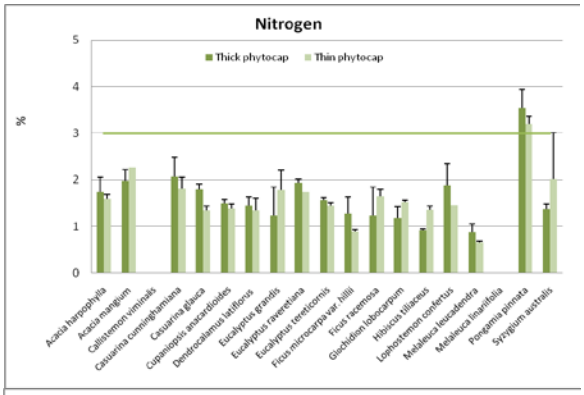
The foliar nutrient concentrations differed significantly (P<0.001) from year 2 to year 3 (Table 4.5). The 3 year-old stand contained higher concentrations of nutrients than those sampled in year 2. The concentrations of sulphur and sodium remained the same, but the concentrations of other nutrients showed a gradual increase in uptake (Table 4.6). The 3 year-old plants contained slightly elevated concentrations of iron, zinc, manganese and boron (Table 4.6). On the other hand, phosphorus, calcium, magnesium copper and iron levels slightly dropped or remained the same in most species (Fig. 4.1 and 4.3). Overall there was a marginal increase in certain elements; the levels were well within the threshold to not affect the plants. Nutrient uptake patterns in plants determine the circulation and storage of nutrients. Nutrient concentrations varied with maturity and these variations were related to accumulation of nutrients in the older tissues and mineral shedding (senescence) from one season to the other. Nutrient concentrations decreased from year 2 to year 3 in the cases of potassium, sodium and copper, and this could be associated with exhaustion of nutrients contained in the root zone while new tissues were being produced by the tree (Chapin *et al.* 1980). Potassium is easily removed by leaching (Chapin 1980), while nitrogen and calcium, zinc, magnesium, manganese, iron and boron are gradually accumulated over one year. Differences in nutrient concentrations in the established trees can be attributed to individual species having nutrient storage-pool turnover times ranging from one year to several hundred years (Day *et al*. 1977). Seasonal variation in nutrients within individual species can also be caused by caterpillars feeding on these trees (Feeny 1970).

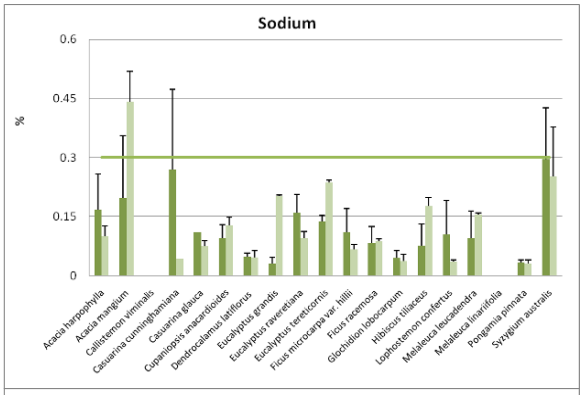
4.4.1.4 Leaf Litter Nutrient Concentration

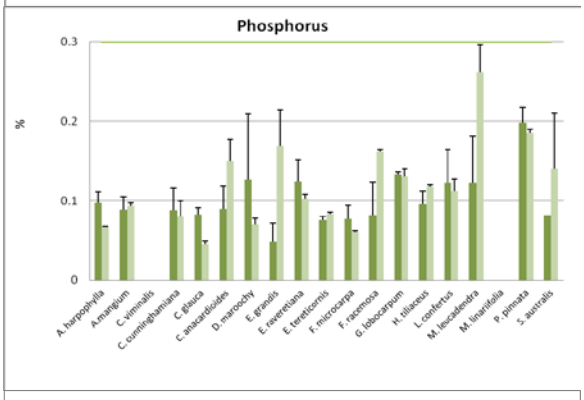
In this study leaf litter was used to determine the nutrient flux from the aboveground vegetation to the soils. Results from the analysis conducted on 3 year-old trees suggested that considerable amount of nutrients were cycled within the phytocaps irrespective of the soil thickness (Fig. 4.4). Species differed significantly (P<0.001) in their litter nutrient composition (Table 4.5) as they were diverse in morphology, growth patterns and physiology. The lower concentrations of nitrogen, potassium, copper and zinc in the leaf litter compared to the live tissues of leaves (Table 4.6) can be attributed to nutrient withdrawal from leaves of many species (Vitousek and Sanford 1986). Phosphorus, sulphur, sodium and magnesium levels were the same as observed in the live tissues of the leaves (Fig 4.4). Manganese, iron and boron

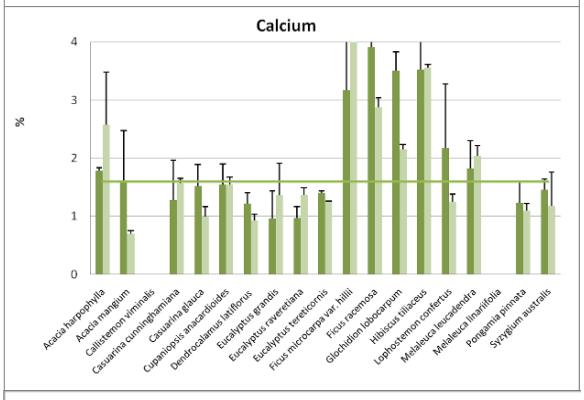
concentrations were elevated in leaf litter compared to the live tissues of the leaves (Fig. 4.4).

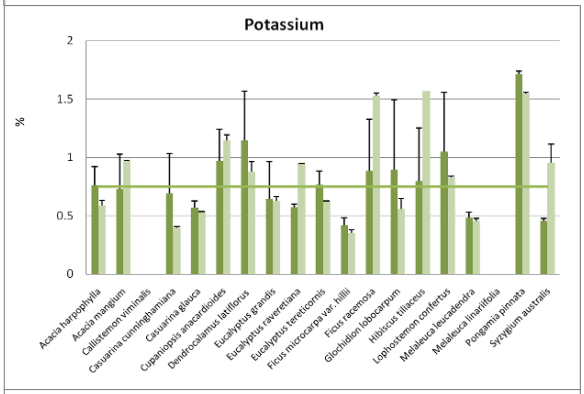
In this study, the leaf litter from the 3 year-old trees contained lower levels of nitrogen, sodium, phosphorus potassium and copper compared to the levels in the live tissues of the leaves (Fig. 4.3 and 4.4). The leaf litter from *P. pinnata* showed high level of nitrogen (Fig. 4.4). *Ficus microcarpa, F. racemosa, G. lobocarpum* and *H. tiliaceus* showed slightly higher levels of calcium (Fig. 4.4) and *E. tereticornis* and *G. lobocarpum* showed slightly elevated levels of zinc (Fig. 4.4). *Pongamia pinnata* showed slightly more elevated levels of potassium in the leaf litter than in the live tissues of its leaves (Fig. 4.4). Magnesium levels remained the same in the majority of the species (Fig. 4.4). *Acacia hapophylla* contained slightly higher levels of calcium and sulphur in the leaf litter than in the live tissues of its leaves (Figs. 4.3 and 4.4). Zinc, manganese, iron and boron were at higher levels in the leaf litter than the live tissues of leaves (Fig. 4.4). The variation in different elements among different species may be associated with differing ability of species to translocate and re-translocate elements within the tree. This would in turn contribute to species differences in nutrient recycling (Vitousek and Sanford 1986). Overall, adequate (90% to 100%) levels of nutrients were being recycled into the soil, which is beneficial for plant growth and the longer sustainability of the phytocaps. This attests that the soil being moderately fertile (Appendix A) and able to support plant growth without any health deficits. Some species showed slightly elevated levels of leaf nutrients, which in this instance were insignificant to their health and growth.

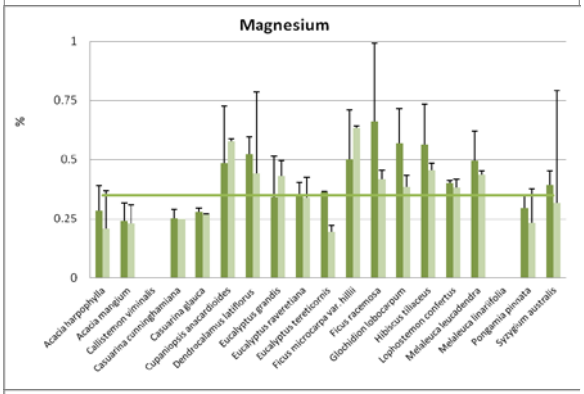

Nitrogen
Thick phytocap
Thin phytocap
%
0
1
2
3
4
5
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Sodium
%
0
0.15
0.3
0.45
0.6
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Phosphorus
%
0
0.1
0.2
0.3
A. harpophylla
A.mangium
C. viminalis
C. cunninghamiana
C. glauca
C. anacardioides
D. maroochy
E. grandis
E. raveretiana
E. tereticornis
F. microcarpa
F. racemosa
G. lobocarpum
H. tiliaceus
L. confertus
M. leucadendra
M. linariifolia
P. pinnata
S. australis


Calcium
%
0
1
2
3
4
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Potassium
%
0
0.5
1
1.5
2
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Magnesium
%
0
0.25
0.5
0.75
1
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
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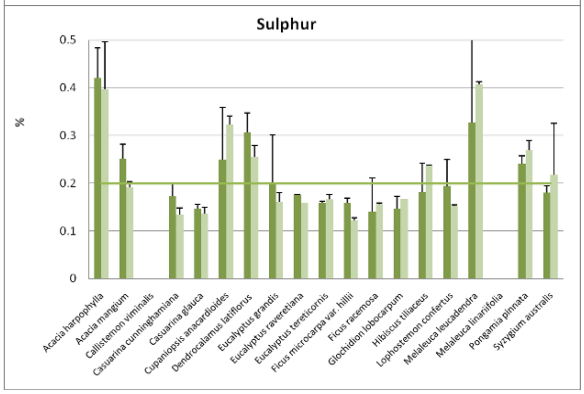
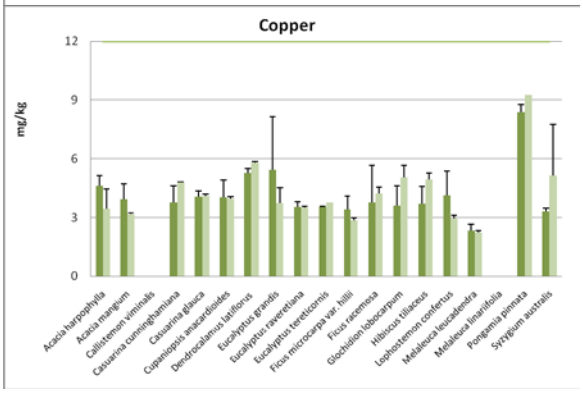
Syzygium australis


Sulphur
%
0
0.1
0.2
0.3
0.4
0.5
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Copper
mg/kg
0
3
6
9
12
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

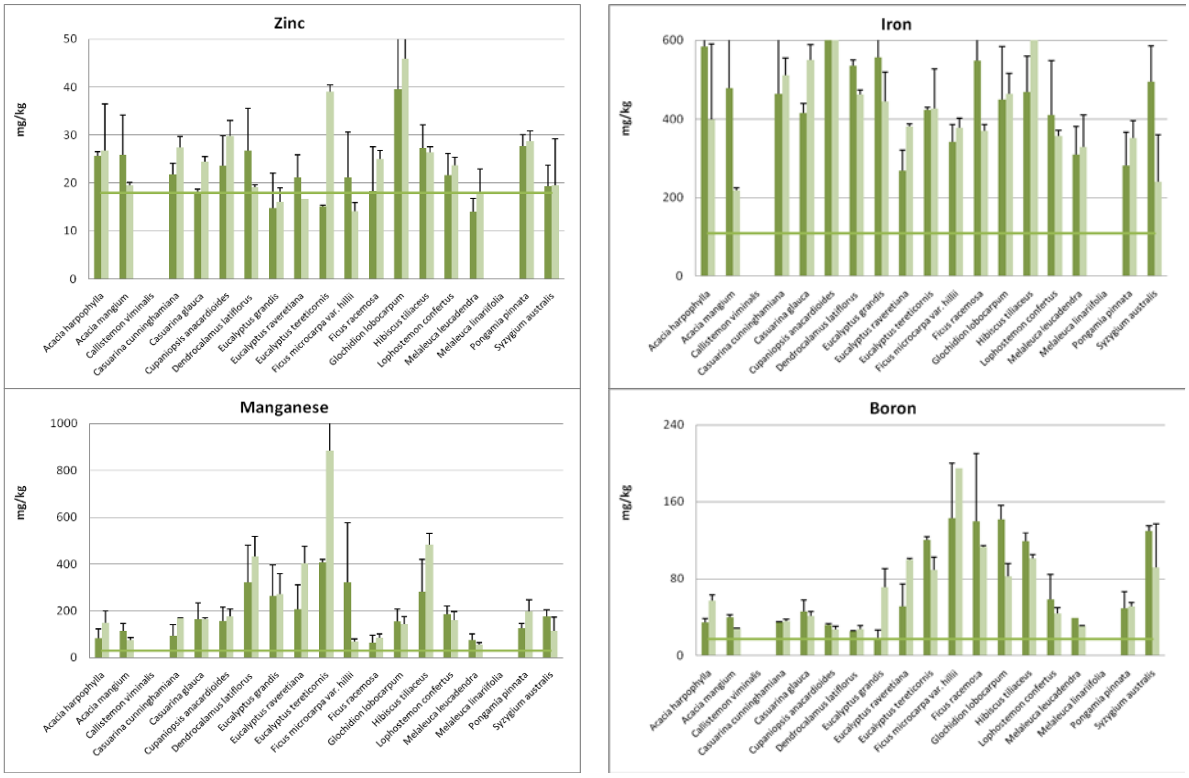


Figure 4.4: Average values of leaf litter nutrient concentrations in 3 year old plants grown in the Thick and Thin phytocapping systems.
Bars represent standard errors. The horizontal line shows the optimum levels recommended for normal growth of plants according to Table 4.3.

4.4.2 Foliar and Leaf Litter Heavy Metal Composition

4.4.2.1 Foliar Composition of Heavy Metals in 2 and 3 Year-Old Trees

Overall, the 2 and 3 year-old stands showed no elevated concentrations of heavy metals (Table 4.7) except in *G. lobocarpum*, which showed high levels of cobalt. In this study, species differed significantly (P<0.001) (Table 4.5) in heavy metal concentrations. This may be associated with the ability of different tree species to translocate heavy metals from root to shoot. Zinc, cadmium and nickel are translocated to the leaves, while chromium, lead and copper are usually retained in the roots (Pulford and Watson 2003).

Table 4.7: Lowest, highest and mean heavy metal concentrations (Mg/kg) in 2 year and 3 year-old trees

		As	Cd	Co	Cr	Hg	Mo	Ni	Pb	Se
Leaves (2005)	Lowest	86.1	24.5	74.2	417.1	50.6	43.6	628.8	681.3	63.5
	Highest	1383.9	130.5	10208	1521.0	298.5	978.1	14202	5257.8	248.2
	Mean	380.0	11.4	755.0	770.4	127.3	253.5	3690.8	2250.4	123.7
Leaves (2006)	Lowest	101.1	10.5	86.2	415.1	51.6	47.6	625.8	684.3	65.5
	Highest	1398.9	134.5	10220	1519.0	299.5	982.1	14199	5260.8	250.2
	Mean	395.0	13.8	767.0	768.7	128.3	257.5	3687.8	2253.4	122.0
Leaf Litter (2005)	Lowest	220.5	24.5	166.7	681.6	65.9	140.3	963.5	1475.0	66.7
	Highest	3101.5	136.1	9609	1800.8	175.3	1067.8	6811.8	6238.5	166.0
	Mean	654.4	8.5	978.9	956.1	105.1	321.2	2967.9	2590.8	109.6
Leaf Litter (2006)	Lowest	211.5	5.2	129.1	744.8	68.3	142.8	868.3	1765.9	84.4
	Highest	4425.3	149.3	10824	1829.9	185.6	1276.5	5866.5	5726.5	179.4
	Mean	703.8	24.7	1005.4	1041.8	115.3	398.0	2355.8	2894.5	118.3

At the sampled growth stages (2 and 3 year-old), most species did not accumulate excessive amounts of heavy metals, most likely due to very shallow penetration into the soil (approx. 600 mm, Chapter 3) and the restricted location of metals into the roots and low uptake into foliage, which is a very common resistance trait of trees (Dickinson and Lepp 1997). Overall, levels of mercury, cadmium, chromium, lead, and selenium were well within the threshold limits (Figs 4.5 and 4.6). However, the 3 year-old *E. grandis* showed slightly higher concentrations of mercury in the thin phytocap (Fig 4.6) and *G. lobocarpum* accumulated very high levels of cobalt in both Thick and Thin phytocaps. The reason for high accumulation of cobalt by *G. lobocarpum* is unknown and requires further investigation on this species. Deeper root penetration and the possible access to heavy metals may vary from landfill to landfill and within landfills in space and time (Fitter 1994). But, *G. lobocarpum* showed elevated concentrations of cobalt in both Thick and Thin phytocap, which may be associated with its genetic ability to hyperacumulate cobalt. Numerous researchers have reported that the species that possess the ability to develop tolerance to heavy metals will take up heavy metals (hyperaccumulators; Jonnalagadda and Nenzou 1997).

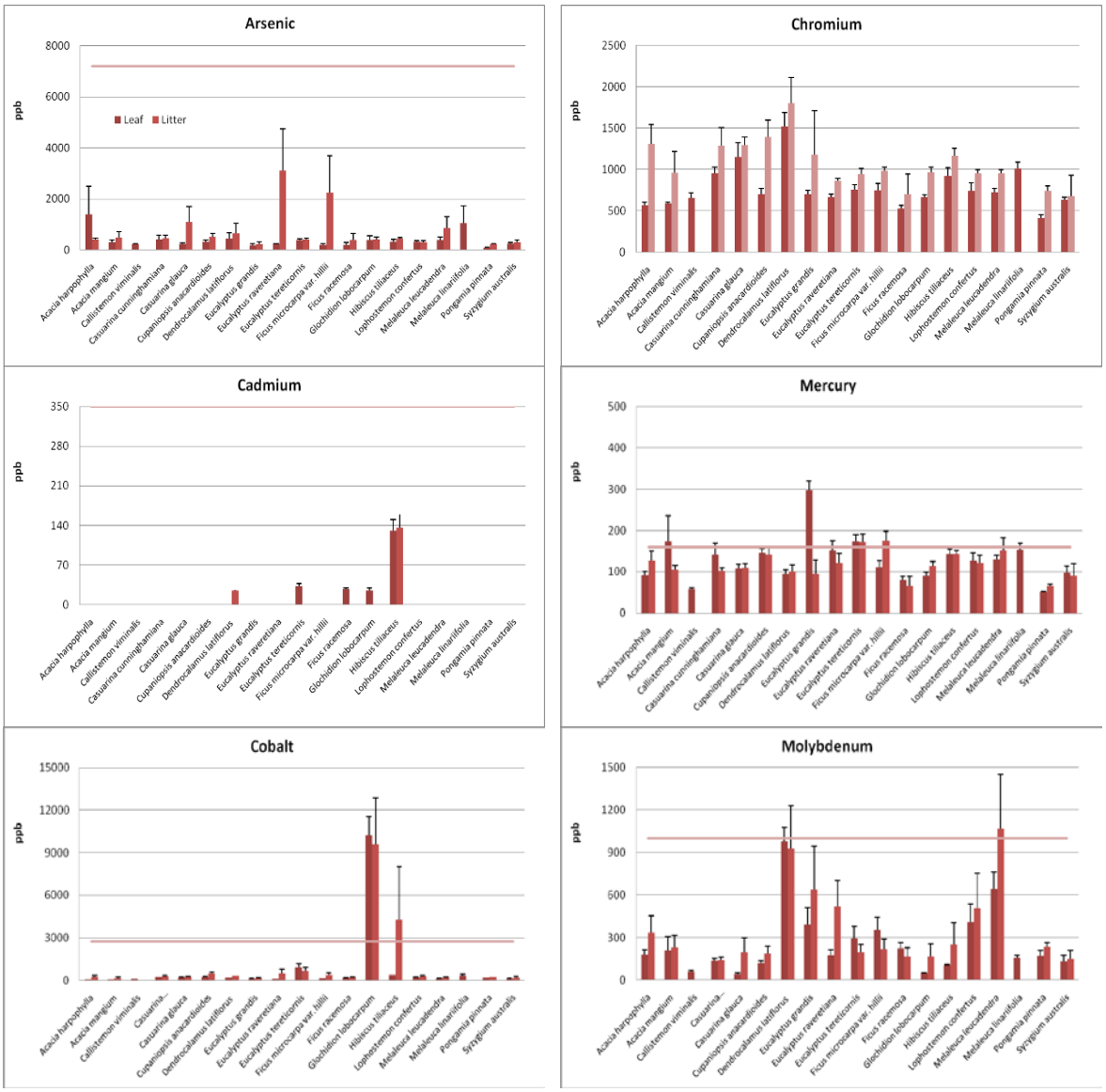
However, even at elevated levels of heavy metals in the soil, trees evolve a few metal-tolerant ecotypes (Khale 1993) which restrict the uptake of heavy metals. The lack of toxicity symptoms in trees also indicate their tolerance to withstand higher heavy metal concentrations than for agricultural crops (Riddell-Black 1993). Several studies in the past have reported good growth rates of trees despite their root penetration into the spoil, waste and mine tailings (Borgegård and Rydin 1989, Dickinson *et al*. 1991, Turner and Dickinson 1993, Landberg and Greger 1994, Punshon *et al*. 1995, Punshon and Dickinson 1997 and Kopponen *et al.* 2001). In this study, however, the 3 year-old *H. tiliaceus* showed slightly higher levels of mercury (517 Mg/kg) in the Thin phytocap (Fig. 4.6) but the levels are not likely to affect the plant. Mercury is readily available to plants (Millan *et al*. 2006) as it has a great affinity to organic matter (Grigal 2003).

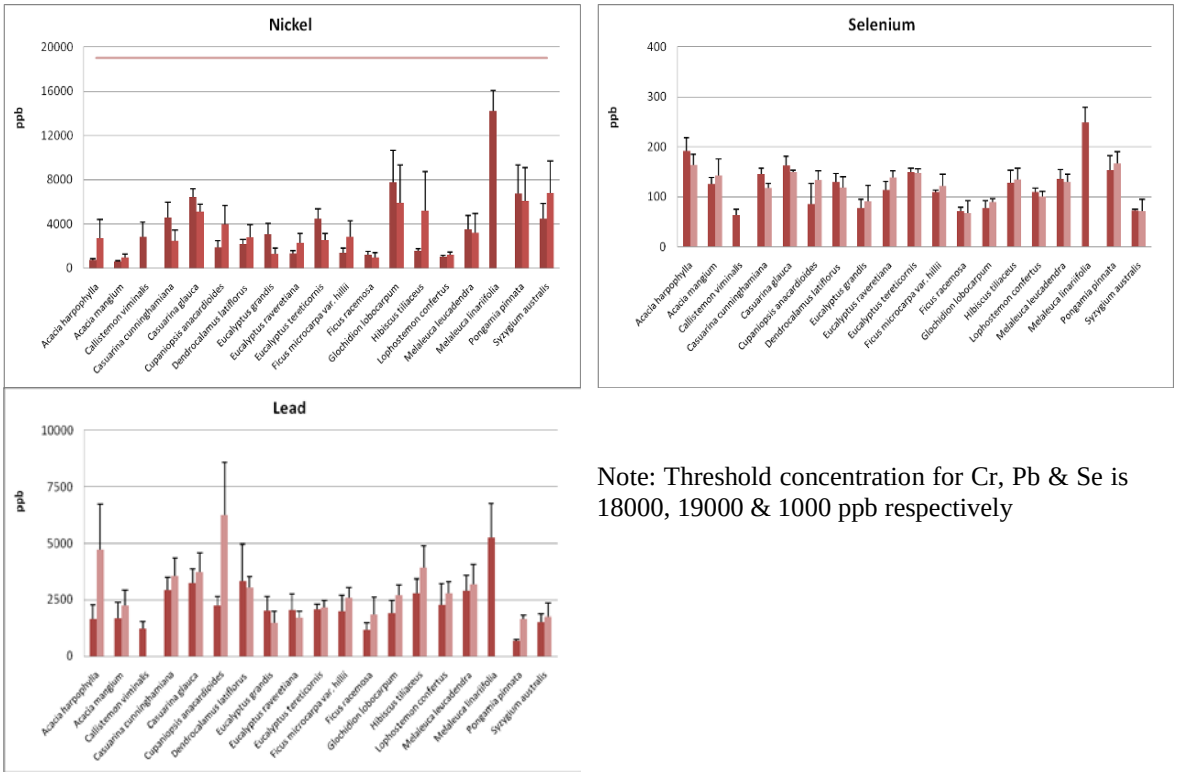
4.4.2.2 Effect of Maturity on Heavy Metal Composition

Seasonal variations in the foliar heavy metal concentrations in trees have been confirmed by various studies in the past, but results from this study revealed no significant (Table 4.5) changes in the foliar heavy metal concentrations over one year (at ages 2 and 3 years, respectively) (Table 4.7). It is too early to make any discrete statements on the observations made as the trees established in this system are in their initial growth phase and have shallow roots. However, based on previous reports and findings, roots of trees grown on landfills and landfill covers do not tend to develop deep roots due to high internal soil temperatures and landfill gases. However, trends in heavy metal uptake will vary as the trees mature and develop deep roots. Riddell-Black (1994) reported consistent increases in foliar heavy metal concentrations shortly before senescence in willow grown on a metal-contaminated substrate.

There was no significant increase (Table 4.5) in heavy metal concentrations over time as the roots were well within the soil profile and most roots did not penetrate the waste by year 3. However, this may not be the case as the trees mature. The roots of the trees may penetrate deep into the soil over time and they may access the waste below taking up heavy metals and releasing them into the environment. It is possible that the soil and trees in the landfill site may constitute a threat to the environment. However, these risks may not be as serious as the threats of trees grown on metal

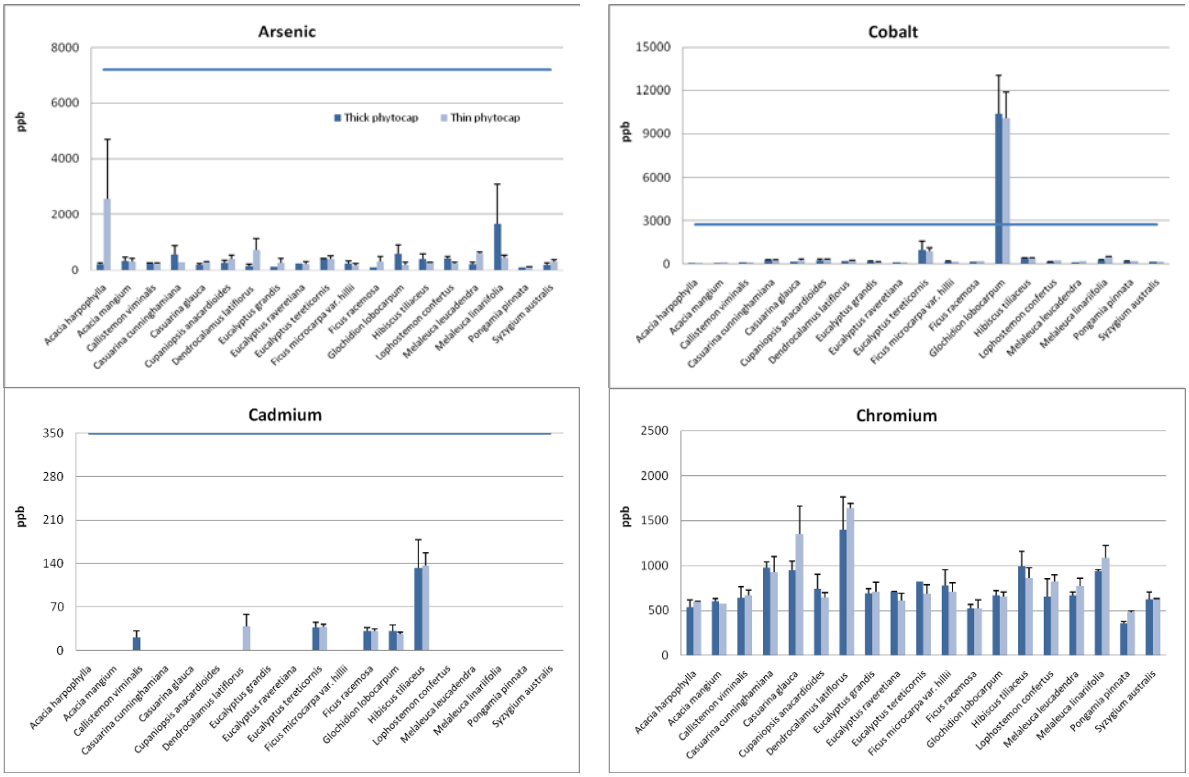
contaminated sites (Fernandes and Henriques 1991), mine sites (Grant *et al.* 2002, Maddock *et al.* 2009), ultramafic mineral sites (Koppittke *et al.* 2008), agricultural sites (Merry et al. 1986), industrial sites (Phillips and Chapple 1995), coastal areas and waterways (Hanley and Couriel 1992) and in soils that contain naturally elevated levels of metals (Lottermoser *et al.* 1999).

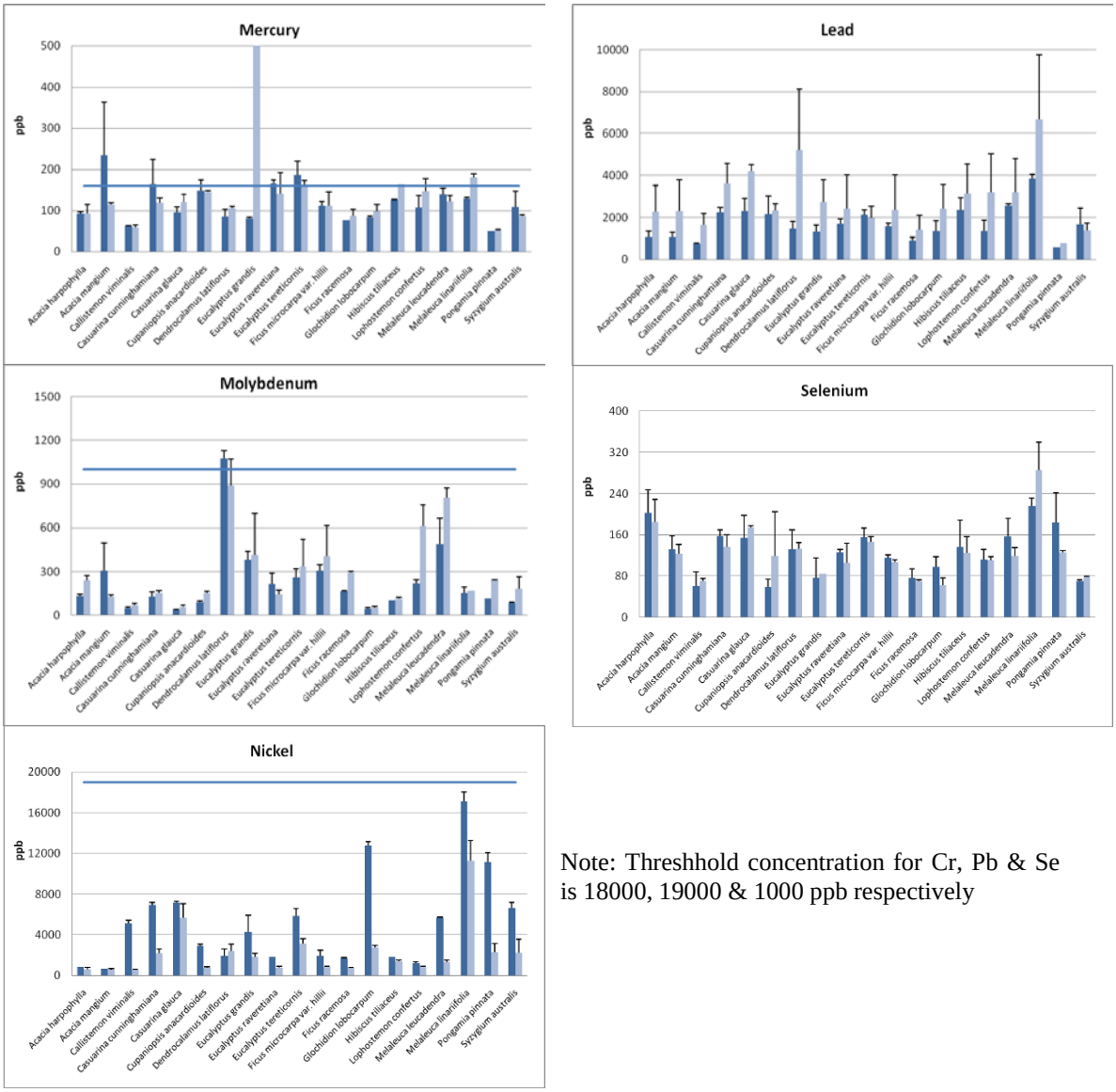




Note: Threshold concentration for Cr, Pb & Se is 18000, 19000 & 1000 ppb respectively

Figure 4.5: Foliar and leaf litter heavy metal concentrations in 2 year-old species averaged over two phytocapping systems.
Bars represent standard errors. The horizontal line shows the optimum levels recommended for heavy metals in plants/soil (Table 4.4).





Note: Threshhold concentration for Cr, Pb & Se is 18000, 19000 & 1000 ppb respectively

Figure 4.6: Foliar heavy metal concentrations in the 3 year-old species averaged over the Thick and Thin phytocapping systems.
Bars represent standard errors. The horizontal line shows the optimum levels recommended for heavy metals in plants/soil (Table 4.4).

4.4.2.3 Leaf Litter Heavy Metal Concentration

Leaf litter of 3 year-old trees showed no elevated (Fig. 4.7) concentrations of heavy metals. Species varied significantly (P<0.001) in their leaf heavy metal concentrations (Table 4.5). Overall, heavy metal concentration in leaf litter was higher than that found in live tissues of leaves (Table 4.7). *Eucalyptus tereticornis* had high concentrations of arsenic compared to other species (Fig. 4.7), but levels were well below the threshold limit (2700 ppb). Similarly, leaf litter cadmium composition of *H. tiliaceus* and *L. confertus* were higher (Fig. 4.7) than those in other species, but was well within the acceptable limit. *Acacia harpophylla* and *H. tiliaceus* showed higher levels of arsenic and cadmium (Fig. 4.7), respectively, than

other species. Overall, the leaf litter from the majority of the plants did not accumulate heavy metals in excessive quantity and the current concentrations are not expected to have an adverse impact on soil, flora and fauna in the phytocapping system. However, cobalt accumulation of *G. lobocarpum* is of some concern as the high levels were also found in the leaf litter (Figs 4.7 and 4.8). Overall, levels of heavy metals being recycled into the system via leaf litter fall are well within the limits the limits reported to affect the environment.

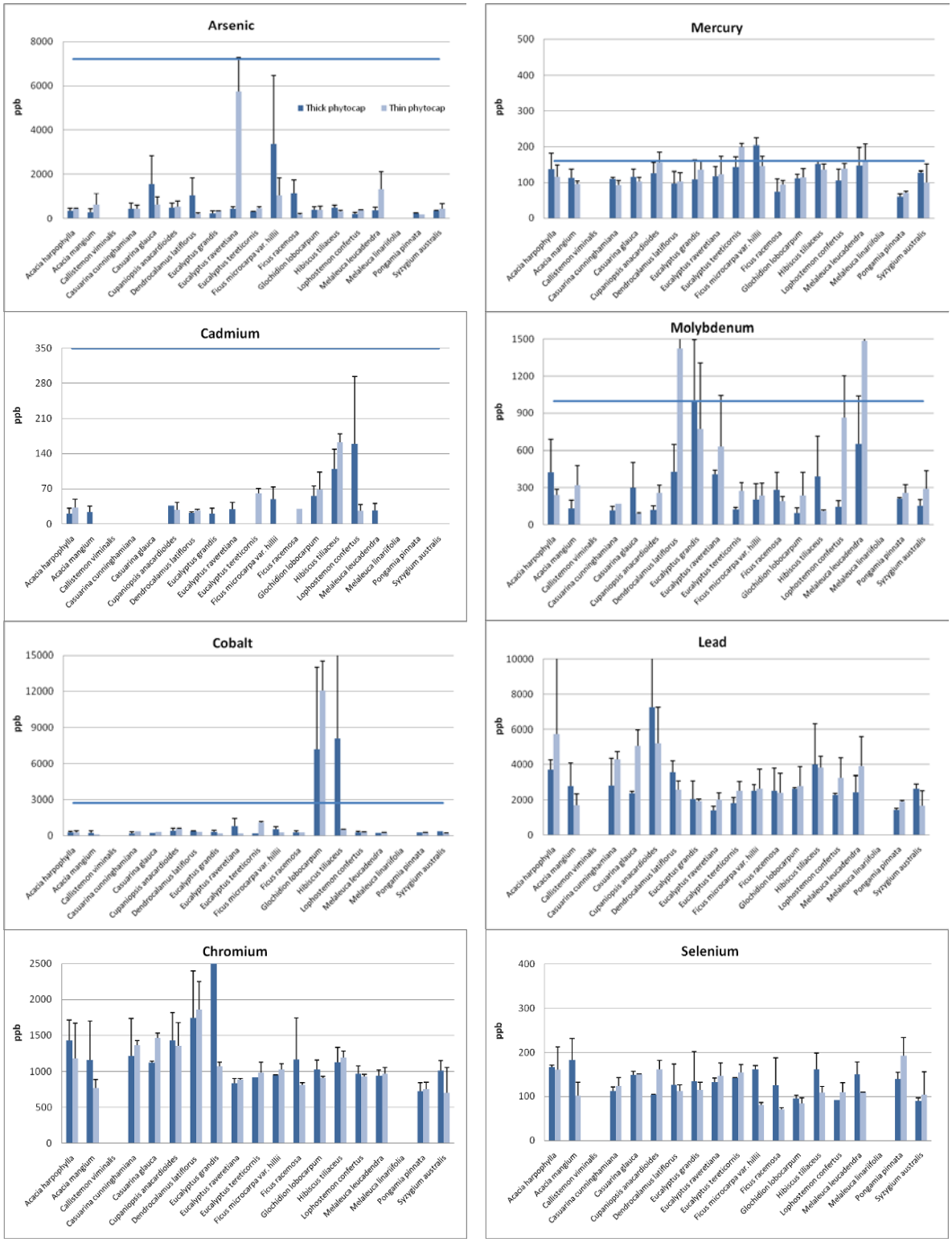
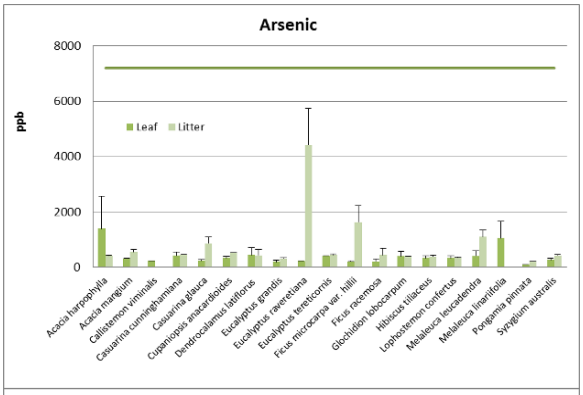
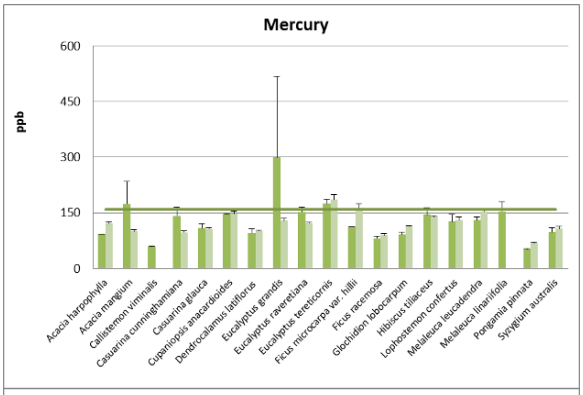
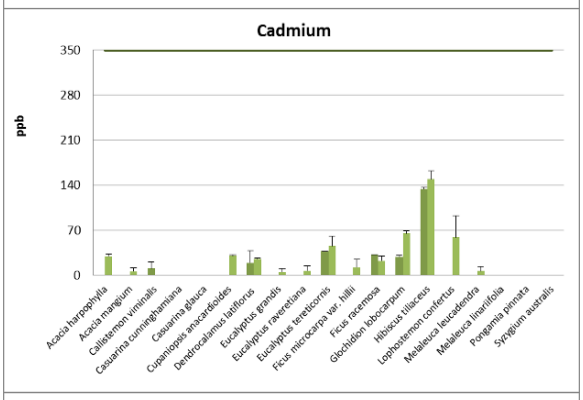
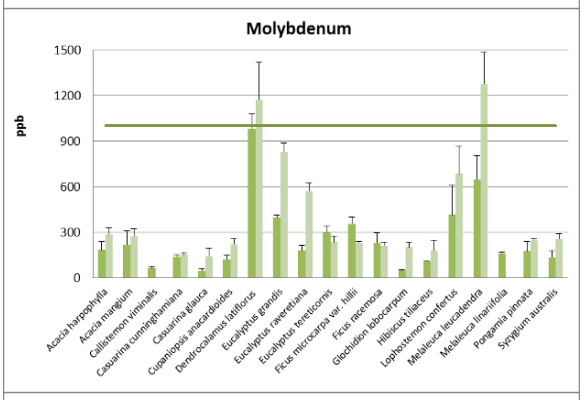


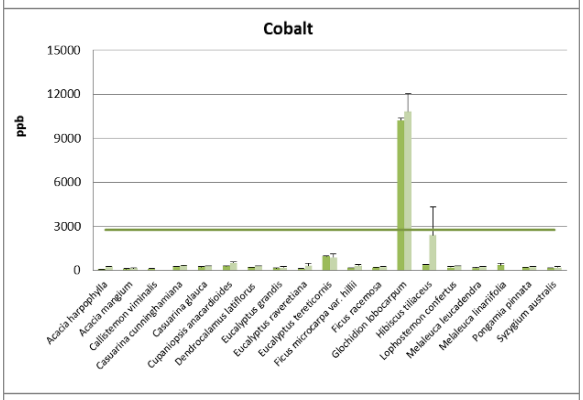
Figure 4.7: Leaf litter heavy metal concentrations in 3 year-old species averaged over the Thick and Thin phytocapping systems.
Bars represent standard errors (n=4). The horizontal line shows the threshold levels recommended for heavy metals in plants/soil (Table 4.4).

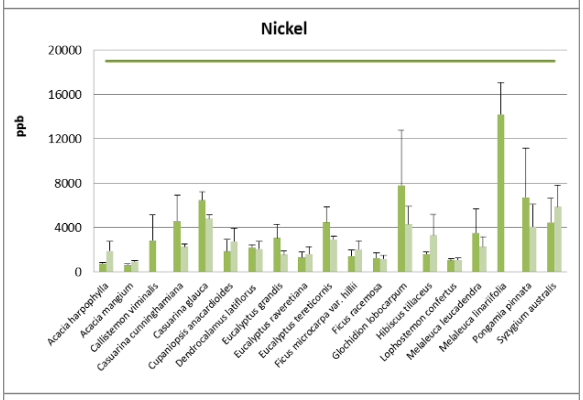

Arsenic
ppb
8000
6000
4000
2000
0
Leaf
Litter
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

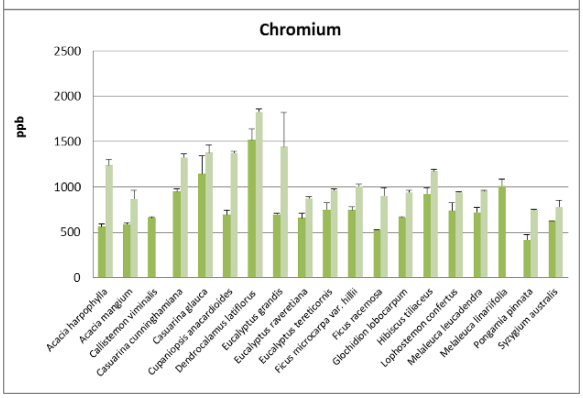

Mercury
ppb
600
450
300
150
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Cadmium
ppb
350
280
210
140
70
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Molybdenum
ppb
1500
1200
900
600
300
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Cobalt
ppb
15000
12000
9000
6000
3000
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Nickel
ppb
20000
16000
12000
8000
4000
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis


Chromium
ppb
2500
2000
1500
1000
500
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
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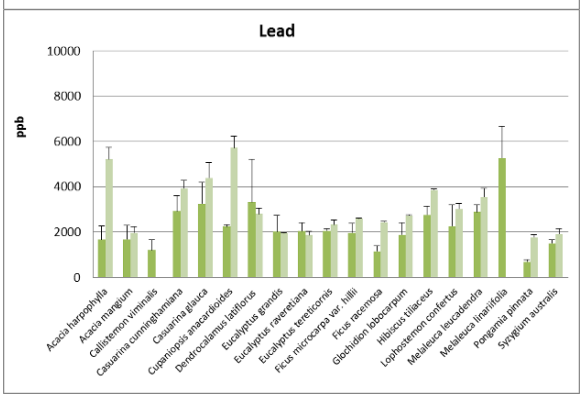
Pongamia pinnata
Syzygium australis


Lead
ppb
10000
8000
6000
4000
2000
0
Acacia harpophylla
Acacia mangium
Callistemon viminalis
Casuarina cunninghamiana
Casuarina glauca
Cupaniopsis anacardioides
Dendrocalamus latiflorus
Eucalyptus grandis
Eucalyptus raveretiana
Eucalyptus tereticornis
Ficus microcarpa var. hillii
Ficus racemosa
Glochidion lobocarpum
Hibiscus tiliaceus
Lophostemon confertus
Melaleuca leucadendra
Melaleuca linariifolia
Pongamia pinnata
Syzygium australis

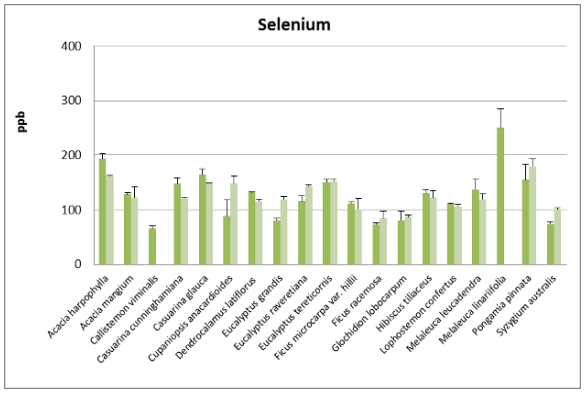


Figure 4.8: Comparison between foliar and leaf litter heavy metal concentrations in 3 year-old species grown in the phytocapping systems.
Bars represent standard errors (n=2). The horizontal line shows the Threshold levels recommended for heavy metals in plants/soil (Table 4.4).

4.4.3 Effect of Soil Depth on Foliar Nutrient and Heavy Metal Composition

There was a significant (P<0.001) influence of soil thickness on foliar nutrient levels (Table 4.5). Trees grown in Thin soil cover accumulated more nutrients than those grown in Thick soil cover (Figs. 4.1 and 4.2). This could potentially be due to proximity of roots to the waste in Thin phytocap than in Thick phytocap. The Thin cap had only 700 mm of soil cover as compared to the Thick cap which had 1400 mm of soil cover. However, root depth (Chapter 3) in Thin and Thick phytocaps was in the range of 500 mm to 700 mm, with a few species showing a root depth on 600 mm in both Thick and Thin soil covers, which in thin soil cover is very close to the underlying waste. Difference in the nutrient levels between the trees grown in Thick and thin phytocaps are likely to diminish as trees mature and send their roots deep down to the waste as indicated by these observations.

Heavy metal concentrations varied significantly (P<0.01) between Thick and Thin phytocaps. Trees grown in the Thin soil cover contained slightly elevated levels of heavy metals compared to those grown in the Thick soil cover (Figs 4.5 and 4.6) and this may be associated with closer proximity of their roots to the buried waste. At this stage, the trees have developed shallow roots to avoid high soil temperature and anaerobic conditions and also due to irrigation supply to support their growth in the initial stages. Hence the availability of water in the upper layers of the soil may not have encouraged the roots to penetrate into the buried waste.

4.4.4 Overall Trend

An overall trend in nutrient and heavy metal concentrations in foliage and leaf litter of 2 and 3 year-old trees established in the phytocapping system is summarised in Table 4.8. In this study, the exotic species such as bamboo showed good growth (Chapter 3); were healthier and grew faster than many Australian native species (Chapter 3). Similar observations were made in a study conducted by Solberg *et al.* (1996) in China.

Table 4.8: Overall trends in foliar and leaf litter nutrient and heavy metal concentrations in the phytocapping system (at 2 and 3 years)

	Foliar (2005)			Foliar 2006			Leaf litter (2006)			Remark
Element	Normal	Low	High	Normal	Low	High	Normal	Low	High	
N	*			*			*			Slightly high in *P. pinnata*
P		*			*			*		
K	*			*			*			
S	*			*			*			Slightly high in *A. harpophylla* & *M. leucadendra*
Ca	*			*			*			Slightly high in four species
Mg	*			*			*			
Na	*			*			*			Slightly high in *A. mangium* foliar (2005)
Cu	*			*			*			
Zn	*			*			*			Slightly elevated in *G. Lobocarpum* leaf litter
Mn	*			*			*			
Fe	*			*			*			
B			*			*			*	
Mo	*			*			*			Slightly high in six species
Co	*			*			*			Very high in *G. lobocarpum*
As	*			*			*			
Cd	*			*			*			
Hg	*			*			*			
Ni	*			*			*			
Pb	*			*			*			
Cr	*			*			*			
Se	*			*			*			

4.5 Conclusions

Overall, trees grown in two phytocapping systems contained adequate levels of nutrients to support growth. Low phosphorus levels are a concern and can be overcome by fertilizing trees at regular intervals. However, Australian trees are known to withstand phosphorus deficient conditions (Phillips 1994). Significant quantities of nutrients are recycled into the soil via leaf litter which will enhance the supply of nutrients to the trees over time. The 3 year-old trees showed slightly elevated levels of nutrient and this will continue as the trees mature and develop more roots. The trees in the thin soil cover contained slightly higher leaf concentrations of nutrients due to the possible closer proximity of their roots to the waste. It is too early to conclude about heavy metal uptake by the trees grown in the phytocaps, as the roots are shallow and are yet to penetrate the buried waste. However, trees may develop tolerance to heavy metals contained in the waste.. With time, trees grown on the Thin soil cover are expected to accumulate larger quantities of heavy metals than those grown in Thick soil cover.

Leaf litter from the majority of the species accumulates low levels of heavy metals, and therefore is unlikely to affect the soil, flora or fauna in the phytocaps. It will be interesting to see if the heavy metal concentrations of the leaf litter will increase as the trees mature. Further tests on mature trees will establish the role of trees in mobilising heavy metals from the soil and releasing these metals into the environment. However at this stage the established of trees on phytocaps to pose no concerns to the environment.

Cobalt accumulation by *G. lobocarpum* is of some concern and this needs to be investigated further, particularly for ecological implications, as the leaves of this species may be completely decimated by caterpillars (Vandecasteele *et al.* 2002), and predation of these caterpillars by birds may lead to adverse ecological consequences. For the time being, it is recommended that this species be not used in phytocaps.

In this Chapter, trees established on 2 phytocapping systems were tested for foliar nutrients. The results show no symptoms of deficiciency or excess of heavy metals; except in one species. The next chapter presents some interesting findings on transpiration and how the water uptake potential differs between species over different rainfall events.

5 Transpiration*

5.1 Introduction

An understanding of the movement of water from the soil to the atmosphere via trees is important with regard to phytocaps, as trees grown on phytocaps make a significant contribution to the hydrological balance of the site on which these are grown. Trees primarily help restrict rainwater entering the buried waste via canopy interception and transpiration. This Chapter discusses the role played by different tree species.

Transpiration is the amount of water taken up (upward movement) by a plant for its own use, with the excess being released into the atmosphere, and is one of the key processes that helps maintain the hydrological balance of a site (Weand *et al.* 1999). For phytocapping to be effective, the plants must transpire sufficient water so as to reduce its percolation into the waste (USDoE 2000). Trees generally transpire water during the day as part of photosynthesis (Eamus *et al.* 2006). The transpiration rates vary between species due to variation in stomatal density and climatic conditions (Vose *et al.* 2003). Transpiration has been expressed in a number of ways and most scientists and hydrologists express transpiration as mm d^{-1} (Eamus *et al.* 2006), as this takes into consideration the area covered by the tree.

* Some data from this chapter have been included in the following papers:

Venkatraman, K. and Ashwath, N. (2007) Phytocapping: an Alternative for reducing technique for leachate and methane generation from municipal landfills. *The Environmentalist*, 27: 155 – 164.

Venkatraman, K. and Ashwath, N. (2009) Phytocapping: importance of tree selection and soil thickness, *Journal of Water, Air and Soil Pollution*, 9: 421-430.

Ashwath N and Venkatraman K (2010). Phytocapping: An alternative technique for landfill remediation. *International Journal of Environment and Waste Management*, : 51-70.

Venkatraman, K. and Ashwath, N. (2010) Field performance of a phytocapped at Lakes Creek Landfill, Rockhampton, Australia, *Management of Environmental Quality Journal*, 21: 237-252.

Sunlight is the main source of energy for trees to transpire, as this process involves the flow of water against gravity (Eamus *et al.* 2006). The amount of solar radiation incident on top of a canopy varies from the minimum (or zero) in the night to a maximum at noon. The transpiration rate and rate of evaporation would be expected to follow the same pattern. However, this is not the case due to "the resistance to water flow that exists between soil and leaf" (Eamus *et al.* 2006). The presence of this resistance results in a time lag between increasing transpiration and increasing rates of water uptake by the roots. In most cases, transpiration increases as the sun rises and decreases by late afternoon, as the sun starts setting. However, resistance in the xylem and leaves does not allow transpiration to take place in the early part of the day in many instances, and it increases during the latter part of the day (Eamus *et al.* 2006).

Night time transpiration of 0.8 mm was recorded by Benyon (1999) during a study at Wagga Wagga (NSW), and this could make a significant contribution to the overall water use of the trees especially during dry season. TDP sensors do not record night transpiration. There are numerous reports of water loss at night. Sapflow measurements indicate that night time loss ranges from 5 to 30% of daily water loss in *Actinidia, Eucalyptus, Malus, Populus, Prosopis, Salix, Taxodium***,** and *Dipterocarp* (Green *et al***.** 1989, Cleverly *et al***.** 1997, Hogg and Hurdle 1997, Becker, 1998, Benyon 1999; Oren *et al***.** 2001). For example, *Arabidopsis, Betula, Brassica***,** *Chrysothamnus, Fraxinus, Picea, Rosa, Tarcobatus and Tilia* have substantial night time water loss (Whitlow *et al.* 1992, Wieser and Havranek 1993, Matyssek *et al***.** 1995, Assaf and Zieslin 1996, Donovan *et al***.** 1999, Lascève *et al***.** 1999). Seasonal changes contribute to the change in transpiration rates. Dye (1995) reported an average uptake of 30 L d^{-1} during winter and around 90 L d^{-1} during peak summers by a *E. grandis* tree that was 14.7 m tall with a diameter of 147 mm. Kalma *et al.* (1998) reported an average consumption of 14.5 L d^{-1} and 10 L d^{-1} in a five year-old *E. grandis* trees in Toolara, near Brisbane, which had a height of 12.8 m and 12.9 m respectively. Similar findings were reported by Dye (1996) for an *E. grandis* tree and Soares and Almedia (2001) for a eucalyptus plantation in Brazil. They reported 1.1 to 5.8 mm d^{-1} (9 year-old), 4 mm d^{-1} (5 years-old) and 2 to 4 mm d^{-1} (9 year-old) respectively. Benyon (1999) predicted 1 to 2 mm d^{-1} of water consumed by *E. grandis* under well-watered conditions.

Leaf temperature, a function of the amount of solar radiation received by the leaf, also affects transpiration rates (Eamus *et al.* 2006). Leaf temperature increases with leaf size (Eamus *et al.* 2006). For example, casuarinas, which have needle, shaped leaves, show very small surface areas as compared to broad leaved species (such as eucalypts). Transpiration rates decrease with a reduction in soil temperature from 45°C to 11°C (Clements and Martin 1934). Transpiration rates of whole plants are also influenced by the soil moisture potential (Forde *et al.* 1974), the extent of soil volume explored by the root system (Kramer 1969), the architecture of the tree (Pruitt *et al.* 1972), and anatomical and physiological features of the tree (Bjorkman 1971). Transpiration rates vary among species (Goldstein *et al.* 1998), season (Benyon *et al.* 1996), soil water availability, leaf area (Ryan *et al*. 2000), leaf biomass (Worledge *et al.* 1998), climatic conditions, root development, the age of plantation (Roberts *et al.* 2001) and geographical region (Vose *et al.* 2003). It is also influenced by shallow water tables (Landsberg 1997). Various studies have shown that transpiration rates of different trees are largely influenced by LAI (Eamus *et al.* 2006, p 109). Transpiration is also influenced by an increase in stem diameter at DBH and root growth (Eamus *et al.* 2006).

The ability of plants to acquire water depends on root distribution (Jarrell *et al.* 1990) which also depends on the above ground responses of plants such as leaf area and leaf biomass (Ryan *et al*. 2000). Tree roots respond quickly to rain (Ansley *et al.* 1989) and the degree to which lateral roots influence water uptake may relate to the availability of water in the soil (Ansley *et al.* 1990). At first, the trees may use the rainwater stored in the upper profile of the soil, followed by streams adjacent to the site, and lastly the available groundwater (Eamus *et al.* 2006).

Water uptake in any system can be determined by methods based on pan evaporation and the Penman-Monteith equation (Milne *et al.* 1984), soil-water balance (Allen *et al.* 1998), lysimeters (Weight 1971), portable gas exchange chambers (Reicosky 1990) and heat pulse methods (Hatton *et al.* 1995). However, sapflow and sap velocity have been extensively used by researchers, scientists and engineers (Lundblad *et al.* 2001) and the values are comparable to those of other methods of estimating transpiration, such as the heat pulse method (Saugier *et al.* 1997). The Thermal Dissipation Probes (TDP) (Huber 1932, Cohen *et al.* 1981)

developed by Granier in the 1980s (Granier *et al.* 1996) require careful installation of the sensors within the sapwood while avoiding the heart wood (James *et al.* 2002).

In the TDP method, the xylem sap is heated continuously; unlike in the heat pulse method where heat is applied as a pulse at regular intervals (typically 1 to 100 seconds; depending on the temperature of the sap) (Belby *et al.* 2004, Eamus *et al.* 2006). Burgess *et al.* (2001) further developed an improved heat pulse technique known as the Heat Ratio Method (HRM) to measure low and inverse transpiration rates in woody plants. This method can quantify low levels of transpiration as well as night time transpiration in woody plants. This method also allows monitoring of water flows in stems and roots of a wide range of species and stem sizes under varied environmental conditions.

In the Heat Balance Method (HBM) or Dynagauges), sensors are generally wrapped around the stem (Vieweg and Zielgler 1960) and a small quantity of heat is applied continuously to raise the temperature of the stem (Braun 1997). This method has been successfully adopted by Sakuratani (1981), Velancogne and Nasr (1989) and Weibel and Vos (1994). The commercial version of the Dynagauges (Braun 1997) (Dynamax, Inc, Houston Texas, USA) was used in this study, with a few dynagauges and HRM probes used for comparison.

A TDP sensor consists of two needles that are inserted into the sapwood of the tree at a fixed distance of 40 mm between the two needles. A copper constantan thermocouple is located within each needle at half way or 15 mm from the base of each needle. The needles have a Teflon coating to assist in the removal from the stem. The needles are usually inserted into the stem at around 1 m height. One needle is inserted at the lower position (reference needle) and the other at the upper position. The upper needle contains a fixed line heater that is constantly heated. When the sapflow occurs, the heat produced by the upper needle will be diluted by the sapflow. When this occurs the upper needle produces more heat as it tries to maintain constant temperature. Thus, the current required to maintain a pre-determined heat will be measured and this will be correlated with sapflow, after correcting for the sapwood volume.

The HRM sensor consists of three 30 mm long needles (probes) integrally connected to a 16-bit microprocessor. The top and bottom probes each contain a set of two very fine copper constantan thermocouples placed at 7.5 mm and 22.5 mm from the tip of each probe. The third and centrally located probe is a line heater that runs the full length of the probe to deliver a uniform pulse of heat through the sapwood. In the HRM technique, three probes are inserted into the sap wood, such that the middle probe releases a pulse of heat, and the probes located above and below will record the heat dissipated from the central probe. Thus, the ratio of the heat dissipated from the central probe to the two symmetrically placed temperature sensors will determine the magnitude and direction of water flux. The rate of dissipation of heat is proportional to the rate of sapflow and this will be corrected for sapwood volume, and expressed in mm per tree per day. The raw data were imported to an excel spreadsheet and the sap velocity readings from individual trees were multiplied by the sapwood area of each tree to obtain sapflow in mm d^{-1}. Wounding coefficient is the value derived from the finite element model to determine the effect of wounding on sap velocity (Burgess *et al.* 2001).

The TDP and HRM techniques are automated (Smith and Allen 1996), moderately invasive (Marshall 1958) and are widely used in transpiration and water relations studies of woody plants (Nadezhdina and Cermak 2003). The TDP technique was developed to overcome limitations encountered by dynagauges in measuring radial profiles of sapflow in large diameter woody trees having deeper layers of sapwood (James *et al.* 2002). Sapwood depth varies widely between species (Whitehead and Jarvis 1981; Wullschleger *et al.* 1998) and can be identified based on xylem water content and/or colour (Nadezhdina *et al.* 2002). The TDP sensors are extensively used in various fields (Braun 1997, Lundblad *et al.* 2001) and are preferred because of their simplicity, low energy requirement, accuracy and low cost (Andrade *et al.*1998, Braun and Schmid 1999). Granier's (1987) TDP sensors have several advantages, and these include: i) the lack of a requirement to calibrate sensors for each species and ii) a more representative measurement of the sapflow flux density through integration of flux density along the length of the probe (Lu 1997, Lundblad *et al*. 2001, Kucera and Tatarinov 2003, Tatarinov *et al*. 2005).

Developed at the University of Western Australia, Perth, Australia, the HRM sensors were validated against gravimetric measurements of transpiration, and have been used in published sapflow research since 1998 (Burgess *et al.* 2001). To the best of our knowledge this is the first time simultaneous performance of the two probes has been tested. This study also compares the performance and accuracy of HRM and TDP in estimating transpiration losses.

The ability of Australian native species to transpire when established on a landfill has not been evaluated before. Nevertheless, this information is critically important to determine the water balance of phytocapped landfill sites. Thus, TDP sensors, HRM sensors and dynagauges were installed in a range of native species that were grown on a phytocapped landfill. This Chapter focuses on species differences in transpiration rates, seasonal variability and the difference between probes in determining sapflow.

5.2 Materials and Methods

Details of methods used in establishing the phytocapping trial are given in Chapter 2. In summary, fifteen species with stem diameters of more than 50 mm (minimum requirement to install sapflow sensors) were selected for this study. Sapflow measurements were recorded for fifteen species grown in Thin phytocap, due to practical constraints. From nine planted trees of each species in the experimental plot, a representative tree was selected depending on the stem diameter. Among the fifteen species selected at each batch of the study (5 to 30 days per batch), sapflow in fourteen species were recorded using TDP sensors (Fig 5.1) and sapflow in *D latiflorus* was recorded using a dynagauge (Filho *et al.* 2005). During installation, the bark was removed until the sapwood was visible. Two 1.5 mm holes were drilled 40 cm apart (Fig. 5.1A). Hydrogen peroxide was applied to the holes to restrict growth of the wood and to allow easy removal of the probes on completion of monitoring. The sensors were then installed carefully and covered with polystyrene shields (Fig. 5.1B) to avoid any damage to the needles during high winds. The sensors were then wrapped with aluminium foil (Fig. 5.1C) to minimise the effect of the external environment on sensor readings. Each sensor was connected to a smart

logger via a monibus bar, and the sensors were powered by a 12V DC external battery which was continuously charged by an 80W solar panel (Fig. 5.2). Sap flow measurements were recorded for 24 hours on an hourly basis to check the functionality of the instrument. For bamboo, the dynagauges (collar sensor) was wrapped round the stem and wound tightly with the aid of velcro strips. A total of 49 observations were taken using various tree species, with each measurement ranging from 5 to 30 days.

Figure 5.1: Installation of a TDP sensor
A) TDP sensor installed, B) probes protected with polystyrene shield, and C) set up sealed and covered with aluminium foil

Figure 5.2: Power and sensor system used to measure sapflow
A) solar panel, B) smart logger, C) monibus bar and D) battery

Sapflow was calculated from the measured sap velocity data. The raw data were imported to an Excel spreadsheet and the sap velocity readings from individual trees were multiplied by the sapwood area of each tree to obtain sapflow in mm d^{-1}. Parameters such as sapwood depth and wounding coefficient were used to calculate sapflow in individual trees (ICT International, 2007). The sapwood area was determined by destructive methods, when the selected trees were felled for biomass measurements in 2006. This was repeated in 2007. In the first instance, alternate trees from each Thick and Thin phytocap (nine trees) and from both replications were harvested. Of 9 trees harvested, 3 were studied for their above ground biomass, and the remaining 6 were harvested and measured for their D_{50}, DBH, sapwood depth and height. In 2007, 3 trees per species (representative of the entire stand) in Thick and Thin phytocaps and from both replications were selected and studied for their above ground biomass, and the rest were measured for their height, D_{50}, DBH and sapwood depth. Sapwood depth measurements were taken at three locations (Fig. 5.3) and the average of these was used to calculate the sapwood area (Fig. 5.4) according to the method of Sharma (2008), as shown.

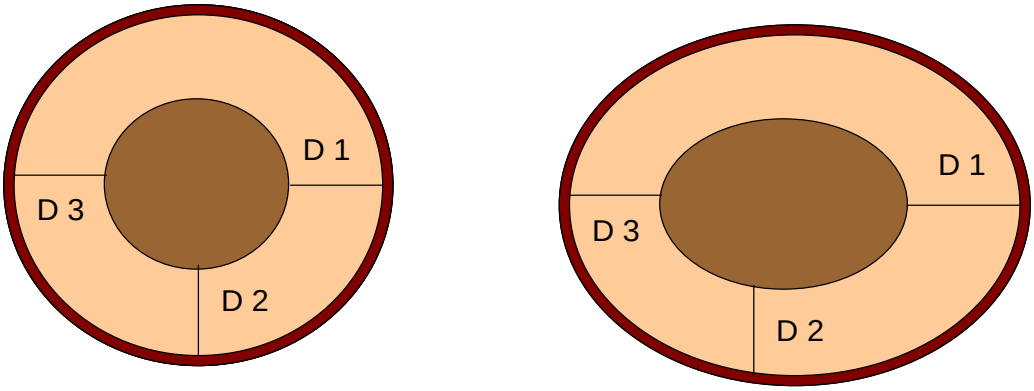


Figure 5.3: Sapwood depth measurement in cylindrical and non-cylindrical stems

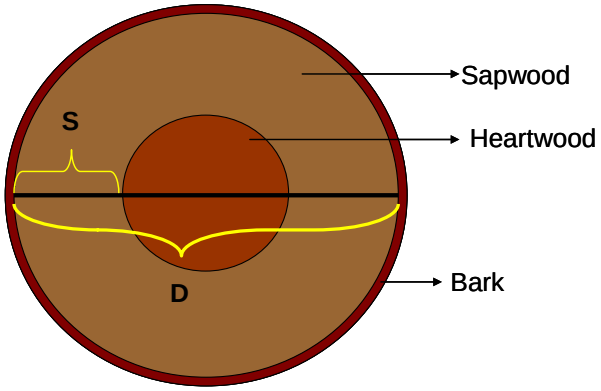


Figure 5.4: Cross section of a typical tree trunk
(S = sapwood depth, D = heartwood diameter)

Sapwood area = Cross sectional area of the stem (excluding bark) – heartwood area

$A = л\ (D/2)^2 – л\ (D/2 - S)^2$

$= л\ \{D^2/4 – (D^2/4 + S)\ 2)\}$

$A = лS\ (D-S).$

Where D is under bark diameter of the tree and S is the sap wood depth

5.2.1 Comparison Between HRM vs. TDP Sensors

Six *E. grandis* trees that had similar stem diameter and height were selected for the study. From the six selected trees, three randomly selected trees were installed with TDP sensors and the other three were installed with HRM sensors.

HRM sensors were installed using similar procedures as for the TDP sensor (Burgess *et al.* 2001) (Fig. 5.5). The installed sensors could cause mechanical

damage or they may interrupt flow by occlusion or blocking of the plant's vascular tissues (ICT international, 2007), resulting in growth of non-conducting tissues directly surrounding the probe. This type of growth, if any, was corrected to achieve accurate results. Wound correction coefficients applicable to a range of wound sizes were generated using numerical models to obtain accurate values (Alec Downey, personal communication). These corrections can be implemented either automatically or they can be introduced manually after collecting the data of raw heat pulse velocities (Alec Downey, personal communication).

Figure 5.5: Installation of HRM sensor

5.2.2 Testing Sensor Accuracy

Three *G. lobocarpum* were carefully excavated from the phytocap and transferred into large planter bags which were then filled with soil collected from the same field and placed on wooden pallets (Fig. 5.6). The planter bags were then mulched to reduce soil evaporation. TDP sensors were installed in each tree and then connected to the smart logger. After 4 weeks of establishment, the initial weight of each pot was taken using a pallet scale. A known amount of water (up to 30L) was added to the pots and they were re-weighed. The pots were then weighed after 24 hours. The difference in initial and final weight was used to estimate the water taken up by each tree in 24 hours. Simultaneously, sapflow readings as determined by TDP sensors were calculated, assuming that the soil evaporation from the mulched pot was minimal and uniform amongst the three tested plants.

Figure 5.6: Transplanted *Cupaniopsis anacardioides* saplings placed on a pallet scale

5.2.3 Soil Moisture Determination

Soil moisture in this study was monitored using micro-gopher, which has a logger and 1 m long (< 25 cm diameter) calibrated (10 cm intervals) rod with a sensor at the tip of the rod. 84 gopher access tubes made of PVC were installed throughout the experimental plot (21 gopher access tubes per experimental plot). Each gopher tube had a diameter of 25 mm was 1.2 metre long. The bottom end of the access tubes was sealed and the top end was capped while not in use (Fig. 5.7). During measurement the micro gopher rod was inserted into tubes upto 1 m each time, with 10 cm interval. Soil moisture levels at 10 cm increments were taken from each access tube. Monitoring of soil moisture was continued at regular intervals (every month). As discussed, changes in soil moisture were recorded at 10 different depths for each of the 84 tubes in the plot. The results in soil moisture were related to plant growth and their capacity to transpire.

Figure 5.7: Micro Gopher access tube

5.3 Statistical Analysis

The sapflow data were subjected to ANOVA using Genstat ver. 8.0, after testing for outliers and homogeneity of error variances. Least significance differences were used when ANOVA tests for species, capping, season or their interactions were found significant. The effects of various tree parameters on transpiration rates were assessed using regression equations (GraphPad Prism v 4.03 and Genstat ver. 13) and the linear equation was chosen as it produced the highest r^2 values.

5.4 Results and Discussion

5.4.1 Transpiration Rates

Fifteen of the 21 species were tested for transpiration rates (over 2 years) as they were the only species that grew to 50 mm diameter (minimum requirement for installing TDP or HRM sensors). Transpiration rates ranged between 0.9 mm d^{-1} to 2.1 mm d^{-1} (average of 49 observations), with an overall average (for all species and all seasons over 2 years) of 1.4 mm d^{-1} (Fig. 5.8). *Acacia mangium*, *H. tiliaceus*, *C. cunninghamiana* and *E. raveretiana* had high transpiration rates (2 mm d^{-1}) (Fig. 5.8). Transpiration rates monitored over 2 years were as low as 0.1 mm d^{-1} and as high as 6.25 mm d^{-1} (Table 5.1). This large range in transpiration rates in individual species may be due to variations in growth rates amongst seasons, rainfall, temperature, wind velocity, vapour pressure deficit and solar radiation as explained by Eamus *et al.* (2006). These variations demonstrate the ability of the species to

transpire copiously during high rainfall period and very little during dry seasons. Such behaviour in plants is highly sought after for species to be grown of phytocaps, as these systems lack access to sub soil moisture unlike those present on natural landscapes.

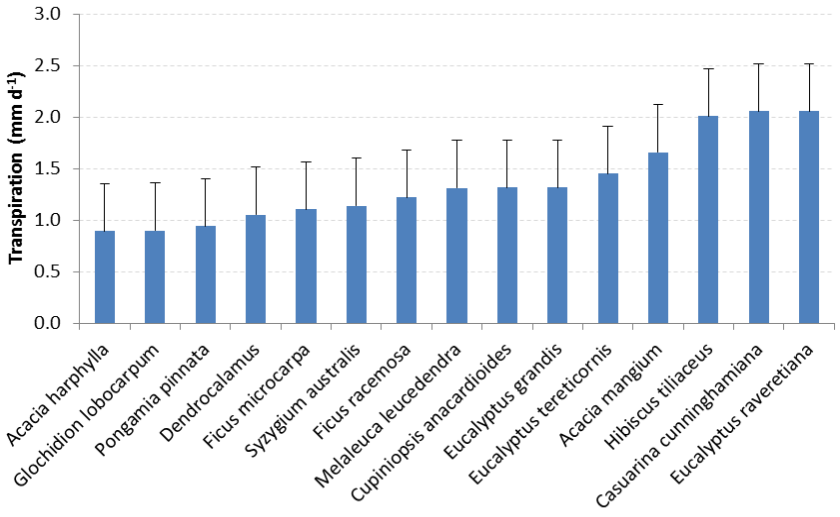


Figure 5.8: Transpiration rates of 15 species grown on a phytocap for 2 years (49 observations) (Bars represent l.s.d. 0.46)

Table 5.1: Range in transpiration rates of 15 species grown on phytocaps over a 2 year period (n = 49)

Species	Transpiration (mm/day)
Acacia harpophylla	0.35 - 2.69
Acacia mangium	0.45 - 4.0
Casuarina cunninghamiana	0.36 - 3.93
Cupaniopsis anacardioides	0.6 - 5.26
Dendrocalamus latiflorus	0.36 - 4.5
Eucalyptus grandis	0.22 – 4
Eucalyptus raveretiana	0.35 - 3.7
Eucalyptus tereticornis	0.28 - 4.0
Ficus microcarpa	0.26 - 4.0
Ficus racemosa	0.2 - 2.7
Glochidion lobocarpum	0.1 - 1.53
Hibiscus tiliaceus	0.36 - 6.25
Melaleuca leucadendra	0.2 - 2.67
Pongamia pinnata	0.1 - 2.64
Syzygium australis	0.3 - 2.56

Trees grown on phytocaps varied significantly ($P<0.01$) in their transpiration rates (Fig. 5.8) and this can be attributed to differing performances of these species in different seasons associated with variations in soil moisture regimes (Benyon *et al.* 1996), leaf area (Ryan *et al*. 2000), leaf biomass (Worledge *et al.* 1998, Roberts *et al.* 2001), solar radiation (Collatz *et al.* 1991, Vose *et al.* 2003) and root development. Although all species experienced the same climatic conditions and seasonal change, they differed largely in their LAI, leaf biomass and root development (Chapter 3). Furthermore, factors such as soil water availability and solar radiation reaching the tree canopy would have varied seasonally and at different times during the day. Variation in transpiration rates between species and among trees of the same species was due to height increments within stand, as the slow growing species were shaded by the faster growing/dominating species, resulting in them acquiring minimum sunlight and energy to transpire.

In many cases, canopy rainfall interception plays a vital role in controlling soil moisture levels (Benyon *et al*. 1996). However, in this study, no significant effect of canopy interception on soil moisture levels both in Thick and Thin phytocaps was noticed. Variations in the quantity of leaf litter produced by the 15 species (Chapter 3) may also have an influence on the soil moisture levels over small distances. However, the differences in soil moisture levels were prominent due to rainfall interception by the tree leaves and leaf litter that may have reduced the quantity of water reaching the soil. Similarly, variation in the height of species due to competition for sunlight and other resources greatly influenced transpiration rates.

5.4.2 Diurnal Variation in Transpiration Rates

Diurnal variation in water uptake depends on the incident solar radiation on the canopy (Eamus *et al.* 2006). The diurnal pattern in two 3-year-old *E. grandis* trees grown on the phytocaps with a stem diameter of 107 mm and 105 mm respectively is shown in Figure 5.9. All species tested showed similar transpiration pattern with close to zero or zero at night increasing to a maximum at noon. As explained by Eamus *et al.* (2006), solar radiation increases to the highest level at midday (Fig. 5.10) and hence the transpiration rates are expected to follow the same pattern. However, this is not the case in many species due to a time lag in the water

movement due to resistance to water flow between the soil and the leaves (Eamus *et al.* 2006). There was a time lag in water uptake by *E. grandis* grown in the phytocapping system (Fig. 5.11).

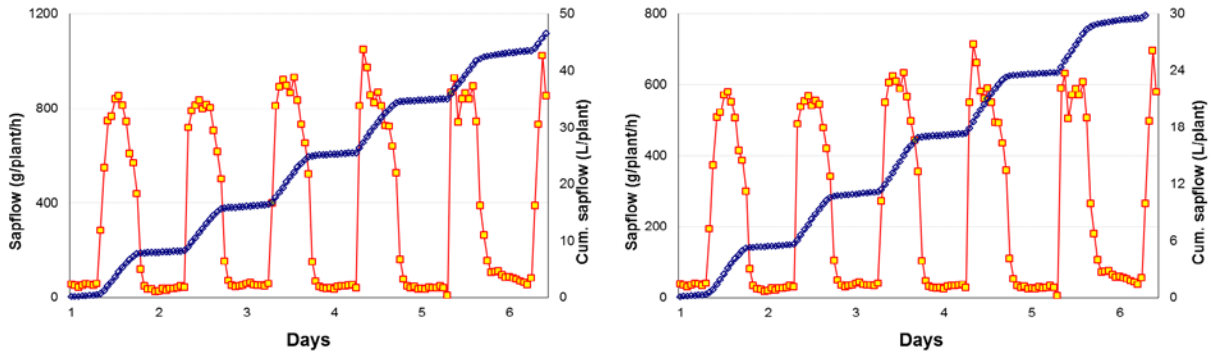


Figure 5.9: Diurnal variations in transpiration rates of two 3-year-old *Eucalyptus grandis* trees

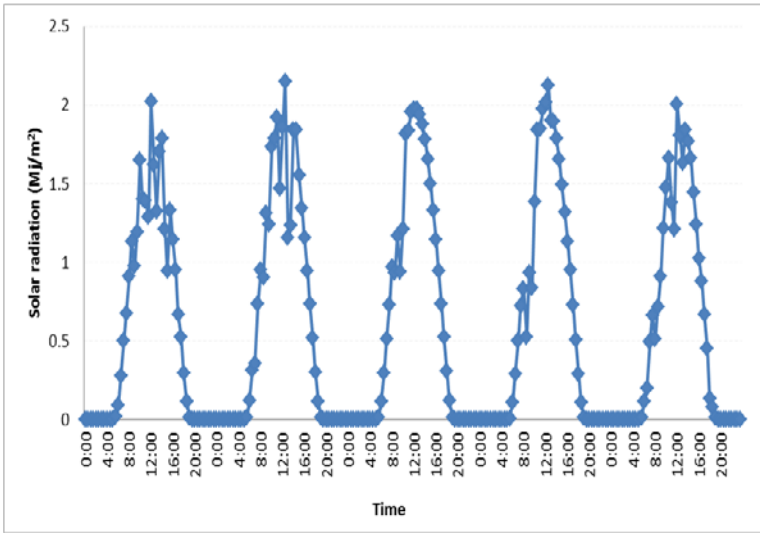


Figure 5.10: Solar radiation observed in January 2006 over five days (1/01/2006 – 5/01/2006)

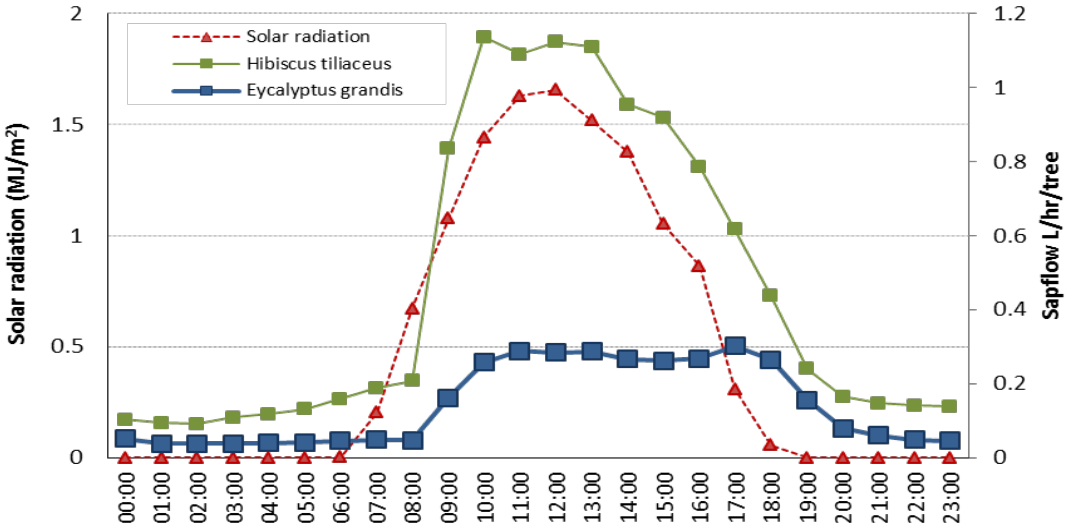


Figure 5.11: Solar radiation and associated transpiration pattern in *H. tiliaceus* and *E. grandis* observed in April 2007

In this study, the influence of seasonal changes on transpiration rates has not been clearly understood because of the lack of continuous monitoring of the same tree over several seasons. However, data of some species have showed their ability to respond to wet and dry cycles. For example data of *H. tiliaceus* shows its ability to adapt to site moisture conditions by transpiring as high as 15 L d^{-1} after a rainfall event and as low as 0.4 L d^{-1} during dry periods (Figs. 5.12 and 5.13). Similar trends were observed by Eamus *et al.* (2006) in *E. grandis* and *E. globulus* in the Victorian climate. *E. grandis* was able of take up 0.89 mm d^{-1} of water. *E. globulus*, on the other hand, transpired 2.2 mm d^{-1} during late spring (rainy season) and only 0.33 mm d^{-1} during summer (drought). This behaviour of the species is extremely important in phytocapping, as seasonal availability of water in Australia is highly variable. This is illustrated in *H. tiliaceus* (Figs. 5.12 and 5.13). A medium size tree such as *H. tiliaceus* could survive in both wet and dry cycles, and this demonstrates the capacity of the species to both persist on the site during drought and rapidly remove water during the rainy season.

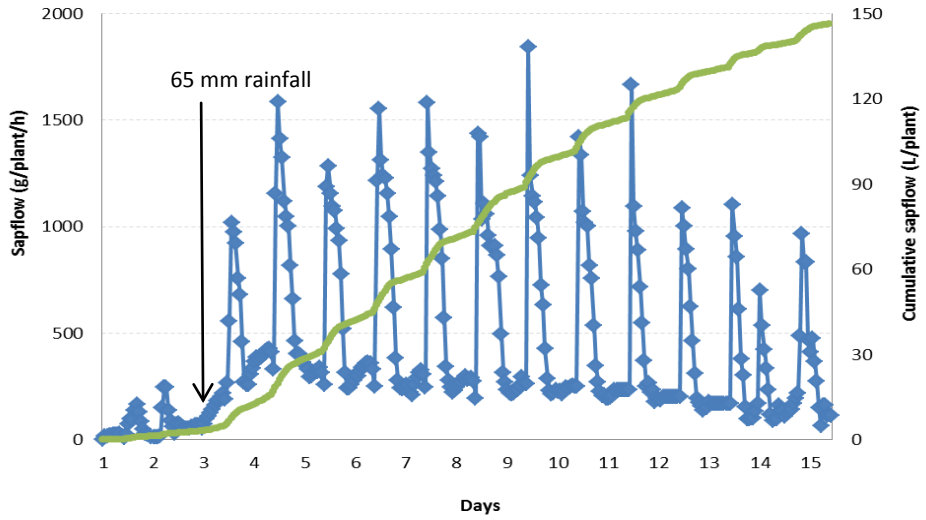


Figure 5.12: Diurnal transpiration pattern in *Hibiscus tiliaceus* during a rainfall event in January 2005

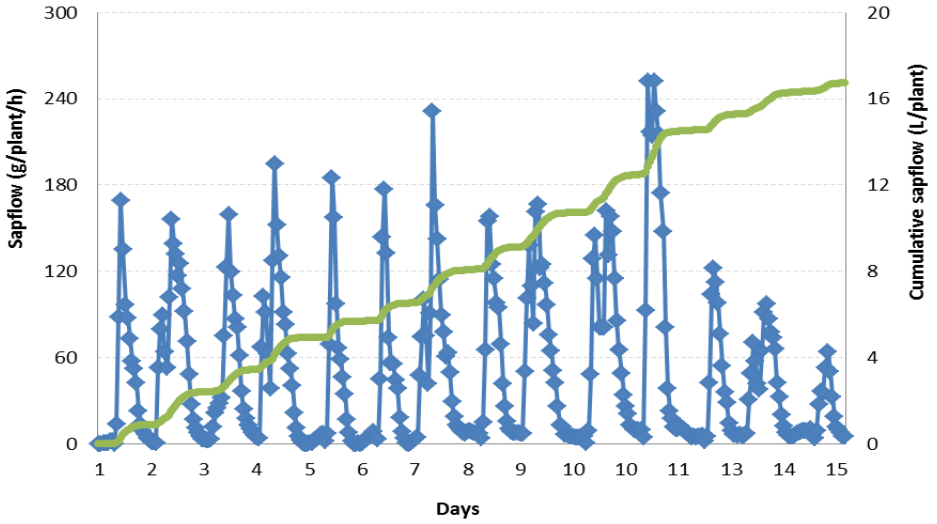


Figure 5.13: Diurnal transpiration pattern in *Hibiscus tiliaceus* during a dry period in May 2005

Transpiration rates in bamboo ranged between 0.36 mm d^{-1} to 4.5 mm d^{-1} (Table 5.1). Results from twelve observations spanning over 18 months (July 2005 to Dec 2006) suggest that the bamboo transpired on an average 1.2 mm d^{-1} (Fig. 5.8). These patterns further illustrate that transpiration increases during wet seasons, maintains averages during normal seasons and declines severely during dry seasons. Similar results have been reported by Li *et al.* (2002) in corn and by Katul *et al.* (1997) in oak.

5.4.3 Species Response to Rainfall

How quickly a tree responds to each rainfall event is important in judging the suitability of a species to be grown on phytocaps. Thus a short experiment was conducted in June 2006 on a bright sunny day. Six species that were installed with sapflow sensors were irrigated (100 L) between 7.00 am and 7.30 am and their response to this irrigation was monitored. Most species were able to take up water within the first two hours of irrigation (Fig. 5.14). A few species such as *A. harpophylla, A. mangium* and *G. lobocarpum* responded to the changed conditions and were able to take up water within one hour of irrigation (Fig. 5.14). *M. leucadendra* showed the unique trend of a very high uptake followed by a steep decline (Fig. 5.14), which could be attributed to environmental factors such as insects, wind and/or fluid in the wound affecting the sensor. The other species showed a sharp increase in water uptake followed by a gradual decrease over 4 to 5 hours. This immediate response to rainfall events is very important in maintaining the hydrological balance of the phytocaps. Similar research is needed over a longer term to test the inherent ability of the species to respond to rainfall and drought. These results clearly suggest that the trees grown on this phytocapping system have the ability to adapt and respond well to frequent wet and dry cycles thereby taking up water quickly and avoiding excess water flowing through the soil layers into the buried waste.

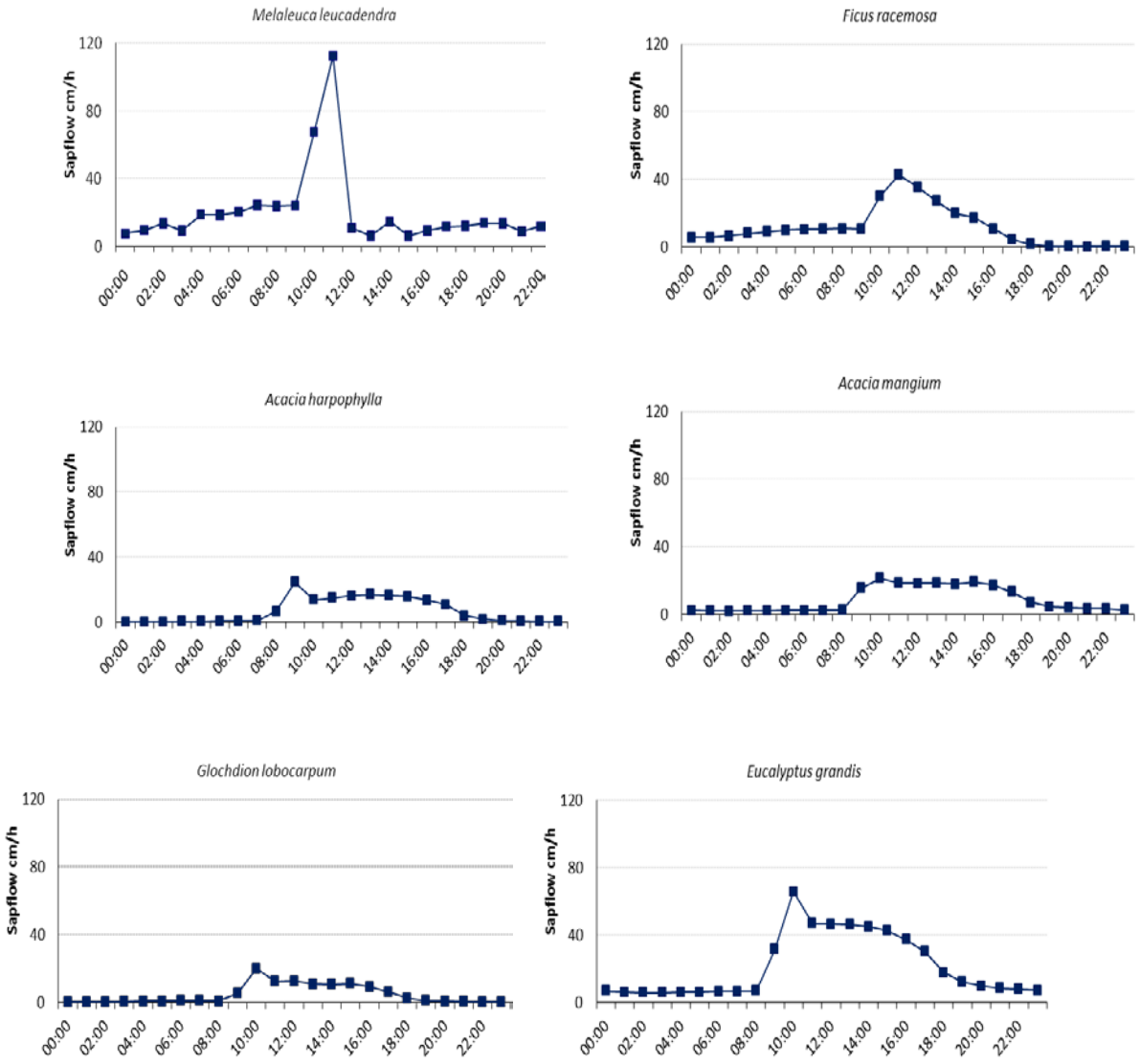


Figure 5.14: Response to irrigation by 3-year-old trees grown on the Thin phytocap

Another experiment conducted in early 2007, during the wet season when a rainfall event (37.5 mm) occurred after a dry period, is shown in Figure 5.15. Sapflow patterns were monitored for 3 consecutive days following the rainfall event. The results showed an increase in transpiration rates in *H. tiliaceus*, *P. pinnata*, *E. raveretiana* and *Ficus macrocarpa* after the rainfall event on the fourth day (Fig. 5.15). Similar trends were observed in a number of other species. This indicates that the trees grown in the phytocapping system rapidly enhanced their transpiration rates within hours of a rainfall event; thereby removing stored water from the soil layer. The extent to which these remove water will show superiority of one species over the other. This concept was explained by Ansley *et al.* (1990) in honey mesquite in Vernon, Texas. This cyclic nature of trees to increase, maintain and lower transpiration rates in response to rainfall and moisture limitation is critically important, not only for judging the effectiveness of the phytocaps, but also for the long-term survival of the species on the phytocaps.

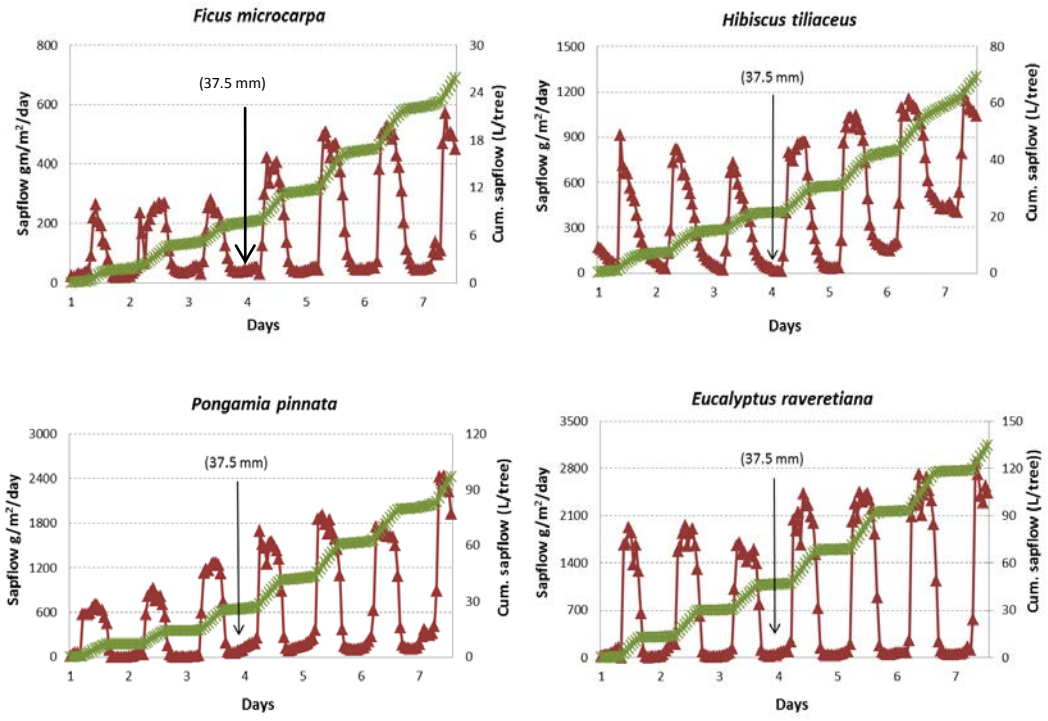


Figure 5.15: Water uptake by four species after a 37.5 mm rainfall event

5.4.4 Factors Influencing Transpiration

Transpiration rates are determined by the size of the plant and its potential to transpire water rapidly. Hence the observed values of transpiration were correlated with growth parameters. Transpiration rates showed a significant correlation with tree height, DBH, D_{50}, LAI and shoot biomass (Table 5.2). Previous studies have demonstrated positive relationships between transpiration rates and tree height or stem diameter (Reich *et al.* 1997, Liu *et al.* 2008). Biomass accumulation also directly correlates with water uptake (Therakan *et al.* 2000, Singh and Bhati 2003). Taller trees have a greater canopy exposure to solar radiation, allowing them to increase in stem diameter (Kammesheidt *et al.* 2003).

In a mixed stand, competition exists between species, and among trees of the same species for resources such as light, water and food. Fast growing trees such as *A. mangium*, *H. tiliaceus*, casuarinas, bamboo and eucalypts grew more than 6 m tall in 3 years (Chapter 3) and most species grew over 2 m (Chapter 3). *Callistemon viminalis* and *M. linariifolia* were the only two species that showed very slow growth rate. Similar findings were reported by Wright and Westoby (1999). This large variation in growth rate is partially genetic and partially due to competition for light between tall and short species (Tilman 1988, Herwitz *et al.* 2000).

Table 5.2: Correlation between transpiration rates and various tree parameters for the 15 species grown on Thick and Thin phytocaps

Parameter	r^2 value
Height	0.54**
DBH	0.50**
D_{50}	0.55**
Shoot biomass	0.38*
Root Biomass	0.076^{ns}
Root depth	0.07^{ns}
LAI	0.40*
Canopy area	0.18^{ns}
Leaf Area	0.051^{ns}

Note: ***; P <0.001, **; P <0.01 and *; P <0.05

Transpiration in the 15 species tested varied significantly ($P<0.001$) between seasons (Fig. 5.16) primarily due to moisture availability, solar radiation, leaf area and season. Transpiration was higher during the low rainfall period (April to October) than during the high rainfall period (November to March). This could be attributed to high evaporation during low rainfall periods and fluctuation in other environmental factors such as VPD and solar radiation during rainfall events (Eamus *et al.* 2002, Medhurst *et al.* 2002).

Differences in transpiration between species and those between individuals of the same species are expected to reduce with tree height and age as explained by Ryan and Yoder (1997). Such differences may be caused by differences in hydraulic resistance of the soil-to-leaf pathway, which may be due to the variations in the path length (stem height) (Walcroft *et al.* 1996) and the sapwood permeability (Mencuccini and Grace 1996).

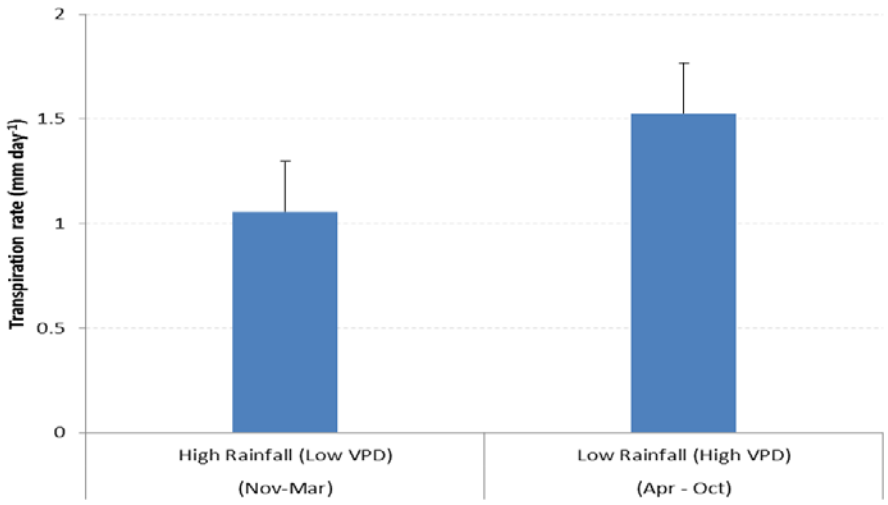


Figure 5.16: Effect of season on transpiration rates
(Bar represents l.s.d. 0.242)

The amount of solar radiation increases from zero at night to the maximum during the day. Since solar radiation increases in the morning and decreases by afternoon (Price and Black 1990, Eamus *et al.* 2002), it is expected that the transpiration in trees would follow a similar pattern. However, resistance in water movement between the soil and the leaf does not allow transpiration to take place instantaneously and a time lag

between increasing transpiration rates and increasing water uptake by the roots was observed (Fig. 5.17). This pattern has been reported by Eamus *et al.* (2002) as well.

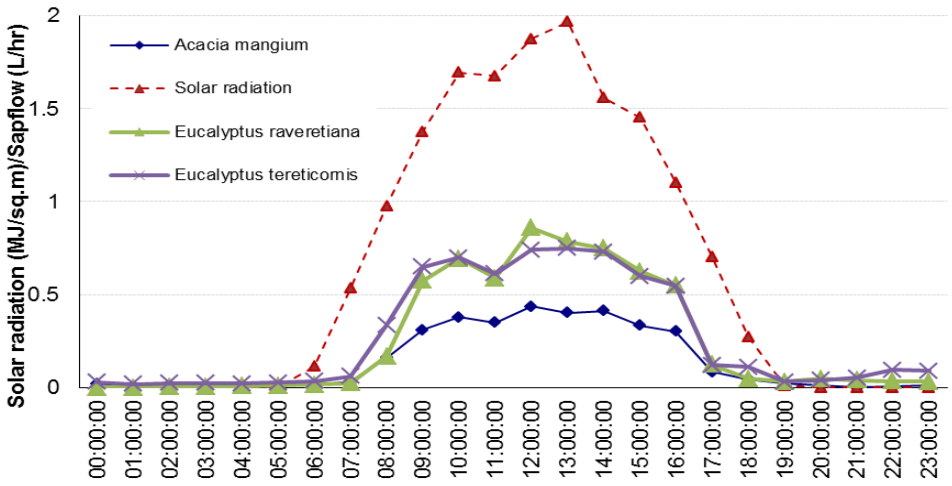


Figure 5.17: Effect of solar radiation on sapflow in 3 year-old *Eucalyptus raveretiana*, *Eucalyptus grandis* and *Acacia mangium* (March 2007)

Sapwood is the primary component of the stem that conducts water (Wullschleger and King 1999). The 3-year-old species differed in their sapwood depth (Fig. 5.18) which showed a strong correlation ($P<0.001$) with transpiration rates (Figs 5.18 and 5.19) as the quantity of water absorbed by the sapwood is influenced by its density and size (James *et al.* 2002). However, transpiration rates are expected to decrease with the age of the tree. As the trees age, the sapwood depth increases, thus decreasing density of the sapwood.

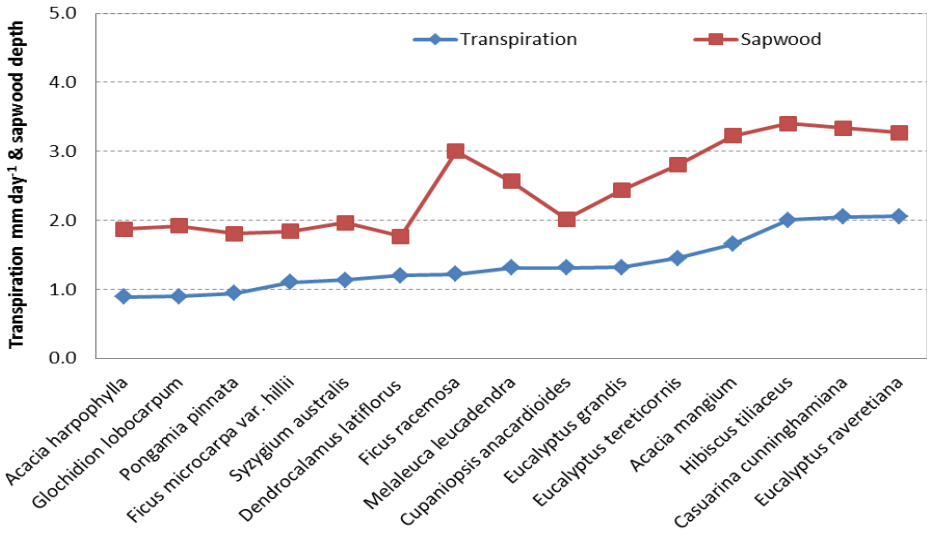


Figure 5.18: Variations in sapwood depth (cm) and transpiration in 15 species grown on a Thick and Thin phytocap for 3.5 years

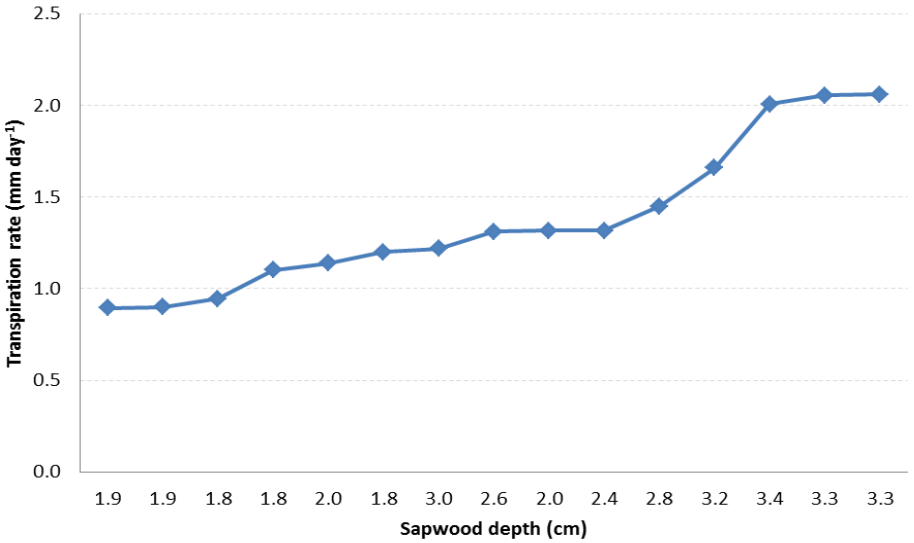


Figure 5.19: Relationship between sapwood depth and transpiration rate

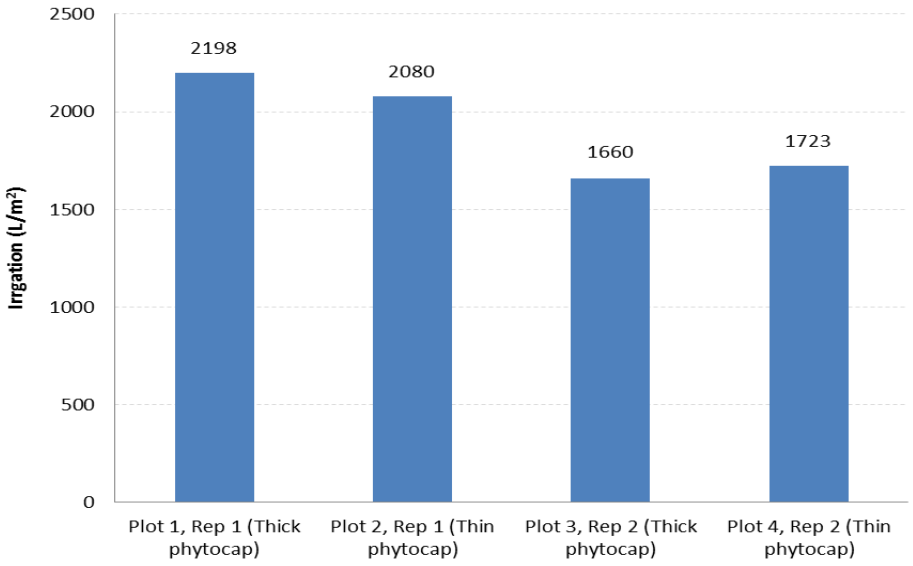


Figure 5.20: Water supplied (L/m^2) to various plots during the study (Total 3 years)

In the current study, trees grown in phytocaps were regularly irrigated (Fig. 5.20) during establishment (first 15 months from 2003 to 2004) and then when they showed severe wilting symptoms. The frequency of water supply was reduced after 15 months of planting (in 2005). Thus the plants were unlikely to be exposed to regular dry periods when they were monitored for transpiration rates. Higher transpiration rates during low rainfall periods may be due to higher VPD.

5.4.5 Effect of Transpiration on Soil Moisture Profiles

Average soil moisture content (100 cm depth) under each species varied significantly ($P<0.001$) between species and between Thick and Thin phytocaps (Fig. 5.21). This was primarily due to variation in rainfall pattern as clearly reflected in the soil moisture content and quick response of trees to take up water during rain events. Similar observations were made by Ansley *et al.* (1989) and Ansley *et al.* (1990).

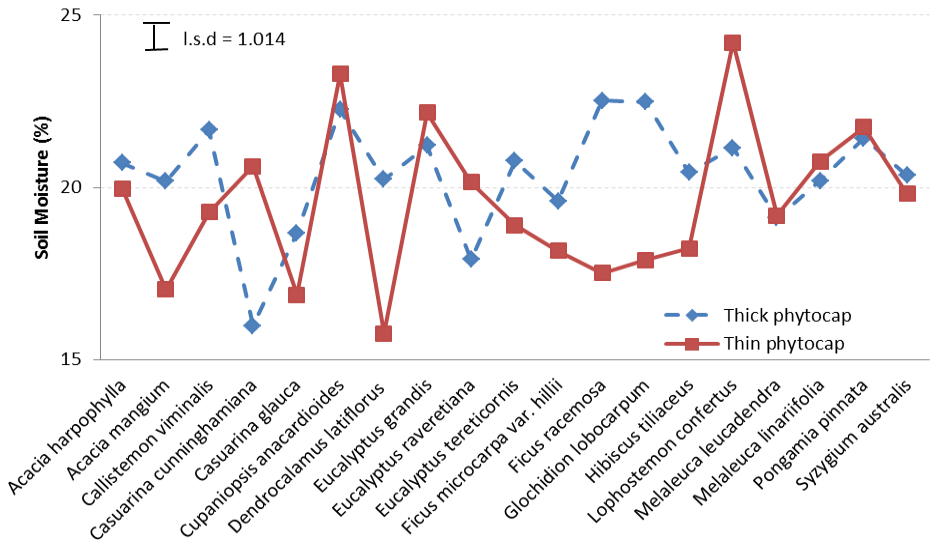


Figure 5.21: Variation in soil moisture content in the root zone of each species in Thin and Thick phytocaps (Average of 51 observations)

Soil moisture content varied significantly ($P<0.001$) between Thick and Thin phytocaps (Fig. 5.22) and was due to difference in plant growth rates (Chapter 3) and soil depth as well as the variation in the chemical and physical properties of the soils used in the two phytocaps.

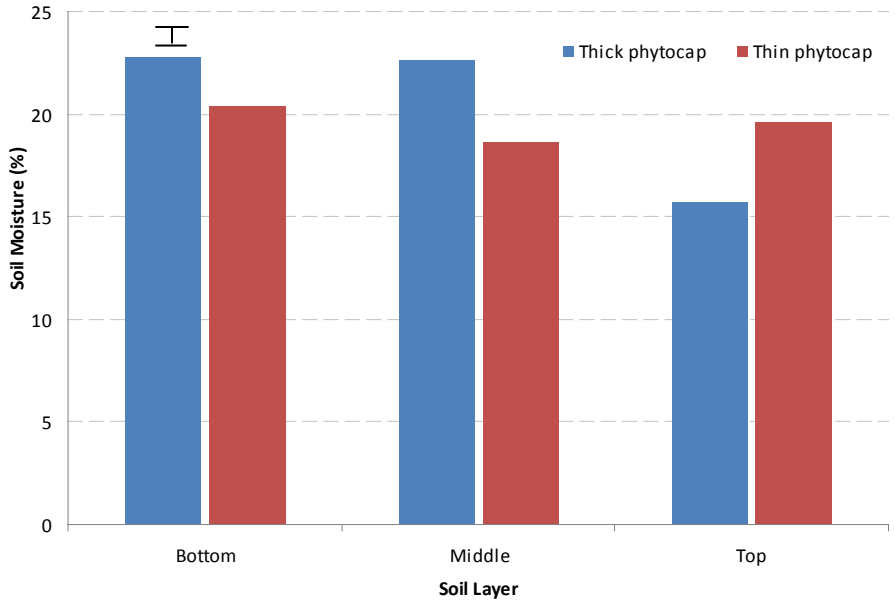


Figure 5.22: Soil moisture content at the top (0 – 300 mm), middle (301 mm – 600 mm) and bottom (601 mm – 900 mm) of the Thick and Thin phytocaps.
Bar shows the l.s.d for soil thickness and soil layer interactions (l.s.d. 2.32)

Soil moisture levels also varied significantly ($P<0.001$) with rainfall (Fig. 5.23), and this is because the potential evapotranspiration exceeds rainfall in Rockhampton (Chapter 2). In this situation, soil dries to the point where hydraulic conductivity becomes very low and any rainfall that occurs will wet the soil uniformly and is lost through transpiration and evaporation before any significant lateral redistribution takes place (Grayson *et al.* 1997). As evapotranspiration decreases and rainfall increases, the soil surface gets saturated and thereby generating more runoff. In the wet to dry transitional period, a rapid increase in potential evapotranspiration (and possibly a decrease in rainfall) causes drying of the soil (Grayson *et al.* 1997).

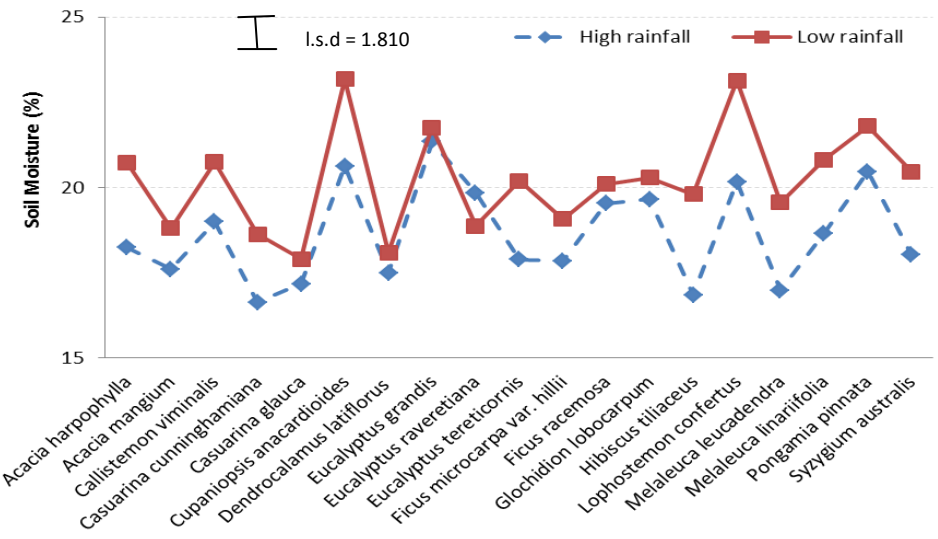


Figure 5.23: Soil moisture content in 100 cm depth soil during low and high rainfall events

Soil moisture data taken after a 16 mm rainfall event showed significant difference between phytocaps and the non-vegetated site. The difference in soil moisture content was 40% (Fig. 5.24), clearly showing the role that trees play in reducing water infiltration into the buried waste. Another test conducted to examine the effectiveness of phytocaps to reduce soil moisture levels after a rain event suggests that the species grown on phytocaps can transpire the water received within days; thus contributing to reduction in percolation (Fig. 5.25). Figure 5.25 also shows species that responded quickly to increase in soil moisture levels.

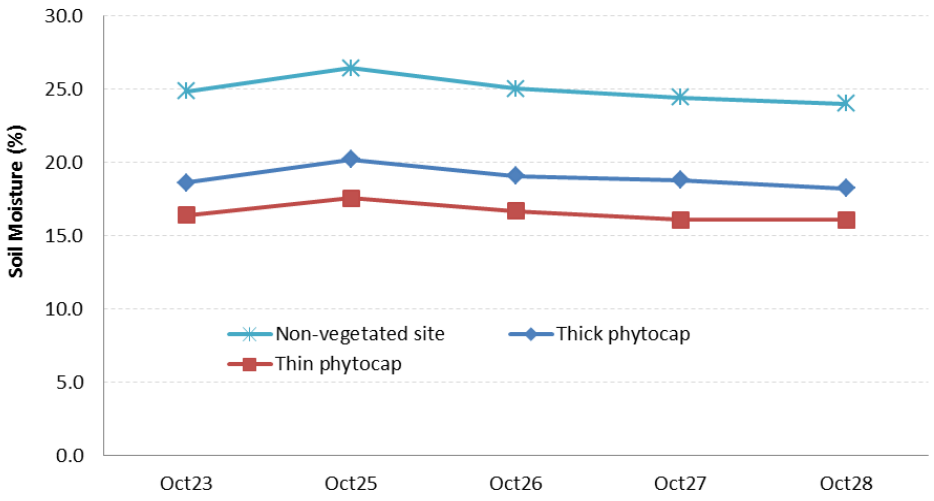


Figure 5.24: Soil moisture content of phytocaps and a non-vegetated site before and after a 16 mm rainfall in October 2006 (Average of 38 access tubes)

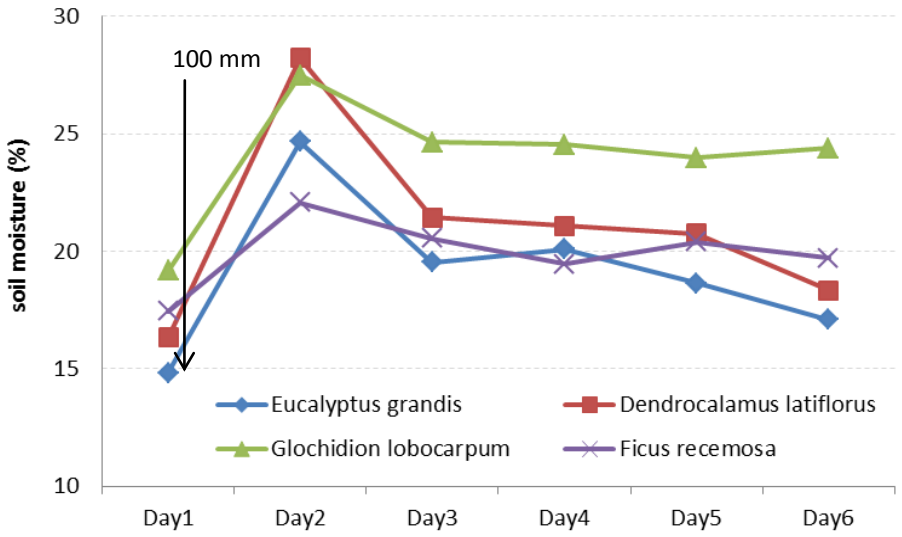


Figure 5.25: Water uptake in four tree species after supplying 100 L of water to each species

5.4.6 Comparison between TDP and HRM Sensors

Therman Dissipation Probe and Heat Ratio Method can be effectively used to measure sap flow in tree stems. Although the heat pulse technique has previously been shown to provide accurate estimates of sap flow in *Eucalyptus* species (Olbrich 1991, Dunn and Connor 1993), errors associated with the estimation of sapwood area (as high as 38%;

Hatton *et al.* 1995) could make the results highly variable. Experiments conducted during this study suggested that the three-year-old *E. grandis* with slightly variable stem diameter and canopy spread was able to take up 0.80 to 2.5 mm d^{-1} $tree^{-1}$ (Table 5.3).

Table 5.3 shows the sapflow values obtained for six *E. grandis* trees with varied stem diameter and Canopy spread. Sapflow measurement calculations in this instance were based on the Canopy spread of individual trees.

Table 5.3: Sapflow measurements in *Eucalyptus grandis* obtained by TDP and HRM sensors

Tree no.	Tree Age (years)	Canopy spread (m^2)	Sensor type	Stem diameter (mm)	Sapflow (mm d^{-1})
1	3	4.15	TDP	105	1.9
2	3	4.09	TDP	107	2.4
3	3	4.16	TDP	124	2.3
4	3	5.02	HRM	76	0.8
5	3	4.60	HRM	116	2.5
6	3	4.12	HRM	124	1.8

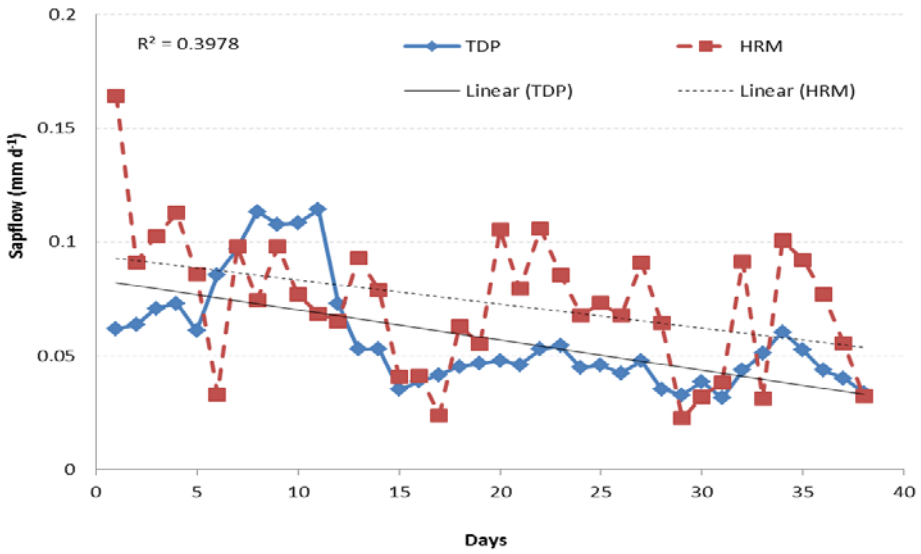


Figure 5.26: Comparison of TDP and HRM sensors for sapflow measurements recorded over 38 days

Tree to tree variability of sap flow was significant (P<0.001), with trees and day interactions (Fig 5.26). This is expected as sap flow varies with season, climatic

conditions, size of the plant and root development (Landsberg 1997). Sapwood depth varied widely among species and within the species which may also have a significant influence on sap flow.

Significant (P<0.001) variations were found between TDP and HRM sensors (Fig. 5.26). Both TDP and HRM sensors are moderately destructive techniques due to the insertions of the probes into sapwood. Insertion of probes is time consuming and sensors must be airtight and completely water proof to avoid damage to the system. Although Green *et al.* (2003) reported that the heat pulse sensors can produce accurate measurements of sapflow in plant stems, provided a reliable procedure is adopted, utmost care must be taken while installing these sensors due to their delicate nature and the high cost of the probes. TDP sensors are fragile and are prone to damage during high winds or storm periods. This was a major drawback and hindrance during this study as these sensors were broken due to bending of trees during wind, storm and cyclonic events. The breakage of sensor needles also resulted in short circuits and other technical problems. The needles in the HRM sensors are more robust and durable than those in the TDP sensors, yet there have been cases where the needles of the HRM sensors were bent by wind that affected stem movements. In open field experiments, especially in the landfill site, the wires connecting the sensor and the logger were damaged by rodents and/or birds. This caused malfunction of sensors on several occasions.

5.4.7 Testing Sensor Accuracy

Results from an experiment conducted to test accuracy of sensors suggest that although there may have been various external factors such as soil evaporation and wind speed affecting weight of pots, the overall impact was very minimal. The sapflow trend (Table 5.4) suggests that the sensors were functioning well, as the values were comparable to those obtained by the gravimetric method.

Table 5.4: Validation of sapflow readings obtained by TDP sensors

	Sensor method		Gravimetric method			
TDP sensor	Stem girth (mm)	Sensor reading (L d^{-1})	Initial weight (kg)	Weight after watering (kg)	Final weight (kg)	Difference (L d^{-1})
1	84.66	8.5	292	320	312	8
2	83.28	8.8	294	323	314.5	8.5
3	85.91	9.3	296	324.5	315.4	9.1

5.5 Conclusions

Results from the current study demonstrate the ability of 15 native species to transpire in landfill conditions in addition to showing their adaptation to the seasonal variation in rainfall. The long-term sap flow monitoring data also show that the trees can remove up to 2.1 mm d^{-1} (= 792 mm yr^{-1}) of water and can survive on as low as 0.1 mm d^{-1}. Overall, the species differed significantly in their transpiration rates as expected; as they differ in various factors such as their growth rate, canopy structure, root depth and distribution. Transpiration rates of the tested species ranged between 0.9 to 2.1 mm d^{-1}, with an overall average of 1.4 mm d^{-1}. The TDP sensor data were verified by the gravimetric method. This short term test showed that the transpiration data from TDP sensors were comparable (5 mm d^{-1}) to those determined by the gravimetric method. This data and changes in soil moisture and increase in sap flow after rainfall events or irrigation clearly show that TDP data was reliable. Furthermore, values obtained from this study were similar to those reported by other researchers.

Diurnal variation in water uptake depends on VPD, which in turn depends on solar radiation, relative humidity and wind velocity. The species showed similar sapflow values as those reported by other researchers. Seasonal variation is an important factor and has not been clearly highlighted in this study due to lack of continuous monitoring and frequent technical interferences. However, in a few instances the same species has been continuously monitored, and these have demonstrated the ability of species to

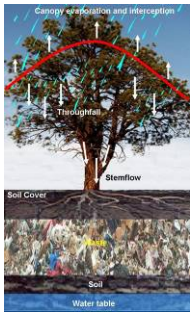
adapt to landfill conditions, where moisture supply may be excessive during certain seasons and deficient during other seasons.

The selected species also showed good responses to irrigation and rainfall events. The species increased transpiration rates within two hours of irrigation. Further research is needed to determine how species respond to variables such as water stress, temperature, wind velocity, light intensity and root depth and distribution, to determine the overall transpiration potential of a species in a phytocap or landfill.

In this study, transpiration had a positive correlation with tree height, DBH and $D_{50.}$ Other factors that influenced transpiration were rainfall, soil moisture availability, sapwood area and solar radiation. Transpiration data using TDP and HRM sensors revealed that the 3-year-old *E. grandis* transpired 2.5 mm d^{-1} with an average uptake of 1.9 mm d^{-1} during the experiment, with the highest potential uptakes of up to 4 mm d^{-1}. Sapflow of a given species varied significantly within the day and between different seasons and the transpiration rates also varied between trees of the same species.

Field experience suggests that TDP sensors are fragile and are prone to damage during high winds or storm periods. This was a major drawback and hindrance during this study as these sensors were broken due to bending of trees during wind and storm. Precautions are therefore needed to be taken while using these sensors. Tests indicated that sapflow readings recorded by TDP sensors are realistic and are comparable with each other and with those reported by other researchers. The high variability between species for water uptake and canopy interception (Chapter 6) offers an excellent opportunity to select best species for a given site to achieve an effective site water balance.

The next Chapter discusses the critical characteristics of plants “Canopy Rainfall Interception” that has not been explored in the history of phytocapping. This special characteristic feature plays a vital role in maintaining site water balance of phytocaps.



6

Canopy Rainfall Interception*

6.1 Introduction

In a vegetated site, not all of the rain that falls on the canopy reaches the ground. Part of the rain is intercepted by the canopy which evaporates directly from the leaves, twigs, branches and bark directly into the atmosphere (Tate 1995, Fig. 6.1). The phenomenon by which rain is captured by the foliage and stems is termed canopy rainfall interception (Steinbuck 2002). This has been examined predominantly in forest canopies (Dykes 1997) and rarely in a landfill environment.

Rainfall that falls on the canopy can disperse in two ways: canopy interception or throughfall. Canopy rainfall interception is the most underestimated process of rainfall analysis (Savenije 2004) and can be partitioned into (i) canopy evaporation, where part of the intercepted rain directly evaporates into the atmosphere and (ii) stemflow, where a portion of the rain that comes in contact with canopy flows through the stem, before finally reaching the ground. Throughfall is the portion of rain that reaches the ground through gaps in the canopy or via water that drips from leaves (Crockford and

* Some data from this chapter have been published in the following papers:
Venkatraman, K. and Ashwath, N. (2007) Phytocapping: an Alternative for reducing technique for leachate and methane generation from municipal landfills. *The Environmentalist*, 27: 155 – 164.
Venkatraman, K. and Ashwath, N. (2009) Phytocapping: importance of tree selection and soil thickness, *Journal of Water, Air and Soil Pollution*, 9: 421-430.
Ashwath N and Venkatraman K (2010). Phytocapping: An alternative technique for landfill remediation. *International Journal of Environment and Waste Management*, 6 : 51-70.
Venkatraman, K. and Ashwath, N. (2010) Field performance of a phytocapped at Lakes Creek Landfill, Rockhampton, Australia, *Management of Environmental Quality Journal*, 21: 237-252.

Richardson 1983). Modelling the water balance of a site therefore requires quantification of the rainfall intercepted by vegetation as well as an estimation of the throughfall (Crockford and Richardson 1983). The current study was initiated to evaluate intercepting properties of different species and to understand factors that influence canopy rainfall interception. Rainfall intercepted by 19 three-year-old tree species grown on a phytocap at Rockhampton, Australia was examined.

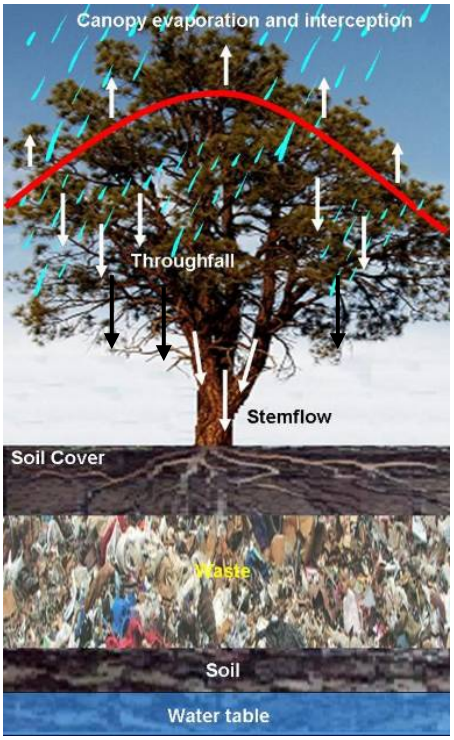


Figure 6.1: Conceptual model of canopy rainfall interception

Canopy rainfall interception has been widely studied in Australia and overseas. A number of interception studies have been completed on tropical forests (Asdak *et al.* 1998)**,** temperate broadleaf forests (Hormann *et al.* 1996**)** and temperate conifer forests (Valente *et al.* 1997)**.** This is the first Australian study to quantify canopy rainfall interception in a landfill environment.

Canopy rainfall interception varies between species and geographical location. For example, Crockford and Richardson (1990) reported canopy rainfall interception of

18.3% in *Pinus radiata* in southeast Australia. Valente *et al.* (1997) found 17.1% and 10.8% in *Pinus* sp. and *E. globulus* plantations in Portugal (annual rainfall 800 mm). Singh (1987) studied rainfall interception in a *Pinus wallichiana* plantation in India and found that 21% of the rain evaporated from the canopy, with stemflow contributing up to 2.7% at a site having an average rainfall of 859 mm. Opakunle (1989) found interception to be 24% and stemflow to be 1.8% in a cacao plantation in Nigeria that received an annual rainfall of 1169 mm. Manokaran (1979) studied the lowland tropical forest in Malaysia in a rainfall zone of 1757 mm and found interception ranging from 0.15% to 100% on a per storm basis. Giacomin and Trucchi (1992) measured an interception of 17% in a study of interception in a beech forest in Italy (annual rainfall 2027 mm). The Forest Science Department in British Columbia also conducted a study of rainfall interception on yellow cedar, red cedar, shore pine and *Sitka spruce* plantations in Smith Island (1862 mm) and Diana Lake (1943 mm), and found the interception to be 25% and 21%, respectively. A similar study by Spittlehouse (1998) reported an interception of 30% by a mature coastal hemlock forest on Vancouver Island. Another study by Llorens *et al.* (1997) on *Pinus sylvestris* showed an interception loss of 24%. On an annual basis, the interception loss from pine plantations can be 20% to 30% of the annual rainfall, while for eucalypts the loss is evaluated at 10% to 20% (Nambiar and O'Loughlin 2001). Similar findings reported by various researchers are presented in Table 6.1. Crockford and Richardson (1999) and Carlyle-Moses (2004) suggested that stemflow and throughfall are directly affected by Canopy area, leaf area, branch angle, bark texture, and rain angle and rain intensity.

Table 6.1: Rainfall intercepted by various forests

Tree	Interception (%)	Ann. Rainfall (mm)	Location	Reference
Rain forest	12.4	2115	Columbia	Vaneklass and Vanek (1990)
Rain forest	18	800	Brunei	Dykes (1997)
Laurel forest	30	733	Canary Island	Aboal *et al.* (1999)
Shrubs	27	230	Mexico	Jose and Rorke (2005)
Ficus benjamina	59	548	Mexico	Guevara-Escobar *et al.* (2007)
Montane forest	25 to 52	591 to 2561	Ecuador	Katrin *et al.* (2005)

Errors in throughfall readings, particularly those in rain gauges located in proximity to the canopy periphery, could account for anomalies in the data (Slayter 1965). Hence, utmost care was taken to reduce such anomalies while installing such gauges at the study site. During storm events, rain directly reaches the ground surface through gaps in the canopy in the form of throughfall. With prolonged rain events, the intercepted rain can accumulate on the foliage and also drip from the leaves and branches in the form of throughfall.

Stemflow is often considered to be an insignificant contributor to the hydrological cycle and hence it is not accounted for in most site water balance determinations (Liu 1997). This is due to a relatively small percentage of the gross rainfall (up to 5%, in most cases) reaching the ground via stemflow (Zinke 1966). However, in some forests stemflow was significant enough not to be ignored (Kovda *et al.* 1979). In fact, areas near the stem received 5 times more rainfall compared to places under canopy and the periphery of the canopy (Navar and Bryan 1990) and could even reach as high as 22 times (Matsubayashi *et al*. 1995). Durocher (1990) found that rainfall input at the base of some trees was 30 to 40 times higher than the mean throughfall. In Australia, stemflow has contributed significantly to soil infiltration and generated overland flow (Herwitz 1986), with stemflow fluxes as high as 31.4 cm^3 min^{-1} per cm^2 of basal area during low intensity rainfall events (2 mm min^{-1}).

There have been several practical implications of stemflow in recent years. For landfill, especially those with a phytocap, the main aim is to reduce water infiltration into the waste, stemflow that may contribute up to 5% of the total rainfall should be considered whilst calculating total hydrological balance of a site. Many stemflow studies have been conducted in the context of a large scale rainfall interception budget (Gash *et al*. 1995, Llorens 1997, Klaassen *et al*. 1998).

Accurate measurement of stemflow is difficult to quantify. In addition, small contributions to gross rainfall received are often neglected (Liu 1997). Crockford and Richardson (1990) found stemflow to be 4.1% of the total rainfall for eucalypts and

8.9% for pines; with the latter peaking at 13% for rainfall events greater than 25 mm. Asdak *et al.* (1998) found stemflow to be 1.4% in an unlogged plot in a rainforest in Indonesia, which received an annual rainfall of 3,563 mm. Large variations in stemflow of trees within species and between species are commonly observed. Lloyd and Marques (1988) found in a study of 18 rainforest trees that 15 trees each contributed 14% stemflow and the remaining three trees contributed 7%, 23% and 56%. This variation in stemflow could be due to variations in DBH (Asdak *et al.* 1998) or the number of branches and their angle in an individual tree. Stemflow is also markedly affected by rainfall intensity and angle (Crockford and Richardson 1987).

Stemflow also has the ability to recharge local groundwater (Taniguchi *et al*. 1996) and soil water (Durocher 1990). This can also affect the spatial distribution of fine roots (Herwitz and Levia Jr 1997). Results from a study in permeable carbonate bedrock suggested that stemflow rapidly infiltrated the soil matrix through macropores and root channels to produce subsurface drainage (Steinbuck 2002).

Variations in stemflow volume, which result from a variety of factors, have been investigated in many forested environments. Rainfall intensity, canopy area, height, bark texture and branch angle have all been shown to influence variability in stemflow volume (Steinbuck 2002). Stemflow generation is higher during moderate rainfall events (Ford and Deans 1978) than during low intensity rainfall events, as most water evaporates from the canopy during low intensity rainfall events. During high intensity rainfall events, the canopy reaches a point where conducting channels become saturated. At this stage water drips as soon as rain events occur. Thus, during high intensity rainfall events the throughfall values increase, while stemflow values remain constant or even decrease (Xiao *et al*. 2000).

Tree morphology and its distribution in the field can control the fate of rain water (Schroth *et al*. 1999). Canopy area is a significant factor in determining variability in stemflow (Aboal *et al*. 1999). Essentially, trees with large Canopy areas have the ability to capture maximum rainfall. Clark (1985) reported two types of branching habits in

species: i) the base of the branch that generally slopes downwards towards the stem (this would greatly increase likelihood of water being conducted to the stem); and ii) the upper branches that may slope down towards the stem but the lower branches will slope away from the stem. In the second situation the upper canopy generates stemflow, but the lower branches do not contribute to stem flow or they may even conduct water away from the stem. The second branching pattern is common in conifer trees, while the first pattern is common in deciduous trees (Clark 1985). The presence of trees with these patterns of branching may cause variation in stemflow.

Herwitz (1987) studied the relationship between branch inclination and branch flow. Laboratory experiments were conducted on dry and wet branches at inclinations ranging from 2.5° to 60°. A linear relationship was found between branch flow and branch angle for dry branches. Wet branches that were subjected to the same experiment revealed that not only were wet branches more efficient in conducting water, but the relationship between branch flow and branch angle was logarithmic. This was because, as dry branches become wet, the intercepted water on the branch surface falls from the branch at a drip point (Fig. 6.2) (Herwitz 1987). Only after sufficient water has flowed over the branch surface, will the drip points combine and the branches develop conducting channels that efficiently direct the water towards the stem (Herwitz 1987).

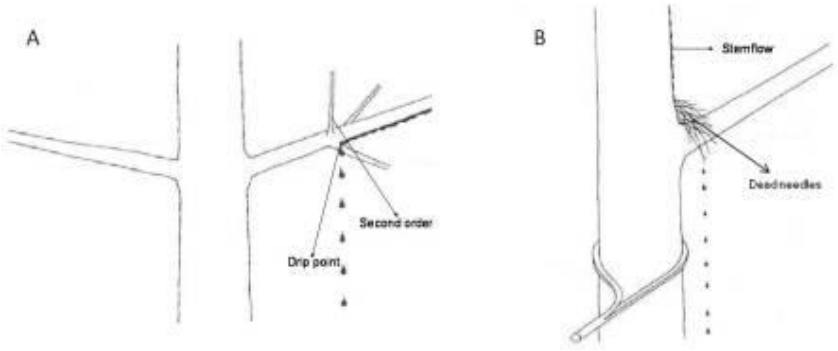


Figure 6.2: Stem drip due to stemflow
A) Stem drip due to secondary branches; B) Stem drip due to presence of dead needles at the nodes (e.g. Casuarina); (Source: Crockford and Richardson (1987)

Stemflow decreases with the age of the tree as well as in trees that do not provide ideal conditions for stemflow generation (Johnson 1990). For example, as conifers age, the

branches slope downward from the stem and this eliminates the pathway for intercepted water to be routed to the stem. However, the intercepted precipitation in the upper canopy will still contribute to stemflow (Johnson 1990). Johnson (1990) also observed an increase in throughfall with the distance from the stem. This was attributed to down-sloping branches, which would be likely to conduct water away from the stem and it would eventually become throughfall at fixed drip points.

Cape *et al*. (1991) conducted a study in pine (*P. sylvestris*), Norway spruce (*Picea abies*), Sitka spruce, larch (*Larix decidua*), oak (*Quercus petraea*) and alder (*Alnus glutinosa*) forests in northern Britain. They concluded that the branching habit of different species was the main determinant of stemflow. This can be quite important when quantifying the relationship between species and stemflow in a region, as there could be marked variations between provenances.

In the literature, tree stem properties have been reported to control stemflow variability (Aboal *et al*. 1999). Trees with smooth bark will conduct more water to the ground, with reduced loss occurring in the form of stem drip. Stem drip is the water that falls directly from the stem to the ground due to a localised drip point, typically on a rough textured bark surface (Fig. 6.2).

To quantify bark roughness, researchers take measurements on small pieces of bark to obtain a bark roughness index which is used to classify the bark for a given species (Aboal *et al*. 1999). In addition to affecting stem drip, bark roughness influences the amount of initial storage of precipitation prior to stemflow (Steinbuck, 2002). Durocher (1990) implemented a study on a red oak (*Quercus rubra*) and sweet chestnut (*Castenea sativa*) forest in Bristol, England, and found large differences in stemflow generation not only between the species but also between individuals of the same species having similar stand positions, shape, and dimensions. High resolution stemflow recording gauges showed that once the initial interception storage was exceeded, stemflow timing reflected the rainfall dynamics almost perfectly (Duroacher 1990). Very small quantities of stemflow were collected once precipitation had ceased, indicating little water was

stored. It was noted, however, that the conclusions drawn from the smooth bark trees could not necessarily be applied to rough bark types. Helvey and Patric (1965) have reported storage of intercepted rainfall prior to stemflow in many studies conducted in the eastern United States. Stemflow starts after 1.27 mm of rain on beech (*Fagus grandifolia*) (Voigt 1960), but in other species stemflow does not occur until rainfall exceeds 5.08 mm (Black 1957) or 22.86 mm (Gilbert 1953).

6.2 Materials and Methods

To determine canopy rainfall interception potential of 19 species grown on two phytocaps and the factors affecting the interception, a series of experiments were conducted as outlined below.

6.2.1 Throughfall

Throughfall was determined using a standard rain gauge (4 cm wide and 51 cm high) (Fig. 6.3) (Crockford and Richardson 2000). Four randomly selected plants of a species in Thick and Thin phytocaps were monitored by placing the rain gauges under the canopy of each tree at 30 cm, 40 cm and 50 cm from the main stem. Trees from one replication of each capping system were monitored and a total of 456 rain gauges were used within an area of 2500 m^2. An additional 20 rain gauges were placed around the experimental plot in an open area to record total rainfall received at the site. The rain gauge readings were recorded for almost all rainfall events during a 24 month period (50 rainfall events). The canopy rainfall interception was calculated as follows:

$$\text{Canopy rainfall interception (\%)} = \frac{\text{Total rainfall} - \text{Throughfall} - \text{Stemflow}}{\text{Total rainfall}} \times 100$$

Figure 6.3: Standard rain gauge used in measuring throughfall

6.2.1.1 Leaf Area and Specific Leaf Area

Five leaves (Westoby 1998 and Weiher *et al*. 1999) were collected from each species from (from top, middle and bottom part of the canopy) both Thick and Thin phytocaps and from both replications soon after sunrise or just before the sunset as recommended by Garnier *et al.* (2001). The leaves were then gently wiped with a paper towel and scanned using a scanner and the scanned image was analysed using Delta – T leaf area meter (Bouma et al. 2000, O'Neal *et al.* 2002, and Cornelissen *et al.* 2003). The same leaves were dried at 70°C for 3 days and the dry weight was determined. Specific leaf area was calculated as follows:

$$\text{Specific leaf area} = \frac{\text{Leaf area } (mm^2)}{\text{Leaf weight (g)}}$$

6.2.1.2 Leaf Adsorption

Ten leaves were sampled from each species from Thick and Thin phytocaps and from both replications (40 leaves/species). Their fresh weights were determined and then the leaves were dipped in rainwater for about 10 seconds. The leaves were removed and the droplet collected at the tip of the leaves was removed using a filter paper prior to determining the wet weight of the sample (Liu 1996, Liu 1998). The difference in weight

of the leaf before and after dipping in water provided the adsorption capacity of the leaf (Haines *et al.* 1985, Liu 1998).

6.2.1.3 Leaf Thickness

Leaf thickness was measured using a micrometer. A minimum of five readings were taken for each species in each of Thick and Thin phytocaps and from both replications. The measurements were taken from the centre of the leaves. Major veins and midribs were avoided on all the leaves as recommended by Wright and Cannon (2001) and Vendramini *et al.* (2002).

6.2.1.4 Leaf Number

Leaf number was determined using the formula:

$$\text{Number of leaves/plant} = \frac{\text{Total fresh leaf weight/plant}}{\text{Average fresh weight/leaf}}$$

The average fresh weight was determined from five randomly selected leaves/species in each of thick and thin phytocaps from both replications. At harvest (2007) total fresh weight was determined for 3 plants per species from thick and thin phytocaps and then the average fresh weight of each leaf was calculated.

6.2.1.5 Leaf Toughness

Leaf toughness was measured using a leaf fracture toughness tester, built to the same specifications as the machine described by Wright and Cannon (2001) and Darvell *et al.* 1996) (Fig. 6.4). In short, this machine measured the force required to push a razor blade (held at a constant angle) through the leaf lamina, at the widest point of the leaf. It measured the force required to cut a leaf at constant cutting angle and speed. A penetrometer (Vogel 1988, Choong *et al*.1992, Edwards *et al.* 2000) was used to estimate leaf toughness. In this study three leaves per species from each of Thick and Thin phytocaps and from both replications were used to determine leaf toughness.

Figure 6.4: Leaf toughness tester

6.2.1.6 Leaf Hairiness

The procedure followed was similar to that adopted by Velkama *et al.* (2003). Three leaves of each species from both Thick and Thin phytocaps and from both replications were collected, rinsed with distilled water to remove organic debris and dust, and blotted dried with a paper towel. Once dried, the samples were treated with liquid nitrogen and observed under a scanning electron microscope (JSM 6360LA) (Fig. 6.5) for hair density. Leaf hair density from an average of three leaf samples was recorded for each species. Images of the samples were scanned and the total number of hairs counted. Hair density was calculated based on the scale given for each image.

Figure 6.5: Scanning electron microscope

6.2.1.7 Canopy Storage Capacity

A total of 12 randomly selected leaves per species from Thick and Thin phytocaps and from both replications were used to determine canopy storage capacity. The leaf samples were brought to the laboratory in an insulated storage container. The leaves were gently wiped with a paper towel to remove dust and the initial weight of the leaves was recorded. The leaves were then sprayed and saturated with distilled water until water started dripping from leaves. Once saturated, leaves were weighed (Final weight). The difference in the weight between the initial and the final weight gave the adsorption capacity of leaves. Canopy storage capacity was calculated by extrapolating leaf adsorption potential (artificial wetting) of each species by Canopy area (Liu 1998):

$$\text{Canopy storage (mm)} = \frac{\text{Leaf adsorption (ml/leaf) x number of leaves}}{\text{Canopy area } (m^2)}$$

6.2.2 Stemflow Measurement

To measure stemflow, a split plastic hose was stapled around the tree using galvanised staple pins with one of its ends tapering downwards to discharge water into a graduated jar (Fig. 6.6; Crockford and Richardson 2000). The gaps between the hose and the bark were sealed with neutral silicon sealant and the sealant was left to dry for 24 hours

before measurement. Once dried completely, the tapering end was inserted into the jar through the lid (Fig. 6.6). The measuring jar was closed and anchored to the stem to restrict movement and to prevent evaporation. During rain events, the water flowing through the stem entered the cup shaped split hose and ran through the tapering end into the jar. Three such stemflow gauges were installed per species in each of the Thick and Thin capping systems, and measurements were taken over several rainfall events that spanned over a year. Stemflow was calculated as follows:

$$\text{Stemflow (mm)} = \frac{\text{Volume of rainwater collected (L) in a rainfall event}}{\text{Canopy area of that tree } (m^2)}$$

In most species that were studied for canopy interception, the canopy was not fully closed at one year or at one and half years after establishment. Canopy area was recorded each year for each species and the stemflow was calculated based on canopy area. Stemflow collars were installed 9 months after the beginning of the collection of rainfall interception data. Thus the stem flow values for the early part were estimated based on the values obtained from the later measurements (based on the rainfall received per event and its intensity) so that both stem flow and interception data were derived for the entire measurement period. Stemflow values for the period January 2005 to September 2005 therefore include extrapolated values according to rainfall intensity and duration.

Figure 6.6: Stem flow collar

6.2.2.1 Branch Angle

The angle between the main stem and the lateral branches, measured above the lateral branch, was measured using a protractor to represent branching angle (Honda *et al.* 1997).

6.2.2.2 Bark Texture

This experiment was conducted on 12 species grown on the Thick phytocap only. A bark texture experiment was conducted at 1 meter above the measurement collar (approximately 1.5 m above the ground). One tree from each 12 species was selected. Water was added in one-liter increments and was allowed to flow for 10 minutes before the recovered volume at the collection point was measured. Immediately after measuring the recovered volume, another liter was applied. A manual sprayer with an adjustable nozzle was modified for this purpose. The container was calibrated to 0.25 litres accuracy. The bark was divided into four equal segments (four pie of the stem) and 0.25 litres of water was applied to each segment. After several trials, a steady even spray was able to be produced and this was found to be the most effective in preventing loss from the impact of water on the bark surface (Steinbuck 2002). This experiment was repeated three times on the same tree on three different days during low rainfall periods.

6.3 Statistical Analysis

Data was analysed (ANOVA) using Genstat ver. 8.0. Least significance differences (l.s.d.) are presented in figures where the F values of the treatment, capping, species or their interactions were significant ($P<0.05$). Regression analysis was carried out to determine interrelationships between tree traits and interception data using GraphPad Prism v 4.03. A polynomial equation was chosen for all graphs as this produced the highest r^2.

6.4 Results and Discussion

6.4.1 Canopy Rainfall Interception

The 3-year-old trees that were established on Thick and Thin phytocaps were able to intercept up to 50% of the rainfall on a per storm basis, with an overall average of 30% of the total rainfall received at Rockhampton (Fig. 6.7). Data represent the average performance of trees across a variety of rainfall events ranging from 0.6 mm to 80 mm. Furthermore, data was recorded over 26 months representing winter, spring, and summer seasons and windy, still and humid climates. With current performance of trees, only 546 mm of the total rainfall would reach the ground surface, and this does not include the rainfall intercepted by leaf litter. This effective reduction of rainfall from 780 mm to 546 mm will have practical implications. This is quite a significant contribution towards the hydrological balance of the phytocapping system. For example, a reduction in effective rainfall could mean the use of shallower depths of soil caps. The use of shallow depths of soil in Thick and Thin caps will eventually contribute to reduced costs for landfill construction.

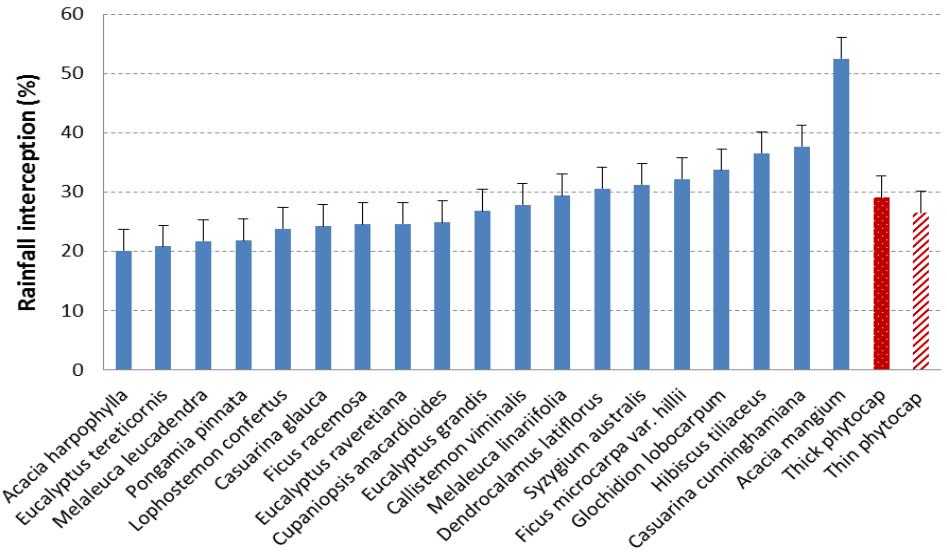


Figure 6.7: Canopy rainfall interception by 19 species grown in a Phytocapping system
Data are means of 50 measurements taken over a period of 26 months (Bars represent l.s.d. 3.587)

6.4.2 Effect of Species on Canopy Rainfall Interception

The species used in this study showed a significant (P<0.001) difference in rainfall interception between species (Fig. 6.7) which can be attributed to the differences in leaf characteristics. Few species had needle shaped leaves (e.g. *Casuarina* spp.), a few had broad leaves (e.g. *Hibiscus tiliaceus*) and most species had medium sized leaves (Table 6.2). The list of species examined and their leaf type and their rainfall intercepting capabilities are given in Table 23.

Table 6.2: Rainfall intercepted by 19 tree species with different leaf size and shape

Species	Leaf type/size	Canopy rainfall interception (%)
Acacia harpophylla	Medium	20.10
Acacia mangium	Broad	52.50
Callistemon viminalis	Narrow	27.82
Casuarina cunninghamiana	Needle	37.68
Casuarina glauca	Needle	24.22
Cupaniopsis anacardioides	Medium	24.87
Dendrocalamus latiflorus	Broad	30.60
Eucalyptus grandis	Medium	26.80
Eucalyptus raveretiana	Medium	24.62

Eucalyptus tereticornis	Medium	20.78
Ficus microcarpa var. *hillii*	Medium	32.14
Ficus racemosa	Medium	24.56
Glochidion lobocarpum	Medium	33.68
Hibiscus tiliaceus	Broad	36.54
Lophostemon confertus	Narrow	23.78
Melaleuca leucadendra	Medium	21.74
Melaleuca linariifolia	Narrow	29.36
Pongamia pinnata	Medium	21.82
Syzigium australis	Narrow	31.24

Trees grown on the phytocaps were of varied nature, shape and size. A few were fast growing such as *A. mangium*, *Eucalyptus* sp., *H. tiliaceus* and *Casuarina* spp., while some were moderately fast-growing such as *F. racemosa* and *P. pinnata* and the others were slow growing, such as *A. harpophylla* and *C. viminalis*. Yet, a few slow and moderately growing species such as *A. harpophylla* and *S. australis* intercepted rain better than some fast growing species like eucalypts, *C. glauca* and *D. latiflorus*.

6.4.3 Factors Influencing Canopy Rainfall Interception

Apart from size, tree species also differed in leaf morphology, leaf texture, leaf shape, leaf angle, canopy spread, branch orientation and branch angle. Crockford and Richardson (2000) also noted similar variations between plant species in their study. Canopy spread, leaf morphology, leaf area and other tree traits influenced rainfall interception at various levels (Table 6.3). The polynomial equation was used to test the relationship ($P<0.05$) between canopy rainfall interception and various tree characteristics as it gave the highest r^2 values. Statistical analysis of various tree parameters revealed that canopy rainfall interception was primarily influenced by LAI and the number of leaves in both Thick and Thin phytocaps (Table 6.3). Similarly, higher interrelationship was noticed between canopy rainfall interception and leaf thickness.

Overall, from Table 6.3, factors such as LAI, canopy spread, leaf number, leaf thickness, leaf adsorption, leaf toughness, leaf area and canopy storage had an influence on canopy

rainfall interception. Leaf adsorption, leaf toughness and leaf thickness showed a significant correlation (Table 6.3). LAI has shown a positive correlation with canopy rainfall interception. Similar correlation was observed by Kang *et al.* (2005); however, during heavy winds and high storm events rainfall interception was reduced, which may be associated with canopy saturation. Canopy spread, leaf adsorption and leaf number increase the surface area to allow more water to be held during rainfall events thereby increasing the water holding capacity of the canopy. Leaf toughness plays a vital role in increasing interception during heavy rainfall events. This characteristic in leaves allows the leaf to hold more moisture in heavy rainfall events without leaning. There is a positive correlation between canopy rainfall interception and leaf traits.

Table 6.3: Relationship between canopy rainfall interception and various tree components (r^2 values)

Parameter	Thick phytocap	Thin phytocap	Combined
LAI	0.16***	0.19***	0.16***
Canopy area	0.13**	0.03**	0.06**
Leaf number	0.17**	0.15***	0.15***
Leaf thickness	0.42***	0.23**	0.26**
Leaf adsorption	0.04**	0.04**	0.14***
Leaf toughness	0.17**	0.13**	0.14**
Leaf area	0.23***	0.23**	0.1**
Canopy storage	0.12**	0.05**	0.12**
Means of all parameters	0.75*	0.7*	0.65*

Note: **denotes significance at 0.01 probability and ***= 0.001 probability

Specific leaf area also varied significantly (P<0.001) between species (Fig. 6.8). This was due to genetic differences between species in leaf morphology, size and shape. Leaf toughness (force of fracture) showed a marked variation (P<0.001) between species (Fig. 6.9) and between phytocaps (P=0.002) (Fig. 6.9). This was due to higher leaf thickness, dry matter content and specific leaf area of the trees grown in the Thick phytocap than in those grown in the Thin phytocap. Leaf thickness totally depends on nutrients and moisture content (Witkowski and Lamont 1991) and the availability of photosynthetically active radiation (Nobel and Hartsock 1981). Leaves of species grown on the thick phytocap were thicker than those grown in the thin phytocap (Fig. 6.10) and

this is likely to be associated with availability of more water and nutrients as the trees in Thick cap had deeper roots and had access to black clay which would have had better nutrients than the sandy loam soil. Physically stronger leaves can survive better in hail and wind (Cornelissen *et al.* 2003), and they intercept better during high intensity rainfall events. They also have greater lifespan (Cornelissen *et al.* 2003). Stronger leaves give a lower splash effect, and subsequently intercept more rainfall (Kang *et al.* 2005).

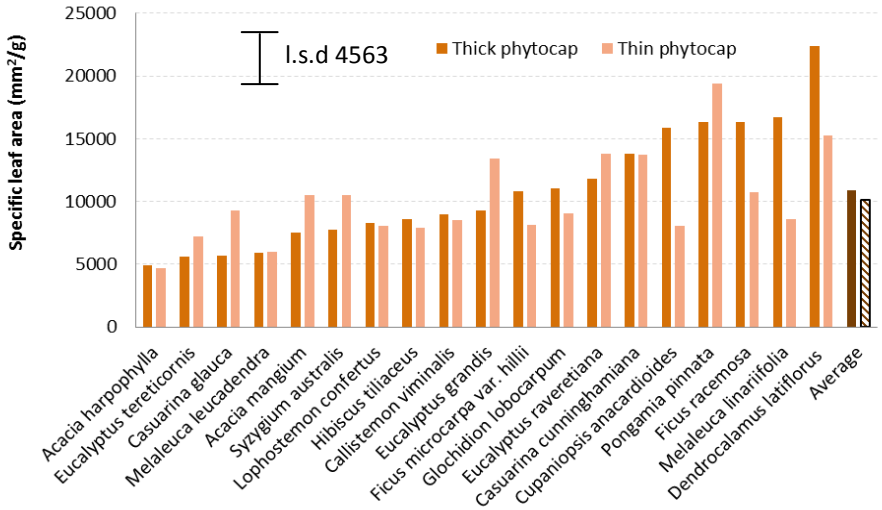


Figure 6.8: Specific leaf area of 19 species grown on Thick and Thin phytocaps

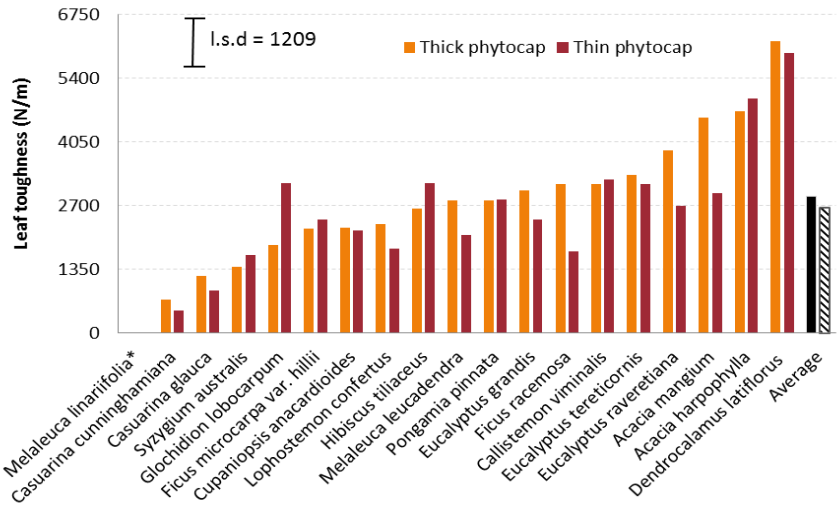


Figure 6.9: Leaf toughness measured in 18 species grown on Thick and Thin phytocaps
* Leaves too small to determine leaf toughness

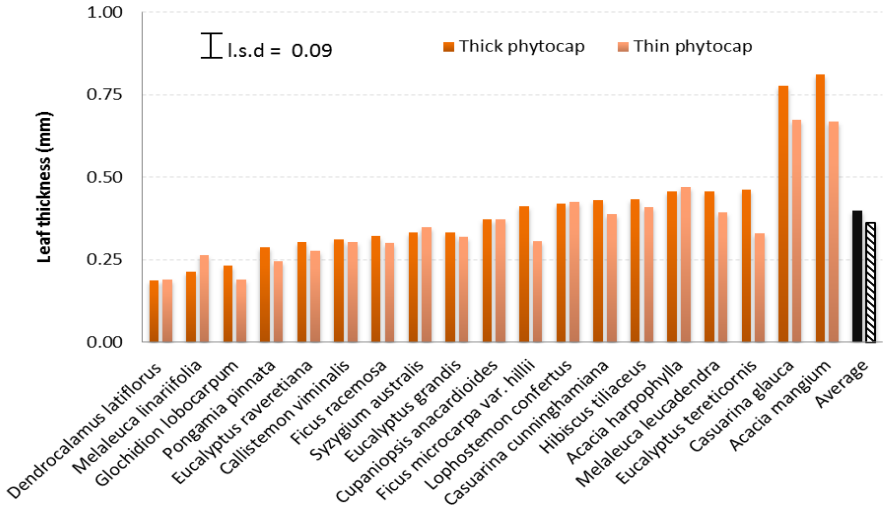


Figure 6.10: Leaf thickness in 19 species grown on Thick and Thin phytocaps

Canopy rainfall interception is also influenced by canopy storage (Crockford and Richardson 2000) which in turn is governed by leaf adsorption capacity. Leaf adsorption capacity varied between species and phytocaps (P<0.001) (Fig. 6.11). Leaf adsorption capacity is the water retaining strength of individual leaves (Juniper and Jeffree 1983), which depends on various leaf morphological parameters like leaf hairiness, cuticle thickness, wax deposition, leaf angle and leaf area (Baker and Hunt 1981), and other environmental factors such as temperature, leaf contact angle and drop size distribution (Haines *et al.* 1995).

Leaf hair density differs between species and within species due to their genetic makeup (Stoner 1992, Paliniswaby and Bodnaryk 1994). It is interesting to note that smaller leaves in *M. linariifolia*, *G. lobocarpum*, *F. microcarpa* and *S. australis* had higher water retention capacities than the broad-leaved species such as *H. tiliaceus*, *D. latiflorus*, *A. mangium* and the *Eucalyptus* species (Fig. 6.11). Overall, trees grown on the Thick phytocap had better adsorption capacity than the ones grown on Thin phytocap and were primarily due to differences in leaf size and abundance of leaf trichomes on the upper leaf surface. This indicates that leaf area, although significant, may not contribute towards water retention. Variations in water retention capacities of

different trees could also be due to leaf surface, droplet size distribution and leaf contact angle.

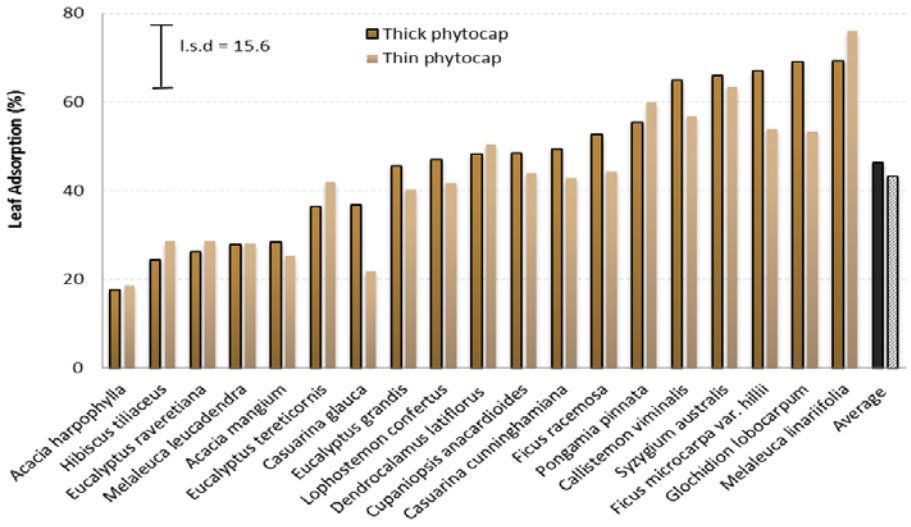


Figure 6.11: Leaf adsorption capacity of 19 species grown on Thick and Thin phytocaps

Interestingly, species with higher leaf hair density (Fig. 6.12) (e.g. *H. tiliaceus* and *D. latiflorus*) retained low water levels compared to species with lower leaf hair density (Fig. 6.13). *C. cunninghamiana* (needle shaped leaf) (Fig. 6.14) with low hair density had higher water retention capacity than high leaf hair density *H. tiliaceus* and *D. latiflorus* and this could be due to lower leaf contact angles (Fig. 6.15) (less than 90°C) (Fig. 6.15A) that covered a larger area of the leaf surface (Haines *et al.* 1985). However, water retention capacity was improved by number of leaves and the canopy spread. *Dendrocalamus latiflorus* and *H. tiliaceus* had better canopy storage capacity because of the presence of larger number of leaves than in other trees grown on the phytocap (Fig. 6.16). Results showed that the leaf size had influence on canopy interception, as the cladodes (modified branches that look like leaves) of *C. cunninghamiana* intercepted large proportion of the rain. In comparison, *E. tereticornis*, which has broader leaves, had low canopy interception (Fig. 6.16). There are two primary reasons for this to happen, one being the canopy storage capacity and the second being stem drip that was not taken into account during this study. In this instance, *C. cunninghamiana* (0.45 mm per tree) had greater storage capacity than *E. tereticornis* (0.28 mm per tree) and many

other species (Fig 6.16). These values refer to the static storage capacity that is retained in the canopy and this does not drain to the ground (Dunkerley 2000). A study conducted in the US on canopy interception reported a canopy storage capacity that ranged from 0.25 to 9.4 mm per canopy (Zinke 1965).

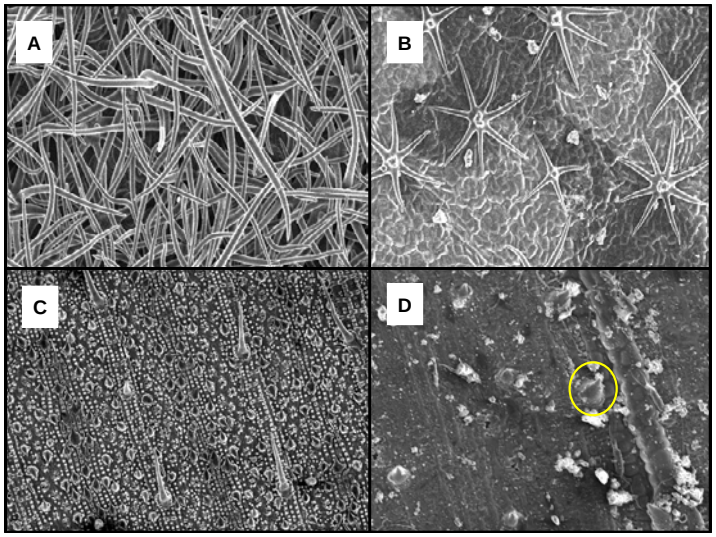


Figure 6.12: Leaf trichomes in *H. tiliaceus* and *D. latiflorus*
A) Abaxial surface of *H. tiliaceus*, B) Adaxial surface of *H. tiliaceus*, C) Abaxial surface of *D. latiflorus*, D) Adaxial surface of *D. latiflorus* (scale 1mm)

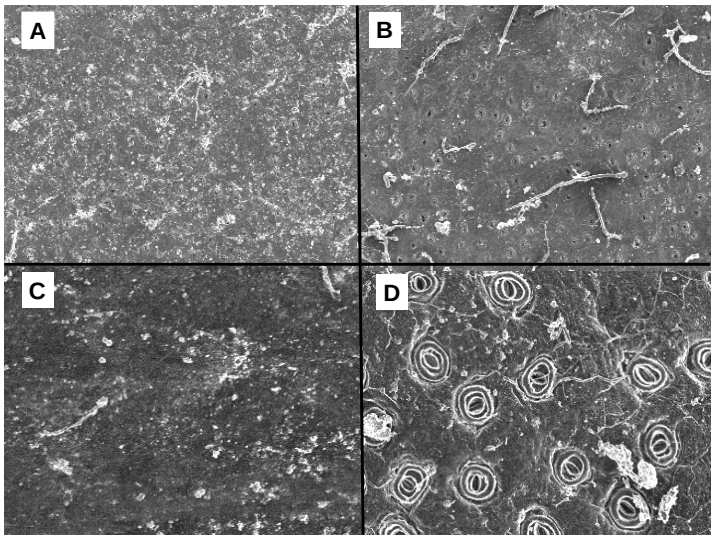


Figure 6.13: Leaf trichomes in *M. linariifolia* and *F. microcarpa*
A) Adaxial surface of *Melaleuca linariifolia*, B) Abaxial surface of *Melaleuca linariifolia*, C) Adaxial surface of *Ficus microcarpa*, D) Abaxial surface of *Ficus microcarpa* (scale 1mm)

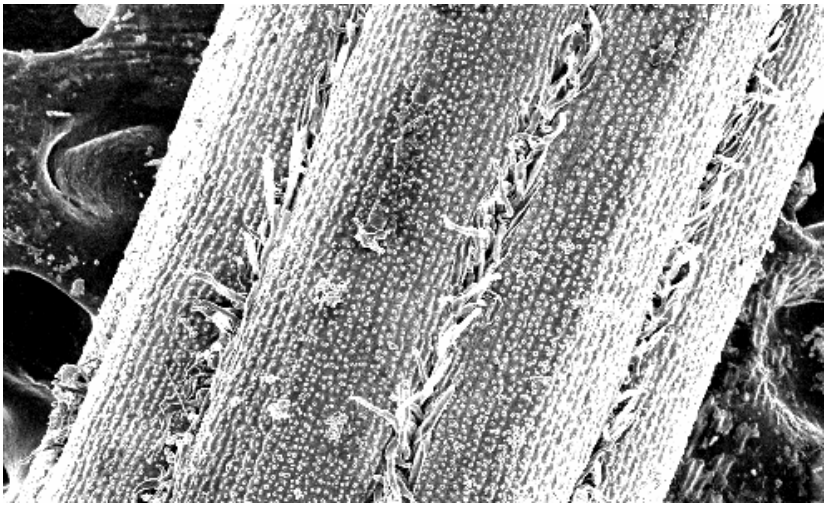


Figure 6.14: Hair density in *C. cunninghamiana* (scale 1mm)

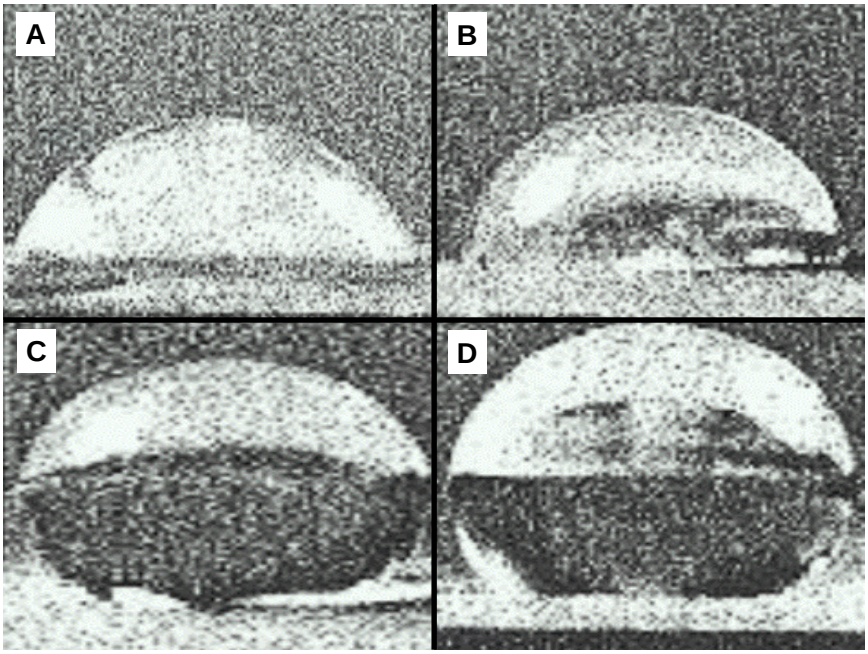


Figure 6.15: Side views of 10 µl droplets of water on the adaxial leaf surface (Contact angle increasing from A – D) (Source: Haines *et al.* 1985)

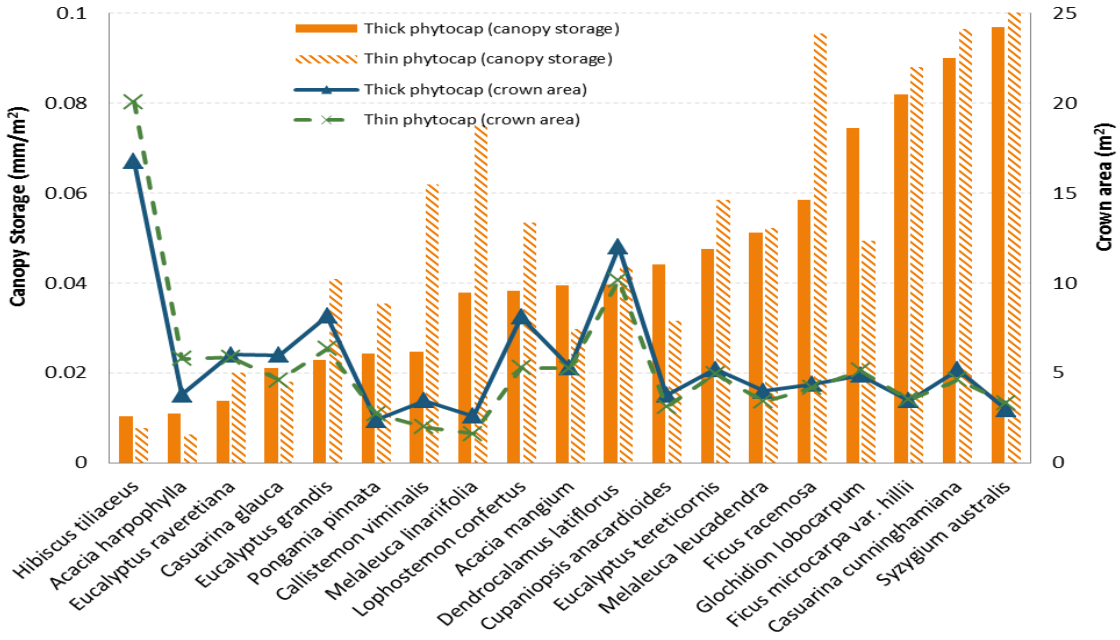


Figure 6.16: Canopy storage capacity and canopy area of 19 species grown on Thick and Thin phytocaps

Variation in canopy interception amongst tested species could also be associated with the variation in rain angle and intensity which are likely to result in significant amount of water reaching the ground via throughfall and stemflow (Dunkerly 2000, Silva and Rodriguez 2001). The overflow patterns from leaves depend on rain events that exceed storage capacity of the species. On an average, 15 mm to 25 mm rainfall was received in Rockhampton per rainfall event, but the intensity varied from 2 mm h^{-1} to 490 mm h^{-1}. Another reason for the presence of large variations between species in canopy interception is wind speed, which shakes leaves and lowers the effective storage capacity in the canopy (Dunkerly 2000). Canopy interception is also affected by tree spacing (Teklehaimanot *et al.* 1991). Closely planted trees intercept less water than those spaced widely. Teklehaimanot *et al.* (1991) proved that a Sitka spruce tree with 2 m spacing intercepted less water than those planted at 8 m, due to variations in the branch angle, canopy morphology, tree size, rainfall intensity and rainfall duration. However, spacing was kept constant in the current study, so this would have not influenced the results, although the size of the tree would have.

Rainfall interception was strongly influenced by the amount of rain received, rainfall intensity and rainfall duration (Table 6.4). Most trees intercepted higher amounts during moderate rainfall periods than during light intensity rainfall events (Fig. 6.17). Most rainfall events in Rockhampton ranged between 15 mm to 25 mm during the study period (2005 to 2006) with many events having less than 10 mm and very few exceeding 30 mm. Rainfall intensity also had some influence on canopy rainfall interception (Fig. 6.18). In Figure 6.18, a few values of zero explain the relevance of the graph suggesting that interception occurred even for very low rainfall and that the interception increased during low rainfall events. Rainfall duration also had some effect on canopy rainfall interception (Fig. 6.19). Rainfall and duration had a combined effect on canopy rainfall interception. Low intensity rainfall events of longer duration increased canopy interception, but low intensity rainfall events of shorter duration had lower rainfall interception. This interception is governed by the canopy storage capacity and evaporation.

Table 6.4: Interrelationships between rainfall, rain duration and rainfall intensity on canopy rainfall interception (r^2 values). Note: ***denotes significance at 0.001 probability

	Interception
Rainfall	0.54***
Rain duration	0.38***
Rain Intensity	0.55***

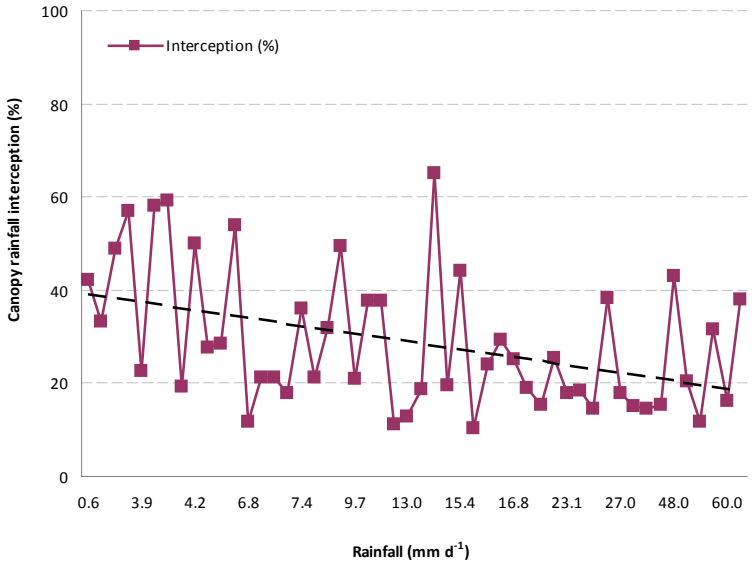


Figure 6.17: Rainfall intercepted by 19 species during high rainfall (>20 mm), moderate rainfall (11 – 20 mm) and low rainfall (0.6 – 10 mm) (2005 – 2007) events

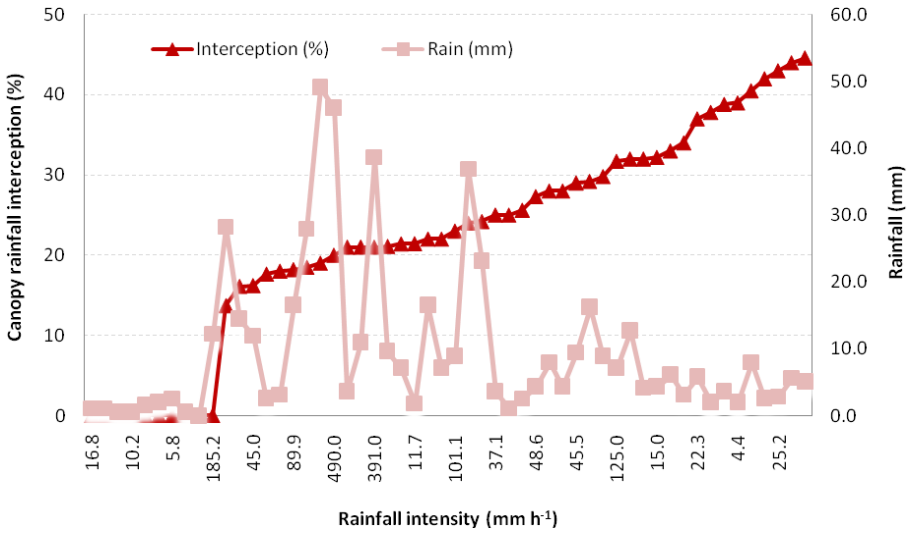


Figure 6.18: Rainfall intercepted by 19 species during low, medium and high intensity rainfall events in 2006

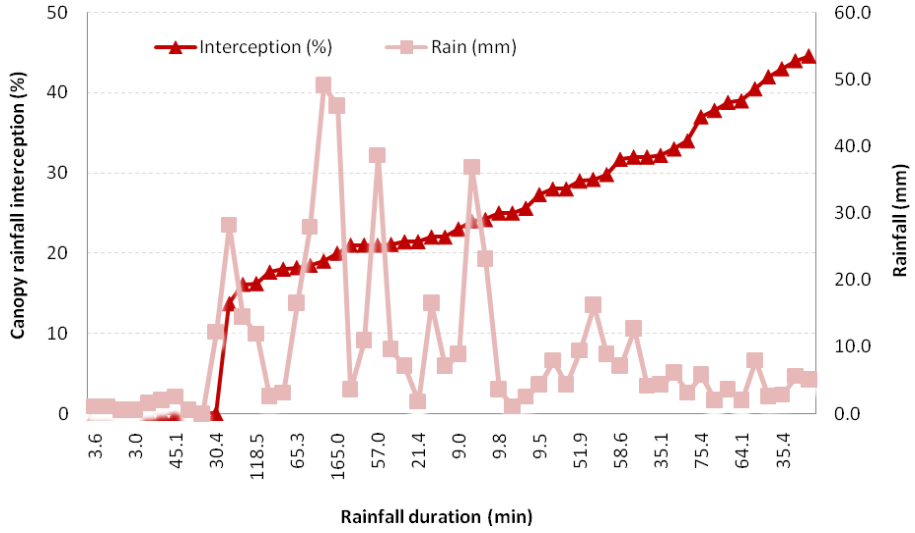


Figure 6.19: Rainfall intercepted by 19 species during rainfall events with varied durations (2006)

6.4.4 Effect of Capping Thickness on Canopy Rainfall Interception

Canopy rainfall interception also showed significant (P<0.001) variation between thick and thin phytocaps (Fig. 6.20). Species in the Thick phytocap generally intercepted more rain than the same species that were in Thin phytocap (Fig. 6.21) and this may be due to better growth in the thick phytocap than in the Thin phytocap (Chapter 3). Better growth in Thick phytocap was possibly due to greater availability of resources such as nutrients (Maurice 2005), better root development (Moffat and Houston 1991, McGuire *et al.* 2001) and water availability (Fu-sheng *et al.* 2005, Venkatraman *et al.* 2011). Root development was more extensive in the Thick phytocap than in the Thin phytocap (Chapter 3). A few species such as *E. grandis*, *E. raveretiana*, *E. tereticornis*, *A. mangium* and *C. cunninghamiana* grew more rapidly in the Thick phytocap than in the Thin, as they had better root growth in Thick cap. Deeper rooting systems meant better access to larger quantity of nutrients and water.

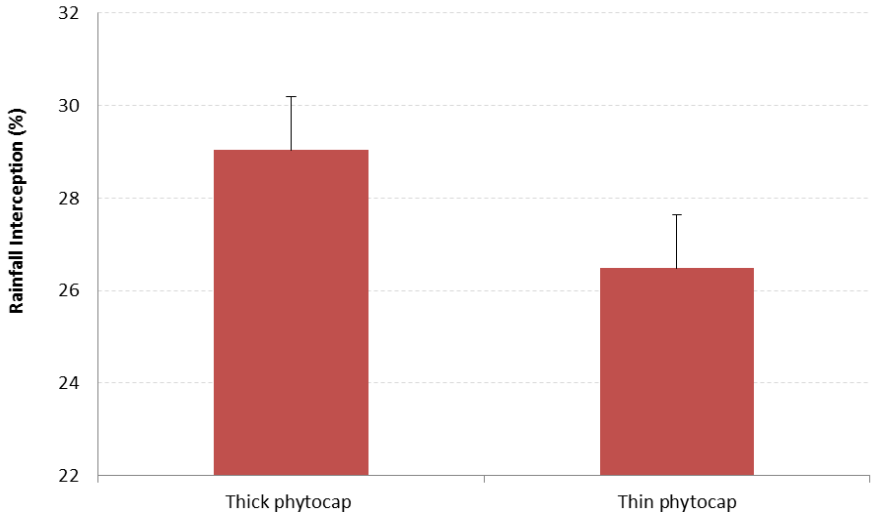


Figure 6.20: Rainfall interception in Thick and Thin phytocaps
(n = 50 rainfall events) (Bars represent l.s.d 1.164)

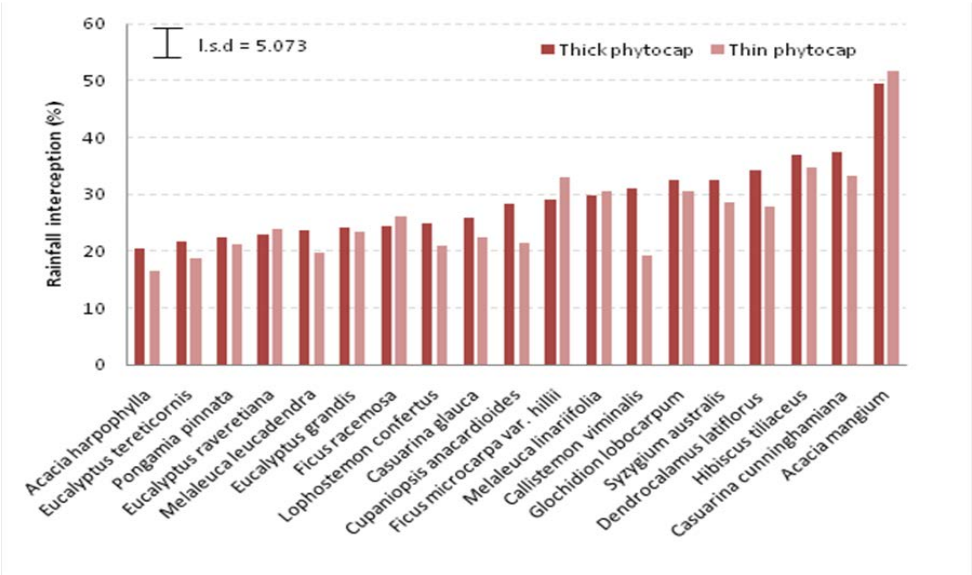


Figure 6.21: Rainfall interception by 19 species grown on Thick and Thin phytocaps
(n = 50 rainfall events)

6.4.5 Stemflow

Variability (P<0.001) in stemflow between species (Fig. 6.22) was associated with a variety of factors such as stem diameter, canopy morphology and other features such as branch angle, rain intensity and bark texture (Steinbuck 2002, Aboal *et al.* 1999). The stemflow in these species contributed up to 4.5% of the total rainfall and is similar to

the 5% reported by Zinke (1966) in hardwood species in the US. Studies in the past have shown that stemflow may not start in many species until 1.27 mm to 22.86 mm of rain has fallen (Steinbuck 2002). Hence variation in stemflow could also make a significant contribution to variation in canopy interception.

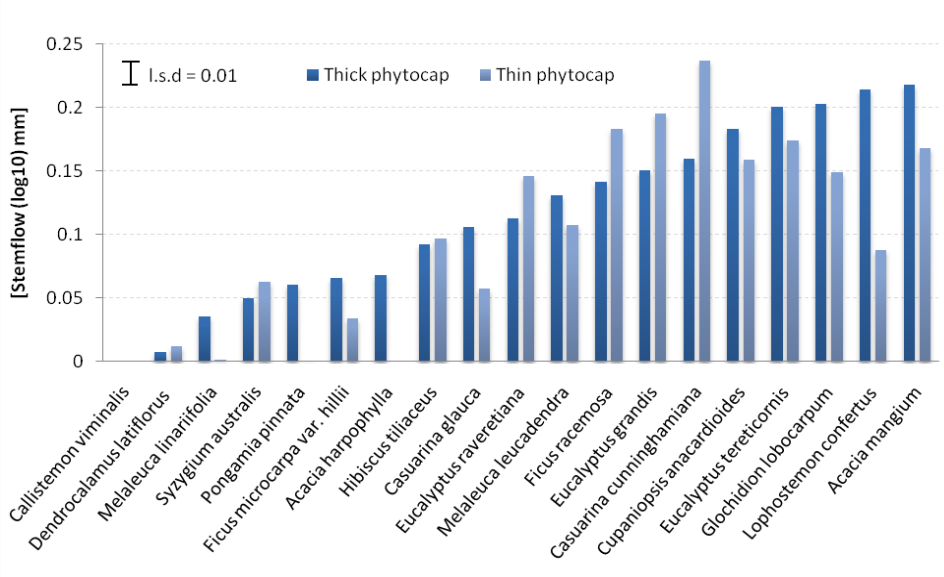


Figure 6.22: Stemflow in 19 species grown on Thick and Thin phytocaps
(Bar represented species.cap interaction)

There was a significant difference (P<0.001) in stemflow generated by the same species in the Thick and Thin phytocaps and this can be attributed to tree shape and angle from the ground, as well as tree spacing and canopy dimension. Stemflow was higher in the Thick phytocap (Fig. 6.23) due to better tree growth, which could be due to availability of resources such as nutrients, better root development and water availability. More importantly, the Thick phytocap had 300 mm of black cracking clay with 600 mm of clay soil, which was more fertile (Appendix A) than the soil present in the Thin phytocap. High fertility leads to tougher leaves, which in turn gives strength to the leaves to hold more water on their surface. If the leaf is not tough then they drop water quickly thereby allowing more water to reach the ground surface.

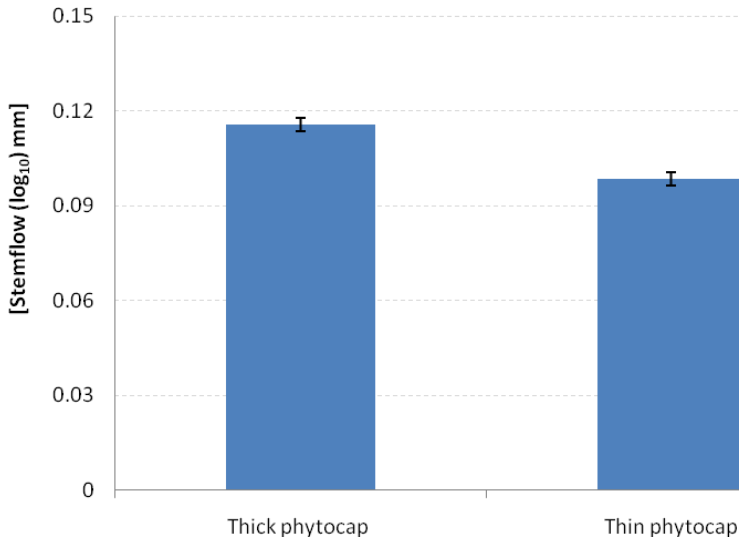


Figure 6.23: Comparison of stemflow generated in Thick and Thin phytocaps
(Bar represents l.s.d. 0.002)

6.4.6 Factors Affecting Stemflow

Polynomial equations were used to test the relationship between stemflow and growth parameters and tree characteristics as it gave the highest r^2 values. Regression analysis revealed a significant correlation between stemflow and tree characteristics such as canopy spread, branch angle, height, D_{50}, and DBH (Table 6.5). Stemflow increased with an increase in D_{50}. According to Crockford and Richardson (1999) and Carlyle-Moses (2004), stemflow and throughfall were directly affected by canopy spread, height and DBH.

Species differed significantly (P<0.001) in their water holding capacity in their bark (Fig. 6.24), and for the water recovered from stemflow for the same amount of water applied (Fig. 6.25). *Acacia mangium*, which had a very rough textured bark, was able to hold more water than other species (Fig. 6.25). On the other hand, *D. latiflorus*, which had a smooth bark, retained the lowest quantity of water (Fig. 6.25). Bark texture as a stemflow controlling variable is quite evident. Stemflow increased with the saturation of bark. *Dendrocalamus latiflorus* and *P. pinnata* allowed the most recovery of water. Approximately 99% of the volume applied was recovered (Fig. 6.25). At this point of

time the stem was completely saturated. Water loss in the form of stem drip was observed, although the volume lost was negligible.

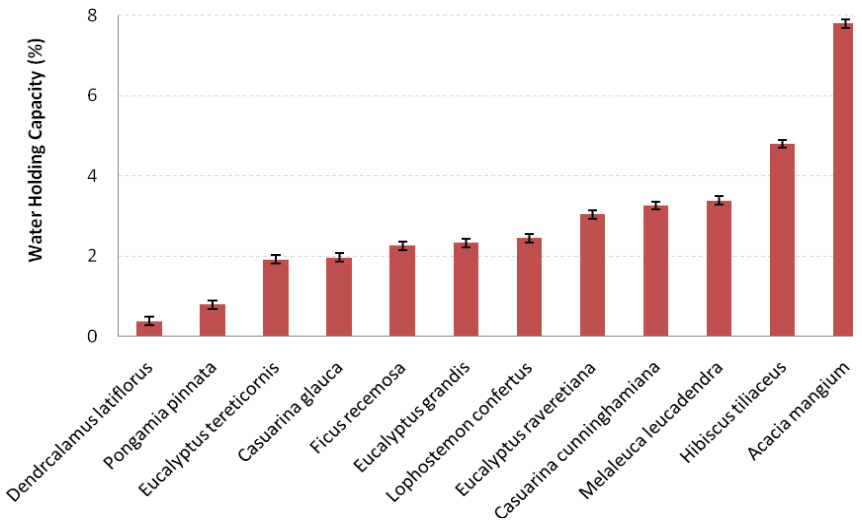


Figure 6.24: Water holding capacity of bark in 12 species grown on a phytocap
(Bars represent l.s.d. 0.104)

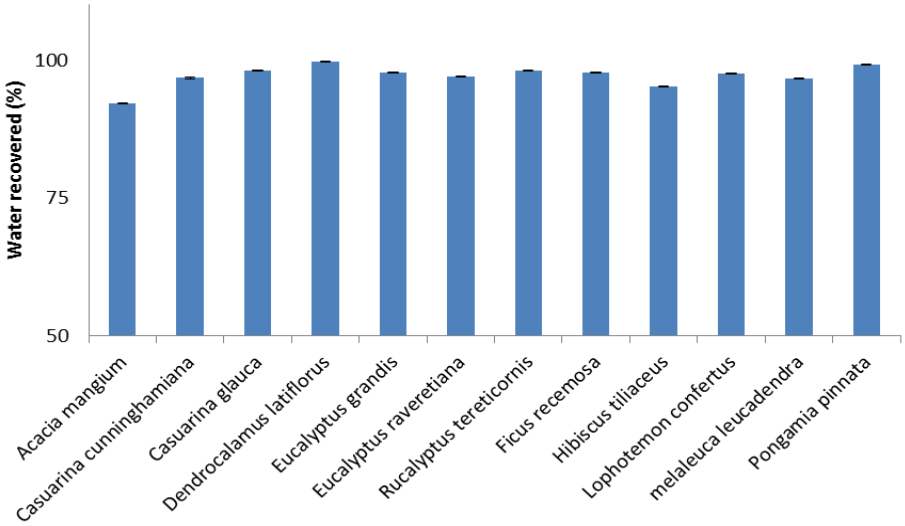


Figure 6.25: The proportion of water recovered from stemflow in various species grown on a phytocap
(Bars represent l.s.d. 0.104)

Table 6.5: Correlation between stemflow and selected tree parameters (r^2 values)

	Canopy area	Branch angle	Height	D_{50}	DBH	Combined
Thick phytocap	0.52**	0.13**	0.26**	0.49**	0.43**	0.49**
Thin phytocap	0.62**	0.42**	0.39***	0.64***	0.54***	0.83***
Thick & Thin phytocaps	0.37**	0.03**	0.33***	0.57***	0.46***	0.77***

Note: *** denotes significance at 0.001 probability

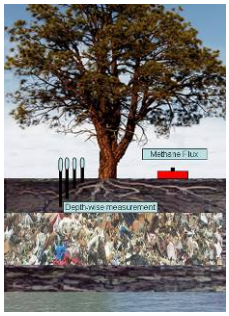
6.5 Conclusions

The 3 year-old trees that were established on Thick and Thin phytocaps were able to intercept up to 50% of the rainfall on a per storm basis, with an overall average of 30%. This is a significant contribution towards hydrological balance of the phytocapping system. A study conducted by Putuhena and Cordery (1996) showed that a pine leaf litter can hold 0.97 mm kg^{-1} m^2 while eucalypt leaf litter can hold 1.13 mm kg^{-1} m^2. The present study also demonstrated that canopy rainfall interception, including stem flow, varied significantly between species due to various factors such as their growth habit, leaf characteristics, canopy characteristics, bark texture, tree position and rainfall patterns. Canopy rainfall interception also varied between Thick and Thin phytocaps. The species that were established in the Thick phytocap intercepted more rain than the same species grown in Thin phytocap. This was due to better growth of trees in the thick phytocap with respect to height, stem diameter, larger canopy area and biomass. This is most likely as a result of deeper soil and better quality of soil in the thick phytocap compared to the thin phytocap. Canopy rainfall interception is also influenced by various leaf traits, contact angle of the droplets, drop size, bark texture and rainfall intensity.

The stemflow was estimated at 4.5% of the total rainfall and is similar to previous reports (e.g. 5% – Zinke 1966). Stemflow varied between species as they varied in their growth rate, canopy size, canopy structure, leaf morphology and leaf number. The species differences in stem diameter, canopy area and other features such as branch angle and bark texture contributed to variations in stemflow. Overall, the results clearly demonstrate the need to consider plant canopy interception as an essential parameter in

modelling water balance for sensitive sites such as landfills. Consideration of canopy interception when modelling the cost of constructing landfills can help reduce the gross costs. This is because volume of soil required to be used for phytocaps can be reduced considerably due to reduction in the effective rainfall reaching the surface.

By combining the results of Chapter 4, 5 and 6, it is clear that the trees grown on the phytocapping system have the potential to reduce water input into the buried waste. Trees and soil together are also shown to have a potential to reduce methane emission from landfills. The next Chapter shows how phytocaps can reduce methane emissions from landfills.



7

Methane Emissions[†]

7.1 Introduction

In Australia more than 50% of the municipal solid waste (MSW) is placed in landfills (EPA 2007, McLaughlin *et al.* 1999) as it is still the most economical method of waste disposal (CSIRO 2001, Tonini *et al.* 2009, Izzo *et al.* 2009). Landfills contain mostly putrescible wastes that degrade and produce leachate and gasses, predominantly methane when they come in contact with water (Jones and Nedwell 1993). Degradation of waste in landfills depends on factors such as soil moisture content (Ramaswaby 1970), oxygen levels in the soil (Christensen and Kjelden 1989), soil temperature (Farquhar and Rovers 1973) and pH of the soil (Ehrig 1983).

7.1.1 Methane Production

Methane is the dominant hydrocarbon present in the atmosphere with an average concentration of 1.7 ppm (Borjesson and Svanson 1997, Humer and Lechner 1999) and is found to be high around landfills (Giani *et al.* 2002). Methane is a very important greenhouse gas and currently accounts for 15% to 20% of the greenhouse gas economics (US EPA 2003). The total positive climate-forcing attribute of methane over

[†] Some data from this chapters have been included in the following papers:

Venkatraman, K. and Ashwath, N. (2011) Phytocapping: an innovative technique to reduce methane emission from landfills, *Perspectives in Environmental Research*, pp 245-258.

Venkatraman, K. and Ashwath, N. (2010) Phytocaps Reduce Methane Emission from Landfills, *Environmental Research Journal*, 4: 321-334. (Cross Publication)

Venkatraman, K. and Ashwath, N. (2009) Phytocaps Reduce Methane Emission from Landfills, In *Handbook of Environmental Research*, pp 341-363.

Venkatraman, K. and Ashwath, N. (2009) 'Can phytocapping technique reduce methane emission from municipal landfills?' *International Journal of Environmental Technology and Management*, 10: 44–55.

the last 150 years has been estimated to be 40% of that of carbon dioxide (Hansen *et al.* 1998).

Methane concentration in Australian landfills is comparable to levels emitted from landfills globally (Bogner *et al.* 1996). Methane production in landfills normally occurs in two stages - the non-methanogenic stage and the methanogenic stage (Whalen *et al.* 1990, Czepiel *et al.* 1995, Borjesson and Svensson 1997). The non-methanogenic stage is initiated by hydrolytic process (capturing energy) which reduces complex organic matter to smaller soluble compounds such as fatty acids, simple sugars, amino acids, water, hydrogen and carbon dioxide (Imshenetsky 1968, Farquhar and Rovers 1973). In the methanogenic stage, anaerobic bacteria decompose the waste to produce methane (Gebert *et al.* 2009). Gases such as nitrogen and hydrogen sulphide are also produced during this stage (Alexander 1971).

Microorganisms that are active in the methanogenic stage are generally the bacteria of the genus *Methanobacterium*, common inhabitants of soil and sewage (Czepiel *et al.* 1996, Bogner *et al.* 1997). However, methanogenesis is greatly influenced by soil moisture (Visvanathan *et al.* 1999), oxygen availability in the soil (Farquhar and Rovers 1973) and pH (Ehrig 1983, Cadillo-Quiroz *et al.* 2006).

Normally, a steady rate of methane production is reached after 80 to 500 days of waste deposition, and this rate is maintained for 10 to 20 years (Moore *et al.* 1998). Time required for the degradation of waste in landfills and the amount of gas formed depends on the type and quantity of waste buried; its water content and waste compaction (Farquhar and Rovers 1973). Waste composition differs considerably between countries. Municipal solid waste in Australia is dominated by garden and organic material (c. 50%), leading to a high potential for methane generation (Moore *et al.* 1998). Although the waste contained in landfills produces large quantities of methane, a considerable proportion of this gas is oxidised by *Methanobacterium* sp. in the soil covers placed over the waste (Czepiel *et al.* 1996, Bogner *et al.* 1997).

7.1.2 Methane Emission

A study conducted by Gebert *et al.* (2009) showed that methane flux in landfills was predominantly driven by the type of methanotrophs present and their population structure. Results from their study of five different landfills in West Germany suggested that the lack of nitrogen in soils influenced methanotrophic activity and that flux varied within a site and among the five sites due to the variations in species composition and their population structure. In spite of the variability in the soil, *Methylocystis* species was found to be dominant in all five landfills.

Bingemer and Crutzen (1987), Richards (1989) and Bogner (2003) estimated 9 to 70 Tg yr^{-1} methane emissions from landfills alone. Minimising methane emission from landfills will make a substantial contribution to reducing global warming. Generally, the rate of emission of methane (methane flux) from landfills varies over orders of magnitude being from less than 0.0004 g m^{-2} d^{-1} to 4000 g m^{-2} d^{-1} (Bogner *et al.* 1997b). Methane fluxes observed in various landfills in different countries are presented in Table 7.1.

Table 7.1: Methane flux rate in various landfills worldwide

Country	Methane flux g m^{-2} d^{-1}	Reference
Europe	0.1 – 0.3	Galle *et al.* (2001)
France	10.5	Pokryszka *et al.* (1995)
Germany	45	Jager and Peter (1995)
Japan various locations	-0.31 – 384	Ishigaki *et al.* (2005)
Moscow	0 – 38.4	Nozhevnikova *et al.* (1993)
South Africa	0.1 -0.3	Morris (2001)
Sweden	0.12 – 7.6	Borjesson and Svensson (1997)
Tokyo	200	Tohijima and Wakita (1993)
United Kingdom	0 – 38.4	Jones and Nedwell (1993)
USA	10776 (highest)	Bogner and Spokes (2003)

7.1.3 Methane Oxidation

Methane oxidation rates have been reported to be very high at 15% moisture content and temperatures ranging from 25°C to 30°C (Boeckx and van Cleemput 1996, Visvanathan *et al.* 1999). Municipal solid waste is often exposed to such conditions and hence they produce methane as one of the major components of landfill gases (Giani *et al.* 2002). Landfill gas comprises methane (45 to 60% v/v) and carbon dioxide (40 to 60 % v/v) (Swarbrick and Dever 1999). Studies conducted by Duffy *et al.* (1996) and Yuen (1999) in different landfills across New South Wales (NSW) and Victoria reported methane concentrations ranging from 50% to 62% in NSW, and up to 61% in Victoria. Carbon dioxide concentrations were 35% to 42% in NSW and 38% in Victoria.

Generally, methane oxidation rate increases with increasing soil temperature (De Visscher *et al.* 2001). Low soil temperatures inhibit methane oxidation (Whalen *et al.* 1990, Visvanathan *et al.* 1999). Borjesson and Svensson (1997) reported that soil temperature is the controlling factor of methane oxidation and can explain 85% of the variation in measured methane oxidation. Methanotrophic bacteria are favoured over a certain range of temperatures. Czepiel *et al.* (1996) found that the oxidation rate increased as the temperature increased to 36°C and stopped at 45°C. Humer and Lechner (2001) reported that methane oxidation rate was 70% to 80% of maximum at

18°C. At a lower temperature of 4°C, little oxidation was observed. Most studies have reported an optimum temperature 25°C to 35°C for methane oxidation (Whalen *et al.* 1990, Nesbit 1992, Dunfield *et al.* 1993, Boeckx *et al.* 1996, Borjesson and Svensson 1997, Visvanathan *et al.* 1999).

Methane oxidation depends on incident methane concentration (Singh 2000), soil type, soil compaction, soil aeration, soil temperature and soil moisture content (Czeipiel *et al.* 1996). In general, presence of plants enhances methane oxidation (Maurice *et al.* 1999). The depth at which maximum oxidation occurs varies with landfills. Czeipiel *et al.* (1996) and Whalen *et al.* (1990) found maximum oxidation rate occurring at depths of 5 cm to 15 cm and 3 cm to 6 cm, respectively, whereas Knightly *et al*. (1995) found the maximum oxidation rate occurring at a depth of 20 cm. The depth at which the maximum methane oxidation occurs depends on the type of soil used (Borjesson and Svensson, 1997). Methane oxidation rate is higher in coarse sand (166 g m^{-2} d^{-1}) than in fine sand or clayey top soil (110 g m^{-2} d^{-1}) (Knightly *et al*. 1995). In the current study methane concentrations were significantly lower in top layers (0 to 15 cm) due to the presence of mulch and the porous soil used providing optimum levels of oxygen required to oxidise methane. The Thick and Thin phytocaps were also vegetated with a wide range of species, which may have the ability to diffuse oxygen into the root zone (e.g. *M. leucadendra*). As this species naturally occurs in waterlogged soils, the diffusion of oxygen via the vascular system or specialised parenchyma with interconnected air spaces (Chan *et al.* 1991) helps aerate the root zone for its survival and growth.

Methane oxidation was reported to be high in the uppermost layers (15 to 40 cm) of soil (Visvanathan *et al.* 1999). Kallistova *et al.* (2005) found that 60% to 70% of methane was concentrated in the 45 to 60 cm layers of the soil cover, and decreased sharply in the uppermost layers. The highest recorded methane oxidation rate of 166 g m^{-2} d^{-1} was reported by Knightley *et al* (1995). Visvanathan *et al.* (1999) found a methane oxidation rate of 100 g m^{-2} d^{-1}. Both these values are much higher than the rates (45 g m^{-2} d^{-1}) reported by Whalen *et al.* (1990). Whalen *et al.* (1990) derived their values

from column testing which cannot be compared with the values of other studies as the column test did not include soil layers and the methane emission was not variable. Several researchers reported different maximum methane oxidation zones at different depths: between 40 cm and 60 cm by Nozhevnikova *et al.* (1993) and Borjesson and Svensson (1997), 15 cm and 60 cm by Barratt (1995), 3 cm and 12 cm by Whalen *et al.* (1990), and 20 cm and 30 cm by Knightley *et al.* (1995).

Soil compaction decreases soil aeration (Kozlowski 1999) leading to a slow oxidation process because oxygen is the key factor that activates the oxidation process and the rate of oxygen diffusion is directly proportional to soil porosity (Kozlowski 1999). Porosity can provide channels for oxygen penetration as well as increasing contact surface area for methanotrophic bacteria. Borjesson *et al.* (2004) reported a significant inverse relationship between methane oxidation and soil compaction. Soils with high porosity retain methane and oxygen longer in the pores leading to a higher oxidation rate (Humer and Lechner 1999). Soil oxygen is consumed by the roots and is replaced by diffusion process from the atmosphere. This diffusion is impeded by soil compaction (Kozlowski and Pallardy 1997). On the other hand, tree roots play a vital role in reducing methane emission by inducing root exudates, oxygen supply and also by supporting beneficial microbial populations that help to oxidise methane in the soil layers. Some species in the present study may have the ability to release monoterpenes from roots into the soil. Monoterpenes are known to increase methane oxidation in soils. Such observations were made by Maurer *et al.* (2008) in a study conducted in Europe. Optimal levels of nutrients help release exudates that act as catalysts to enhance the methane oxidation process within soil layers. However, in some instances, root exudates have been known to increase methane emission (Neue and Sass 1994, Chidthaisong and Watanabe 1997), and this is also regulated by nutrient status (Wassmann and Aulakh 1998). Some species may also have the ability to control methane flux by diffusion, ebullition and vascular transport (Gauci *et al.* 2010).

7.1.4 Mitigation of Methane from Landfills

There are primarily two ways by which methane emissions from landfills can be reduced. One is to use landfill gas recovery systems (Borjesson *et al.* 2000) and the other is to enhance methane oxidation in soil covers placed over the waste. Methane from landfills have been extracted, recovered and used for various purposes such as bio-energy and electricity generation (DeWalle *et al.* 1978). Gas recovery techniques have proven to be very efficient as they can reduce landfill methane emissions by up to 90% (Humer 1999). Bogner *et al.* (1993) found that methane emission had decreased by more than three orders of magnitude on sites where gas recovery systems have been installed as compared to adjacent landfills that had no recovery systems. However, installation of gas recovery systems is expensive and is not practical for small-scale operations in Australia (Craig Dunglison; personal communication).

Oxidation of methane in the soil covers that may or may not contain vegetation would offer an economical way of reducing methane emission from landfills (Ashwath and Venkatraman 2007). Firstly, the soil cover acts as storage of water and prevents the waste from coming in contact with the water. Secondly, it helps assist the growth of microorganisms and plants, which together assist in methane oxidation. Soil covers can oxidise 7% to 50% of the methane generated from landfills (Knightley *et al.* 1995). Czepiel *et al.* (1996) reported a 10% methane oxidation in a landfill site in north-eastern USA during winter, and a higher rate of 20% during summer in a similar landfill site. Under certain conditions, landfills can even absorb atmospheric methane and oxidise methane to produce carbon dioxide (Borjesson and Svansson 1997, Bogner *et al.* 1997, Bogner *et al.* 1997a). Establishment of phytocaps could offer an economical way of reducing methane emission from landfills. This Chapter reports the findings of a field study conducted to test the effect of two phytocaps (700 mm and 1400 mm) on methane emission.

7.2 Materials and Methods

Methane emission from the Rockhampton experimental site was measured within thick and thin phytocaps using a portable methane gas meter (Gastech, Australia, 2004).

7.2.1 Diurnal Variation

Diurnal variations in methane concentrations were determined by monitoring methane continuously separately underneath 19 species over 24 hours at 17 months (19 February 2005), 18 months (15 March 2005) and 19 months (22 April 2005) after planting. PVC tubes were used for these measurements and they were buried around plants to a depth of 30 cm (root zone) (Fig. 7.1A). The tube was left open at the bottom and closed with a lid at the top. During measurement, the lid of the tube was removed and the probe of the gas meter (GT series) (Gastech Australia 2005), which could detect oxygen, carbon dioxide, methane and hydrogen sulphide simultaneously was inserted into the PVC tube to a depth of 20 cm below the soil surface. Methane concentrations were recorded after the meter readings were stabilised. Concentrations of carbon dioxide, oxygen and hydrogen sulphide were also recorded. Based on the observations recorded in this preliminary study, all further methane readings were recorded between 9 am and midday.

7.2.2 Surface and Root Zone Concentrations

Methane concentrations were monitored in the adjacent areas of the experimental site that were kept devoid of vegetation (bare site that contained 500 mm to 1000 mm of interim uncompacted soil cover over the waste). Methane concentrations were measured at two depths, one at the surface and the other in the root zone (30 cm below the surface). Root zone methane was measured following the same procedure as described in section 7.2.1. PVC tubes were installed at a distance of approximately 50 cm from the tree trunk. One tube was installed per plot (or species) in each of the thick and thin phytocaps and from both replications.

For surface methane measurements, the inlet tube of the methane meter (Gastech Australia Pty LTD) was connected to a 70 mm diameter plastic funnel, which was placed inverted on the surface (Fig. 7.1A). The funnel was twisted left and right to ensure proper positioning and sealing to minimise the meter pulling air from outside the enclosed area. For surface measurements, up to five readings were taken randomly within each species in thick and thin phytocaps and from both replications.

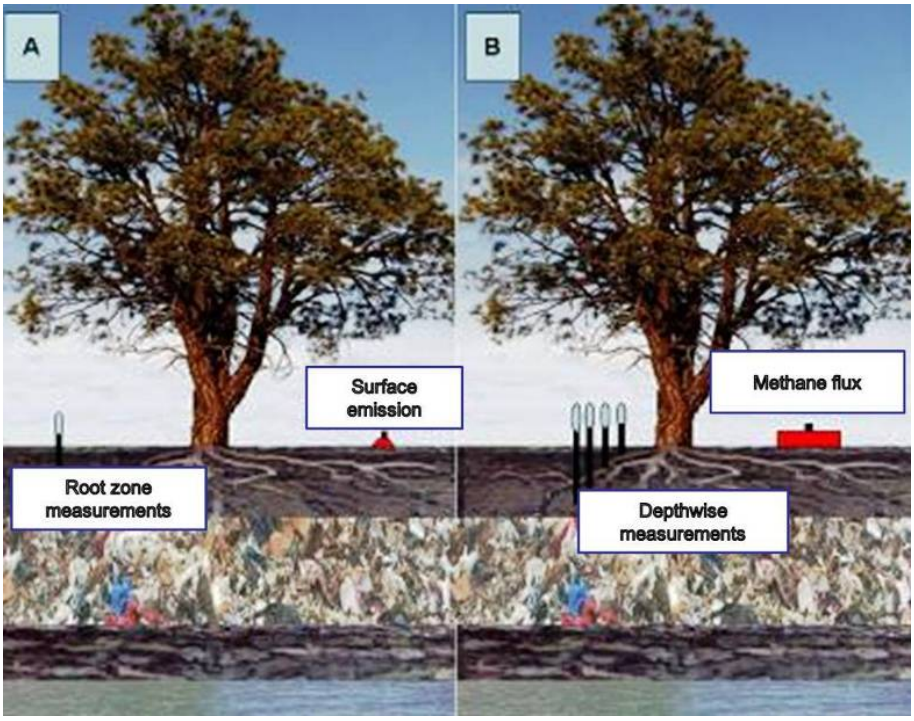


Figure 7.1: Various techniques used to measure methane emission from phytocaps
(A) Methane measurement at root zone (left to the tree); Methane surface measurement (right to the tree), (B) Depth wise methane concentration (Left to the tree) and Methane flux measurement (right to the tree)

7.2.3 Effect of Species and Soil Depth on Methane Concentrations

For species and depth wise methane studies, only 5 of 21 plots (each plot represent 1 species x 18 seedlings) were selected each in thin and thick phytocaps of the first replication. Methane emission was measured within each of the 5 selected plots using a portable methane gas meter (Gastech, Australia 2004). Methane concentrations were measured at five different depths: at the surface and below the surface at the depth of 15 cm, 30 cm, 50 cm and 90 cm respectively (Fig. 7.1B). The surface methane

concentrations were recorded using an inverted funnel placed airtight on the surface with the tapering end connected airtight to the probe of the gas meter as before. The measurements at 15 cm, 30 cm, 50 cm and 90 cm were achieved using PVC tubes (20 mm diameter) that were buried to the desired depth 4 weeks prior to the measurement (Fig. 7.1B). The tubes were left open at the bottom and were closed with a lid at the top. The experimental set up of the depth-wise estimation of methane concentration is shown in Figure 7.1B. The PVC tubes were installed 30 cm away from the tree trunk, with a spacing of 10 cm between tubes. Depth-wise methane measurements were taken on nine different days during January 2007 to April 2007 (39 to 41 months after establishing the trial).

7.2.4 Methane Flux

Several methods have been employed to determine methane flux in landfills (Diot *et al.* 2002). These include accumulation chamber, static chamber, infrared thermograph method, external recirculation chamber and tracer method. The chamber methods consider an increase in gas concentration within a known volume of air, during a measured time period (Fischer 1999). These measurements are less expensive and yield data in a short time (Fischer 1999). Chambers may be used statically or dynamically (Fischer 1999). In the current study, the static chamber technique was employed (Fig. 7.1B). A static chamber consists of a sealed container of a known volume placed over the landfill surface to measure increase in methane concentration over a short period of time (Fischer 1999). Since the gas flow is caused by diffusion, methane concentrations reached its maximum and stabilised after a certain time depending on the size of the container and the gas flow rate. The change in concentration was plotted as a function of time and methane flux was calculated as ppm based on the slope.

The gas concentrations in ppm values were later converted to that in g m^{-2} d^{-1} using the ideal gas equation (Sawyer *et al.* 1994) as shown below:

Volume of the container = $\pi r^2 h$ = 3.14 x 10 x 10 x 4 = $\underline{1256\ cm^3 = 0.001256\ m^3}$

Surface area = πr^2 = 3.14 x 10 x 10 = 314 cm^2 = $\underline{0.0314\ m^2}$

Ideal gas equation: PV = nRT, Where

P = Pressure = 1.01325 x 10^5 Pa = 101325 Pa

V = Volume of the container = 0.001256 m^3

n = no of moles of gas

R = gas constant = 8.314 m^3. Pa. K^{-1} mol^{-1}

T = Temperature = 302.05 °K (or 28.9 °C; average of three days)

Two sets of observations were taken from each of five plots randomly selected plots on thick and thin caps. Similar observations were taken from adjacent areas of the experimental site that had no vegetation but had soil cover of similar depth as the two phytocaps. Changes in methane concentrations were measured at one minute intervals for a total time period of five minutes. In most cases methane concentrations stabilised after 2 to 3 minutes. The observations were then plotted against time to determine the slope.

7.2.5 Root Depth and Soil Compaction Measurements

Protocols followed to determine root depth are given in Chapter 3 Section 3.2.1.5 and those for soil temperature are given in Chapter 2. Soil compaction was determined using the standard proctor compaction test at 20 cm, 30 cm, 40 cm, 50 cm, 60 cm, 70 cm and 80 cm soil depth under each species in plot 3 (Thick cap) and plot 4 (Thin cap). This exercise was conducted during tree harvest in 2007.

7.3 Statistical Analysis

Data obtained from this study was tested for outliers, normality and homogeneity of error variances before subjecting these to ANOVA using Genstat ver. 8.0 (Wass 2011, Payne 1997). Parameters that showed significance for the F test (P<0.05) were subjected to t test. Effects of time were also tested for some parameters that were measured repeatedly. Least significance differences (l.s.d.) are presented in figures where the treatment, capping, capping depth, species effect, time or their interactions

were significant (P<0.05). Standard errors of differences are provided for some parameters where there were insufficient data available for ANOVA, or when the F test was found to be not significant (P<0.05).

7.4 Results and Discussion

7.4.1 Diurnal Variation

Twenty four hour monitoring of the soil surface methane concentrations revealed that the measurements taken around 9 am were found to be high and consistent in both Thick and Thin phytocaps (Fig. 7.2). This time of the day also coincided with the cooler periods for field work in the tropical climate, thus all further methane monitoring was carried out between 9 am and midday Australian Eastern Standard Time (AEST).

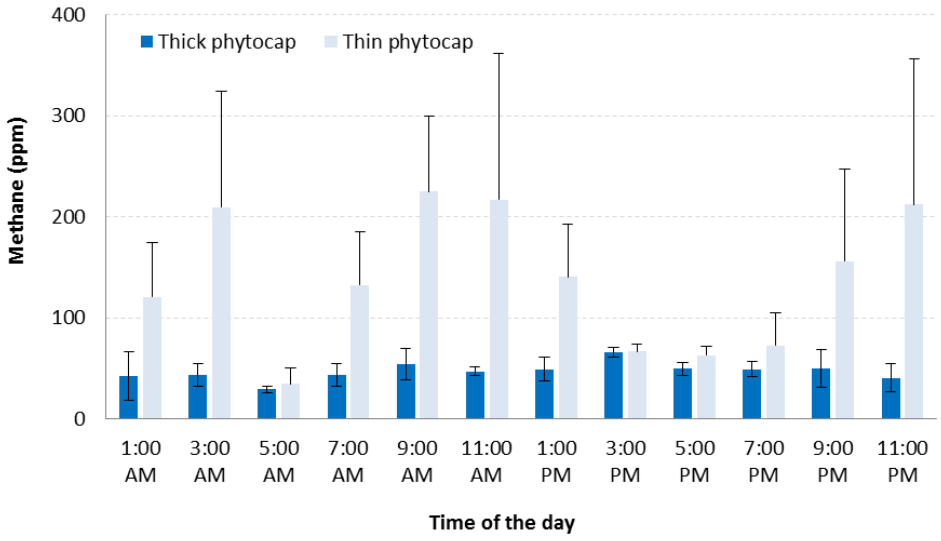


Figure 7.2: Diurnal variations in root zone methane concentrations in Thick and Thin phytocaps Note that methane concentrations were up to four times higher in Thin cap than in Thick cap at most times of measurements. The data were collected on 19 February 2005, 15 March 2005 and 22 April 2005). (The bars represent standard errors; n = 3)

7.4.2 Methane Concentrations at Surface and Root Zone

Methane concentrations in Thick and Thin phytocaps showed a consistent trend in both surface and root zone measurements. Root zone methane concentrations were consistently lower in the Thick phytocap than in Thin phytocap for all tested species

(Fig. 7.3A) as the roots zone in the thin phytocap was at a closer proximity to the buried waste than that of the root zone in the Thick phytocap. The surface methane concentrations were also lower in theThick phytocap for the majority of the tested species (Fig. 7.3B) due to different composition of soil layers and soil thickness in the Thick phytocap; this presumably allowed more time for oxidation of methane to occur.

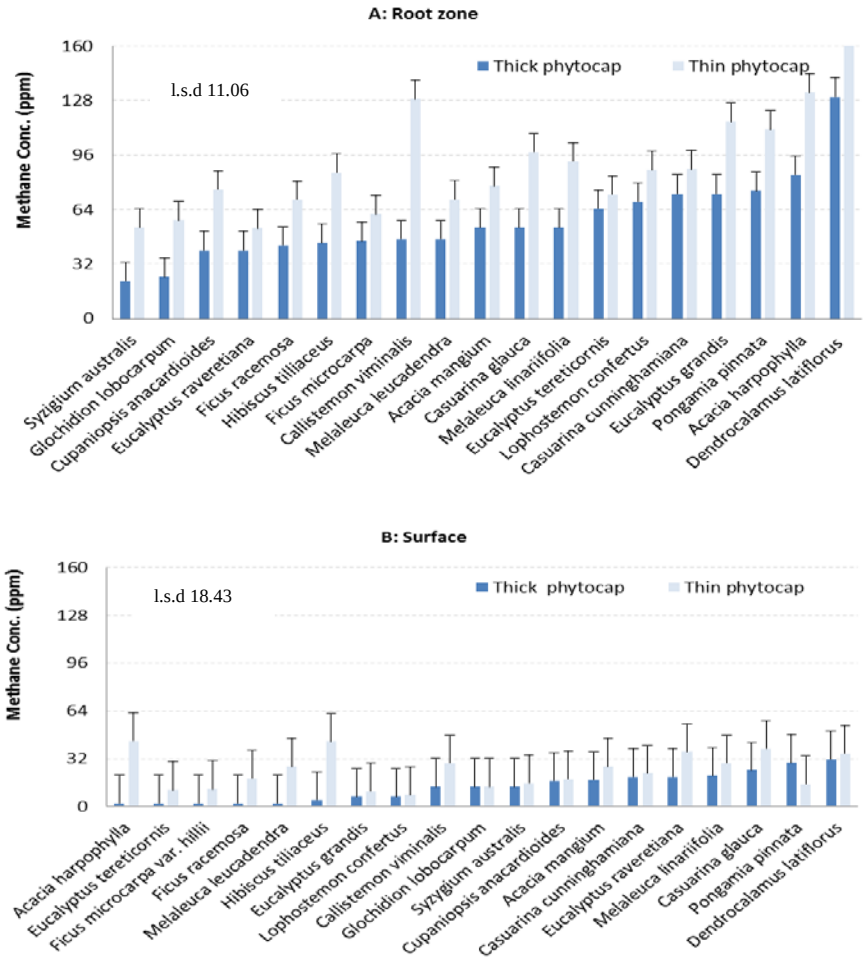


Figure 7.3: Methane concentrations in surface and root zones of a Phytocapping system
A: root zone (30 cm below the surface) and B: surface of a Thick and Thin phytocap. The values are means of nine observations collected over three seasons (Mar 05-Jan 06). (The bars represent l.s.d for species capping interaction

The methane concentrations were significantly (P<0.001) lower at the surface than in the root zone for majority of the tested species was associated with methane oxidation in the soil (Fig. 7.4). These findings are supported by those of Bogner *et al.* (1997) and Christopherson *et al.* (2000).

Figure 7.4 shows a large variation between species in their root zone methane concentrations. Some species such as *L. confertus* and *D. latiflorus* showed marked differences between the surface and the root zone methane levels, indicating their contribution to methane reduction. This variation in root zone methane levels could also be due to the type and composition of the buried waste. General inferences can be made about contribution of tree species to methane reduction but no firm conclusions can yet be drawn about the role of species in methane oxidation because of the lack of information on the concentrations of methane passing through the root system, the spatial variability in methane emission in the root zone, and most importantly, lack of information on species rooting patterns in the uppermost layer of the soil (up to 1 metre in depth).

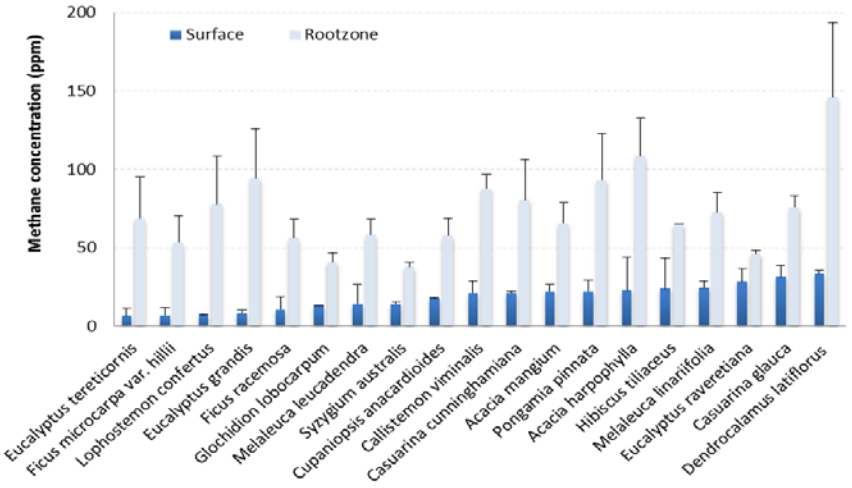


Figure 7.4: Comparison between root zone and surface methane concentrations
The values are average methane emission from Thick and Thin phytocaps (18 observations over 3 seasons), and the bars represent standard errors of means; n=2)

7.4.3 Effect of Soil Thickness on Methane Concentrations

Overall, the Thick phytocap was 55% more efficient in reducing methane emission compared to the Thin phytocap (Fig. 7.5) Significantly lower (P<0.001) levels of surface and root methane concentrations (Figs. 7.4) in the Thick phytocap was due to greater exposure of methane to a larger volume (depth) of soil which may have an increased rate of oxidation by soil bacteria (Bogner *et al.* 1997, Khalil *et al.* 1998, Kallistova *et al.* 2005) and/or diffusion with the atmospheric oxygen. Consideration of

the soil thickness required to reduce percolation of water into the landfill is more important than that needed for reducing methane emission, as landfill operators are required by law to limit the entry of water into the landfill, while they are not required to reduce methane emission.

Surface methane concentrations in the non-vegetated landfill site were significantly (P<0.001) higher compared to those in the Thick and Thin phytocaps (Fig. 7.5). This shows potential of phytocaps to reduce methane emission by 4 to 5 times that of a non-vegetated site. However, the role of trees in reducing methane emission from landfills is unclear. Several researches in the past have confirmed the role of tree roots in enhancing methane oxidation. Some plants also provide channels for oxygen via their roots or via indirect benefits to methanotrophic bacteria (via root exudates) to oxidise methane to carbon dioxide (Ding *et al.* 2005).

The methane concentrations measured at the surface and in the root zone of the phytocaps were influenced by the nature of waste buried and the soil moisture content. Since the type of waste buried under a phytocap could differ markedly (e.g. car bodies, timber or pure domestic waste), large spatial variations in methane emission can be expected in the root zone.

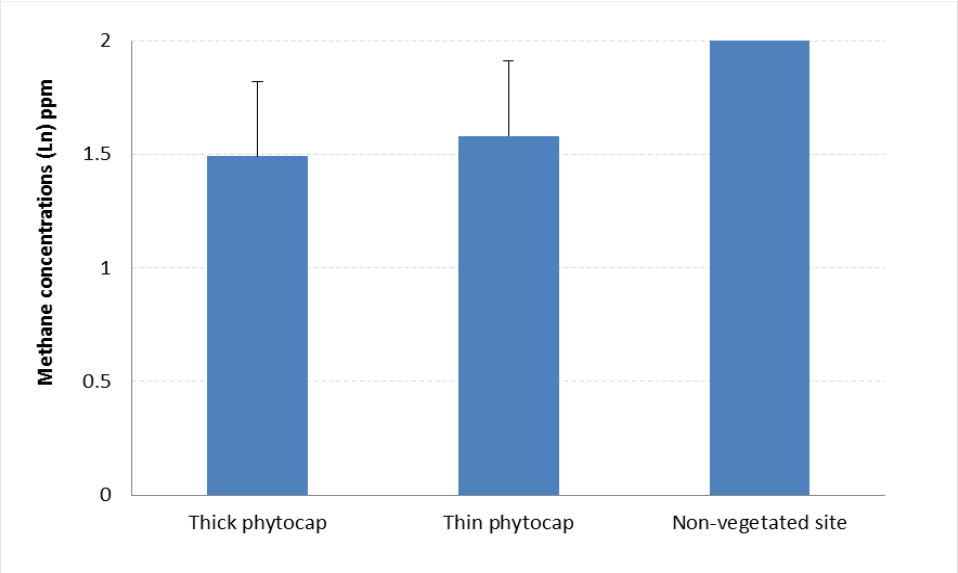


Figure 7.5: Surface methane concentrations in the phytocapped site and its adjacent bare site (Bars represent l.s.d. 0.331)

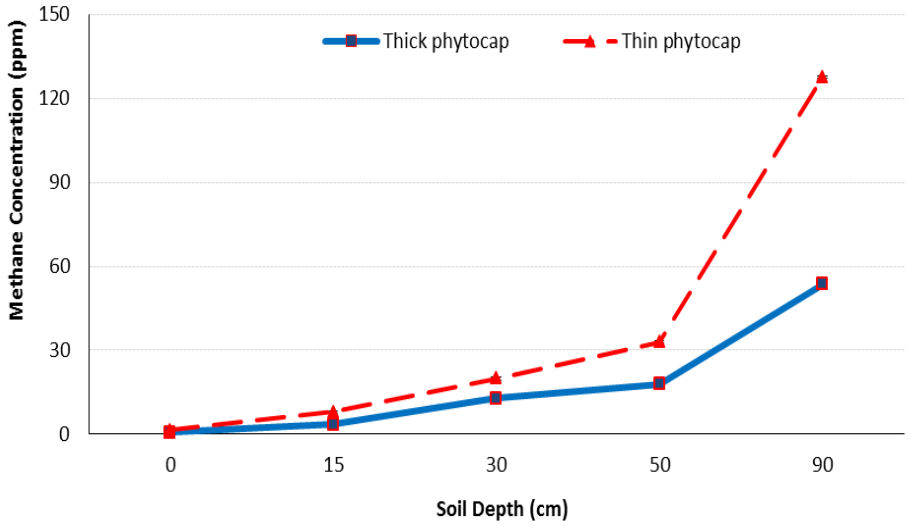


Figure 7.6: Methane concentration at different soil depths in Thick and Thin phytocaps (Data are means of 5 measurements taken for 5 species over 9 days. (l.s.d. 0.6)

Methane concentrations decreased significantly (P<0.001) from 90 cm depth to the surface in both the Thick and Thin phytocaps (Fig. 7.6). Overall, methane concentrations were higher in Thin phytocap than in the Thick phytocap. Higher concentrations of methane are expected in the deeper layers as these layers are the first to come in contact with the gas from the waste. Depth-wise decrease in methane concentrations may be due to methane oxidation in the soil and/or due to diffusion with the atmospheric air.

Methane concentration in Thin phytocap was on an average 49% more than that in the Thick phytocap (Fig 7.6). This difference in methane concentrations may be attributed to the proximity of the sampling points to the waste in Thin phytocap as compared to the sampling points in the thick phytocap. Similar observations were made by Nozhevnikova *et al.* (1993) in a closed landfill in Moscow, Russia.

Approximately 98% of the methane that was observed at 90 cm soil depth was reduced before it reached the surface (Fig. 7.6). This was associated with the combined effects of roots in supporting the microbial population (Gregory 2006) and soil in oxidising

methane (Berger 2005). Root depth in the different species grown on Thick and Thin phytocaps ranged from 40 cm to 80 cm (Fig. 7.7). Preliminary results from the Australian Alternative Cover Assessment Program (A-ACAP) indicated that vegetation can alter soil physical properties to enhance oxygen availability, which will increase the methane oxidation capacity of the phytocap (Sun *et al.* 2011). However, De Visscher *et al.* (1999) emphasised that some species grown on landfill soil cover may inhibit methane oxidation by nitrogen uptake. A study conducted by Bodelier and Laanbroek (2004) confirmed that the application of NH_4NO_3 to soils reduced the uptake of atmospheric methane by up to 33%.

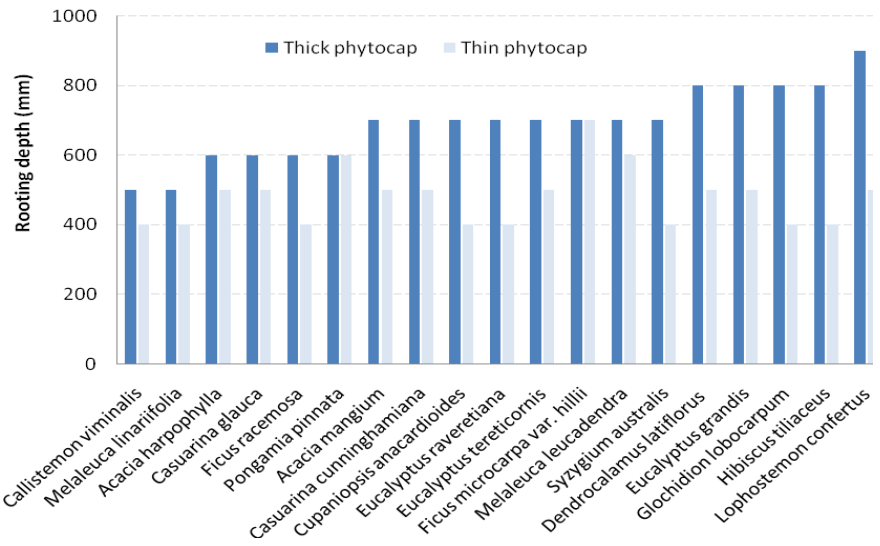


Figure 7.7: Root depth in various species grown on Thick and Thin phytocaps (3 years after planting)

Soil compaction also reduces the movement of methane into the atmosphere, which also inhibits diffusion of methane with oxygen in the soil. Figure 7.8 shows soil compaction levels at each depth of soil. The soil at 20 to 40 cm depth had higher compaction levels (120 kPa to 199 kPa) than deeper layers, possibly contributing to methane oxidation.

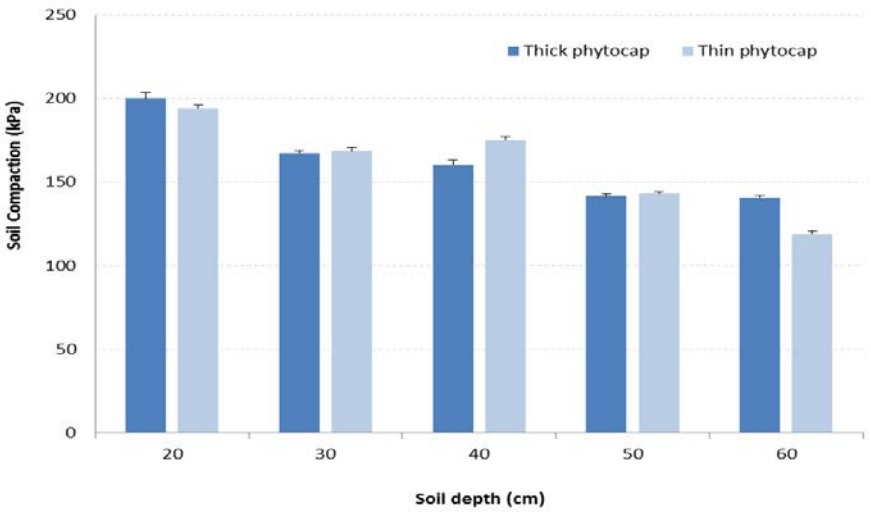


Figure 7.8: Soil compaction levels at different soil depths
(Bars represent SE, n=2)

Soil temperature, another parameter impacting on methane oxidation, was recorded between 2 m and 5 m depth. Soil temperature in this study averaged 30°C with a <2°C difference in temperature between the 5 m and 2 m soil depth (Table 7.2). Overall, the results from this study suggest that phytocaps can reduce methane concentrations by 7 times (P <0.001) compared to the incident methane concentration at 90 cm depth (Fig. 7.9).

Table 7.2: Soil temperature at 2 m and 5 m depth (2006)

Month	2 m	5 m
March	31.6	31.2
April	31.5	31.2
May	30.2	36.2
June	30.5	31.3
July	30.2	28.2
August	24.1	30.7
September	24.6	30.4
October	28.7	29.9
December	31.3	30.0
Average	29.2	31.0

Methane concentrations decreased significantly (P<0.001) from deep to shallower layers of soil. Similarly, a study by Rose and Mahler (2009), using varying percentages of compost in the soil, found that 100% compost had an average methane oxidation rate of 43% and a maximum of 97%. In contrast, soils containing 25% and 50% compost had average methane oxidation efficiencies of 20% and 25%, respectively. Methane concentrations also varied significantly (P<0.001) between the five examined species used in this study. The variability was highest in the surface emissions. Amongst the five species that were studied, *F. microcarpa* and *D. latiflorus* had a higher ability to oxidise methane in their root zone (Fig. 7.9). This was possibly due to the presence of profuse rooting and the ability of the tree roots to support methane oxidation.

Higher concentrations of methane at 90 cm soil depth under *F. microcarpa* and *D. latiflorus* (Fig. 7.9) may be associated with the type of waste buried and possibly due to the initial inhibition of methane oxidation, owing to the lower oxygen availability or due to a delay in supply of root-exuded substrates for anaerobic bacteria, or by both.

Addition of mulch also enhances methane oxidation in landfill cover (Rose and Mahler 2009). The initial size of the mulch used ranged from 75 mm to 100 mm, providing a highly porous cover, and this may have allowed optimal oxygen diffusion into the soil. Similar observations were made by Chanton and Liptay (2000), Hilger and Humer (2003) and Streese and Stegmann (2003) who found that mulch and other sources of organic content in the soil can reduce methane emission into the atmosphere.

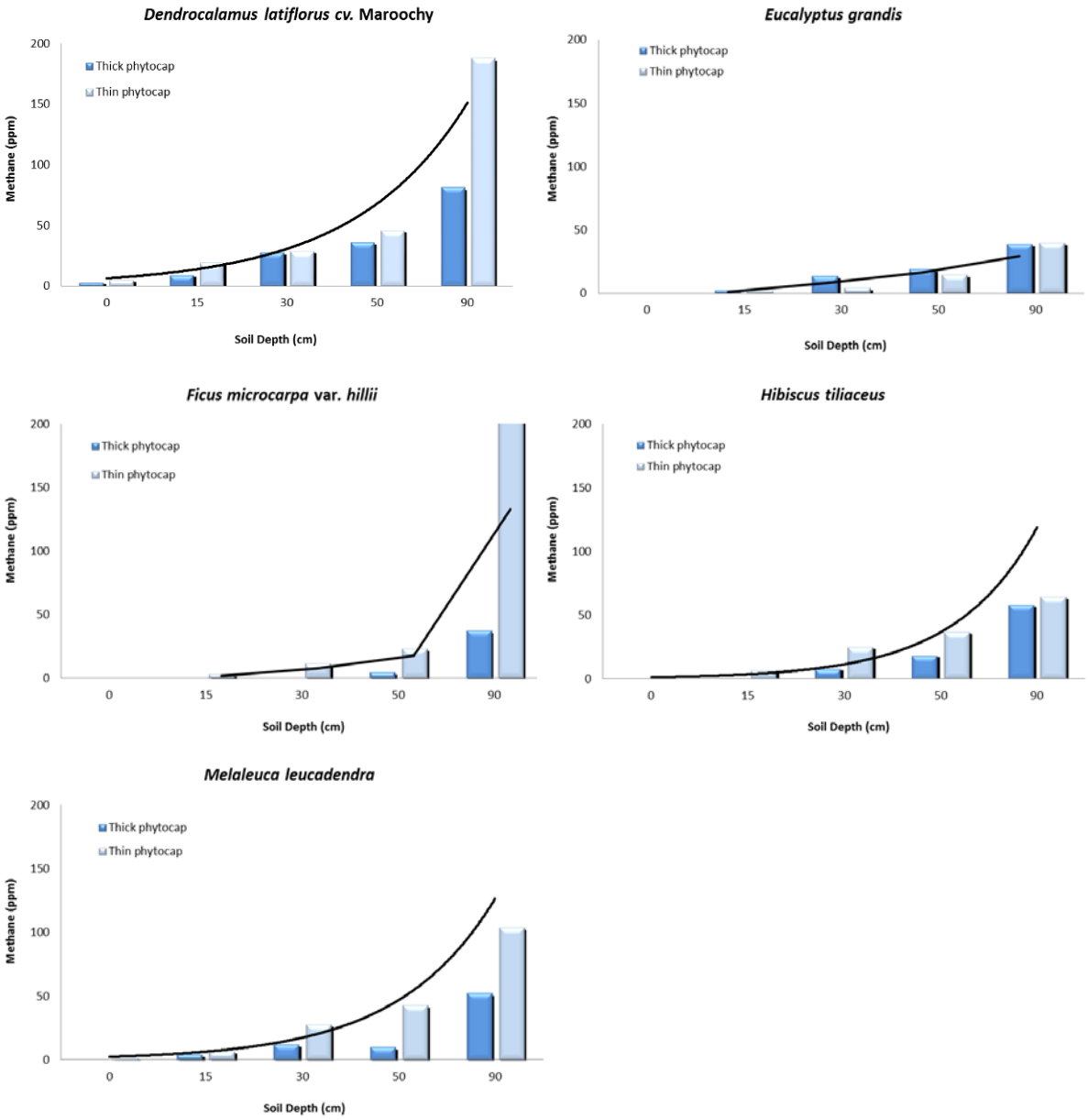


Figure 7.9: Depth wise methane concentrations in the root zones of five species grown on Thick and Thin phytocaps
(Values are average of nine observations over three months) (L.s.d. for species.depth = 0.894)

7.4.4 Methane Flux

Methane flux rates in Thick and Thin phytocapping systems ranged between <0.0007 g m^{-2} d^{-1} and >0.0009 g m^{-2} d^{-1} as compared to >0.0036 g m^{-2} d^{-1} in the adjacent non-vegetated landfill. These values are on the lower side than those reported by Bogner *et al.* (1997) (0.0004 g m^{-2} d^{-1} to 4000 g m^{-2} d^{-1}) due to the age of waste (c. 25 years), which may have reached its peak degradation potential and may be attributed to tidal pressure created, which may have exhausted most of the generated methane. It has been observed that during low tide there is a suction pressure, which sucks in the

methane concentrations and releases these at once during high tides. This phenomenon and its frequent occurrence over the years would have exhausted methane concentrations within the Lakes Creek Road landfill. At this stage methane generation from this waste is quite low. Normally, a steady rate of methane production is reached after 80 to 500 days of waste deposition, and this rate is maintained for 10 to 20 years (Moore *et al.* 1998) after which it gradually declines.

Methane flux also varied significantly (P<0.001) with the day of measurement (Fig. 7.10) and the phytocapping treatments (P=0.031) (Fig. 7.11). This may be associated with soil moisture, soil temperature (Boeckx and Cleemput 1996), soil properties (Urmann *et al.* 2009) and the types of methanotrophs (Gerbert *et al.* 2009). High methane oxidation can only occur if supply of oxygen and the populations of methanotrophs are adequate (Gerbert *et al.* 2009, Rose and Mahler 2009). Seasonal variability of methane flux is reflected by the composition of soil gas profiles (Rachor *et al* 2009). Among the three treatments herein measured, the non-vegetated site showed a greater methane flux than the Thick or Thin phytocaps. This to a certain degree demonstrates the beneficial effect of tree roots on methane oxidation.

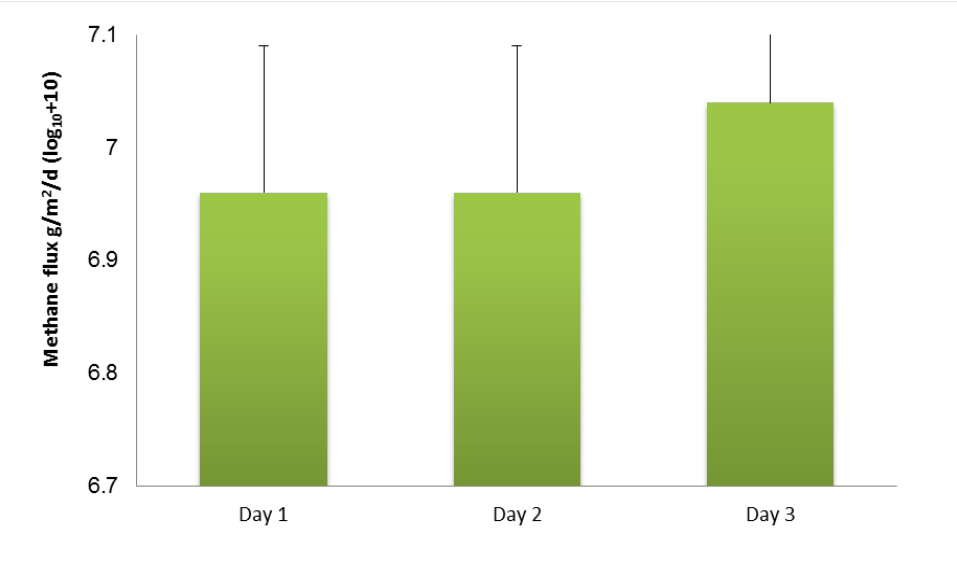


Figure 7.10: Methane flux estimated in Phytocapping systems
(n = 3) (Bar represents l.s.d. 0.1316)
Note: All data were converted to $\log_{10}$+10 to facilitate ANOVA, which does not accept 0 values

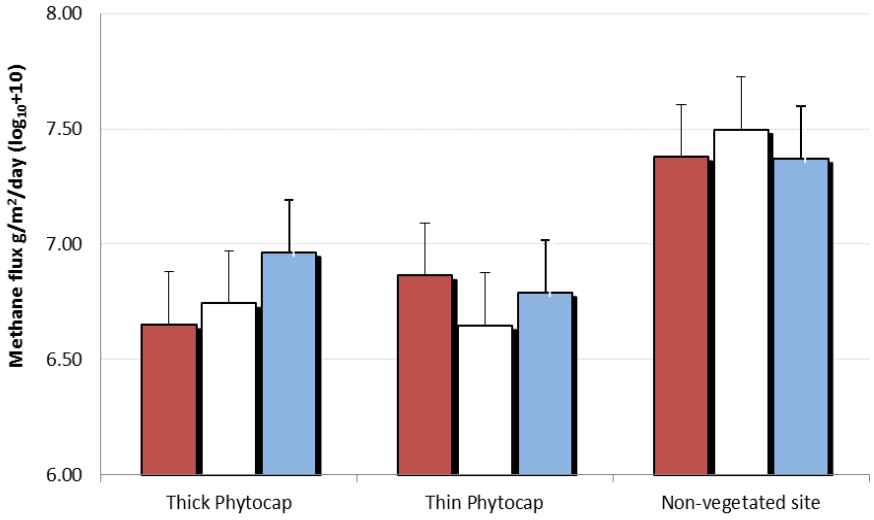


Figure 7.11: Methane flux from Thick and Thin phytocaps and their adjacent non-vegetated site
The three bars in each category correspond to methane flux on three consecutive days (l.s.d. 0.22)
Note: All data were transformed to $\log_{10}+10$ to facilitate ANOVA, which does not accept 0 values

7.5 Conclusions

Methane concentrations were monitored with a view to testing the effects of Thick and Thin phytocaps on methane emissions. Overall, phytocapping reduced methane emission from landfills (by 4 to 5 times). However, several factors such as methane oxidation rate in the soil, microbial population count in the two phytocaps and the spatial and temporal dynamics of methane needs to be investigated in detail to provide clarity on the role of soil thickness and trees in minimising methane emission.

Based on results obtained in this study, it is recommended to use 1000 to 1500 mm of unconsolidated soil, because the Thick phytocap was 55% more efficient in reducing methane emission compared to Thin phytocap. Depth-wise monitoring of methane concentrations showed a significant decrease in methane emission. Species differences were noticed for surface and root zone methane concentrations, but species effects could not be separated due to confounding effects of other uncontrollable factors.

Methane flux in the Thick and Thin phytocapping systems ranged between <0.0007 g m^{-2} d^{-1} and >0.0009 g m^{-2} d^{-1} as compared to >0.0036 g m^{-2} d^{-1} in the adjacent non-

vegetated landfill. Although the measured flux was lower than those reported in the past, there is a clear indication that phytocaps will reduce methane emission. The lower flux in the current study may be attributed to the age of the landfill and/or due to the type of waste buried beneath the two phytocaps. Certain species grown in the phytocapping system have the ability to pump oxygen into the soil and also produce exudates that may enhance methane oxidation; however the rate of methane oxidation in these soils is unknown and requires further investigation.

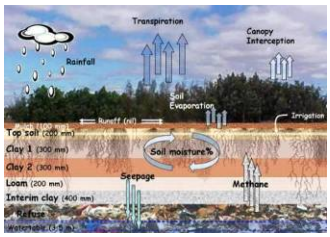
This study shows the potential of phytocapping systems to minimise methane emission from landfills. According to Falzon (1997), methane from landfill accounts for 13.5% of Australia's total emissions, with an estimated 710,1000 tonnes of methane being released into the atmosphere annually. With current technology (gas extraction), it is feasible to recover up to 90% of the methane (Falzon 1997) but the economic viability of landfill gas extraction depends on size of the landfill and the quantity of waste disposed. Gas recovery systems are expensive and not practical to be installed at landfills that are accepting less that 100,000 tonnes of waste per year. Investigation on the role of phytocaps in methane oxidation will provide a new horizon for small and medium sized landfill operators to tackle the greenhouse emission economically.

Secondly, the proposed carbon tax/Emissions Trading Sceme under the Clean Energy Act 2011 will be introduced Australia wide sometime in 2013, which will have a significant impact on landfill operators. Landfills emitting 25000 t Co_2^{-e} or more will be obligated to pay a carbon tax of approximately $23 per tonne of Co_2^{-e}. Landfills emitting as low as 10000 t Co_2^{-e} may also be included. However, as part of the Carbon Farming Initiative there are opportunities to offset carbon via gas recovery, flaring or diversion of organic waste from landfills. There are other opportunities for landfill operators to purchase carbon credits and sequester carbon via afforestation programs. Phytocapping, an alternative landfill capping technology using tree has a great potential to offset carbon through sequestration. However this needs to be further explored in detail.

Overall, phytocaps can be employed in all landfills, which could reduce methane emission. Currently, flaring or gas lighting has been accepted as a practical option

for smaller or older landfill sites (Treloar 1998). Results from the current study clearly demonstrate that phytocaps can abate methane emission while also maintaining the hydrological balance of landfill site.

The next Chapter presents an outcome of a water balance simulation undertaken using HYDRUS 1D code. The Chapter discusses percolation rates of Thick and Thin phytocaps using the selected ten species which performed well on landfills.



8

Site Water Balance*

8.1 Introduction

Landfill leachates are generated as a result of water percolating into waste and dissolving contaminants (Christensen *et al.* 1994). Water sources in a landfill include water contained in the waste, soil cover, rain (Bengtsson *et al.* 1994) and groundwater, or even tidal water intrusion such as at the site used in this study where the Fitzroy river tides were influencing the water balance of the site (Ashwath and Venkatraman 2010). It is important to assess effectiveness of phytocaps with respect to the amount of water that percolates into the waste. For these reasons numerous water transport models have been used to analyse and measure percolation rates based on Richard's equation and the water balance method (Albright *et al.* 2002, Williams 2005). Models using Richard's equation have shown more accuracy (Albright *et al.* 2002) and have proven to be better than the water balance method due to their ability to describe water flow in any direction (Jirka Simunek, pers. comm.).

Researchers have used a number other models to measure percolation rates in many situations and scenarios. Several recent modelling studies have also been undertaken to assess hydrological performance of landfill covers (Chai and Miura 2002, Ho *et al.* 2004), with the majority of them measuring seepage production (Dho *et al.* 2002, Ham 2002). Models used in previous studies include UNSAT-H, HYDRUS 1D, HYDRUS 2D, Simulation of Heat and Water (SHAW), Vadose/W, Soil Water Balance and Infiltration Model (SWIM), The Hydrologic Evaluation of Landfill Performance (HELP), TOUGH-2, MACRO and The Leaching Estimation And Chemistry Model (LEACHM) (Fayer *et al.* 1992, Fayer and Gee 1997, Khire *et al.*

* Some data from this chapters have been included in the following papers:
Venkatraman, K., Ashwath, N. and Su, N. (2009) Performance of a phytocapped landfill in a semi arid climate, *In Technologies and Management of Sustainable Biosystems*, pp 195-208.

1999; Johnson *et al.* 2001, Scanlon *et al.* 2002, Albright *et al.* 2002, Benson *et al.* 2004). Amongst these models, HYDRUS 1D, HYDRUS 2D, UNSAT-H and Vadose/W are used most frequently in evaluating the effectiveness of phytocapping systems (Benson 2004).

The Erosion Productivity Impact Model (EPIC) (Williams 2005) uses the water balance method and has been extensively used in agriculture, but was not found robust compared to models that use Richards's equation. Nevertheless, EPIC has been proven to be better than HELP (Hauser and Gammon 2001). HELP was compared with the Vadose/W (Chammas *et al.* 1999) and UNSAT-H (Khire *et al.* 1997) models during studies conducted by the Alternative Cover Assessment Program (ACAP) and the Alternative Landfill Cover Demonstration (ALCD) (Khire *et al.* 1997) project. It was found that the percolation rates were over-predicted by HELP in comparison with UNSAT-H (Khire *et al.* 1997). Hauser and Gimon (2001) compared HELP, HYDRUS and UNSAT-H and reported that UNSAT-H and HYDRUS were more accurate than HELP for phytocaps. Another study by Scanlon *et al.* (2002) compared HELP, HYDRUS, SHAW (Albright *et al.* 2002), Vadose/W, SWIM (Dwyer 2003) and UNSAT-H using water-balance data from covers in semi-arid Texas, New Mexico and Idaho, over periods ranging from one to three years. Scanlon *et al.* (2002) concluded that models employing Richard's equation such as UNSAT-H, SWIM and HYDRUS 1D/2D predicted water balance more accurately than the HELP model.

Other models such as MACRO (Johnson *et al.* 2001) were not as robust as HYDRUS and TOUGH-2 (Albright *et al.* 2002), and Vadose/W did not effectively predict drainage (Albright *et al.* 2002, Benson *et al.* 2004). A few models such as LEACHM, and Model for Effluent Disposal using Land Irrigation (MEDLI) (Tillman and Surapaneni 2002) and WATLOAD have not been used in landfill studies to date. Amongst all the above-mentioned models, it appears that UNSAT-H and HYDRUS predicted drainage effectively (Hauser and Gimon 2001Albright *et al.* 2002, Scanlon *et al.* 2002, Benson et al. 2004). A comparison of various models in terms of their parameters and use is shown in Table 8.1.

Table 8.1: Comparison of different models used in predicting site water balance

Model Acronym	Name	Application	Plant Growth	Transpiration	Solute Transport	Water Retention Method	Reference
EPIC	Erosion-Productivity Impact Calculator	Agriculture	Yes	Yes	No	Water Balance	Williams (2005)
HELP	The Hydrologic Evaluation of Landfill Performance	Landfills	No	No	Yes	Water Balance	Scanlon (2002)
TOUGH-2	Transport Of Unsaturated Groundwater and Heat	Nuclear Waste	No	No	Yes	Richard's Equation	Albright *et al* (2002)
MACRO	MACRO	Soil Water Balance	Yes	Yes	Yes	Richard's Equation	Johnson *et al* (2001)
UNSAT-H	UNSAT-H	Landfills	Yes	Yes	No	Richard's Equation	Albright *et al* (2002)
HYDRUS	HYDRUS 1D/2D	Landfills	Yes	Yes	Yes	Richard's Equation	Albright *et al* (2002)
LEACHM	The Leaching Estimation And Chemistry Model	Agriculture	Yes	Yes	Yes	Richard's Equation	Albright *et al* (2002)
SWIM	Soil Water balance and Infiltration Model	Landfills	Yes	Yes	Yes	Richard's Equation	Dwyer (2003)
MEDLI	Model for Effluent Disposal using Land Irrigation	Piggeries, Sewage Treatment Plants	Yes	Yes	Yes		Tillman and Surapaneni 2002
WATLOAD	WATLOAD	Vegetation Management, Effluent Disposal	Yes	Yes	Yes		Byers *et al.* 1999
STOMP	Subsurface Transport Over Multiple Phase	Nuclear Waste	Yes	Yes	Yes		Oostrom *et al.* 2004
Vadose/W		Agriculture	Yes	Yes	No		Benson *et al* (2004)
SHAW	Simulation of Heat And Water	Landfills	Yes	Yes	No	Richard's Equation	Albright *et al* (2002)

Due to the contradictions and inaccuracies in the models used to date, the Subsurface Transport over Multiple Phase (STOMP) (Oostrom *et al.* 2004) was trialled during the present study. This model takes into account gaseous, aqueous and solid phases in one single model. However, due to the complexity of STOMP, it was subsequently decided to use HYDRUS 1D, a model that can simulate water, heat and solute movement in the saturated zone (Simunek *et al*. 2005). HYDRUS 1D has been extensively used for site water balance studies and is continuing to gain popularity amongst hydrologists and environmental engineers. HYDRUS 1D has a finite element solution to Richard's equation for one dimensional flow in variably saturated media (Simunek *et al.* 2005).

Performance of various landfill caps has been evaluated either by qualitative or quantitative methods (Albright *et al.* 2002). Qualitative methods include groundwater monitoring and leachate collection using leachate collection systems. Quantitative methods are divided into indirect quantitative techniques that involve empirical estimates, mass balance methods and unsaturated flow process methods based on Richard's equation (Albright *et al.* 2002); and direct quantitative techniques that are based on measurements using lysimeters (Albright *et al.* 2002). Lysimeters are reference instruments for estimating drainage in agriculture (Allen *et al.* 1991) and in engineered soils (Benson and Khire 1995). These are the most reliable tools to measure percolation (Gee and Hillel 1988). However, lysimeters are very expensive to construct and monitor (Albright *et al.* 2002) and hence soil moisture measurements (qualitative) and HYDRUS 1D (indirect quantitative methods) were used to estimate percolation in the current study.

8.2 Input Parameters

Soil hydraulic tests conducted by Dr Ian Philips (Griffith University) (Appendix A) and tree parameters such as transpiration and rooting depth, and climate data (rainfall and evaporation) were used to predict site water balance using HYDRUS 1D. A number of plant and soil parameters essential for predicting the site water balance were measured during the study. A detailed report of soil hydraulic parameters and the protocol adopted to determine each parameter is given in Appendix A. Canopy rainfall interception was measured for 50 rainfall events over

two years. Transpiration in various species was determined using Thermal Dissipation Probes (TDP) and dynagauges. In this simulation, an average transpiration of 1.5 mm day^{-1} was used. This average figure represented values of the ten better performing species established at the trial site. Rooting depth in both Thick and Thin phytocaps was measured after 3.5 years of establishment. The average rooting depth of 19 species (700 mm for Thick phytocap and 500 mm for Thin phytocap) was used in the simulation. Irrigation values were recorded for each plot over three years and were added to the daily rainfall values. Soil hydraulic parameters were taken from studies conducted by Phillips in 2004 and 2005 (Table 8.2), and mulch hydraulic parameters were obtained from Findeling *et al.* (2007) (Table 8.2). Precipitation and evaporation data were obtained from Bureau of Meteorology (BOM) and the weather station located at the landfill site in Rockhampton. Final simulations were completed using average values obtained for the selected ten species grown in the phytocapping system.

Table 8.2: Values of various parameters used in HYDRUS 1D simulation

Soil type	θ_r	θ_s	α	n	K_s cm/day
Mulch	0	0.53	0.015	1.185	259.2
Sandy Loam	0.13	0.41	0.0186	1.675	165.6
Andersite Clay	0.28	0.61	0.0181	1.698	2551.5
Yaamba Clay	0.23	0.59	0.03	1.612	1219.2
Sandy Loam	0.13	0.41	0.0186	1.675	165.6
Black Cracking Clay	0.28	0.61	0.0198	1.665	6451.2

Where:

θ_r is the residual soil water content;

θ_s is the saturated soil water content;

α is the parameter in the van Genuchten soil water retention function;

n is the parameter in the van Genuchten soil water retention function, and

K_s is the saturated hydraulic conductivity.

Before running the model, canopy interception (32%) was deducted from the actual rainfall data to derive effective rainfall. Irrigation values were added to rainfall data, and rate of soil evaporation was taken as 50% of that of the non-vegetated site (worst case scenario) of total evaporation values, as evaporation under agro-forestry

systems were found to be lower (23% to 40%) than evaporation from an open area (Wallace *et al.* 2000, Jackson and Wallace 2000, Albright *et al.* 2002). Merta *et al.* (2006) found that the soil evaporation from agricultural crops was considerably lower under a high leaf area index (LAI). For example, the soil evaporation was 50% at an LAI of 1.5, in comparison with 5% for denser crops with an LAI >3.0. Based on these data, soil evaporation was taken as 50% of that reported by the BOM. Various steps and parameters included in the model are given in Table 8.3.

Table 8.3: Important steps and parameters used in HYDRUS 1D model

Step	Selection/Parameter(s)
Main Process	Water Flow, Root Water Uptake
Soil Hydraulic Model	van Genuchten – Mualem
Water Flow Parameters	See Table 2
Water Flow Boundary Conditions	Upper Boundary: Atmospheric BC with surface run-off
	Lower Boundary: Free Drainage
Root Water Uptake Model	Water uptake reduction model: Feddes
	Solute Stress Model: No solute stress
Root Depth	Measured values
Time Variable Boundary Conditions	Time
	Rainfall (measured) + irrigation - canopy interception
	Soil evaporation: 50% of the measured value
	Transpiration

Since established species grew at different rates with some growing faster than the others, data of only the ten better performing species, for canopy rainfall interception and transpiration (Table 8.4) were used in the simulation. These selected species can be grown in the greater Rockhampton Region as they perform well in a landfill environment, reduce incident rainfall via canopy interception and take up appreciable amounts of water. In addition, these are also resilient to drought and fire; both of which are very common in landfills.

Apart from predicting site water balance, several infiltration tests, soil compaction test and soil moisture test were conducted in 2007 (after 3.5 years of phytocapping). The results of these are provided in Appendix A.

8.3 Modelling Scenarios

HYDRUS 1D predicted runoff, soil storage and percolation of water for the site. Initially, the site water balance was simulated for 6 years from 2000 to 2006, and then same parameters were extended for 15 years from 1992 to 2006. Site water balance was predicted for both Thin (700 mm) and Thick (1400 mm) phytocaps and with and without vegetation for both soil caps.

Results from simulations for the 15 years are presented in this section. Apart from modelling the percolation rates using the actual measured values of precipitation, evaporation and transpiration, an additional eight scenarios with different combinations of hypothetical transpiration and canopy rainfall interception rates (based around the measured values) were created for Thick and Thin phytocaps, which included:

Scenario 1: transpiration 0.5 mm d^{-1} and canopy interception 32%;
Scenario 2: transpiration 1.0 mm d^{-1} and canopy interception 32%;
Scenario 3: transpiration 1.5 mm d^{-1} and canopy interception 32%;
Scenario 4: transpiration 0.5 mm d^{-1} and canopy interception 20%;
Scenario 5: transpiration 1.0 mm d^{-1} and canopy interception 20%;
Scenario 6: transpiration 1.5 mm d^{-1} and canopy interception 20%;
Scenario 7: transpiration 0.5 mm d^{-1} and canopy interception 10%;
Scenario 8: transpiration 1.0 mm d^{-1} and canopy interception 10%;
Scenario 9: transpiration 1.5 mm d^{-1} and canopy interception 10%;

8.4 Results and Discussion

8.4.1 Modelling Using Measured Values

The water balance simulation without vegetation estimated a cumulative percolation of around 2000 mm over 15 years (133.3 mm yr^{-1}; 17% of the annual rainfall received at Rockhampton) for the Thick phytocap compared to 2300 mm (153 mm yr^{-1}; 19.6% of the annual rainfall received at Rockhampton) for the Thin phytocap (Fig. 8.1). The difference reflects the role played by the soil depth in retaining water. The Thick phytocap could hold up to 660 mm moisture in comparison with 350 mm by the thin phytocap (Fig. 8.1). Surface runoff predicted for this site was negligible

in both phytocaps (ranging from 0.20 mm to 4 mm) in 15 years (Fig. 8.1) due to flat surface of the experimental plots, and, most importantly, to the presence of 100 mm of mulch. Lack of runoff also tested efficiency of phytocaps in a worst case scenario, wherein all the precipitation was considered in the calculation. Thus, performance of the phytocaps will be much more effective if the runoff component is added as this proportion of the rain does not go through the soil layer.

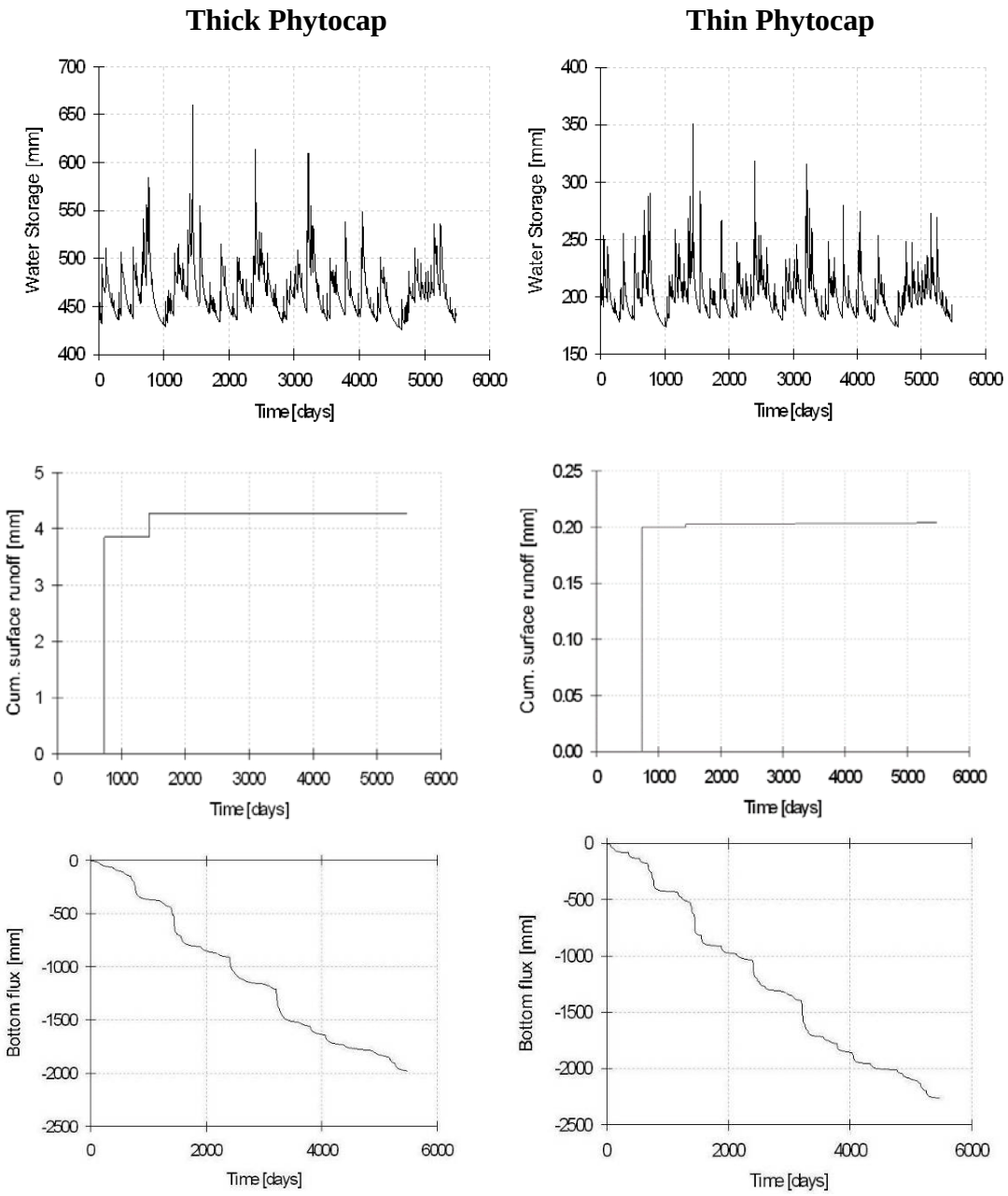


Figure 8.1: Simulated storage capacity of soil (top), cumulative runoff (middle) and percolation of water (bottom) in Thick and Thin phytocaps, respectively in the absence of vegetation
(Cumulative of 15 years data from 1992 to 2006)

In the second simulation, site water balance was simulated for the same phytocap as that used in the first simulation, but additional component "vegetation" was introduced. In this simulation, the average transpiration of 1.5 mm day^{-1} was used. The average rain intercepted by the ten selected species was 32%. Therefore, the incident rain was reduced by 32% for each event, and the corrected rainfall was used in the simulation. The water added via irrigation was also added to Rainfall.

The HYDRUS 1D simulations for the vegetated site showed a percolation of 251 mm (16.7 mm yr^{-1}; 2.14% of the annual rainfall received at Rockhampton) for the Thick phytocap and 358 mm (23.8 mm yr^{-1}; 3% of the annual rainfall received at Rockhampton) for the Thin phytocap over 15 years (Fig. 8.2). The 15 year rainfall data (with a total rainfall of 9006 mm) also included very dry and very wet period (300 mm rain in three consecutive days in 2003). The percolation for vegetated phytocaps was 6 to 8 times less than that simulated for non-vegetated sites. It is also notable that percolation on Thin phytocap occurred more frequently than in the Thick phytocap (Fig. 8.2) and this could make a major difference to leachate generation and methane gas emission. These data clearly demonstrates the role played by vegetation in phytocapping. Benson *et al.* (2004) demonstrated significance of vegetation in the site water balance, particularly the role in soil moisture depletion and the relationships between the root depth and the soil moisture depletion.

The maximum soil storage capacity of the phytocaps reduced from 350 mm to 320 mm in the Thin phytocap and from 660 mm to 570 mm in the Thick phytocap in the presence of vegetation (Fig. 8.2). This reduction is soil storage capacity may be associated with change in soil structure due to root penetration, change in bulk density and change in pore size which in turn affect water retention properties as does the the spatial variability of the soil. Roots of fast growing trees can penetrate tough soil layers thereby creating macropores (Auge *et al.* 2001 and Glinski and Lipeic 1990) thus allowing for free water movement.

Surface runoff drastically decreased in the Thick phytocap but increased slightly in the Thin phytocap (Figs 8.1 and 8.2) in comparison with the same parameters in Scenario 1. Decrease in surface runoff may be due to increased water uptake by trees thus creating more space for water storage. The slight increase in surface runoff in the Thin phytocap may be associated with soil saturation and periods where rainfall rates exceeded infiltration rates.

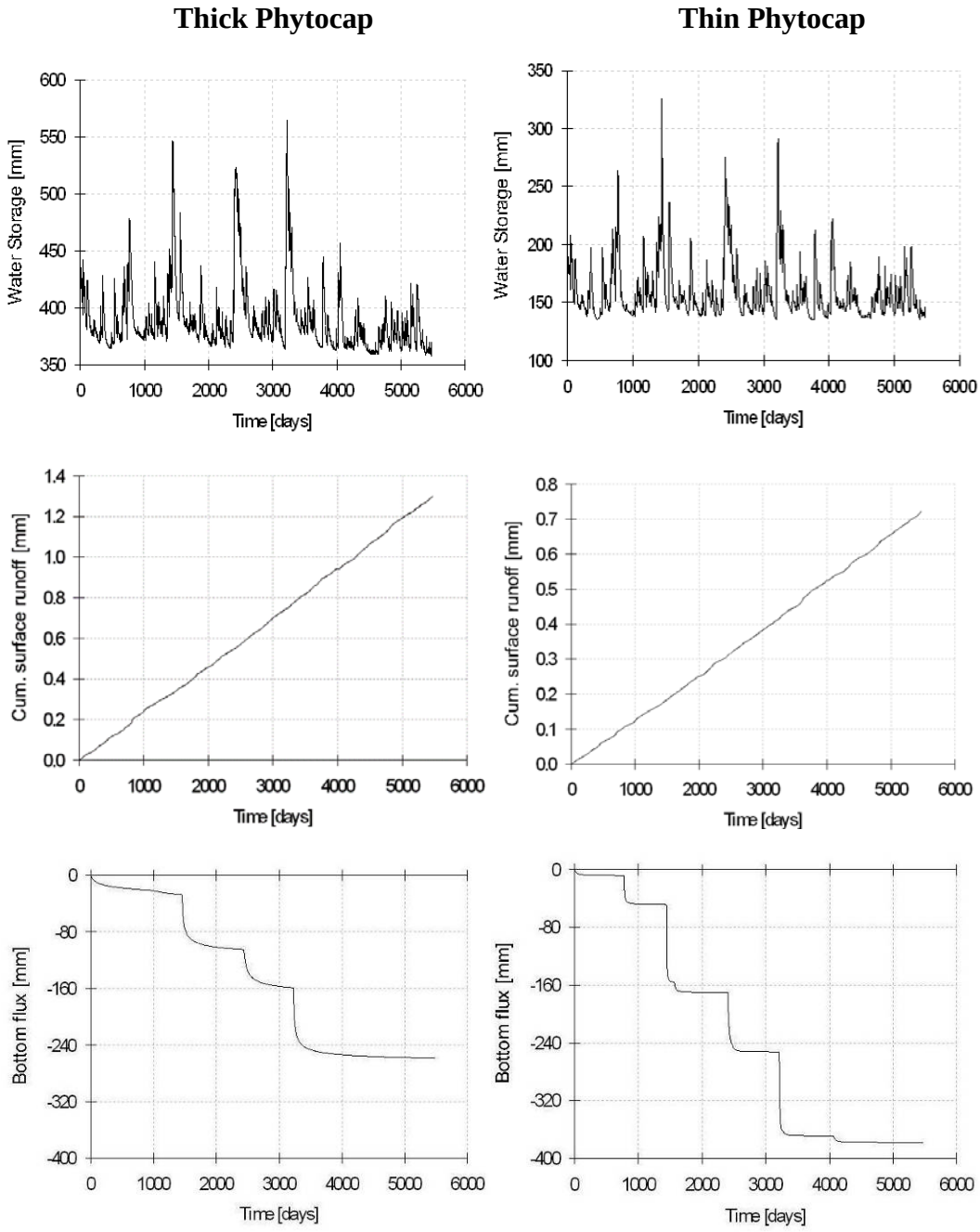


Figure 8.2: Simulated storage capacity, runoff and percolation in the Thick and Thin phytocaps in the presence of vegetation
(15 years of data from 1992 to 2006)

Results from simulations suggest that phytocaps are very effective in reducing percolation of water into the waste. In these simulations, establishment of ten selected species using 1400 mm layer of unconsolidated soil would allow for a percolation of

251 mm over 15 years. This is equivalent to 16.7 mm yr^{-1}. This value is significantly lower than the percolation rate expected for a clay cover (c. 10% of the incident rainfall) (Mr Geoff Thompson, pers. comm.2007). Results also show that reduced percolation was due to the presence of deep rooted trees (Chapter 3) (comparison of Figs 8.1 and 8.2).

A comparison of results from this study with those reported for ACAP suggests that the percolation estimated for the phytocaps in this study are within the values reported for alternative landfill caps in the US (12 to 128 mm yr^{-1}) (Benson *et al.* 2002). For example, the percolation rate at the ACAP study site at Omaha, Nebraska, is comparable to that in Rockhampton in terms of rainfall (760 mm yr^{-1}), and the measured percolation (using lysimeters) at this site was 60 mm yr^{-1}. The simulated percolation rate at Rockhampton ranged between 16.7 mm yr^{-1} to 23.8 mm yr^{-1} for Thick and Thin phytocaps, respectively. This rate, therefore, is much lower than percolation rates reported for the Omaha site. Less than 3% of the incident rain percolated during the Australian Alternative Cover Assessment Program (A-ACAP) phytocapping trial at Townsville, Queensland, Australia (Table 8.4). A comparison of the features of the A-ACAP trial sites and the Rockhampton site is shown in Table 8.4.

Table 8.4: Comparison of Rockhampton site with other sites in terms of simulated or measured percolation

Parameters	[1]Rockhampton, Qld, Australia	[2]Omaha, NE, USA	[2]Townsville, Qld, Australia	[2]Lismore, NSW, Australia	[2]Adelaide, SA, Australia	[2]Melbourne, Vic, Australia
Rainfall	780 mm yr^{-1}	760 mm yr^{-1}	2212 mm yr^{-1}	1710 mm yr^{-1}	468 mm yr^{-1}	843 mm yr^{-1}
Soil thickness	1400/700 mm	1100 mm	1500 mm	1300 mm	1500 mm	1700 mm
Vegetation type	Trees	Grasses	Trees/Grasses	Trees/Grasses	Grasses	Trees/Grasses
Drainage	16.7/23.8 mm yr^{-1}	60 mm yr^{-1}	47 mm yr^{-1}	46.96 mm yr^{-1}	7.65 mm yr^{-1}	14.97 mm yr^{-1}

[1]simulated based on vegetation, soil and climate date [2]measured using lysimeters

8.4.2 Modelling Using Pseudo Transpiration and Canopy Interception Values

During this simulation, eight scenarios with different combinations of transpiration and canopy rainfall interception were created for Thick and Thin phytocaps. Table 8.6

shows that both transpiration and canopy rainfall interception are vital for sustainability and performance of a phytocap. Among the eight scenarios, scenario 3 in the Thin phytocap, scenario 6 in both the Thick and Thin phytocaps and scenario 7 in the Thin phytocap showed slightly higher percolation than the 10% allowable limit (Mr Geoff Thompson 2007 Pers. Comm.). However, this could be overcome by selecting species with good transpiration and canopy rainfall interception attributes. From Table 8.5 it is evident that a soil depth ranging from 700 mm to 1400 mm is sufficient to restrict percolation of water into waste in the Rockhampton Region.

Table 8.5: Percolation rates for eight different scenarios for Thick and Thin phytocaps in Rockhampton 1992-2006

Scenario	Transpiration (mm d^{-1})	Canopy interception (%)	Percolation (mm yr^{-1}) Thick phytocap	Percolation (mm yr^{-1}) Thin phytocap
1	0.5	32	38.33 (4.91%)	54.44 (6.9%)
2	1.0	32	24 (3%)	35.33 (4.5%)
3	0.5	20	71.33 (9.1%)	96.66 (12.3%)
4	1.0	20	44.66 (5.7%)	60 (7.7%)
5	1.5	20	32.66 (4.1%)	46.66 (6%)
6	0.5	10	96 (12.3%)	141.66 (18%)
7	1.0	10	72 (9.2%)	88.66 (11.3%)
8	1.5	10	50.66 (6.5%)	70 (8.9%)

8.5 Conclusions

HYDRUS 1D predicted percolation rates of 16.7 to 23.8 mm yr^{-1} in Thick and Thin phytocaps respectively. Additionally, percolation rates in most scenarios were less than 10% of the total rain received in Rockhampton (Table 8.5). Percolation rates in the two phytocaps were significantly lower than that in the non-vegetated site (Figs. 8.1 and 8.2). These results demonstrate the role played by the vegetation in maintaining site hydrological balance. Selection of appropriate species based on their ability to transpire well during high rainfall events and minimise their water uptake during dry periods is extremely important for the sustainability of the phytocapping system. Species with good canopy interception must be considered to effectively reduce the rain water reaching the ground surface and thereby reducing percolation of water into the landfill.

Predicted percolation rate for the Rockhampton site is much lower than that expected from well-constructed and maintained clay-capped landfill (which was equivalent to 78 mm in Rockhampton; at 10% of incident rain). This shows equivalent or better capacity of the phytocapping system to limit entry of water into the landfill. Lower cost of establishing phytocaps on landfills (c. 50% of that of clay cap) underpins the superiority of phytocaps over clay caps.

Thick and Thin phytocaps performed equally well in maintaining a low percolation rate of less than 10% of the received rainfall. There were a few scenarios where percolation rates were slightly higher than the allowable limit, but this problem could be overcome by selecting species with good transpiration and interception features. Results from this simulation are comparable with those from ACAP and A-ACAP studies which used lysimeters. Trends in percolation at Rockhampton are consistent with those observed at Townsville and other A-ACAP sites. This attests accuracy of results obtained from HYDRUS 1D simulation. The current simulation also determined that soil depths ranging from 700 mm to 1400 mm were sufficient to restrict percolation of water into waste in Rockhampton. The current research therefore supports the recommendation for use of the phytocapping technique for landfill remediation in many parts of Australia, especially in the drier regions where PET exceeds precipitation.

This Chapter demonstrated that Phytocapping system is as effective as the clay capping system in maintaining site water balance. The next Chapter gives an overview of the economics of landfill remediation using clay caps and phytocaps, and is based on various reports produced by various Local Governments and environmental consultants across Australia.

9

Economic Analysis of Phytocapping

9.1 Introduction

Technology and cost of landfill capping in Australia have undergone significant changes in the last 20 years. With stringent State and Federal Government regulations, it is becoming difficult for Local Government "Councils" in Australia, particularly small and regional councils, to manage and remediate their landfills. The capital, construction, operation and maintenance costs of clay caps (the manadated technique of remediating landfills) are very high due to limited availability of clay materials locally. Cost of clay is determined by the distance it has to travel to reach landfills . Furthermore, studies in the past have found that many medium and small sized councils have not budgeted for landfill capping, and in most cases landfill gate fees does not include landfill postclosure expenses. Despite conducting studies on waste management for the past 30 years, from both technical and economical points of view (Clarke *et al.* 1999, McDougall *et al.* 2001), hardly any work has been undertaken on waste management issues that are challenging Local Governments (Qian and Burritt 2004).

Landfilling is by far the most economical and common method of Municipal Solid Waste (MSW) disposal in Australia (Xu *et al.* 1997, Scott *et al.* 2005), especially in regional areas due to low population densities and high costs of transporting the waste to suitably constructed landfills. When landfill cost is calculated, costs associated with its ongoing monitoring of groundwater, air pollution, methane gas emission and landfill capping are usually ignored (Xu *et al.* 1997) due to lack of awareness of future liabilities and/or due to the difficulties associated with quantification. (Stanley 1992). Hence landfill costing is underestimated in the landfill pricing and in setting up of gate fees.

Recently, a nine-hole golf course was opened at the site of the old Lucas Heights landfill in Sydney; part of a $83 million sporting complex funded by the New South Wales (NSW) Government (Lamb 2009).

A large number of landfills have been constructed in the past 30 years, in Australia due to availability of abundant land and large distances between towns and cities. Currently, more than 600 landfills (licensed and unlicensed) are in operation (Bateman 2005). In many parts of Australia, local councils are responsible for the operation of waste management facilities such as landfills, Waste Transfer Stations (WTS) and Material Recovery Facilities (MRF). With increasing environmental awareness, and introduction of waste levies and stringent regulations, cost of landfill operation, management and maintenance is getting higher. This has placed economic pressure on many regional councils as they have not accounted for post closure management and monitoring costs in their budgets. This is primarily due to a lack of waste data and inadequate record keeping and reporting (Gauthier 1998), as many of the landfills in regional Australia lack good infrastructure such as weighbridges, compactors, signage, fences and gates.

Future costs of waste management is uncertain, and some costs are more predictable than others, such as landfill capping and maintenance (US EPA 1998). Cost of landfill closure and after-care may be defined based on the life span of the landfill estimated by that council (Qian and Burritt 2004), and good post-closure management depends on the cost allocation for closure of landfills during the active phase of the landfill (Gauthier 1998). Waste management regulation in Australia has changed over the years and this constant change in capping design and materials used can pose economical risks to councils and landfill owners (Qian and Burritt 2004). However, phytocaps on the other hand reduce the cost of landfill remediation by 35% to 72% based on the site climatic condition and availability of local soil (Hauser *et al.* 2001)

Of all states in Australia, Queensland has not had a waste levy until late last year, which in turn has placed waste management low in council budgets to-date. However,

awareness of waste issues is slowly growing among Queensland communities. This has encouraged many councils to introduce recycling, and divert waste going away from landfills. Councils are also actively engaged in reviewing their waste facilities and rationalising their landfills to operate these in a more structured, systematic and environmentally-sound manner.

Central Queensland has over 30 landfills spread across five councils, with a total area of 175,977 m^2 and a population of 210,968 (1% of the total Australian population and 5% of the total Queensland population). Three councils are managing more than five landfills each. Rationalisation of landfills will involve closing and capping of many of the landfills in the Region. This will incur a large expense and post-closure care for up to 30 years. Trans-Pacific Industry (TPI) has approximately $100 million set aside for landfill remediation projects (Lamb 2009) and indicated that $20 to $25 million dollars would be spent over the next two years on the closure of its Tullamarine Landfill in Melbourne; with an expense of $400,000 to $500,000 annually in ongoing costs for the next 30 years (Lamb 2009). This takes the total cost of remediating the landfill site to approximately $35 to $40 million (Lamb 2009).

Typically, clay is used to cap old and completed landfills due to its low hydraulic conductivity (EPA 2005). Main advantages of clay are lower costs (where locally available) as compared to HDPE, low permeability, availability (where locally available), robustness and chemical compatability (Arch 1998). However, where clay is not locally available, its use can be uneconomical, particularly in regions such as Central Queensland where landfills are small, distances are long and availability of clay is limited.

Under new Federal regulations governing landfill closure, landfills must be monitored and inspected and integrity of the clay cover should be maintained for at least 30 years following closure (EPA 2005). This includes operation of the leachate collection system, extensive groundwater monitoring, inspection and repair as needed of the cap

and other protective systems, and maintenance of the financial assurance bond or other security.

A typical clay cap for a Maryland sanitary landfill in the US consists of 300 mm to 500 mm of compacted clay cost approximately US $371,387 per hectare (Maryland Department of Environment 2010). Actual costs depended largely on local availability of materials used to construct the cap, the topography and ease of installation at a particular site, the design selected, and the cost reductions associated with bulk-buying.

The cost of landfill capping depends significantly on its physical characteristics and its licence conditions (BDA 2009) which comprise mainly the type of waste accepted (e.g. inert and/or putrescible), size of the landfill (small, medium or large; refer table 1) and the climatic condition (wet temperate, dry temperate and moist and wet tropical). Phytocapping as an alternative landfill capping technique was trialled at Lakes Creek Landfill, Rockhampton. A three and a half year study demonstrated the effectiveness of phytocaps to reduce water percolation through the waste.

9.2 Methodology and Approach

Cost analysis of phytocapping versus clay capping has been conducted using a collative and desktop research approach. In this approach, real-life case studies and documented facts and figures have been collated and reported. Some figures have been acquired through direct communication with Brisbane City Council (BCC) and Rockhampton Regional Council (RRC).

9.3 Cost Analysis

In 2005, the Waste Management Association of Australia (WMAA) estimated the private cost of a large best-practice landfill (lined and engineered) in an Australian capital city at around $25 per tonne (BDA 2009). A report submitted by the BDA Group (2009) to the Department of Environment, Water, Heritage and the Arts, Canberra, and

the Wright Corporate Strategy (WCS) estimated the full costs of large landfills for the Australian Capital Territory (NOWaste) at around $50 per tonne (including capping costs). It also reported an estimated cost of $25 to $45 per tonne, $40 to 150 per tonne and $40 per tonne for the City of Mount Gambier (Caroline landfill), Great Lakes City Council and Hastings Council (Cairncross landfill), respectively. This estimated cost included site establishment, cell construction, operation, cell closure and post-operations.

In Australia, landfills are categorised into small, medium and large based on the acceptance of waste quantity. Table 9.1 gives three categories of landfills based on quantity of waste received. This in turn helps determine the cost of landfilling (including capping). Table 9.2 gives a comparison of costs reported by the WMAA and the WCS for landfills taking 200,000 tonnes of waste per year.

Table 9.1: Different Categories of landfills in Australia

Landfill	Category (t/yr)
Small	<10,000
Medium	10,000 – 100,000
Large	>100,000

Source: BDA (2009)

Table 9.2: Estimated costs of large best practice landfills in Australia

Type of cost	Cost per tonne of waste (AUD)	
	WMAA	WCS
Land purchase including airspace	2	2
Approvals / site development	2	6
Cell development	6.5	10
Operations	10	18
Capping and rehabilitation	2.5	5
Aftercare	2	8
Total	25	49

Source: BDA (2009)

It should be noted that WMAA estimates are averages for large best practice landfills and these do not include management costs. The WCS estimates were developed in the Australian Capital Territory context and include management costs. Based on the above figures, an indicative cost of capping and after care cost can be derived for different landfills. Landfill capping is undertaken primarily to address pollution or occurrences that may lead to pollution. Clay caps are predominantly being used throughout Australia. It is evident from Table 9.3 that cost of landfilling (including capping) is higher for small and medium sized landfills and hence increasing the financial burden borne by small and medium sized councils. This is a typical scenario in Central Queensland.

Table 9.3: Estimates of cost of landfilling in Australia ($ per tonne)

Type of cost	Small	Medium	Large
Land	5	3	2
Approvals / site development	10	6	4
Best practice liner	13	8	5
Leachate collection	6	4	3
Gas recovery	6	4	3
Amenity management	1	1	1
Operations	34	20	14
Capping & remediation	10	6	6
Post-closure maintenance	15	9	6
Total	100	61	44

Source: BDA (2009)

Table 9.4 gives a cost comparison of two landfills that were clay capped in 2008 and 2009. The Table also shows the price projection for clay caps obtained through an assessment conducted by Phytolink Australia Pty Ltd (Report A) in 2003 for a landfill site at Buderim. Based on figures acquired from two different councils in Queensland, it is clear that phytocapping technique will cost c. 50% less than that of clay capping system (note that clay capping depth depends on the landfill licence conditions).

Table 9.4: Summary of cost of clay capping in Queensland

Council	Year capped	Depth of compacted clay (mm)	Cost per ha (AUD)	Capping profile
BCC	2008	900	233,300	900 mm Clay, 200 subsoil, 150 mm topsoil with turf
RRC	2009	600	295,000 - 350,000	600 mm Clay, 200 subsoil, 150 mm topsoil with turf
Report A	2003	900	197,500	900 mm Clay, 100 subsoil, 100 mm topsoil with grass
Report A	2003	600	148,000	600 mm Clay, 100 subsoil, 100 mm topsoil with grass

BCC: Brisbane City Council, RRC: Rockhampton Regional Council

Table 9.4 shows the cost of clay capping per hectare in Queensland. The large variation in the cost of is chiefly due to the time differences, topography of the landfill, availability of material onsite and capital works cost associated with it at that point of time. In Brisbane, it cost $233,300 per hectare to cap one cell at one of the landfills in 2008 (Table 9.4). The cost does not include the purchase of clay since it was readily available onsite, and this had saved the Council at least $5 to $7 per hectare in capping costs. Conversely, Rockhampton Regional Council has been spending $295,000 to $350,000 on landfill capping (see Report B). Costs given in rows Table 9.5 are an outcome of a detailed study conducted by Phytolink Pty Ltd in 2003 and Maunsel (Report B) in 2005 for Rockhampton Regional Queensland.

Table 9.5: Summary of cost of phytocapping in Queensland

Council	Year capped	Depth of soil/clay cover (mm)	Cost per ha (AUD)	Capping profile
BCC	2003	1500	138,900	1500 mm soil + vegetation
Report A	2003	1000	128,625	1000 mm (soil + topsoil)+ vegetation
Report B	2005	600	30,946	600 mm (soil + topsoil)+ vegetation
Report B	2005	1000	145,431	1000 mm (soil + topsoil)+ vegetation
Report B	2005	1300	366,927	300mm (clay) + 1000 mm (soil + topsoil)+ vegetation

Table 9.5 provides cost of phytocaps per hectare in Queensland. Cost may vary between states and amongst regions due to varied climatic conditions. Based on an

enxperimental trial conducted by BCC would have spent only $138,900 per hectare on its capping and landfill remediation using phytocapping.

By comparing Tables 9.4 and 9.5, it can be shown that phytocaps are up to 50% less expensive than clay caps in Queensland. Similar conclusions can be made with regard to remediating landfill in other parts of Australia. Hauser *et al.* (2001) condusted a cost analysis of phytocaps versus clay caps for Southern Great Plains in the US and reported that phytocaps/ET caps cost 35% to 72% less than a clay cap. A detailed cost analysis conducted by Hauser *et al.* (2001) is given in Table 9.6. Typically, a phytocap costs 50% less than clay caps (Hauser *et al.* 2001), but most of cost analysis conducted till date only evaluates cost of construction (Pers. Comm. Victor Hauser 2012). Since phytocaps are less costly to maintain than conventional covers, the overall cost savings through phytocaps may be more important than the construction cost itself. If the long term maintenance costs are factored in the analysis then phytocaps would have a greater advantage.

Table 9.6: Estimated costs of major components required for construction of conventional and ET caps in Southern Great Plains

Item	Conventional (US$ per hectare)	ET Cover (US$ per hectare)
Soil cover placement	43,400	130,100
Water drainage layer	98,400	-
Geomembrane barrier layer	118,400	-
Compacted clay barrier layer	92,700	-
Gas collection layer	98,400	98,400
Common fill, foundation	43,400	43,400
Grass establishment	2,600	2,600
Total	-	-
One barrier (clay)	378,900	-
Two barriers	497,300	-
ET cover with gas collection	-	274,500
ET cover with no gas collection	-	176,100

Source: Hauser *et al.* (2001)

Apart from financial benefits (low cost), phytocaps also have ongoing environmental and social benefits (Table 9.7).

Table 9.7: Ongoing benefits and limitations of Phytocapping

	Factors	Clay capping	Phytocapping	Comments
1	**Economic Benefits**			
	Cost of capping (average)	$250,000/hectare	$160,000/hectare	Clay cap will be a cheaper option where clay is locally available
2	Permits	Required	Regulatory barrier	
3	Design	Required	Required	Expensive to design
4	Administration cost	Required	Required	
5	Top soil	Required	Required	Soil characterisation is required
6	Clay	Required	Not Required	
7	Irrigation	Not Required	Required	Irrigation may sometimes be expensive
8	Transportation	Required	Not Required	
9	Recyclable material	Not Required	Required	Use of tyres, concrete etc.
10	Supervision	Essential	Required	
11	Tree selection	Not Required	Required	Vegetation characterisation is required
12	Establishment	Not Required	Required	Clay caps have a set standard and requirement, phytocaps don't.
13	Maintenance (first 3 yrs)	Required	Essential	
14	Compaction/testing	Required	Not Required	May require hydraulic conductivity testing
15	Geotextile	Required	Not Required	
16	Tax/Insurance	Required	Required	
17	**Environmental benefits**	Limited	Many	
	Bushfire	Not prone	Prone	
	Ecological	No	Yes	
	Biodiversity conservation	No	Yes	
	Native species promotion	No	Yes	
	CO_2 sequestration	Limited	Yes	

	Methane reduction	No	Yes
	Leachate reduction	Yes	Yes
	Groundwater contamination	No	No
18	**Social benefits**		
	Parklands	Yes	Yes
	Sports field	Yes	No
19	Natural ecosystem	No	Yes
20	Aesthecity	Yes	Yes
21	Home for birds and animals	No	Yes
22	Odour control	Yes	Yes
23	Commercial value (Timber production)	No	Yes
24	Breakdown	Reconstruction	Replanting

9.4 Conclusions

From the data shown above, it is evident that phytocapping is less expensive than clay caps, particularly to Local Governments that do not have supply of clay onsite. Ongoing maintenance of clay caps is also very laborious and expensive. On the other hand, phytocaps are less expensive and are easy to maintain and repair. Phytocaps also offer other benefits such as biodiversity conservation, carbon sequestration, providing amenity park values and picnic sites. These sites can also be used for commercial purposes such as cut flower production or hardwood timber production or biofuel production (biodiesel or bioethanol feedstocks). Overall, phytocapping is the ideal option for most landfills in Australia, predominantly in the medium to low rainfall regions.

Typically, a phytocap costs 50% less, but most cost analyses conducted till date only evaluate cost of construction and do not include cost of maintaining phytocaps. Phytocaps cost much lesser to maintain than conventional covers, and hence long-term maintenance cost of phytocaps may be more important than the construction cost itself. If long term maintenance costs are factored in the analysis, phytocaps would have a better advantage that clay cap in most situations.

10

Summary and Conclusions

10.1 Summary

The waste sector in Australia has advanced over the last decade due to developments in technology. This has introduced alternative waste technologies (AWT) and energy from waste (EfW) technologies. In addition, greater emphasis has been placed over recycling, waste diversion and landfill remediation strategies. Despite problems associated with landfills, they are still being built due to their lower capital cost and being the easiest means of disposing waste globally and in Australia (Scott *et al.* 2005, Izzo *et al.* 2009, Tonini *et al.* 2009). Landfilling is particularly common in regional areas due to low population density and high cost of transporting wastes to cities where other methods of disposing or recycling waste have been well established. Due to increased generation of wastes, local councils who own and run most of the landfills have concerns over environmental problems arising from these landfills. Landfills built prior to 1993 in Australia do not have liners (Friends of the Earth 2000) and have become a major threat to the environment (CSIRO 2001). At present, more than 600 landfills are in operation in Australia (Bateman 2005, Johnston 2009) and most of the small and medium sized landfills in regional Australia still operate with bare minimum infrastructure, despite making landfill capping mandatory for all landfills; big or small.

Local governments are responsible for municipal solid waste management; including collection, storage, treatment and disposal. In Australia, it is mandatory to obtain a license to operate and maintain landfills. These licenses specify types of waste to be accepted, means of managing environmental pollution (dust, air, odour and water), landfill design criteria and post closure management and monitoring. Every Local Government organisation that owns and operates a landfill is required to abide by the

license condition to reduce environmental impacts. However, ongoing maintenance and operating costs of landfills are significantly higher for small and medium sized councils (population <200,000) than for large councils. Some regional councils in Australia operate more than 10 landfills/waste tranfer stations (WTS) and maintainance of these landfills is becoming difficult and expensive.

Operational and closure costs of landfills have escalated due to increasing environmental awareness and stringent regulations of the Environmental Protection Agency (EPA). The most popular practice in Australia is the use of a compacted clay cap. (EPA 2005). Costs of construction, operation and maintenance of clay caps, as specified by State Governments, are very high depending on size of landfill and local availability of clay material. Studies in the past demonstrate that many medium and small councils have not budgeted for landfill capping. In most cases waste disposal fees and charges do not include landfill capping expenses. Furthermore, mandatory monitoring of landfill's environmental performance, and managing integrity of clay caps for up to 30 years would create another difficulty for many councils.

Hence phytocapping, a relatively new and alternative landfill capping technology was trialed in Rockhampton, Australia, using 21 tree species. This technology will provide a sustainable alternative to clay capping, especially for small and medium-sized landfills that are commonly found in regional areas of Queensland. Phytocaps have two major components: plants that act as 'bio-pumps' and 'rain interceptors', and soil that acts as 'storage'. Soil is an important component of a phytocapping system. It plays a vital role in supporting tree growth and simultaneously retaining water during rainfall events (Hauser *et al.* 2001), Medalinski *et al.* 2003). Thus soil and plants together minimise percolation of water into the buried waste.

Phytocapping was trailed in 1997 by the US EPA in various agro-climatic conditions using native grasses, shrubs and limited number of tree species (*Salix* sp. and *Populus* sp.). Since then phytocappnig has been used in various forms as capping system in many parts of USA. However, this technology is still in its infancy in Australia. For this

reason, a field trial was conducted at Rockhampton to test effectiveness of phytocapping in minimising percolation of water into the waste. This study is the first of its kind in Australia and had focused on the role of trees and soil depth in reducing percolation of water into the buried waste. The study also evaluated additional benefits of phytocapping, such as reduction in methane emission into the atmosphere. Results of this trial have encouraged authorities to carry out further studies (Yuen *et al.* 2011) on landfill site water balance and amend the legislation (WMAA 2011), a significant contribution to the waste industry.

10.2 Research Outcomes

10.2.1 Trees and Soil

Nineteen of 21 species grown in the trial were evaluated for growth, transpiration and canopy rainfall interception and methane oxidation.. For phytocaps to perform well, established species should survive, grow and contribute to site water balance via transpiration, canopy interception..

Locally available soils were selected for this study as they can support native vegetation due to presence of compatible *Rhizobia* and microrhizal fungi (Hauser *et al*. 2001). These soils were assessed for hydrological properties (Appendix A) which remained unchanged over the study period. However, porosity and bulk density of the soil changed due to root growth, but this did not have marked effect on site water balance. Compaction rates of soils after 3.5 years of tree growth varied with soil depth, ranging from 120 kpa to 200 kpa (Appendix A).

10.2.2 Tree Survival and Growth

The Rockhampton trial involved 21 tree species that were established and tested for their performance on a landfill for the first time in Australia. Of the 21 species grown, 19 survived and grew well. *Populus* sp. and *Salix* sp. did not survive the high temperatures in summer and this was anticipated, given that these are temperate species (Cunningham and Read 2003). Overall, most species survived and grew well and they

closed canopies (2 m x 2 m spacing) within 3 years of establishment. As a result maximum canopy rainfall interception was realised (Sands 2004, King *et al.* 2005) making a significant contribution to site water balance (a reduction of 30% of rainfall).

Trees in this study grew well as shown by height growth, stem diameter, canopy spread and LAI. Fast growing species such as *A. mangium, H. tiliaceus*, casuarinas, bamboo and eucalypts grew more than 6 m tall in 3.5 years. Most species grew over 2 m yr^{-1} (Fig. 3.8). *Callistemon viminalis* and *M. linariifolia* were the only two species that showed slow growth rates, as they have slow growth habits. All species attained a stem diameter of more than 30 mm in 3.5 years of establishment. Established species attained a LAI of 2.4 within 3.5 years. The maximum achievable LAI is in the order of 6 to 8 in a mature forest (Beadle 1993). Fast growing species such as bamboo, hibiscus, eucalypts and casuarinas closed canopies well within the first three years, reflecting their dominance over the medium and slow growing species. The established trees also accumulated significant quantities of biomass over 3.5 years, which correlates to water uptake. Biomass produced by species grown in this system is comparable with that of other related studies and can be used for carbon sequestration (Fang *et al.* 2007). *Hibiscus tiliaceus* and *D. latiflorus* produced more than 100 t ha^{-1} of shoot biomass within 3.5 years, with most species accumulating 30 to 90 t ha^{-1} in 3.5 years. *Melaleuca linariifolia, C. viminalis* and *P. pinnata* accumulated less than 10 t ha^{-1} biomass in 3.5 years. On an average, the total biomass of the trees grown in this phytocapping system constituted 80% shoot biomass and 20% root biomass.

The root distribution study also revealed that all species except bamboo exhibited a well-developed tap root system and their highest root density was concentrated in the top 40 cm of the soil system. Within 3.5 years, roots of most species penetrated as deep as 700 mm in the Thick phytocap and 500 mm in the Thin phytocap. The shallow rooting depth and profuse lateral root growth in this study were influenced by higher soil temperature within and the methane gas emissions from buried waste. Despite this the established species showed appreciable root growth. Overall, these plant growth measuring from height, stem diameter, canopy spread, LAI, biomass production and

root distribution provide evidence that the phytocapping system supports normal growth of a wide range of tree species. These species also contribute to environmental, social and economical benefits of the site. Thus careful selection of species is critically important for the phytocapping system to function to its fullest capacity. Most importantly, the selected species should transpire at a very high rate during the rainy season and minimise their transpiration during dry seasons.

10.2.3 Canopy Rainfall Interception and Transpiration

Plants established on a phytocapping system must provide multifunctional roles to maintain optimum site water balance. Trees should act as bio-pumps as well as canopy rainfall interceptors to reduce the water entering the soil. In addition, established species should be capable of enduring local climatic conditions such as drought, salinity, cyclones, fire, waterlogging and wind.

Some species in this study transpired as much as 6.25 mm d^{-1} during rainy season and as little as 0.1 mm d^{-1} during dry season. Overall, these species transpired 1.4 mm d^{-1}. The majority of the species transpired rapidly after a rain event. This is a significant contribution to the effectiveness of phytocapping system. This attribute of the tree species will help balance site hydrology while also allowing established plants to survive and sustain growth during dry seasons. This attribute is extremely important for plants grown on phytocaps.

Another important feature of tree growth on phytocapping system is canopy rainfall interception. This study has demonstrated that the canopy rainfall interception can be reduce up to 30% of the incident rainfall making a significant contribution to the hydrological cycle of a phytocapped landfill site. Interstingly some species such as *A. mangium* trees grown on phytocaps were able to intercept up to 52.5% rainfall on a per storm basis at Rockhampton with an annual rainfall of 780 mm. With this interception by trees, only 546 mm of the total rainfall would have reached the ground surface. A reduction of 30% of effective rainfall meant approximately 30% reduction in the cost of the soil that would have been used to retain the component of the rainfall.

The native species grown in this phytocapping system differed significantly in their rainfall interception due to differences in their morphological characteristics. Canopy rainfall interception also varied with environmental conditions such as rainfall intensity and rain duration. Overall, the proportion of the rain intercepted by the tree canopy was significant enough to be included in the site water balance modelling.

10.2.4 Methane Oxidation

Tree roots also act a subtrate for methane oxidising bacteria. However, no microbial studies were undertaken in this study. Martin (1996) used landfill gas tolerant plants were as many other plants had failed to survive at sites enriched with landfill gas, due to high soil temperature (>40°C) and lack of oxygen (Waisel *et al.* 1991). It is also reported that most tolerant species avoid zones of soil containing high concentrations of gases by not sending their roots below 200 mm (Martin 1996). Overall, presence of vegetation significantly reduced (4 to 5 times) methane emissions from landfill site.

10.2.5 HYDRUS 1D Prediction

HYDRUS 1D simulations and comparison between Thick and Thin caps suggested that a minimum soil depth of 1000 mm would be required for tree growth in the greater Rockhampton region. This inference is also consistent with the reports of Martin (1996), Moffat and Houston (1991) and McGuire *et al.* (2001) who tested several species on various soil depths and concluded that use of thicker layers of soil will be beneficial, particularly during dry periods. Addition of a thicker layer of soil adds costs to phytocapping and hence efforts should be made to optimise soil thickness based on local soil availability, climatic conditions and the needs of the plant species used.

Predicted percolation rate at the study site is comparable with the percolation rates found at Omaha, Nebraska, USA and Townsville, Queensland, Australia (Table 10.1). Simulated percolation rate at Rockhampton ranged between 16.7 mm yr^{-1} to 23.8 mm yr^{-1} for thick and thin phytocaps, respectively (Table 10.1. These values are, much lower than percolation rates reported for the Omaha site. Less than <2.5% percolation rate was

reported for a phytocapped site at Townsville by the Australian Alternative Cover Assessment Program (A-ACAP). This rate was comparable with the values predicted in the current trial using HYDRUS 1D. This suggests that the predictions made based on above ground monitoring is reliable, provided all plant related parameters are accounted (Table 10.1).

Table 10.1: Comparison of simulated percolation rates at Rockhampton with measured percolation rates at Omaha, USA and Townsville, Australia

Parameters	[1]Rockhampton, Qld, Australia	[2]Omaha, NE, USA	[2]Townsville, Qld, Australia
Rainfall (mm yr^{-1})	780	760	2212
Soil thickness (mm)	1400/700	1100	1500
Vegetation type	Trees	Grasses	Trees/Grasses
Drainage (mm yr^{-1})	16.7/23.8	60	47

[1]simulated [2]measured with lysimeters

10.2.6 Benefits of Phytocaps

Besides practicality, robustness and environmental benefits, phytocaps are also an economically viable solution for landfill remediation as they are less expensive to construct than clay caps, especially in regional areas where the Councils do not have enough local supply of clay. Ongoing maintenance of clay caps can be laborious and expensive. In contrast, phytocaps can be less expensive and easier to maintain as these constitute a natural system. Phytocaps can also offer other social and environmental benefits to councils and the public, such as recreational park and biodiversity conservation. Under the Carbon Farming Initiative, carbon credits are only available to methane gas recovering system or flaring and this option is compatible with phytocapping. Landfills exceeding the threshold of 25,000 carbon equivalent will have to be clay capped to reduce their carbon liability.

10.2.7 Concerns about Phytocaps

Several concerns have been raised by the Department of Environment and Heritage Protection (DEHP) on the state of phytocapping during an event of storm, cyclone or bushfire. In the event of a bushfire, native trees will regenerate as they are genetically

adapted to cope with bushfires. This ensures integrity of phytocaps and their ability to withstand bushfires. However this may not be the case during storms/cyclones as the uprooted trees may leave the cap exposed. Under these circumstances, soil thickness will play a vital role. As a preventative measure, designers and planners should consider the region's climate, average windspeed, elevation, soil and rooting depth as indicated by Martin (1996) before deciding on landfill capping or the selection of species (Ashwath. N 2011, Pers Comm.). Pruning of trees could be another solution to reduce breakage of trees due to storms. Maintenance of diverse species can minimise the above problem and loss of one species will be compensated by another species. In a worst case scenario, the site can be reestablished with new species within a year (Ashwath 2011)

Concerns have also been raised about the dispersion of contaminants (heavy metals) from the buried waste through litter fall and/or into the food chain. It is possible that the soil and trees of a phytocapped landfill site constitute a threat to the environment, but this is a lesser threat than that of soil and trees grown on mine sites (Merry *et al.* 1986, Fernandes and Henriques 1991, Lottermoser *et al.* 1999, Grant *et al.* 2002, Kopittke *et al.* 2008, Maddocks *et al.* 2009) and industrial sites (Phillips and Chapple 1995). The current study demonstrated that all plants grown on this phytocapping system (except *G. lobocarpum*) did not contain unusual levels of heavy metals suggesting that it is unlikely that this will be a serious concern. Furthermore most plants do not transport heavy metals as they limit the entry of heavy metals into theirroot system (Dickinson and Lepp 1997).

10.2.8 Economic Viability of Phytocaps

Economic analysis of constructing a phytocap based on the limited available information showed that phytocapping system cost half as much as the clay capping system when the construction phase was considered. Phytocaps become much more attactive as the maintenance costs of phytocaps are lower than those of clay caps

10.3 Conclusions

This study demonstrates that Phytocaps are as effective as a clay cap in restricting percolation of water into the buried waste in Queensland and many parts of Australia. Phytocaps offer other benefits such as recreational parks, biodiversity conservation and improve aesthetic value of urban site. Cost of construction of a phytocap is approcimately 50% of that constructing a compacted clay cap. Phytocaps can also reduce methane emission and most importantly, phytocaps allow only a small proportion of the rain to percolate (2% to 3%), as found in this study and also reported by the A-ACAP at various locations in Australia). This limited rate of percolation in phytocaps is considered necessary to induce slow decomposition of the waste. Slow decomposition of waste will minimise the risk of buried waste causing a severe environmental hazard should it remain mummified and accidentally come in contact with water. Slow decomposition would also reduce the burden to future generations as the decomposed waste is less likely to hamr the environment compared to the waste that is in its virgin condition.

The rsults of the current study paved the way to develop further studies via A-ACAP. These results were also instrumental in gaining approval from DEHP to use phytocap as the final cap at Lakes Creek Road Landfill at Rockhampton, Australia.

10.4 Further research

Following on from this research, other research has been conducted to extend benefits of this novel method of landfill remediation to other agro-climatic regions of Australia (http://www.wmaa.com.au/aacap/aacap.html, Michael *et al.* 2007, Salt *et al.* 2007 and Wong *et al.* 2007, Salt *et al.* 2008, Yates *et al.* 2008, Sun *et al.* 2009, Yuen *et al.* 2010, Sun *et al.* 2010).

Although this research has addressed many concerns on the effectiveness of phytocaps, tree growth and methane reduction, other impending factors effects of elevated soil temperature and its impact on root growth needs further investigation. Such investigation will help optimise soil thickness in order to enhance overall performance of the phytocapping system and to prevent tree mortality caused by high root zone temperature and high landfill gas concentrations.

Resources in the form of shredded tyres, crushed glass, mulch and crushed concrete can be explored for their physical and chemical properties, and their use as a capillary barrier, which in turn will increase water retention capacity of the soil cover.

Lastly, a detailed study is warranted to predict how quickly trees transpire water after rainfall events and to quantify the ability of established species to survive when soil moisture becomes a limiting factor during dry season.

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Appendix A
Soil Properties

Introduction

Soil is an important component of a phytocapping system. Soil plays a vital role in determining sustainability of phytocaps; particularly in the initial stages of tree growth, as soil needs to hold maximum water during rainfall events (Hauser and Gimon 2001) and simultaneously provide nourishment to plants growing in phytocapping system (Madalinski *et al.* 2003).

Assessment of soil properties in-situ is complex and highly dependent on selection of the spatial borders (Hailing 1997) and indicators to evaluate their sustainability (Smyth and Dumanski 1995). In this study, the primary focus was on the movement of water through soil cover. Hence, the soil used in this study was subjected to hydraulic analysis (Dwyer 2001, Hauser and Gimon 2001) were also characterised for chemical properties. Properties such as hydraulic conductivity, infiltration rate, sorptivity, aggregate stability and soil-water retention properties were tested and analysed by Dr Ian Philips (Griffith University). The mathematical model CHEMFLO that uses Richards's equation (Nofziger and Wu 2003) was used to simulate water movement in the capping systems. The RETC code was used to derive the van Genuchten model parameters, which were then used in the CHEMFLO model to simulate water movement in the phytocaps.

Soil Hydraulic Properties

Locally available soils were selected for this phytocapping study as they could support native vegetation due to presence of native microflora (Hauser and Gimon 2001). The soil hydraulic properties were determined twice during the study: once at the beginning of the experiment and the other a year later. Both in-situ and laboratory experiments were conducted to determine hydraulic conductivity and water holding capacity of soils used. In-situ soil surface infiltration tests were conducted using disc permeameters (Fig A1) at the end of the study to evaluate effects of root development (Graham and Syvertsen 1984) and soil fauna (Lee and Foster 1991) on soil hydraulic conductivity.

Two disc permeameters in foam lined buckets were used (Fig. A1). Other equipment included: two 20 L capacity open buckets, a suction pump and plastic tube to fill the permeameter, two polystyrene foam pads (30 cm square), two 20 L water containers, bags of moist contact at 3% water contact (sand), pad rings, two 200 mm diameter x 3 mm height, a small level, a recording tape and player, recording sheets, four clipboards, pens, a soil moisture sheet, two scrapers for levelling sand pad, a large syringe with fine plastic tube to fill permeameter bubble tubes, clippers for trimming stubble, two small buckets to cover sand pads, soil moisture containers and a piece of Wettex and a small dry towel.

A large bucket was filled with water as the water supply for the permeameter. The second bucket was half-filled with water to rinse the permeameter after measurements. The permeameter was placed on the polystyrene foam pads and all bubble taps were open (Fig. A1). This is to be performed prior to commencing the experiment, to avoid errors. The side tube was then filled with water to below the 4 cm mark using the syringe. This was done by inserting a fine plastic tube syringe down to the bottom of the open 4 cm bubble tube. The bubble taps were closed and water was drawn into the disc permeameter by placing the permeameter just under the surface of the water in the full bucket and using the suction pump to draw water into the stand pipe. The pump was connected by means of tubing to the non-return valve on the tap of the standpipe. The permeameter was tilted to remove air bubbles trapped in the disc. Finally, the

permeameter was placed back on its pad (polystyrene foam). The water level was observed to the 4 cm level mark while bubbling gently. A similar procedure was repeated for the other disc permeameter.

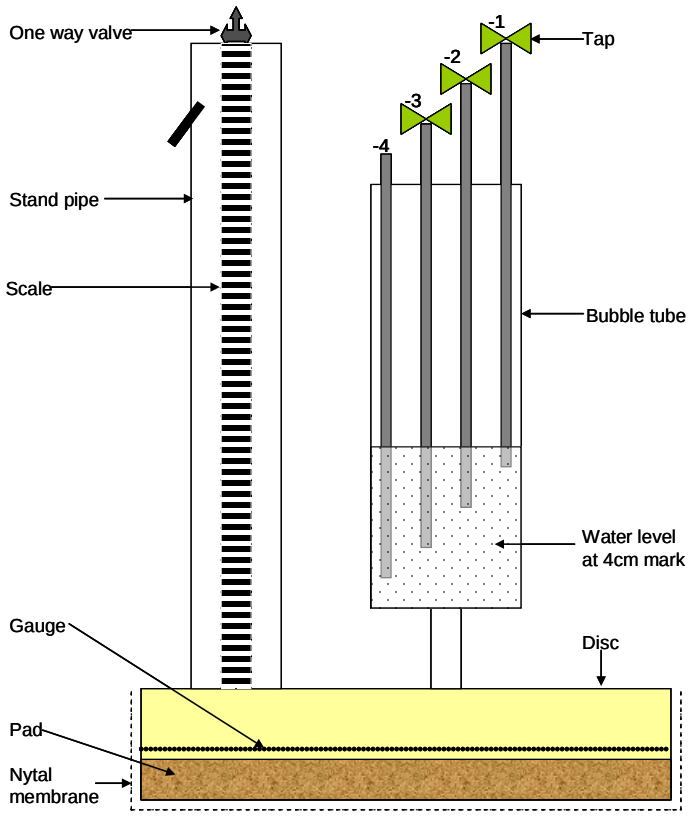

Figure A1: Disc Permeameter

During measurement, assessed surface infiltration was a levelled site was chosen as surface infiltration was monitored. Mulch and other litter were cleared from the surface and vegetation such as small grasses and weeds were clipped to ground level. The 200 mm diameter x 3 mm ring was placed on the surface to fit the surface tightly with no gaps to avoid water loss due to the sloping sand at –1 cm water suction. The ring was filled with moist (3%) sand, which was spread evenly by hand and levelled using a scraper. The small sand piles were carefully brushed away to increase contact area of the sand and the membrane in the disc permeameter. The ring was carefully removed and covered with a small bucket to stop desiccation. A similar procedure was repeated for the other permeameter.

During measurement, the site details such as permeameter number and combination factor as marked on the permeameter (e.g. 15.4) were recorded. Before measurement the buckets were removed from the sand pads and the disc permeameters were placed on each of the pads avoiding any bubbles in the disc. Bubble taps were completely closed except -4 cm tap. The tape recorder including a pre-set cassette player was started. The first reading was taken at 15 seconds with the subsequent reading taken with a difference of 5 seconds to take the second permeameter reading. Readings were taken as per the cassette at 15 seconds interval up to 1 minute 30 seconds, at a 30 seconds interval up to 4 minutes, and then a 60 seconds interval for the last reading. Then, the -3cm water tap was opened and the tape recorded was rewound and the same procedure was repeated. This procedure was repeated for -2cm and -1cm taps as well for both permeameter. If infiltrations rates were relatively high, then readings are -4cm water can be terminated at 5 min. Open -3cm water tap, reset stopwatch or rewind tape recorder and commence readings for -3cm water supply tension. When the water in the stand pipe seemed like it was running out, the permeameter was removed and rinsed and cleaned, placed on the polystyrene foam pads and refilled for the next site. Three such measurements were taken at three different sites in each plot.

In this study in-situ determination of soil moisture status was carried out on a monthly basis using micro-gopher (Bhattarai *et al.* 2006) which provided information on plant water uptake and on influence of ground water table fluctuation on phytocap. Installation of piezometers for testing the groundwater quality also assisted in collecting data on composition of landfill material and their moisture content. This study identified depth of the ground water table from the surface of the phytocap and also characterised its influence on the soil within the phytocapping system during rain events and tidal fluctuations.

Several factors affecting water retention properties of soil; include changes in soil structure due to root development (Ague *et al.* 2001), soil texture (Williams *et al.* 1983), porosity (Bird *et al.* 2005), pore size (Johnson *et al*. 2003), bulk density (Kalman *et al.* 1996) and soil compaction (Willat and Pullar 1983). Additionally, spatial variability

within the landscape also impacts water retention properties of soil (Shouse *et al.* 1995). Soil compaction changes soil physical properties and hydraulic properties and also has adverse effects on tree growth (Assouline *et al.* 1997, Unger and Kasper 1994). Soil compaction can occur naturally by settling of soil or may be induced by tillage, heavy machinery, fire, trampling by animals (Kozlowski 1999) or raindrops (Agassi *et al.* 1985, Morin and Van Winkel 1996). Soil compaction also affects growth behaviour of plants, which in turn may have an adverse impact on plant biomass, leaf area (Batey and Mckenzie 2006) and transpiration (Sadras *et al.* 2005, Carr and Dodds 1983) due to inhibition of root growth (Carr and Dodds 1983). These parameters evidently decide the fate of phytocapping system. Low hydraulic conductivity of the soil used in phytocaps is an important consideration as it has to maintain the integrity and sustainability of the system for up to 30 years. Determining hydrological properties of soil is required prior to establishing a phytocap to ensure sustainability of phytocap throughout its lifespan. Determination of soil hydraulic properties were also required to predict site water balance and were analysed using HYDRUS 1D. Data presented on soil hydraulic conductivity are an extract of results from field experiments conducted by Dr Ian Philips in 2004 and 2005.

Soil Physical and Chemical Properties

Scientists have always relied on knowledge of chemical and physical properties of soils to assess capacity of sites to support productive forests (Schoenholtz *et al.* 2000). Recently, the need for assessing soil properties has expanded because of growing public interest in determining consequences of management practices on the quality of soil relative to plant productivity (Schoenholtz *et al.* 2000). Soil quality includes assessment of soil properties and processes as they relate to ability of soil to function effectively as a component of a healthy ecosystem. Basic soil quality indicators like soil texture and depth are useful for comparing soil quality among soil types, and within a soil type before and after some management practice have been imposed. Soil texture is the most fundamental qualitative soil physical properties controlling water, nutrient, and oxygen exchange, retention, and uptake (Schoenholtz *et al.* 2000). Soil colour can provide quantitative information on the current soil moisture status (Brady and Weil 1999).

Similarly, N, P, K, organic matter content, EC and pH are the first order soil chemical indicator.

Soil organic matter content is commonly recognised as one of the key chemical parameters of soil quality, yet quantitative assessment of its contribution to soil quality is often lacking (Schoenholtz *et al.* 2000). Through its role in aggregate stability it influences soil porosity, and thus gas exchange reactions and water relations. It is a critical pool in the carbon cycle and a repository of nutrients, and through its influence on many fundamental biological and chemical processes it plays a pivotal role in nutrient release and availability (Nambiar 1997). Electrical conductivity is a measure of ion concentration and potentially negative effect of salinity on water relations and nutrient imbalances that inhibit plant growth and productivity (Burger *et al.* 1994). In this study, soil physical characteristics such as soil texture, soil EC, soil pH, soil organic carbon content and soil colour were determined. Soil samples were sampled using an auger and placed in plastic bags, labelled and sent to CSBP Pvt Ltd, Perth for analysis. Soil chemical analysis for various elements was also done using the same sampling technique as described above. Standard methods were used to determine soil physical properties (Table 4.1). Soil colour was determined using Maunsel colour charts. Data on soil colour provided by the soil-testing laboratories (CSBP, WA) was also used.

Samples were collected from different types of soil to be used in Phytocapping using a soil auger (10.2 cm diameter, bucket auger, with detachable 1.2 m handle). Representative samples, weighing about 300 g, from each soil layer were taken to the CQU laboratory and air dried. During drying, soil clods were tapped and crushed. Dried samples were sent to the CSBP laboratory (Westfarmer's Ltd., Western Australia) for chemical analysis (see Appendix B for methodology). The laboratory used standard extracts (Appendix B) to determine nutrient concentrations using Atomic Absorption Spectrophotometer (AAS), Inductively Coupled Plasma Atomic Emission Spectrometer (ICP AES) and segmented flow analysers.

Results and Discussion

Hydraulic properties of the Thick (1400 mm) and Thin (700 mm) phytocaps were assessed both in the field and in the laboratory before planting the seedlings (2004) and after one year after planting (2005) by Dr Ian Philips (Griffith University). These values were subsequently used in the HYDRUS 1D modelling to predict the site water balance. The soil infiltration rate was determined at final stages of the study (2007) using a disc permeameter. Soil compaction was also tested before harvesting the plants.

Soil Hydraulic Properties Determination

Soil hydraulic conductivity obtained using disc permeameter in 2007 were 57.6 cm d^{-1} and 62.4 cm d^{-1} for the sandy loam layer in Thick phytocap and Thin phytocap, respectively. This is higher than that observed in 2004 (16.5 cm/day) (Table A1). This may due to root development and particle size distribution (Shuh and Bauder 1986).

Table A1: Comparison of measured hydraulic conductivity of different soils with default values from HYDRUS 1D

	Default Hydraulic conductivity HYDRUS 1D	Phytocap	Measured hydraulic conductivity (2004-2005)	Measured hydraulic conductivity (2007)
Soil type	cm/day	Soil type	cm/day	cm/day
Sand	7128	Mulch	259.2	
Loamy Sand	3502	Sandy Loam	165.6	60
Sandy Loam	249.6	Andersite Clay	2551.6	
Loam	60	Yaamba Clay	1219.2	
Silt Loam	108	Black Cracking Clay	6451.2	
Sandy Clay Loam	314			
Clay Loam	62.5			
Silty Clay Loam	16.8			
Sandy Clay	28.8			
Silty Clay	4.8			
Clay	48			

Soil Density Determination (2007)

Soil compaction levels varied with depth ranging from 120 kPa to 200 kPa (Fig. A2). This was expected as the top layers of the phytocapping system were greatly influenced by root growth (Assouline *et al.* 1997, Unger and Kasper 1994) and soil water content (Soane and Van Ouwerkerk 1994). At all compaction levels, the penetration resistance increased with decreasing soil water potential (Lipeic *et al.* 2002). Sandy loam had greater compaction than clayey soils in the phytocapping system. In other words, increasing soil moisture content of the clay soils reduced the load support capacity of the soil (Kondo and Dias Jr 1999) thus decreasing permissible ground pressure (Medvedev and Cybulko 1995).

Soil compaction and soil moisture are only significant when comparing soils of the same depth. Considerable variation between depths in the same profile, and between profiles, makes it difficult to compare results (Quiroga *et al.* 1999). At high soil moisture levels, difference in soil resistance between compacted soil and un-compacted soil is low and usually lower than the value that limits root growth (>2000 kPa). Hence, it is clear that the site is compatible with good tree growth and is ideal for phytocapping. The degree of compaction depends on the soil mechanical strength, which is influenced by soil texture and soil organic matter contents (Larson *et al.* 1980, Hettiaratchi 1987).

Since soil compaction mainly decreases soil porosity, and thus increasing soil porosity is a clear way of reducing or eliminating soil compaction. Managing soil compaction, especially in arid and semi-arid regions, can be achieved through appropriate application of some or all of the following techniques: addition of organic matter; controlled traffic; mechanical loosening such as deep ripping and selection of plants with strong tap roots (Hamza and Anderson 2005); and at the same time selecting species that can transpire percolated water.

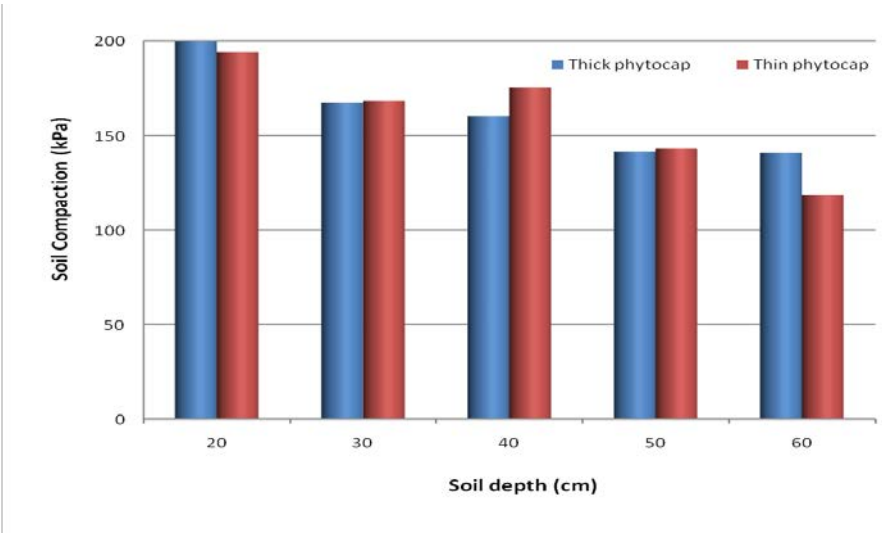

Figure A2: Soil compaction levels at various depths of Thick and Thin phytocaps

Soil Physical and Chemical Properties

Soil physical and chemical properties were analysed to determine composition of the soil used in phytocapping and its properties so as reduce the adverse impacts of the soil on tree growth. The study site has a mixture of brown, brownish grey and grey soil, ranging from sandy loam to clay texture (Table A2). Soils in this study had a field capacity ranging from 30% to 46%, with a pH of more than 7 except for sandy loam soil which was slightly acidic in nature (Fig A3). Bulk density of soils used ranged from 1.32 to 1.58 Mg m^{-3} and the porosity of soils ranged

Table A2: Colour of the soils used in phytocapping

Soil Type	Plot	Cap	Colour
Black Clay	Plot 1	Thick	Grey
Black Clay	Plot 1	Thick	Grey
Black Clay	Plot 2	Thin	Greyish brown
Black Clay	Plot 2	Thin	Brownish grey
Black Clay	Plot 3	Thick	Brownish grey
Black Clay	Plot 3	Thick	Brownish grey
Black Clay	Plot 4	Thin	Brownish grey
Black Clay	Plot 4	Thin	Brownish grey
Sandy Loam			Brown
Yaamba clay			Brown
Andersite Clay			Brown

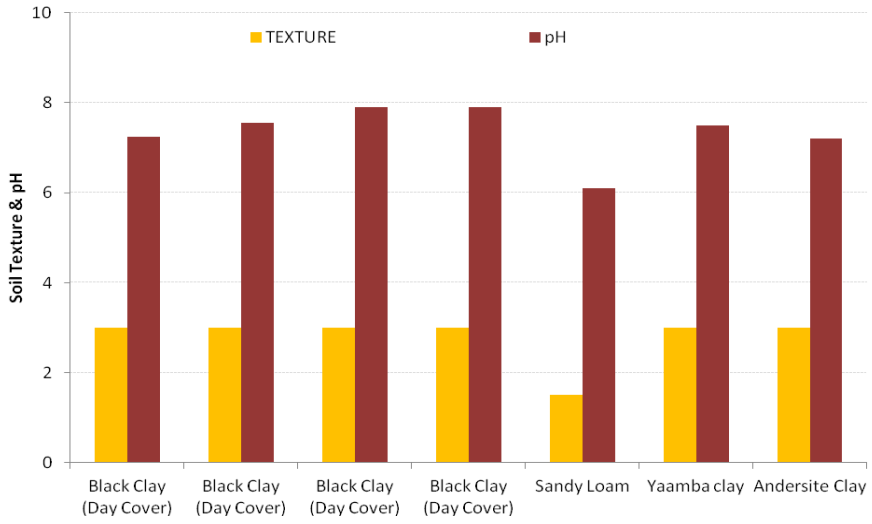

Figure A3: Soil texture and pH
Note: The four Black Clay denote plot 1, 2, 3 & 4 respectively

It is interesting to note that the top layer of the phytocapping system comprised sandy loam soil was slightly acidic in nature (Fig A3) as compared to the other soil material. On the other hand the organic carbon content was very low in the sandy loam soil as compared to that in other soil layers (Fig A4).

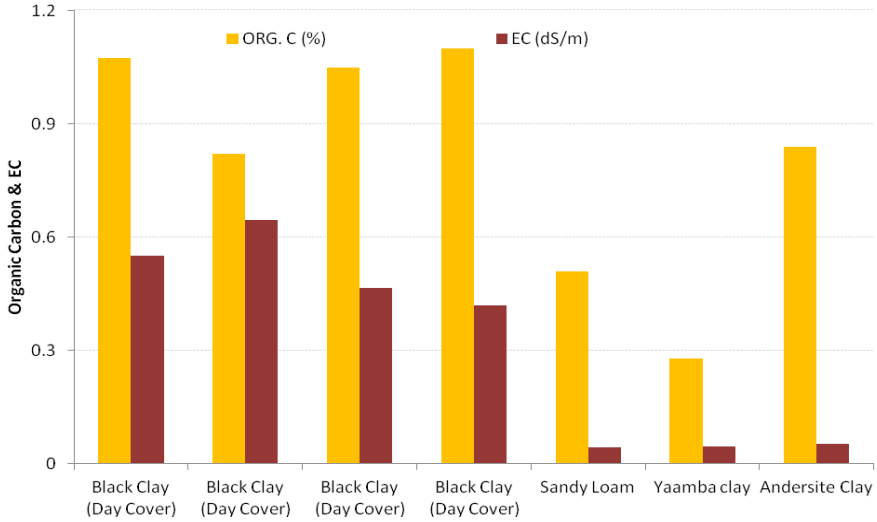

Figure A4: Soil EC and organic carbon content
Note: The four Black Clay denote plot 1, 2, 3 & 4 respectively

Presence of elevated levels of exchangeable Ca and Mg in the black clay soil suggests that these may be taken up by plants with deep roots and this may be evident from their foliar chemical analysis.

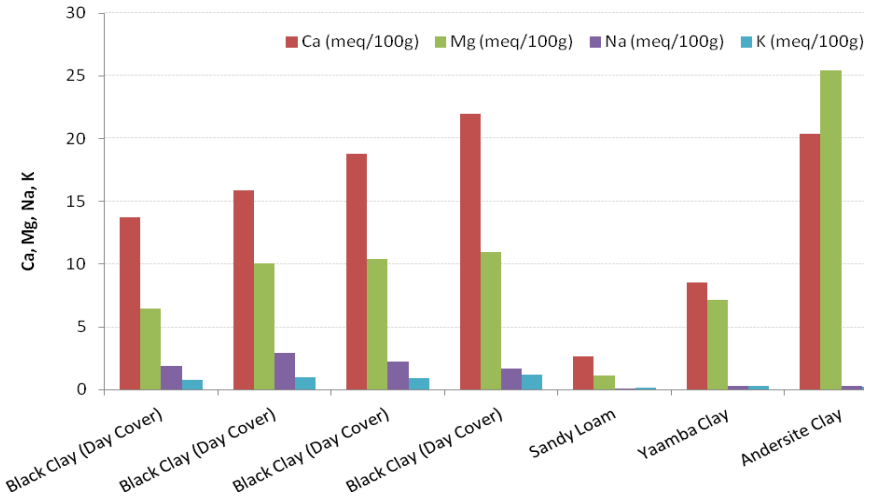

Figure A5: Exchangeable soil Ca, Mg, Na & K
Note: The four black clay denote plot 1, 2, 3 & 4 respectively

Overall, the soil physical and chemical properties showed that the soils used in the study were similar in retaining soil nutrients (Fig A6, A7 & A8), except that the sandy loam and Yaamba clay had exchangeable ions.

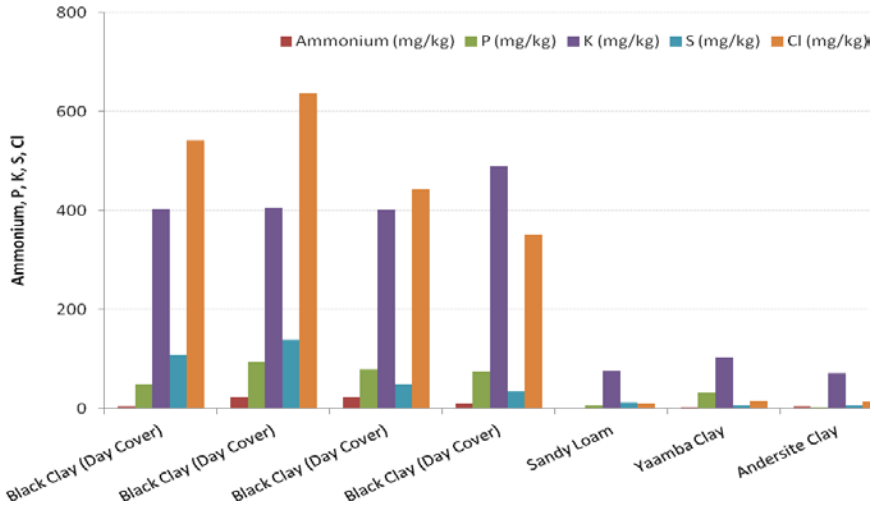

Figure A6: Soil micronutrients
Note: The four Black Clay denote plot 1, 2, 3 & 4 respectively

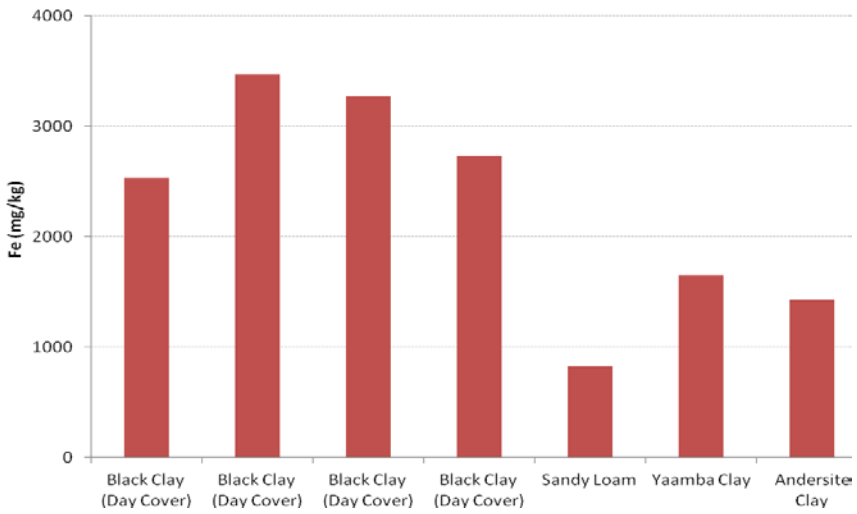

Figure A7: Fe in soils
Note: The four Black Clay denote plot 1, 2, 3 & 4 respectively

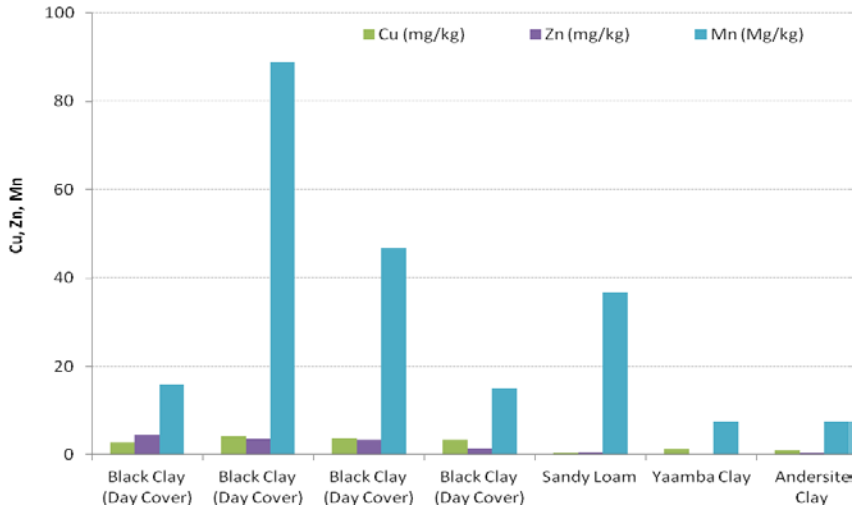

Figure A8: Cu, Zn & Mn in soils
Note: The four Black Clay denote plot 1, 2, 3 & 4 respectively

Conclusions

Results from the study conducted in 2004 reveal that values of hydraulic conductivity were of the order expected for various soils. Variability between field and laboratory conditions was high, which is not an uncommon phenomenon given the heterogeneity of the soil. The irrigation schedule appeared to be appropriate for the trial; however

surface ponding and poor infiltration rates were observed at the site during rainy season. High field water capacity of the sandy loam layers reduces the rate of water entry in to the soil during the early stages of irrigation. It is also apparent that the soil structure may improve due to root penetration, as plants grow and the maintenance of the soil-water deficit through transpiration.

Findings from the 2005 results revealed that the hydraulic conductivity of the capping system improved significantly. Increasing hydraulic conductivity, along with improved sorptivity, aggregate stability and soil-water retention properties, strongly indicates that capping systems change in response to factors such as plant growth and soil fauna. However, results obtained from the above two studies will be incorporated in the HYDRUS 1D code to simulate site water balance of the phytocapping system. This will not only predict the percolation rate but also define the significance of tree selection and soil thickness for increased sustainability of the phytocapping system.

Results from the permeability tests conducted in 2007 revealed a slight increase in hydraulic conductivity of the sandy loam soil, which could be due to changes in soil structure, root growth or due to raindrop effects as reported in the past. However, compaction rates of soils after 3 years of planting are appropriate for retaining maximum moisture adequate to support root growth.

Results from this study suggested an inconsistency in the waste height buried within the landfill. Moisture content data clearly suggests that most of the waste material was affected by tides and were completely soaked in water at a depth of 5 m.

The study site has a mixture of brown, brownish grey and grey soil, ranging from sandy loam to clay texture. Soils in this study had a field capacity ranging from 30% to 46%, with a pH of more than 7 except sandy loam soil which was slightly acidic in nature. Bulk density of the soils used ranged from 1.32 to 1.58 mg m^{-3}.

Appendix B
Foliar and Soil Chemical Analysis

Foliar chemical analysis performed by CSBP Wesfarmers Pty Ltd Perth, Australia. The following protocols were followed by the laboratory:

ROUTINE METHODS

Sample Preparation:

All soils were dried for 24 hours at 40˚C and then sieved and ground to <2 mm.

A. EXTRACTABLE SULPHUR IN SOILS (mg/kg) Soils are extracted at 40°C for 3 hours with 0.25M potassium chloride and the sulphate sulphur is measured by ICP. This method is known as the KCI-40 or Blair/Lefroy Extractable Sulphur method (Blair *et al.* 1991).

B. NITRATES AND CHLORIDES IN PLANTS

This method determines colorimetrically via the Lachat Flow Injection Analyser the Nitrate Nitrogen content in plant material. Nitrate is reduced to nitrite by passage of the sample through a copperised cadmium column. The nitrite concentration of the solution is then determined colorimetrically on Lachat at 520 nm.

The chloride content in the solution is then determined colorimetrically. The liberation of thiocyanate ion from mercuric thiocyanate by the formation of soluble mercuric chloride forms the basis of the reaction. In the presence of ferric ion, free thiocyanate ion forms coloured ferric thiocyanate, and its absorbance is proportional to the chloride concentration. Ferric thiocyanate is read by the Lachat at 480 nm.

C. OXIDISABLE ORGANIC CARBONS IN SOILS (%)

Concentrated sulphuric acid is added to soil wetted with dichromate solution. The heat of dilution is used to induce oxidation of soil organic matter. The amount of chromic

ions produced is proportional to the organic carbon oxidized and is measured colorimetrically at 600 nm.

D. PHOSPHORUS & POTASSIUM IN SOILS (mg/kg)

Available phosphorus and potassium are measured using the Colwell method. Soils are tumbled with 0.5 M sodium bicarbonate solution adjusted to pH 8.5 for 16 hours at 25°C employing a soil: solution ratio of 1:100 (Rayment and Higginson 1992).

The acidified extract is treated with ammonium molybdate/antimony trichloride reagent and the concentration of phosphorus is measured colorimetrically at 880 nm. The concentration of potassium is determined using a flame atomic absorption spectrophotometer at 766.5 nm.

E. TRACE ELEMENTS IN PLANTS

After complete digestion of the plant material with a mixture of nitric acid and hydrogen peroxide, the digests are diluted with hot de-ionised water to dissolve all precipitates. The clear solutions are presented to the ICP-AES for determination of the elements, B, Cu, Zn, Mn, Fe, Ca, Mg, Na, K, P and S (McQuaker *et al.* 1979).

F. LECO TOTAL CARBON & NITROGEN IN PLANTS & SOILS (%)

Total carbon (TC) and total nitrogen are a measure of all the carbon or all the nitrogen in the sample, including both inorganic and organic carbon/ nitrogen. Samples are measured on the LECO analyzers as follows the sample is loaded into a sealed glass combustion tube at 950˚C and flushed with oxygen for a very rapid and complete combustion. All gases generated are collected and a sub sample of this gas is measured on an infrared detector and a thermal conductivity cell to measure total carbon and total nitrogen respectively.

NON-ROUTINE

1. **EXTRACTABLE ALUMINIUM (mg/kg)**

Soils are extracted with 0.01 M calcium chloride solution and the extract analysed for aluminium by a colorimetric method using a catechol violet reagent (Bromfield 1987)

2. **EXTRACTABLE BORON (mg/kg)**

Soils are extracted in boiling calcium chloride solution for 15 minutes and the resulting boron content is determined colorimetrically using Azomethine-H colouring reagent and the coloured solutions are measured by UV/Visible Spectrophotometry

3. **EXTRACTABLE PHOSPHORUS (OLSEN) (mg/kg)**

Soil are extracted at a ratio of 1:20 with $NaHCO_3$ (pH 8.5) for 30 minutes. The acidified soil extract is treated with ammonium molybdate/antimony trichloride reagent and the concentration of phosphorus is measured at 880 nm on a UV/visible spectrophotometer (Rayment and Higginson 1992).

MOLYBDENUM & HEAVY METALS IN PLANTS Plant material is digested in a mixture of nitric acid and hydrogen peroxide before molybdenum, cobalt, selenium, arsenic, nickel, lead and cadmium are measured using ICP-MS. Soils are extracted with Aqua Regia, Tamm's reagent or hot calcium chloride & extracts are measured on the ICP-OES (McQuaker *et al.* 1979).

Appendix C
Tree Canopy Images Using HemiView

Dendrocalamus latiflorus

Eucalyptus grandis

Hibiscus tiliaceus

Casuarina Cunninghamiana

Figure C1: Canopy density of four tree species grown in the phytocapping system after 3 years of planting, as viewed by HemiView