DYNAMICS OF VEGETATION AFTER CABLE AND GROUND-BASED LOGGING

By

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Abstract

The relationship between forest resource management and the conservation of biological diversity is a controversial one, with the steep country logging of old-growth forests in Tasmania being the source of much of this attention, due to the perceived greater impacts of logging on steep slopes and the greater damage caused by the large modern cable systems.

Retrospective (chronosequence) and permanent plot studies of cable and ground-based logging in forests aged 0-54 years on steep slopes in southern Tasmania and comparable unlogged sites indicated that following both logging treatments, there was a gradual recovery in floristic composition towards mature forest. Vascular plant species richness recovered more rapidly in the ground-based logging treatment than the cable logged treatment. Differences between treatments were confined to the initial recovery stages where treatment effects were evident in the extent of soil, woody debris and micro-habitat disturbance. Following canopy closure and self-thinning (i.e. after 20-30 years) sites logged by the two techniques resemble each other in vascular species composition and that of mature unlogged forest. However, differences still existed in forest structure and in the composition of the obligate epiphytic fern flora (particularly that present on *Dicksonia* caudexes).

Generally cable logging did less physical damage to vegetation than ground-based logging. Ground-based logging led to a greater degree of floristic change than cable logging. The pattern of vegetation change was related to the degree and nature of soil and habitat disturbance.

Conventional ground-based thinning of 30 year old regrowth forest led to a greater change in the composition of the ground stratum (an increase in the cover of weed and other herbaceous species) than cable thinning. Cable thinning caused little soil disturbance and ground fern species rapidly regenerated rhizomatously. Herbaceous and weed species were infrequent after cable thinning.

Cable logging practices adjacent to or through retained stands of streamside vegetation can result in significant changes to the vegetation. This can occur during all phases of tree falling, establishment and movement of cable settings, yarding, fire break construction and the regeneration burn. Many of the detrimental effects of cable logging adjacent to retained stands of streamside vegetation are indirect, resulting from actions such as windthrow, tree health decline and crown scorch

from regeneration burning. While there has been an improvement in forest practices since the 1980's when the attempted full suspension logging over streamsides was ended, the on-going health and viability of retained streamside vegetation is still in doubt.

Ground-based logging inflicted greater levels of mortality than cable logging on populations of the tree fern *Dicksonia antarctica*, a functionally important understorey species in wet forests. Approximately 80% of the *D. antarctica* resprouted following cable logging prior to burning. Only 40% survived the regeneration burn. This was reduced to 27% after 10 years. While ground-based logging resulted in greater mortality, it led to a greater degree of new sporophytic recruitment to the *D. antarctica* population. The significant reduction in the abundance of this slow growing species following logging has implications for the conservation of obligate vascular epiphytes that use it's caudex as a substrate.

The vascular epiphytic flora, the primary late successional specialist group, was almost completely lost following cable logging. Epiphytes were slow to recolonise following either cable or ground-based logging, although their recovery was more rapid following ground-based logging. While most of the vascular epiphytic species recovered 30-50 years following either logging method, their abundance was reduced and the diversity of substrates available for colonisation was limited.

The response of vegetation to logging treatment was studied at the three levels; the community, the functional group and the species. No general statement can be made about the logging treatments across these levels as the treatment response was not uniform. However, the individual species level of response provided most of the understanding of processes operating following either logging method in terms of how these treatments differ in their spatial variability, the biological legacies they create and their intensity. The use of permanent plot monitoring associated with the chronosequence studies effectively validated the generalised chronosequence trends, at least for the short term predictions, and also questioned many of the trends described elsewhere in the literature from chronosequence studies which lack the rigour of an individual species level of experimental treatment and monitoring.

The examination of cable and ground-based logging in a controlled manner has provided a depth of understanding to previous logging studies that was generally not available. It has added additional emphasis to the role of logging as both a process of vegetation disturbance and one of establishing biological legacies which mediate the regenerative response. It also questions much of the literature which describes vegetation responses in terms of generalised hypotheses, eg the intermediate disturbance hypothesis, as vegetation responses are not uniform across differing levels of examination.

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Chapter 1 Introduction

1.1 Steep country defined

Forest logging practices on steep slopes (>18^0) involve a choice between cable and ground-based methods. The choice is made depending on economic, engineering and environmental constraints. While, the economic (Verani 1995, Stokes and Schilling 1997, Sauder and Wellburn 1987, Murphy *et al.* 1991) and engineering (Forrester 1995, Shaffer *et al.* 1993, Courteau and Heidersdorf 1991, Bennett 1997) comparisons of these logging practices are well researched, there is little published research that compares the environmental aspects, with the exception of issues related to soil erosion (Watt and Krag 1988, Minore and Weatherly 1988). More specifically, there is little research that relates the choice of logging practice (cable versus ground-based) to the observed patterns of recovery of vegetation following logging. This is because the majority of logging on steep slopes throughout the world is conducted using ground-based methods. Consequently, few studies have had the opportunity to compare steep slope logging methods, in the same place, at the same time, keeping environmental factors (such as slope, soil type and climate), and forest types constant.

Steep slopes and cable logging are not synonymous. Steeply sloping land does not necessarily require cable techniques (e.g. Rogers and MacDonald 1989), and cable techniques are not necessarily restricted to steep slopes (Meek 1996). Apart from those cable systems which rely upon a moderate slope (e.g. >10^0) for their gravity driven carriage return (e.g. cable thinning experiment in chapter 3), most cable systems will also operate on flat terrain, in fact some are particularly suited to environmentally sensitive sites such as wetlands (Aulerich 1990, Shabalin *et al.* 1991). Successful applications of cable logging systems on flat land include Indonesia (Bertault and Sist 1997), Columbia (Aulerich 1990), the United States (Jones *et al.* 2000) and the former Soviet Union (Saniga 1988). Between 1990-95, 11.9% of the areas logged by cable systems in Tasmania were not steep land, according to the slope criteria in the Forest Practices Code (1995).

The aim of this study was to examine the impact of a range of steep country logging practices on the regenerative response of native vegetation. In particular, this study compares cable and ground-based logging in Tasmania, where both practices were implemented on steep slopes.

1.2 The extent of steep country logging systems globally

Steep country logging, and cable logging systems, are employed widely across north America, Canada, the former Soviet Union, northern and central Europe, New Zealand and with limited applications in Australia, Japan and south east Asia. Large complex systems (for example skyline settings) predominate in north America and Canada, while smaller more mobile systems are more common in central and southern Europe and Japan. No data could be sourced on the proportion on steep to non-steep land logged using cable or ground-based systems globally.

1.3 The extent of steep country logging systems in Australia

Cable logging in Australia is almost entirely a Tasmanian phenomenon, with very limited examples of cable logging in New South Wales, in Queensland (Ward 1982) and the A.C.T. (Melmoth 1979). Although similar forest types exist in New South Wales and Victoria to those logged using cable techniques in Tasmania, the widespread use of the technology in Tasmania is seen as a response to the prevailing policies concerning forest utilisation, particularly with large logging units and integrated sawlog and pulpwood markets, which did not exist to the same extent in other Australian states from the 1970's.

1.4 The extent of steep country logging systems and environments in Tasmania

Vegetation management practices associated with native forest harvesting in Australia remain controversial despite an almost continuous series of public inquiries since the late 1970's. Logging practices in Tasmania have received particular scrutiny through these inquiries, with part of this attention (e.g. Madden 1991) being focussed on the increasing proportion of steep land and cable logging since the late 1980s (Figure 1.1), which also coincided with the sharp increase in logging activity from the 1970's associated with the export of woodchips to Japan (RAC 1992). Cable logging operations were also controversial because of the lack familiarity with a technology developed in North America, Canada, Europe and New Zealand, the visual impact of large scale logging operations undertaken using this technique (Forest Practices Code 1993), risks of soil erosion (Laffan 1991), potential for regeneration difficulties and an unknown vegetation response (Madden 1991).

While of particular importance in areas such as the Florentine, Styx, and Tyenna Valleys southern Tasmania and the north east Tasmanian mountains, cable logging only accounted for one-fifth (Figure 1.1) of the total area logged (or approximately

13% of total wood cut) in Tasmania in 1990. However, cable logging has increased relative to ground-based logging since the early 1980's (Figure 1.2).

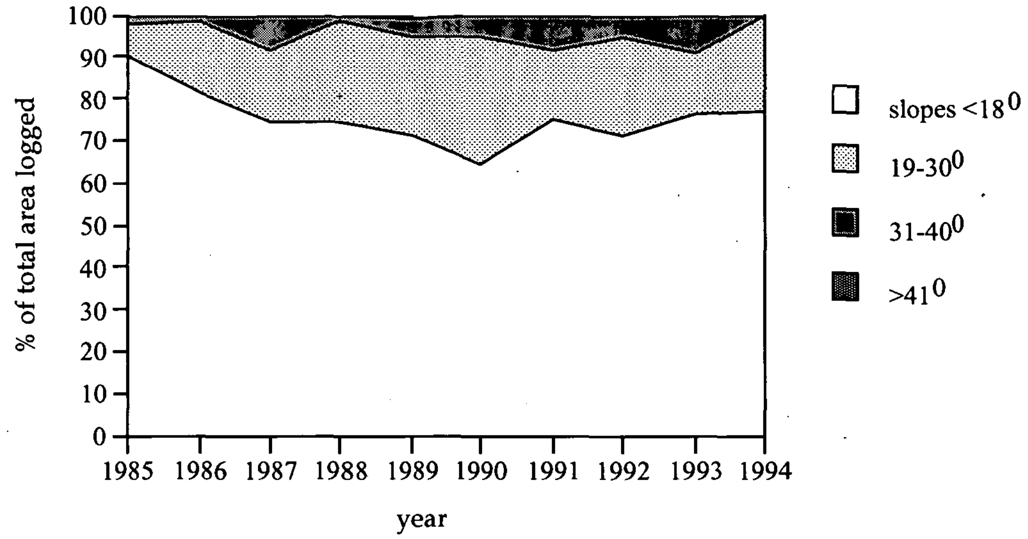


Figure 1.1 *The proportion of steep slopes (>18⁰) relative to shallow slopes (<18⁰) logged in Tasmania between 1985 and 1994, source: Forestry Tasmania. Only a small portion of land (>41⁰) was logged between 1988 and 1990.*

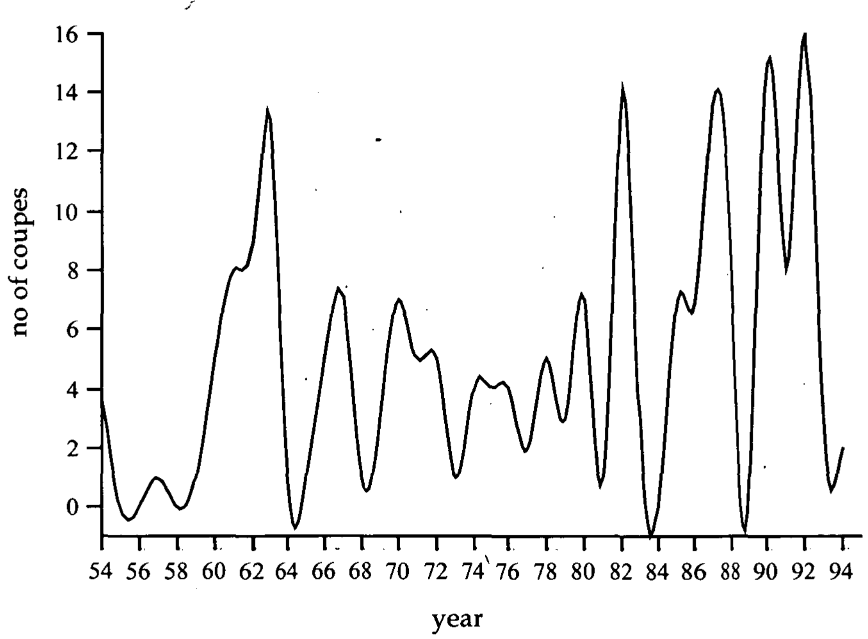


Figure 1.2 *The number of coupes logged using cable systems in the Florentine, Styx and Tyenna Valleys of southern Tasmania 1954-1994.*

More recent applications of cable logging in regrowth thinning have expanded its use to the extent that its future viability in both first and second rotation harvests appears to be assured. Much of the information on the potential environmental consequences of this expansion rests on data from north American studies of soil and regolith (soil-bedrock interface) stability, none of which is based on an explicit consideration of the effect of logging practices on vegetation.

The environments and vegetation of steep slopes in Tasmania

Steeply sloping land in Tasmanian forests available for logging is defined by the Forest Practices Code (1993) as land with slopes above 20^{o} (36%) and a steep coupe is defined as a coupe with more than 50 percent of its area on slopes greater than 20^{o}. A coupe is defined as an area of forest of variable size, shape and orientation, on which harvesting takes place and is usually harvested and regenerated over one or two years (Forest Practices Code 2000). Similar restrictions of $25\text{-}30^{o}$ are placed on ground-based logging on steep slopes in other Australian States. Geomorphologists consider slopes of 18^{o} or 20^{o} to distinguish between steep and moderately steep slopes (Tersleck 1983).

The environmental characteristics of steep slopes are highly variable. An analysis of the range of forested areas available for logging in Tasmania by slope categories revealed that one-third are steep using the above criteria. An analysis of the characteristic slope of wet eucalypt forested communities in Tasmania (Kirkpatrick *et al.* 1988) indicated that only small proportion were recorded predominantly from steep slopes, and the dominant species on steep slopes (*E. coccifera* and *E. delegatensis*) were outside the optimal environments for sawlog and pulpwood production (Figure 1.3). Similarly, no vegetation communities normally associated with logging were described in the literature as being exclusively associated with steep slopes. The majority of the temperate eucalypt forest communities subject to cable logging in Tasmania are not restricted to steep slopes, they occur on a range of steep and shallow slopes, their presence determined by other ecological factors such as moisture and geologies (Kirkpatrick *et al.* 1988), and in the case of wet eucalypt forests, characteristically on south east facing moist slopes (Reid *et al.* 1999). Those vegetation communities directly associated with steep slope environments in south eastern Australia such as rock outcrops (e.g. Ashton and Webb 1978), block streams (e.g. Caine and Jennings 1968, Ashton and Moore 1978), rock cliffs (Coates and Kirkpatrick 1992) and landslides (e.g. Cullen 1991) are generally unsuited to logging.

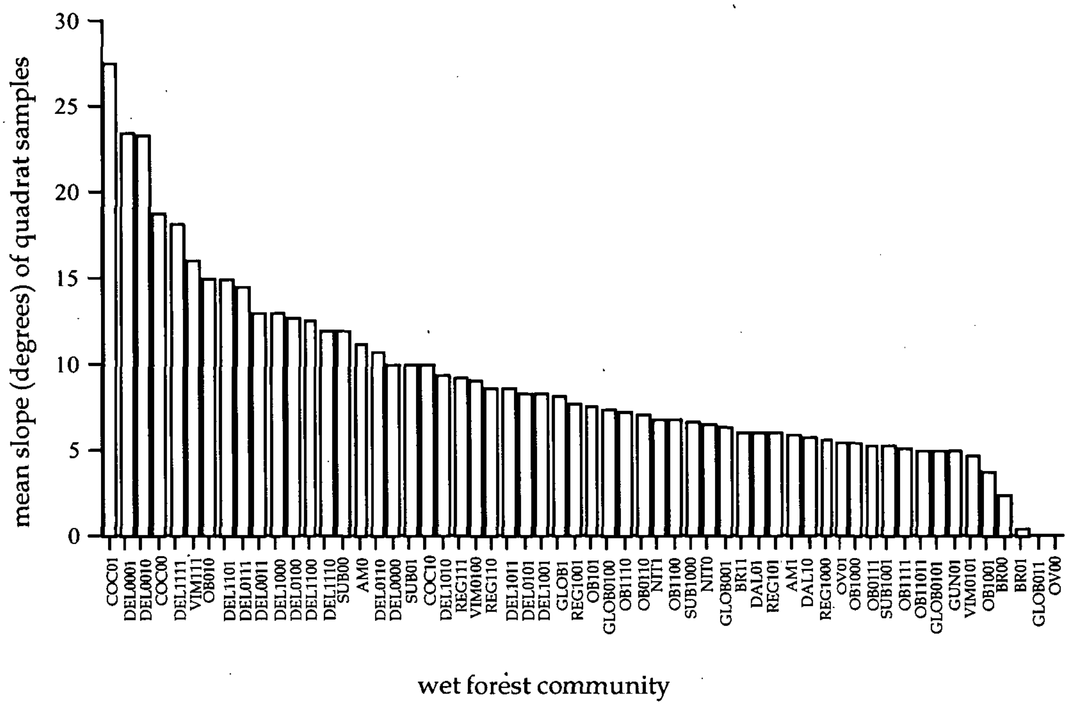


Figure 1.3 *The relationship between slope and the Tasmanian wet forest communities described in Kirkpatrick* et al. *(1988)*

Of the geological types characterising land subjected to cable logging, dolerite and mudstone occupy the greatest area (Table 1.1), with the patterns largely conforming to regional geological differences across Tasmania.

Table 1.1 *Predominant geological types within land cable logged in Tasmania 1990-95.*

geology	number of cable coupes	% of total area cable logged
dolerite	41	35.7
mudstone	46	34.7
granite	13	9.1
other sediments	16	7.8
dolerite/mudstone	7	6.6
basalt	16	6.1
total	139	100

Slope is only one component of the environment for vegetation on steep land in southern Australia, and while it could be assumed to effect insolation, wind exposure, run off, erosion, soil depth, etc. there are few direct references to it in the ecological literature, except for the following examples. On steeply sloping ground, the activities of burrowing and digging marsupials or monotremes may be more pronounced in terms of soil and litter disturbance than on lesser slopes. On sloping ground, litter will collect up to 20-30cm deep against obstructions including tree trunks, logs and rocks, and the soil itself is scratched downhill and heaped at its angle of rest by lyre-birds (Ashton 1975a) at an average of 70 cm/year and 200 tonne/ha (Ashton and Bassett 1997). Foraging by lyrebirds in bare floor areas on steep slopes will therefore result in a complex micro-topography of excavations, accumulations and terracettes (Ashton and Bassett 1997). Slope has also been suggested to be a determinant of the pattern of tree root development in *E. regnans* due in part to wind direction and possibly soil creep (Ashton 1975b). Working in similar forested environments, Lindenmayer *et al.* (1999b) argued that because steep slopes receive less radiation than flat slopes, and less radiation equals increased moisture and a lower probability of catastrophic fires, these steeply sloping environments were more likely to promote the development of multi-aged stands of *Eucalyptus regnans* in high rainfall areas. For a general treatment of the ecological literature relevant to the *E. regnans-E. obliqua* dominated forests of Tasmania studied in this thesis, see Wells and Hickey (1998) and Ashton (1976, 1981) and Ashton and Attiwill (1994)

Cable logging systems on steep slopes in Tasmania

Cable logging systems can be rigged in a variety of ways to accommodate variation in slope profile, log mass, logging area size and shape, the occurrence of streams, and requirements for lateral yarding, etc. Cable logging systems are frequently varied from site to site within one logging area. A brief description of the major cable logging systems applied in the Tasmanian forested areas examined in this thesis (Table 1.2) follows. Since no descriptive accounts of the cable systems operating in Australia are available (with the exception of cable thinning by Cunningham 1997), overseas technical references including Studier and Binkley (1974), McGonagill (1978) and Liley (1983) are followed. Each system is also described diagrammatically in Figures 1.4, 1.6, 1.8, 1.10 and 1.12 and their attributes contrasted with ground-based logging in Table 1.2. Photographs of these logging systems are presented in Figures 1.5, 1.7, 1.9 and 1.11. The essential point here is that cable logging systems, like ground-based logging systems, are highly variable, and as a consequence, so are their environmental effects.

Table 1.2 *Summary of logging systems applied on steep slopes at the experimental sites in this thesis*

Configuration	Experiment	Location	Chapter
	Standing Skyline		
Configuration	**Experiment**	**Location**	**Chapter**
North Bend	Dry forest dynamics following cable and ground-based logging	Wielangta 42a	2
	Vegetation regeneration on draglines	Wielangta 42a	3
	Live Skyline		
Configuration	**Experiment**	**Location**	**Chapter**
Gravity Return (Shotgun)	Cable regrowth thinning	Jungle 3a	3
Running skyline with mechanical grapple	Wet forest dynamics following cable logging and burning	Pearce 04	2
"	*Dicksonia antarctica* response to cable logging and burning	Pearce 04	5
"	Epiphytic species dynamics following cable logging and burning	Pearce 04	6
"	Disturbance to streamside reserve edges by cable logging and burning	Pearce 04 Blue 09	4
"	Disturbance to a streamside reserve by cable logging	Wallaby Creek	4
"	Wet forest dynamics following cable logging and burning	Blue 09	2
"	Vegetation regeneration on draglines	Loatta 209, 217	3
Slack line	Disturbance to streamside reserves by cable logging	Roses Tier	4
"	*Dicksonia antarctica* response to cable logging of streamside reserves	Roses Tier	5
Highlead	Chronosequence surveys of vascular flora following cable logging and burning	Florentine/ Styx River Valleys	2
"	Chronosequence surveys of epiphytic species following cable logging and burning	Florentine/ Styx River Valleys	6
	ground-based		
Tracked feller buncher, and rubber tired grapple skidder	Regrowth thinning	Jungle 3a	3
Bulldozers, grapple skidders and rubber tired grapple skidder	Chronosequence surveys of vascular flora following ground-based logging and burning	Southern Forests	2
	Dry forest dynamics following cable logging	Wielangta 42a	2

Skyline Systems

A skyline system is one where the block or carriage rides on a skyline (Figures 1.4, 1.6, 1.8, 1.10 and 1.12). A skyline is a wire rope hung between two or more supports. The skyline provides vertical lift to the log so that the choked (tethered) end of the log is carried free of the ground, or occasionally the log is fully suspended in the air.

The distance between the two supports that suspend the skyline is the span. Occasionally the span is supported by intermediate supports at some position along its length, usually determined by the slope profile, and particularly if the slope is convex (e.g Loatta 209, 217 see Chapter 3). The key feature of any skyline system is deflection. Deflection is the vertical distance between a chord (an imaginary line drawn between the two supports suspending the skyline) and the skyline at mid-span. A decrease in deflection will put additional tension on the skyline, reducing its load carrying capacity. Increasing deflection (i.e. lowering the skyline) adds to the load carrying capacity by reducing tension, however the logs may not clear ground obstacles adequately and additional soil and vegetation disturbance will result.

The choice of skyline system depends on the topography, the type of carriage (e.g. shotgun/motorized) and the amount of lateral yarding (sideways movement of logs) required. Skyline systems can be either standing or live depending on whether the skyline length can readily change during hauling. On a live skyline, one end of the skyline is attached to one of the yarders' winch drums so it can be raised or lowered as required to assist log extraction.

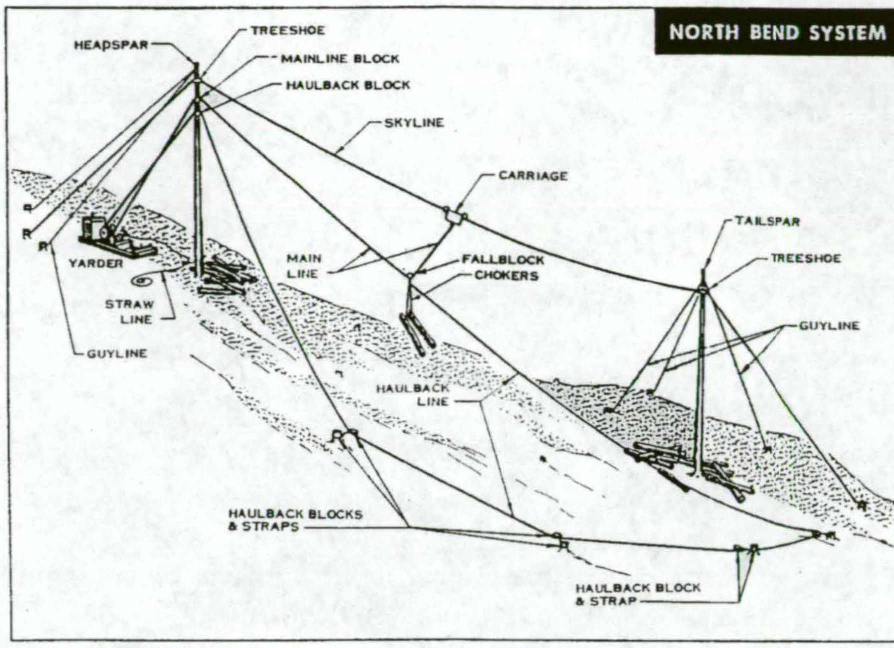


Figure 1.4 *Standing Skyline - North Bend (from Studier and Binkley 1974)*

Figure 1.5 *The experimental area Wielangta 42a was logged with a standing skyline using a North Bend system between April and August 1992. The arrangement of the skyline system was varied between settings and landings occupied by the yarder. The landings and yarding directions are provided in Figure 3.1.*

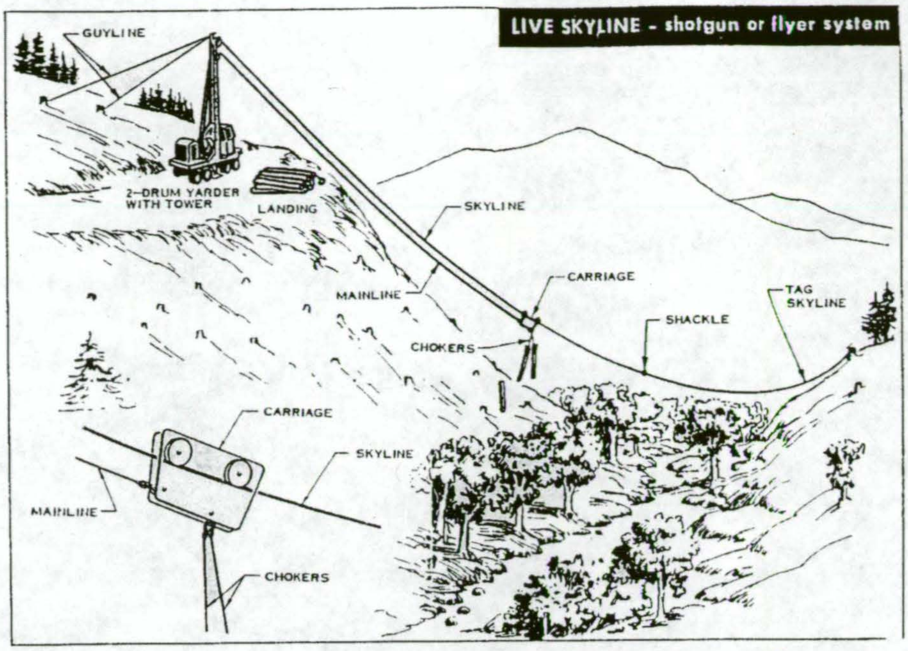


Figure 1.6 *Live Skyline – Gravity return or shotgun system (from Studier and Binkley 1974)*

Figure 1.7 *Cable thinning using a live skyline (Figure 1.6) and shotgun carriage system at the Jungle 3 experimental area in 29 year old* Eucalyptus regnans *regrowth, 1991. This skyline system was applied to the thinning experiment according to the plan in Figure 3.5*

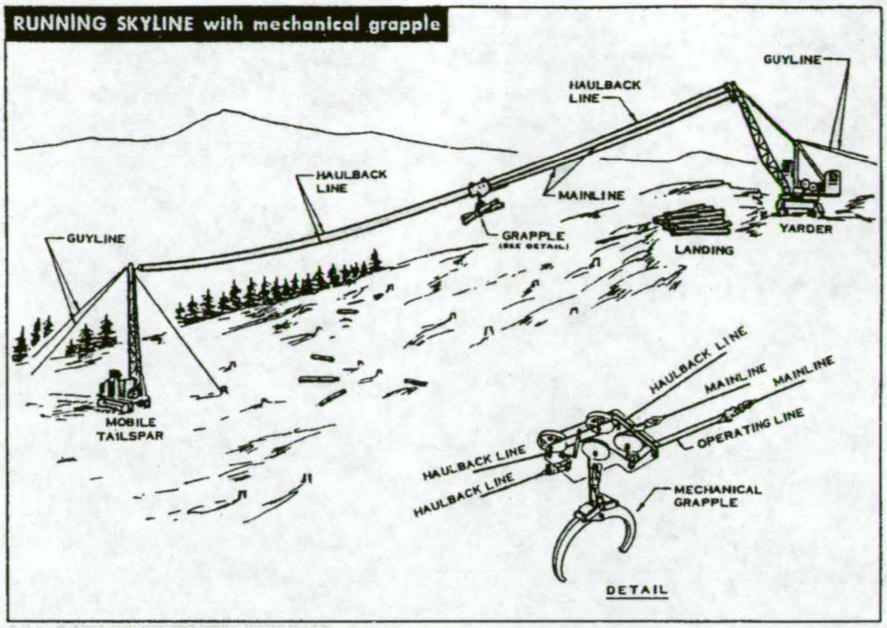


Figure 1.8 *Live Skyline – swing yarder with mechanical grapple (from Studier and Binkley 1974)*

Figure 1.9 *Live Skyline – Madill 122 Swing Yarder with mechanical grapple working at Blue 09 experimental area, March 1992. The machine is yarding logs down slope and landing them continuously along the road. The same system and machine was used at the Pearce 04 experimental area.*

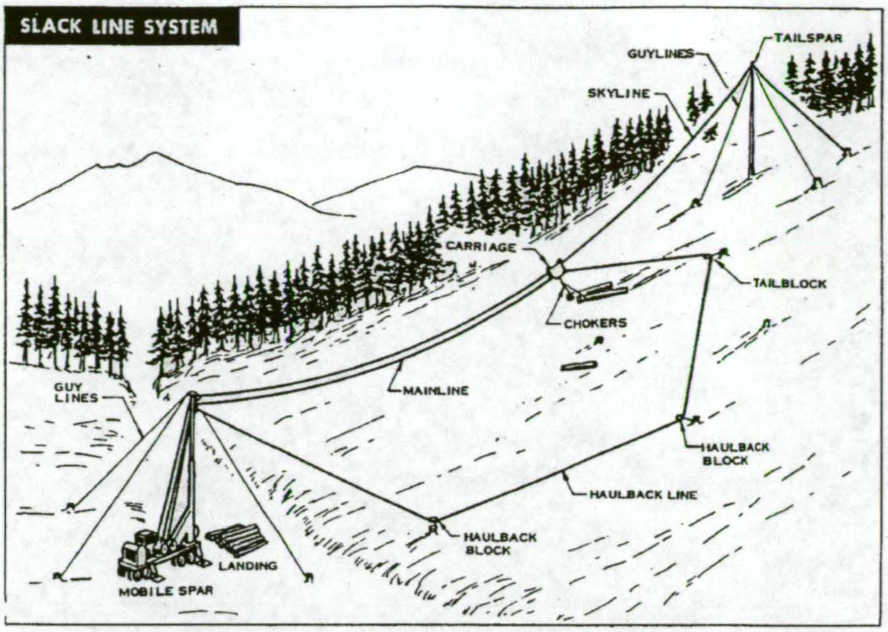


Figure 1.10 *Live Skyline – slackline system (from Studier and Binkley 1974)*

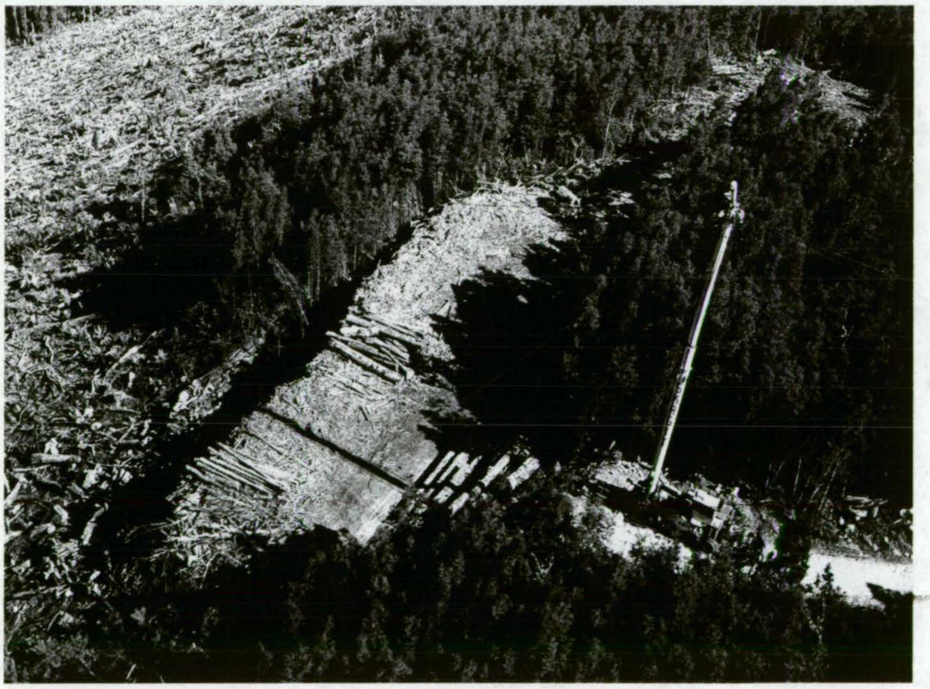


Figure 1.11 *Live Skyline – slackline system in use at Roses Tier 142a, north eastern Tasmania 1991 (Skagitt Bu 739 slackline yarder 110' tower, 8 guylines). This skyline system, and variations of it using both motorised and dropline carriages, was used at the Roses Tier experimental sites examining streamside reserve vegetation in Chapter 4.*

Highlead Systems

The term highlead refers to the position of the mainline lead block that is elevated above the ground by a spar (Figure 1.12). The mainline provides some upward lift to the choked end of the log above the ground, reducing drag and helping it avoid obstacles (such as boulders and cut stumps). In highlead systems vertical lift to the logs depends upon the height of the mainline block, the ground slope profile, the distance of the log from the spar and the breaking capacity of the haul back drum.

Highleading is reported in forest management records as being widely used for cable logging operations in the Florentine Valley of southern Tasmania from the 1960s onward. The precise nature of the cable logging system used is not however described, but it appears to have involved several Madill yarders and either fixed or mobile tall holds (see Figure 1.12). Similar cable logging systems were also used for cable logging relatively flat areas in the Florentine Valley bottoms from the 1960's. Log suspension would have been minimal due to the combination of large logs and poor deflection, and anecdotal reports by forest industry personnel suggest considerable soil disturbance could occur.

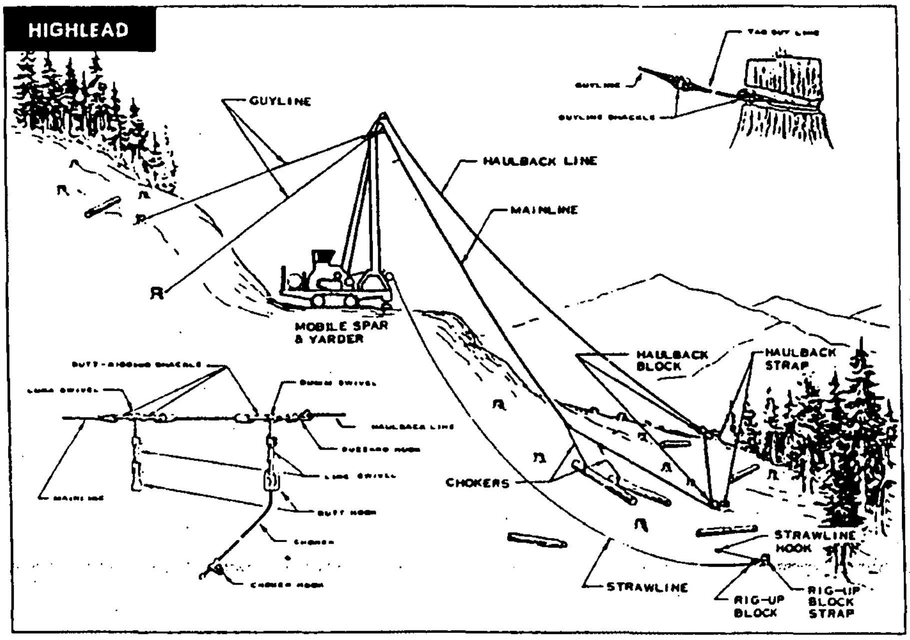


Figure 1.12 *Live Skyline – highlead system (from Studier and Binkley 1974)*

Ground-based logging systems on steep slopes in Tasmania

Mechanised ground-based logging has been used sporadically in Tasmania on steep slopes since the late 1960s, often in association with the logging of more gentle slopes within the one logging operation. The primary areas are the north east Tasmania (Roses Tier, Scottsdale) and the southern forests (Geeveston, Arve, Picton). As an example, the proportion of steep land logged using ground-based systems in the southern forests over a thirty year period averaged only 11.3% (Figure 1.13).

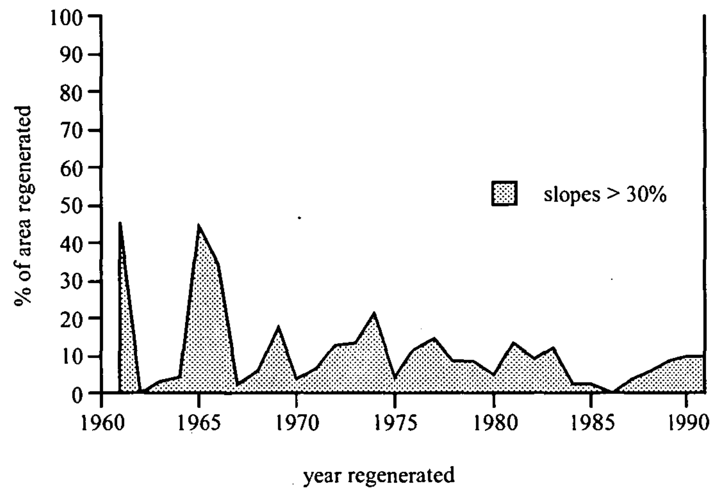


Figure 1.13 *The proportion of steep slopes (>30%) relative to shallow slopes (<30%) logged in the Tasmanian Southern Forests between 1961 and 1991, source: Forestry Tasmania.*

Ground-based logging systems over this period involve three related techniques; bulldozers of the D6 and D7 type for log snigging from the 1960's through to the 1970's, traxcavator bulldozers for snigging and loading from the early 1970's through to the mid 1980's and finally, the introduction of excavator loaders and rubber tired skidders in the mid-late 1980's. Bulldozers pushed and winched logs and used log ramps for loading trucks. Logs were snigged normally by dragging them down or across slopes or at times upslope. Snigging distances of up to 800 m. were common, which is long by current standards. With logs being winched or dragged over the surface, gouging (defined as the removal of the mineral soil surface - van Rees *et al.* 2001) was likely to occur, even with the log being raised at one end above the ground using a wire rope (choker). Traxcavators, with their rigid frames, were less manoeuvrable than bulldozers and grouser lugs attached to their tracks to increase traction resulted in increased ground compaction and soil mixing. Traxcavators were fitted with forks for log loading and were responsible for the gradual replacement of the log ramps used for loading by bulldozers. Modern excavator loaders, with their greater manoeuvrability, wider tracks and swivelling carriage resulted in less soil disturbance. Log snigging still relies primarily on bulldozers. The use of rubber-tyred skidders has been limited and restricted primarily to the period since 1988. Log grapples attached to bulldozers (similar to Figure 1.8) became more common since the 1980's, replacing chokers for snigging logs. Grapples increased lift to logs during snigging compared to the chokers, and were fitted to both bulldozers and rubber-tired skidders. Use of these newer technologies, coinciding with the introduction of the Tasmanian Forest Practices Code in 1987, resulted in reduced soil disturbance, both in extent and intensity.

Silvicultural systems applied to steep slopes in Tasmania

Silviculture can be defined as the theory and practice of managing forest establishment, composition, and growth, to achieve specified objectives (NRE 1996). Cable logging is predominantly associated with clearfell or clearcut logging, however it is also used (though less commonly) in shelterwood systems (Tesch *et al.* 1986), strip felling systems (Saniga 1988), group selection systems (Shaffer *et al.* 1993), and coppice systems (Verani 1995). The Tasmanian experience to date has been limited to applying cable logging to a clearfell silvicultural regime, with the exception of regrowth cable thinning and the retention of wedge shaped portions of rainforest between settings on some cable logging operations in north eastern Tasmania from the 1990's. The standard silvicultural technique for the regeneration of the eucalypt overstorey of the cable logged wet and mixed forests in Tasmania examined in this thesis was to clearfell, burn and broadcast sow on the ash-bed (Pryor 1960, Renbuss *et al.* 1973, Chambers and Attiwill 1994) with eucalypt seed, following the techniques initially described by Gilbert and Cunningham (1972). Clearfelling is also the most operationally efficient logging method as any retained stems (apart from those in the streamside reserves) are likely to interfere with the set up and operation of the cable system.

1.5 Differences between environmental effects of cable and ground-based logging of steep slopes

Cable and ground-based logging systems vary in both their localised and broader spatial effects on vegetation. Ground-based systems disturb a greater proportion of the logging area with landings and trails, and do so with a greater intensity of soil displacement and compaction, and litter disturbance (Table 1.2). The difference relate to the method used to snig logs from the stump to the landing, and the soil disturbance resulting from this practice. Ground-based logging, more than cable logging, results in a more variable pattern of ground disturbance, with areas of intense disturbance separated by areas of no disturbance. Areas of vegetation undisturbed by snigging, referred to as understorey islands by Ough and Murphy (1998) and Hickey *et al.* (1998, 2001) may assist the persistence of species that rely on vegetative regeneration post-logging and burning. The vegetation of clearfelled cable logged areas will usually be uniformly flattened, either by log falling, log yarding, or the sideways movement of the cable skyline or mainline during log hauling.

Table 1.3 *General comparison of cable and ground-based logging systems. Attributes at the planning stage that can impact directly or indirectly vegetation are indicated.*

VARIABLE	STANDING SKYLINE	CABLE SYSTEM LIVE SKYLINE - GRAPPLE, SWING	LIVE SKYLINE - HIGHLEAD	GROUND-BASED SKIDDER
Coupe size	≤50ha	≤50ha	≤50ha	<100ha
Coupe shape	Regular	Regular	Regular	Variable
Productive yarding distance	500m	150-200m	200-300m	<300m
Types of settings	Variable, commonly fan shaped	Tail hold perpendicular to yarder	As for skyline	Variable
Preferred yarding direction	Uphill or downhill	Uphill, moderate downhill OK	Uphill, moderate downhill OK	Downhill
Road intensity	Low	Moderate	Moderate	Moderate to high
Road quality	Steeper side cuts, adequate cross drainage necessary	As for skyline	As for skyline	Not a major consideration, covered by Code
Landing position	Centralised on knolls & ridges	Continuous	Centralised on knolls & ridges	Variable, usually drier areas
Landing size	Relatively large	Smaller or absent when logs are landed on road	Variable	Usually smaller
Landing number/ha	1 per 6.4 ha (Miller and Sirois 1986)	-	-	1 per 0.6ha (Miller and Sirois 1986)
Landing % coupe area	4.1% (Miller and Sirois 1986)	-	-	Greater proportion 10% (Incerti *et al.* 1987)
Shape of slope	Concave or even, important for deflection	Uniform slope for yarder visibility required	Concave or even	Even
Preferred slope	Steep to very steep, minimum slope of 20% with gravity systems	Moderate slopes	Up to moderate slopes	Up to 30%
Downhill yarding ability	Good	Reasonable on moderate slopes	Reasonable on moderate slopes	Up to 30%
Soil strength	Will determine holding capacity of tailblock anchors	Not relevant where running tailholds are used	Will determine holding capacity of tailblock anchors	Relevant on some substrates e.g. granites

Table 1.3 (cont.) *General comparison of cable and ground-based logging systems. Attributes at the logging stage that can impact directly or indirectly vegetation are indicated.*

VARIABLE	STANDING SKYLINE	CABLE SYSTEM LIVE SKYLINE - GRAPPLE, SWING	LIVE SKYLINE - HIGHLEAD	GROUND-BASED SKIDDER
Wet soil conditions	OK	OK	OK	Increased soil disturbance
Terrain conditions	Steep to very steep, broken, rocks and gullies OK	Generally even slopes, rocks OK, up to moderate slopes	Up to moderate slopes, broken terrain, rocks generally OK	Even slopes, dry, up to 30% slope
Soil disturbance	Light to moderate (full or partial suspension)	Light	Moderate, logs 'dragging'	Moderate to severe (rutting and puddling in wet weather)
% of surface disturbed	0.45% (J. Cunningham pers. comm. 1993), 1.75% (Basari *et al.* 1997) 11% (Laffan *et al.* 2001)	3% (Patric and Gorman 1978)	10% (Watt and Krag 1988) 1.4-1.5% (Bennett 1997)	19.6-20.3% (Watt and Krag 1988) 60-80% (Rab 1996) 8-20% (Roberts and Xhu 1998) 23% (Wronski 1984) 25% (Ryan *et al.* 1992) 17-61% (Smith *et al.* 1988)
% soil compacted	3.8% (Miller and Sirois 1986)	-	9.1 (Sarre 1993)	9.7% (Miller and Sirois 1986) 26.9 (Dryness 1965) 20% (Lacey *et al.* 1994) 23% (Wronski 1984) 11-30% (Murphy 1982)
% litter disturbance	0.49% (J. Cunningham pers. comm. 1993)	-	-	65% (Ryan *et al.* 1982) 40-60% (van Rees *et al.* 2001)
% of coupe area as spur roads	2.6%	-	-	3.5% (Miller and Sirois 1986)

Table 1.3 (cont.) *General comparison of cable and ground-based logging systems. Attributes at the logging stage that can impact directly or indirectly vegetation are indicated.*

VARIABLE	STANDING SKYLINE	CABLE SYSTEM LIVE SKYLINE – GRAPPLE, SWING	LIVE SKYLINE - HIGHLEAD	GROUND-BASED SKIDDER
% of coupe areas cable corridors/skid trails	9.2% (Miller and Sirois 1986) 7% (Laffan *et al.* 2001)	-	-	10% (Incerti *et al.* 1987) 18-25% (Rab 1994, 1996) 21.4% (Miller and Sirois 1986) 28% (Dryness 1965) 9-39% (Williamson 1990) 38% (J. Cunningham pers. comm. 1993) 20% (Dykstra *et al.* 2000) 11-30% (Murphy 1982) 25% (Ryan *et al.* 1982) 14% (van Rees *et al.* 2001)
Good deflection	Essential	Essential for visibility	Not critical	Not applicable
Tail holds	Essential	Mobile tailspar	Essential	Not applicable
Damage to retained trees	23.6% (Sarre 1993)	8-10% (Fairweather 1991)	56.7 (Sarre 1993)	45.5% (Sarre 1993)
" thinning	<5% (J. Cunningham pers. comm. 1993) minimal (Han *et al.* 2000)		10-12% and 34-37% (Bennett 1997)	severe (Han *et al.* 2000)
Retention of streamside reserve vegetation	Logs can be fully suspended (but damage to trees in corridors)	-	-	Generally achievable
Logging away from streamside reserve	OK	OK, as continuous landing	OK	(van Rees *et al.* 2001)
Piece size variation	Less difficulty where choker lines used	As for grapple	Can reduce productivity when a grapple is used	Effect varies with method used to attach log
Residue volume per unit area	volume greater than ground-based logging (Woodward and Henry 1985)	As for skyline	As for skyline	10-15% of area (Rosen *et al.* 1987)
Residue distribution throughout coupe area	More continuous	-	Similar to ground-based (Dryness 1965) varies with slope (Hall 1991)	Patchy fuel distribution, varies with snigging patterns

Table 1.3 (cont.) *General comparison of cable and ground-based logging systems. Silvicultural attributes that can impact directly or indirectly vegetation are indicated.*

VARIABLE	STANDING SKYLINE	CABLE SYSTEM LIVE SKYLINE – GRAPPLE, SWING	LIVE SKYLINE - HIGHLEAD	GROUND-BASED SKIDDER
		Silviculture		
Silvicultural system used	Clearfell to selective logging (but high cost of latter)	Clearfell only	Clearfell	Flexible, clearfelling to 600 m
Tree retention	Thinning using a gravity return live skyline is possible	No	No	Yes
Regeneration burn season	Can be burnt later in season as fuels are continuous and on steeper slopes, a better moisture differential between cut or uncut forest is present.	As for skyline	As for skyline	Varies with silvicultural system used
Regeneration burn patchiness	Less patchy due to continuous fuels	As for skyline	As for skyline	Patchy burn possible
Cleared fire breaks	Uncommon	Running tailholds often re-used	Occasionally used	common

Areas not disturbed are usually restricted to portions of the logging area where full or partial log suspension was achieved, for example within streamside reserves.

As a general rule, cable logging results in less soil disturbance on steep slopes than ground-based logging, both in terms of area disturbed, intensity (Dryness 1965, Smith and Wass 1976, Patric 1980, Miller and Sirois 1986, Watt and Krag 1988, Sarre 1993, McMahon 1995, Hans *et al.* 2000, and reviews by Sauder *et al.* 1987, Lousier 1990 and Krag *et al.* 1991) and the amount of organic matter displaced (Minore and Weatherly 1988). The area of soil disturbed during ground-based logging is typically twice that disturbed by cable logging (Watt and Krag 1988), although the effects of soil disturbance on stream habitat due to logging method may be masked by adequate buffers (Davies and Nelson 1994). The process of soil being displaced and compacted during cable logging is poorly known compared to ground-based logging, where most of the soil compaction from yarding occurs in the first 1-3 machine passes (Amaranthus and Steinfeld 1997, Wronski 1984, Williamson and Neilson 2000). However, with cable systems, there are no published studies examining role of the number of cable passes on the creation of draglines or soil and litter disturbance.

Cable logging systems are not uniform in their site impacts. Studies of soil disturbance on steep slopes following cable logging using either skyline (Dryness 1967), highlead (Krag *et al.* 1986, Dryness 1965, Borhan *et al.* 1990) or grapple systems (Schwab and Watt 1981) indicate that skyline systems result in the least soil disturbance. A moderate amount of soil disturbance occurs with the highlead system, and the greatest soil disturbance occurs with a grapple system (Henderson 1993). Variation between these cable logging systems in resulting soil disturbance are primarily a consequence of the way logs are yarded from the stump to the landing. Draglines form when logs are yarded (i.e. hauled) from a stump to a landing and the log is partially suspended and gouges a path or dragline in the soil. Draglines can be sub-divided into two forms, plowing and rutting/gouging (Cummins 1982, Figure 1.14). The term draglines is used in this thesis, although this type of soil disturbance can be alternately referred to as cable ways, cable runs, cable roads (Madden 1991), butt gouges (Smith and Wass 1976), cable corridors (Miller and Sirois 1986) and haul paths (Fransen 1998). Uphill yarded logs tend to form a narrow dragline, whereas downhill yarded logs tend to zig-zag as the weight of the log is not supported by the choker and wider draglines form. Logs yarded parallel to the slope have a tendency to move away from the cable path, resulting in a wider dragline (Sauder and Wellburn 1987).

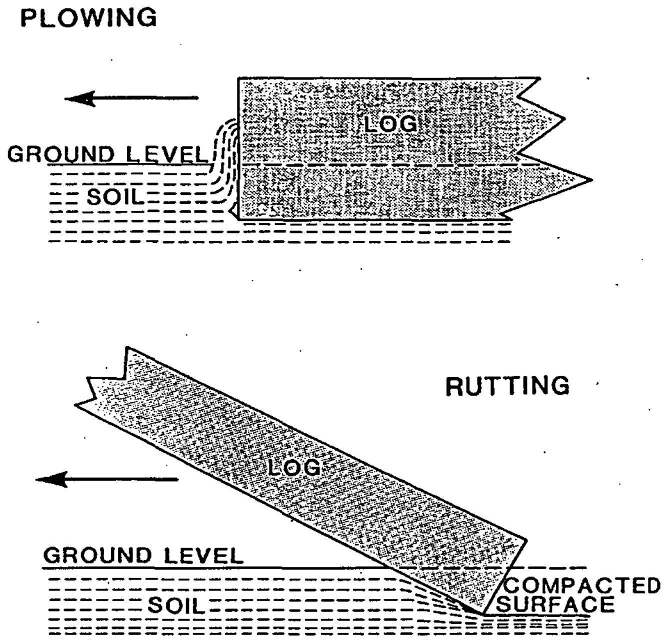


Figure 1.14 *Plowing and rutting of logs during cable logging results in dragline or butt gouges in the soil surface (from Cummins 1982).*

A light scraping of the ground surface is considered inevitable during cable log yarding because one end of the log is usually in contact with the ground at all times (Krag *et al.* 1986). Payload analyses suggest draglines are most likely on slope mid-span ridges (McMahon 1995) and convex slopes (Krag *et al.* 1986) (see Figure 1.16). Draglines typically form a radiating pattern from centralised upslope landings on ridge tops (e.g Figure 1.15) where the yarder is stationary.

Draglines can contribute to mass wasting (Sauder and Wellburn 1987), soil depletion (Fransen 1998) and retard vegetation re-establishment (Smith and Wass 1976, Fransen 1998). Draglines less than 5 cm deep restricted to the litter layer are considered shallow disturbance, while deeper disturbance occurs where a compacted base forms in the dragline that is capable of accumulating and redirecting water into adjacent areas (Sauder and Wellburn 1987). The soils of draglines differ from adjacent soils by their reduced hydraulic conductivity and increased bulk density (Purser and Cundy 1992), reduced organic matter (Guo *et al.* 1997, Laffan *et al.* 2001) and a propensity to actively erode in the short term (Fransen 1998). The vegetation of draglines, compared to adjacent logged vegetation, is typically reduced in vegetation cover, at least in the short term (Fransen 1998), although differences between draglines and adjacent vegetation in attributes such as seedling growth and vegetation composition will decrease with time (Smith and Wass 1976).

Figure 1.16 *Typical pattern of radiating draglines from a central landing, Branches Creek, northern Tasmania, May 1993.*

1.6 Experimental approaches

The principal experimental approaches adopted in this thesis were chronosequence surveys and long term permanent plot or individual plant monitoring. The strengths and weaknesses of these approaches are presented in Tables 1.4 and 1.5. Alternate terms for the chronosequence method include retrospective surveys (Davis 1989), the dynamic approach (Austin 1977), spare-for-time substitution (Pickett 1989) and cross-section analysis. Long terms studies (Pickett 1989) are alternately known as longitudinal studies (Likens 1989) and the static approach (Austin 1977). Chronosequence studies aim to extrapolate a temporal trend from a suite of samples of different ages where the sites are chosen to be as equivalent as possible in all factors which may influence vegetation composition, so that differences between them can be largely attributed to age. It assumes the past (ie. abiotic and biotic factors influencing dispersal, establishment and growth and any priority effects related to previous ecological states of the system) have been the same on the sites examined in the chronosequence. In this case the past also includes the type of harvesting and regeneration practices applied. Chronosequence studies can also generate hypotheses which can be tested by long-term monitoring or ecological experimentation. They are relatively common in forest disturbance studies, providing an ability to rapidly predict plant community responses to disturbance and addressing the immediacy of forest management information requirements (McAninch and Stayer 1989) .

Table 1.4 *Long term monitoring studies, weaknesses and strengths*

Strengths	Weaknesses
• probably the definitive method for studying succession	• studying only one site could result in effects peculiar to that site being recorded, limiting the ability to extrapolate
• ability to expose mechanisms controlling successional dynamics	• outcome of monitoring can be time dependent
• ability to detect autogenic phenomena (with contiguous quadrats)	• lack of clear protocols established for subsequent researchers to trace the initial procedures and measurements,
• ability to distinguish seasonal and short-term fluctuations from long-term trends (with sampling at sufficient intervals)	• subtle differences in research methodology over time can significantly affect the outcomes
• when replicated at different locations, local chance events can be recognised and spatial heterogeneity in harvesting effects measure	• spatial heterogeneity in treatment areas may effect ability to attribute compositional changes to the treatment
• ability to describe seasonal effects in the vegetation response	• studies extending over decades may be impractical or even impossible because of random disturbances

Table 1.5 *Chronosequence analyses, weaknesses and strengths*

Strengths	Weaknesses
• ability to expose general or qualitative trends, regional averages or coarse scale patterns	• fails where the unrecognised effects of the past were of a large magnitude or where controlling the variables such as climate, soils or certain historical factors was not possible across the study
• can be applied with confidence to sites or regions with a history of initial and post-disturbance conditions	• fails where site history determination has been poor or where some of the independent variables are too coarse to control compositional variation which may obscure the experimental response.
• ability to document variability among replicate sites and expose spatial variation	• sufficient replicate sites may be a limiting factor
• works best in systems acknowledged to have strong successional dynamics	• choosing sites to represent the terminus of succession is difficult
• ability to generate hypotheses by correlating vegetation patterns with environmental or structural features of the forest.	• historical data may not be adequate to determine if past harvesting practices were comparable to modern clearfelling
	• sampling intervals are frequently too large to allow trends to be predicted with any known variance
	• the non-random nature of timber harvesting activities makes it difficult or impossible to randomly sample throughout the landscape

Long-term studies, as studies of succession, have the ability to expose mechanisms controlling successional dynamics instead of relying on the correlative approach of chronosequence studies. They also describe finer scale patterns in compositional change. Both of the above approaches have their inherent strengths and weaknesses when applied to field based logging experiments (Peacock and Duncan 1993, Table 1.4, 1.5). Chronosequence studies can provide the general background necessary for interpreting and extending the time frame of long-term studies (Davis 1989), while long term studies, being time consuming to establish, document and design, can provide a particularly powerful means of testing hypotheses (generated from chronosequence studies) concerning logging effects on vegetation. Complementary studies using both chronosequence and long term approaches are uncommon in the forest ecology (e.g. the sand mining regeneration studies of Twigg *et al.* 1989) and have been established in this thesis. The validity of the chronosequences described in this thesis are tested with data from long term monitoring plots, established at the same time, in similar environments, and with the same logging systems.

1.7 Aims of the thesis

The primary objective of this thesis is to examine the response of vegetation cover, richness and composition to the choice of logging method, cable or ground-based, on steep slopes.

The thesis is divided into five experimental chapters.

Chapter 2 examines two forest types; wet eucalypt forest and dry shrubby forest and analyses their response to cable and ground-based logging. The analyses presented are restricted to woody and herbaceous plant species. Variation in vegetative responses are related to logging method and initial forest type using both chronosequence and long term monitoring approaches. The hypothesis tested is that the different nature of ground-based and cable logging impacts on the physical environment will result in differing vegetation responses.

Chapter 3 examines the effect of logging method on soil disturbance. Cable logging effects vegetation through soil disturbance in draglines and light availability following thinning. Soil and vegetation disturbance can either be lessened or increased by the cable logging method employed.

Chapter 4 examines the effects of cable logging method and intensity on the short term health and composition of streamside reserve vegetation. While streamside reserves are normally delineated within production forests in order to protect streamside habitats their ability to also provide for ecologically stable vegetation is questioned. Streamside vegetation provides the interface between the terrestrial and aquatic environment and contains a range of species and growth forms susceptible to logging induced micro-climatic change. Streamside vegetation structure can be significantly altered by some cable logging systems.

Chapter 5 examines the response of *D. antarctica* (Soft tree-fern) to cable and ground-based logging. *D. antarctica* has a poor regenerative response after ground-based logging and burning, attributed by Ough and Murphy (1997) to mechanical disturbance from logging. *D. antarctica* is an important species in wet forest environments as it acts as a host species for a range of vascular epiphytes. Its logging response is hypothesised to have a significant role in the ability of late successional specialist species (dependent on its substrate for germination) to recolonise regrowth vegetation following logging and burning.

Chapter 6 addresses the regenerative response of vascular epiphytes to cable and ground-based logging. The limited existing data indicates that vascular epiphytes are uncommon in regrowth forest, and less common in regrowth forest originating from logging than from wildfire (Hickey 1994, Ough 2001). Observations from overseas studies suggest that forests regenerating following cable and ground-based logging differ in the proportion of woody debris or ecological legacies from the previous forest stand. Epiphytes are known from studies in boreal forests and tropical ecosystems to require a range of substrates, including woody debris and bark surfaces, as host substrates. The logging method employed may therefore restrict the ability of the late successional specialists to recolonise sites post-logging and burning.

Chapter 7 assesses the results of the five experimental chapters in terms of the contemporary understanding of post-logging vegetation dynamics, and the management of vegetation within forests subject to intensive logging regimes.

Chapter 2 Effects of cable and ground-based logging on woody and herbaceous species in Tasmanian forests

2.1 Introduction

Cable logging has been used for decades in north western USA, Canada, Alaska, New Zealand and Europe under a wide variety of operational scales, yet the global literature on its effects on vegetation is limited, compared to the literature available on its effect on soils (eg. Watt and Krag 1988, McMahon 1995, Davis and Nelson 1993, Laffan *et al.* 2001) and damage to retained stems (eg. Fairweather 1991, Sarre 1993, Han *et al.* 2000). Compounding the general absence of studies of cable logging effects on vegetation, the relatively extensive literature initiated in the 1960's (eg. Cremer and Mount 1965, Young *et al.* 1967) on vegetation dynamics following ground-based logging does not address the comparative response of vegetation to cable and ground-based logging.

The comparative review of cable and ground-based logging techniques (Chapter 1) indicated they differ principally in the degree of soil disturbance and the distribution of woody debris across the logging area, assuming all other factors such as wood utilisation standards and regeneration treatments are equal. These differences have been the focus of most of the existing vegetation studies of cable logging. Cable logged and burnt areas are also less effected by soil compaction and humus loss than similar areas logged with ground-based machines where debris is piled prior to burning (Minore and Weatherly 1988). Cable logged sites can also exhibit greater seedling growth potential (Minore and Weatherly 1990).

Studies that compare the effects on vegetation of cable and ground-based logging are scarce (Tappeiner *et al.* 1991). However, their scarcity may be exaggerated by authors from Canada and North America only describing the logging method used as clearcut or clearfell and omitting that it was achieved by cable or ground-based methods. Apart from those studies concerned with draglines (Fransen 1998, Smith and Wass 1976), only Pinard *et al.* (1998) and Woodward (1986) have directly compared the vegetation response of similar environments logged by either cable or ground-based methods. Pinard *et al.* (1998) did so by planting rainforest tree seedlings into a logged forest and concluded the improved growth rates on cable (high-lead) sites 13 years later was the result of less soil compaction compared to the ground-based logged sites. A similar conclusion, ie. that soil compaction from ground-based logging can retard the post-logging vegetation response, was reached by Jones *et al.* (2000) in a comparative study of helicopter and ground-based logging. Compared to ground-based systems, cable logging can minimise damage to advanced regeneration and result in less site invasion by disturbance adapted understorey and herbaceous species (Carleton and MacLellan 1994, Gottfried 1987, Carleton 1988, Pominville 1993). On the other hand, a similar study of seedling mortality, tree damage and recolonisation rates following high-lead cable and ground-based logging gave opposing results (Borhan *et al.* 1990). An examination of the short term (<10 year old) response of understorey vegetation in a relatively species poor forest of coast redwood (*Sequoia sempervirens*) in California indicated that the regenerating understorey following cable logging had almost 50% more cover compared to ground-based logging, and that the vegetation resulting from either logging method began to converge in these characteristics on moist sites seven years post-logging (Woodward 1986). Differences in vegetation response attributed to logging method, whether they are measured in terms of the re-establishment of vegetation cover (Woodward 1986), or of regeneration damaged during log yarding (Pothier 1996) can become less apparent five to seven years after treatment.

Existing studies of the response of vegetation in Australia to logging have only examined ground-based sites, and of those, several have considered mechanical disturbance to the soil and vegetation as being central to their interpretation of the resulting post-logging patterns (Ough and Murphy 1997, 1999). Not only is there a significant gap in the literature examining floristic change to cable logging, there is no published study comparing floristic change to cable and ground-based logging. The current chapter addresses the null hypothesis that there is no difference in the response of woody and herbaceous species in Tasmanian forests to cable or ground-based logging.

2.2 Methods

Two studies examining the response of woody and herbaceous species to cable and ground-based logging are presented in this chapter;

1. a chronosequence analysis of the response of woody and herbaceous species to cable and ground-based logging in wet forest in southern and central Tasmania
2. a long term monitoring study using permanent quadrats examining (a) the impacts of cable logging on woody and herbaceous species in wet forests in the upper Florentine Valley of central Tasmania, and (b) the impacts of cable and ground-based logging on woody and herbaceous species in dry forest at Wielangta in eastern Tasmania

The environments examined in this chapter are steep slopes in Tasmania logged by either cable or ground-based methods in which wet forest (Kirkpatrick *et al.* 1988), or dry forest (Duncan and Brown 1985) are present. This chapter presents research on woody and herbaceous species. Studies in later chapters using these experiments present data on *Dicksonia antarctica* and vascular epiphytes.

Experiment 1: Chronosequence analysis comparing the response of woody and herbaceous species to cable and ground-based logging.

Study Areas

The selection of field study locations is important to a study of this kind. A number of key factors enhanced the overall research design: First, as a member of a forest agency, access to both industry and government forest records and private and government forested areas was possible. Secondly, the forest agencies Geographic Information System (GIS) was used for survey stratification, quadrat selection and mapping. Thirdly, access to current logging operations was possible including an ability to ensure a consistent application of the cable and ground-based experimental logging treatments. Fourthly, Government and industry support enabled the field research to be supported with labour and aircraft hire for large scale aerial photography. Finally, the implementation of this field research was made considerably easier by the authors experience with similar studies elsewhere (Mueck and Peacock 1992, Peacock 1995).

The chronosequence study involved two regions, one forest type and two logging methods, cable or ground-based. Two regions were chosen to conduct the chronosequence studies were the Florentine, Styx, and Tyenna Valleys in central southern Tasmania and the Arve Valley in southern Tasmania (Figure 2.1). The

Florentine Styx, and Tyenna Valleys have one of the longest continuous histories of cable and steep country logging in Tasmania (and therefore Australia) and represents one of the most suitable areas to carry out such a study in south-eastern Australia. The Arve Valley is the most comparable environment to the Florentine Valley and differs by a lack of logging using contemporary cable logging technology. Seventeen coupes were surveyed for the cable logged wet forest chronosequence in the Florentine, Styx, and Tyenna Valleys, and eleven for the ground-based logged wet forest chronosequence in the Arve Valley in southern Tasmania (Table 2.1 and 2.2).

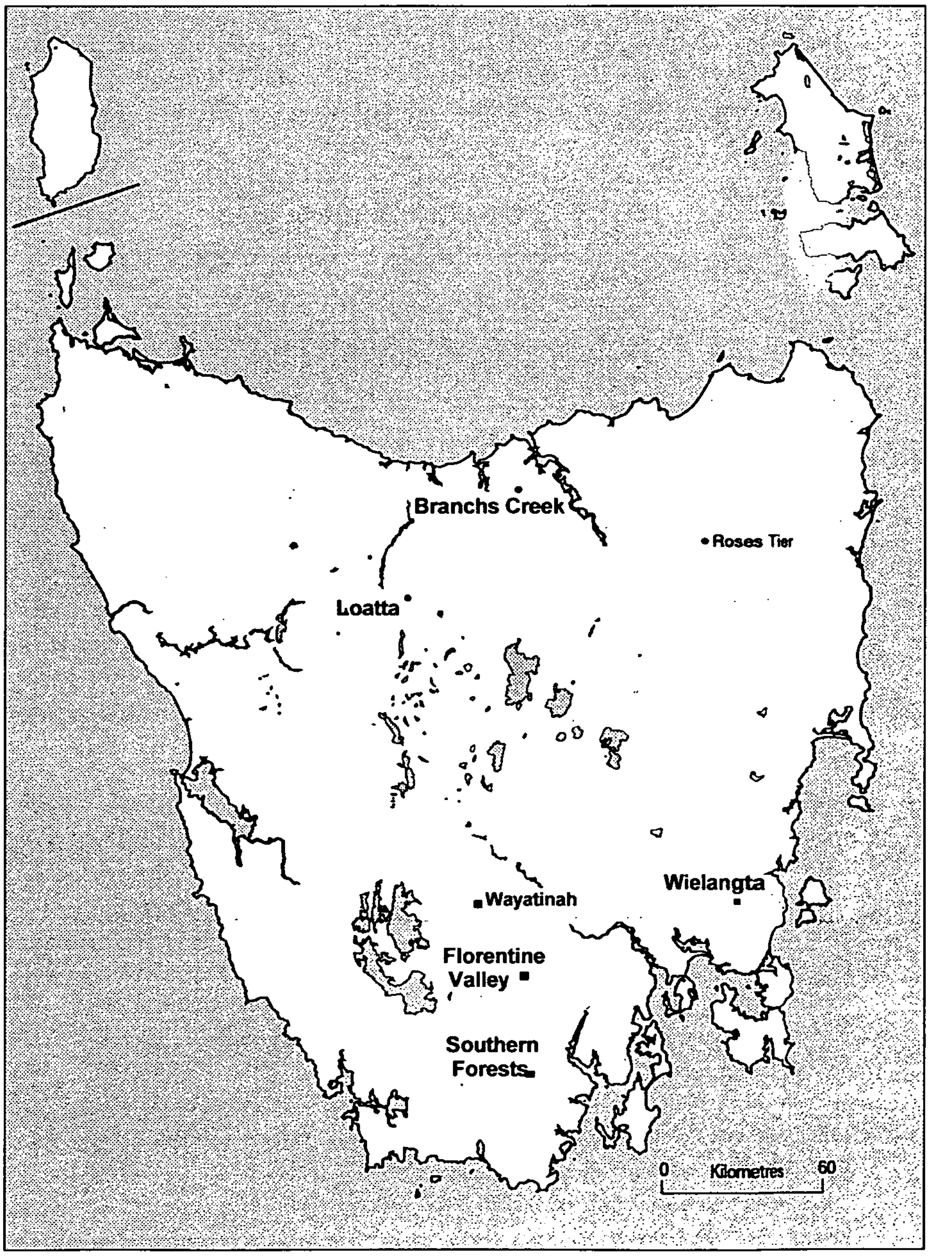


Figure 2.1 *The distribution of the study areas used in this thesis in Tasmania.*

Table 2.1 *Quadrat names, geocodes, pre-logging photo-interpreted forest types (after Stone 1998), age, slope and geology for the cable logged Florentine, Styx and Tyenna Valley wet forest chronosequence.*

coupe	easting	northing	Altitude (m)	ANM map sheet (1: 25 000)	pre-logging forest type	age at 1992	age category (years)	slope (degrees)	geology
Pearce 4a	457800	5303900	310	King William 12	E1b SER	5	5 to 10	26	dolerite
Lower Tiger 6 (E85)	452400	5298700	460	Huntley 4	E2a	7	5 to 10	15	dolerite/mudstone
Islet 2	464900	5293500	460	Ellendale 1	E1b	10	5 to 10	23	dolerite
Parker 23	454300	5292400	380	Huntley 4	E2b.T	14	11 to 20	18	mudstone
Parker 11	455500	5294900	440	King William 16	E1b	15	11 to 20	30	dolerite/mudstone
Parker 5	456200	5291700	540	Huntley 4	E2b	18	11 to 20	28	dolerite
Parker 25	455500	5296500	460	King William 16	E1b+	22	21 to 30	12	dolerite
Misery 26	460300	5295400	610	Ouse 13	E1a	25	21 to 30	18	dolerite
Misery 5	458300	5297100	460	King William 16	E2d S	29	21 to 30	10	dolerite
Westfield 87	459300	5283600	580	Ellendale 5	E1a	32	31 to 40	19	dolerite/mudstone
Snowy 8	468900	5256800	460	Styx 5	f'd E1c	34	31 to 40	23	mudstone
Westfield 11	456500	5284300	400	Huntley 8	f'd E1c. S.	38	31 to 40	9	dolerite/limestone
Risby's Basin 45/47a	469500	5263900	540	Styx 2	f'd E1 & 2c ST	47	41 +	27	limestone
Diogenes 1938/40	482500	5265400	260	Styx 3	E2a	54	41 +	35	dolerite
Pearce 04q29 (control)	458140	5303810	220	King William 12	E1b SER	100	mature	22	dolerite
Islet (control)	464900	5295400	600	Ouse 13	f'd E1a	120	mature	15	dolerite
Mannings Reserve (control)	455100	5294400	440	Huntley 4	E1a	150	mature	19	dolerite

Table 2.2 *Quadrat names, geocodes, pre-logging photo-interpreted forest types (after Stone 1998), age, slope and geology for the ground-based logged Arve Valley wet forest chronosequence.*

coupe	easting	northing	altitude	map sheet (1: 25 000)	pre-logging forest type[a]	age at 1992	age category (years)	slope (degrees)	geology
Franklin 14	489900	531200	360	Glen Huon	O/M E1d ER4d S	5	5 to 10	25	dolerite/mudstone
Picton 1	479900	531700	90	Picton	sev f/d E1d ER2.S	9	5 to 10	16	dolerite/mudstone
Warra 3	477600	532000	110	Weld	E1c MT	10	5 to 10	22	dolerite/mudstone
Arve 53	486600	527400	140	S. Forests	ER1 fire killed E1 f/d E1f	13	11 to 20	12	mudstone
Arve 82	481400	531300	120	Glen Huon	ER p/2 Eucalypt Pole Regrowth	16	11 to 20	17	mudstone
Arve 69E74	483900	530500	210	Glen Huon	E1D O/M/MM/S	18	11 to 20	23	mudstone
Arve 78	484100	530600	160	Glen Huon	E1D O/M/MM/S	21	21 to 30	15	mudstone
Arve 69E66	483400	527800	180	S. Forests	E1c/MM	26	21 to 30	33	mudstone
Arve 23	484700	521900	220	S. Forests	O/M E1B M.M.	30	21 to 30	20	mudstone
Bennetts (control)	486900	520400	260	S. Forests	E1c T	58[b]	mature	25	dolerite
Arve Spur Rd 3 (control)	478900	527100	230	Picton	ER4d S/1	78[b]	mature	23	dolerite

[a]old-growth PI types from 1947 photography - MM = myrtle and associated species, S = scrub, T = understorey species, M = myrtle

[b]presumed understorey age from last wildfire

Survey Design

The experimental design incorporates a regression analysis approach of the vegetation variables of interest (such as floristic composition and richness) against stand age. In the Florentine Valley, stand age was categorised as: 5-10, 11-20, 21-30, 31-40, 41+ years and mature forest, with approximately three stands in each category (Figure 2.2). The surveyed stands encompassed a range of ages within the category (eg. 5, 7 and 10). This was achieved for all categories, except in the 41+ year old category where only two stands per forest type could be surveyed, because of the relative scarcity of silvicultural regrowth of this age. In the Arve Valley chronosequence only the 5-10, 11-20, 21-30 years and mature forest categories were surveyed due to the absence of environmentally similar regrowth aged 30+ years (Figure 2.3).

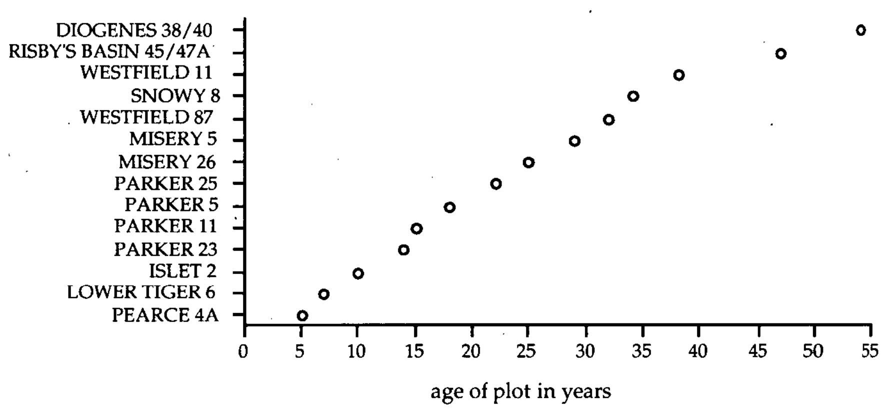


Figure 2.2 *Coupe names and ages at which they were surveyed as part of the wet forest cable logged chronosequence in the Florentine, Styx and Tyenna Valleys.*

Figure 2.3 *Coupe names and ages at which they were surveyed as part of the wet forest ground-based logged chronosequence in the Arve Valley.*

Quadrats were generally limited to altitudes of less than 600m to avoid stands where pre-logging forest canopy was dominated by *E. delegatensis*. The geology of the quadrats selected were mostly pre-carboniferous young rock types (Jurassic dolerite or Permian mudstones and sandstones Figure 2.4, 2.5). A few quadrats were underlain by Ordovician limestone. In most cases, the first two substrates formed a mantle over the limestone in the Florentine Valley.

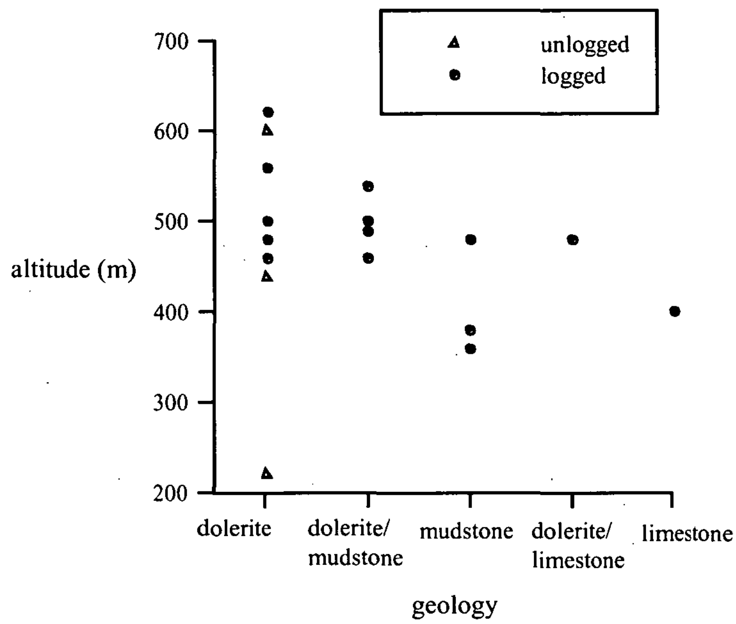


Figure 2.4 *Quadrat altitudes and geologies for the wet forest cable logged chronosequence in the Florentine Valley. A "/" between geologies indicates mixed geologies were present.*

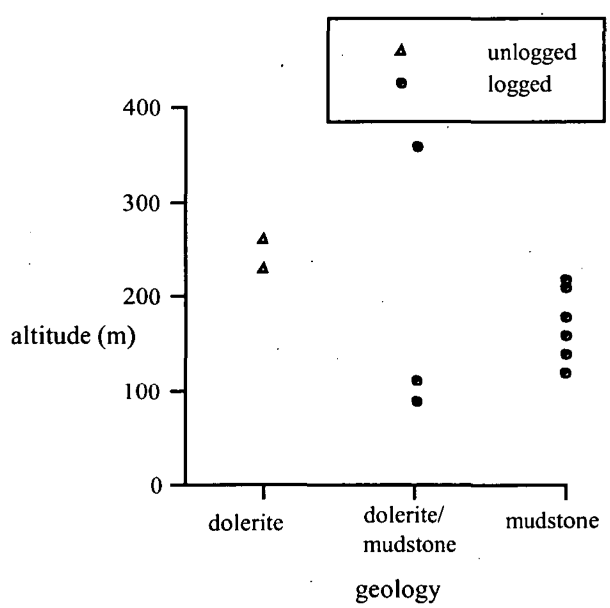


Figure 2.5 *Quadrat altitudes and geologies for the wet forest ground-based logged chronosequence in the Arve Valley.*

These substrates all weather to produce comparatively rich soils, particularly in cool wet forest environments (Jackson 1965, 1968). Oligotrophic substrates such as quartzite were avoided, in most cases pre-cambrian rocks weather to produce soils of poorer nutrient status.

Quadrats were confirmed as wet forest prior to logging using information sourced from pre-logging forest photo interpreted mapping completed in the 1940's, descriptions of the pre-logging vegetation from management records and by comparing quadrats with adjacent unlogged areas with the same photo interpreted type. The quadrats chosen were those with *E. regnans* as the overstorey dominant before logging, although both *E obliqua* and *E. delegatensis* were sub or co-dominant on some quadrats, at least in the regenerating canopy.

Slopes were generally greater than 19°, the slope angle which distinguishes between steep and non-steep country in the Tasmanian Forest Practices Code (1993, 2000). Random sampling of the landscape was not possible, as logging does not occur randomly throughout the landscape, either spatially or temporally. Stratified random sampling (Cochran 1977, Greig-Smith 1983) was implemented, with sampling dispersed (to minimise spatial auto-correlation) throughout the two study regions between coupes that met the survey design criteria.

The silvicultural systems applied to wet forests in Tasmania are described in Forestry Tasmania (1998). The silvicultural treatments applied at each of the surveyed areas was checked for consistency of regeneration method by examining forest management records held by Forestry Tasmania.

The adequacy of the experimental design was evaluated through examination of the correlation between floristic compositional variation (as summarised by ordination analyses) with selected experimental design variables.

Tests of inter-location heterogeneity

Chronosequence studies can be limited by the degree to which spatial variability complicates or confounds the temporal trends recovered in the data. The survey design criteria were established to limit or account explicitly for this potential source of error. Where the chronosequences in the two study regions are compared, tests of inter-location heterogeneity are established. These tests evaluate the validity of comparing the two chronosequences which use separate study regions, ie does the use of two study regions potentially confound the comparison of the two chronosequences. The tests of inter-location heterogeneity examine quantitatively the degree to which the quadrats drawn from the two study regions occupy similar physical environments and climates. In terms of their physical environments, the two chronosequences differed with respect to quadrat altitude, but not aspect, slope or rock cover (Table 2.3)

Table 2.3 *T-test comparison of each of the quadrat variable means for the Florentine (n=17) and Arve Valley (n=11) chronosequence quadrats at a 95% confidence level.*

	Florentine		Arve Valley			
	mean	s.d.	mean	s.d.	t	p
aspect (degrees)	145.5	109.88	129.1	110.45	0.4346	0.6658, ns
altitude (m)	440.2	106.60	189.1	78.40	7.2423	< 0.001
slope (degrees)	21.1	8.04	21.0	5.86	0.0503	0.9601, ns
rock cover (%)	11.0	15.69	9.5	5.22	0.3061	0.7608,ns

A series of tests were also performed to examine the null hypothesis that the two study areas had similar climates and were therefore likely to be suitable for a comparison of chronosequences. Data on temperature, rainfall and radiation parameters were derived for each quadrat in the two study regions from modelled climate surfaces. Forty climatic variables were examined through a series of multivariate and univariate analyses comparing these variables in the two study regions. The two study regions differed with respect to their temperature characteristics, primarily in the difference between the cooler and warmer months (Table 2.4). The Arve Valley study area has warmer summer temperatures than the Florentine study area.

Table 2.4 *Results of t-test of each of the climatic variable means for the Arve Valley (n=11) and Florentine (n=17) chronosequences.*

	Florentine		Arve Valley			
	mean	s.d.	mean	s.d.	t	p
Annual temperature range	18.2	1.16	18.9	0.49	-1.99	0.052
Mean temperature of the warmest quarter	13.3	0.77	14.8	0.45	-5.92	0.000
Temperature seasonality	35.2	2.01	30.3	1.91	7.22	<0.001
Rainfall of the driest quarter	205.8	20.03	215.4	18.71	-1.41	0.163
Rainfall seasonality	23.3	1.14	23.8	0.74	-1.23	0.221
Annual rainfall range	72.0	7.39	76.1	8.73	-1.53	0.131
Radiation with rainfall of driest quarter	54.4	4.07	54.4	3.14	-0.02	0.982
Radiation seasonality	50.7	5.49	50.5	6.21	0.08	0.932
Annual radiation with rainfall range	17.5	0.83	17.6	0.88	-0.57	0.566

Sampling Methods

The unit of vegetation sampling adopted was a square quadrat of 30 x 30m or 900m^2. The unit of treatment (ie. logging) for the chronosequence study is the coupe. With each coupe only being surveyed once, the choice of where to survey the coupe was based on the objective of minimising the variance as no within coupe sampling replication was attempted. The quadrat surveyed within the coupe was selected on the basis of it being representative of a relatively homogeneous, successfully regenerated portion and at least 50m from a road or track. Within each 30 x 30m quadrat, all vascular plant species growing within or projecting into or above the quadrat were recorded and assigned a visually assessed combined cover-abundance value modified from the Braun-Blanquet (Poore 1955) scale as follows:

Field score	Description
+	sparsely or very sparsely present; cover (less than 5%)
1	plentiful but of small cover (less than 5%)
2	very numerous, or covering at least 5-25% of the area
3	any number of individuals covering 26-50% of the area
4	any number of individuals covering 51-75% of the area
5	covering more than 76% of the area.

Cover was defined as foliage projective cover following Walker and Tunstall (1981). Plant species taxonomy follows Buchanan (1999).

Ten sub-plots (2 x 2m) were spaced at 7.5m intervals along the transect either side of the quadrat's mid-point (Figure 2.6). The sub-plots were 1m apart and alternated from side to side along the transect. Sub-plot sampling is used commonly overseas in logging impact studies (eg. Alaback and Herman 1988) or where contiguous plots are required to describe autogenic effects (Austin 1981). It provides a measure of cover and frequency based on repeated observations, potentially minimising sampling error due to operator bias which may occur if the quadrats were re-surveyed at some future stage (see discussion in Leps and Hadincová 1992). If the same quadrats are examined at the beginning and end of the study period, a large proportion of the sampling error of individual points is removed from the estimate of change (Goodall 1952). All vascular plant species present in the 20 sub-plots were recorded and assigned a visually assessed cover-abundance value which was then converted to a presence score. The presence score formed a local shoot frequency score of between 0 and 20 based on the number of times it was recorded within the twenty sub-plots. Local shoot frequency (Greig-Smith 1983) is any aerial part of the plant within, above or over-hanging the sub-plot.

A species can simply be given a presence score if it was within, above or over-hanging the sub-plot. For a detailed discussion on the use and properties of frequency data see Greig-Smith (1983). Data on vegetation structure and quadrat characteristics was also collected in the 30 x 30m quadrat (Table 2.5).

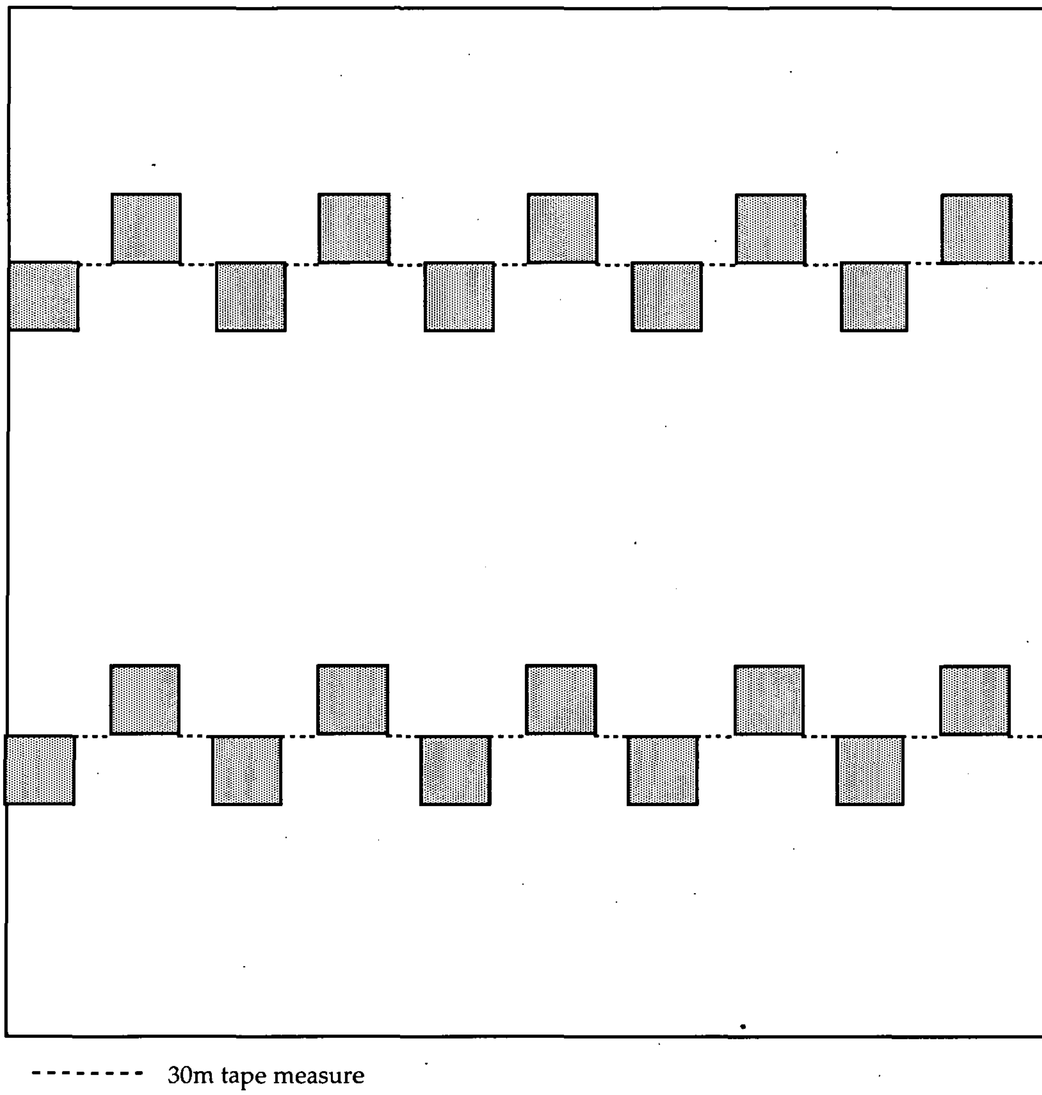


Figure 2.6 *The layout of the 0.09 ha. quadrat illustrating the arrangement and number of 2 x 2m sub-pots (shaded squares).*

Table 2.5 *Quadrat environmental and structural variables.*

Variables	Description
Altitude	extracted from 1:25,000 topographic maps to the nearest 10 metres
Aspect	measured on quadrat in degrees, orientated downslope. The quadrat was perpendicular to the contour
Rock cover	percentage of quadrat covered by rocks
Slope	measured on quadrat in degrees orientated downslope
Geology	recorded from geological mapping compiled by ANM Forest Management and other mapping and verified in the field where possible. The geological formations were Jurassic dolerite, Ordovician limestone and Permian mudstone and sandstone. Several quadrats appeared to have mixed lithologies
Landform	landform type (McDonald et al. 1998)
Cover of bare soil	percentage of quadrat covered by bare ground
Litter cover	percentage of quadrat covered by fine woody debris (<10cm diameter) and leaf litter
Moss cover	percentage of quadrat (rocks, ground surface, litter, coarse woody debris) with a cover of hepatics
Height of canopy	mean dominant height of regenerating canopy.
Cover of ground stratum	projective cover (Walker and Tunstall 1981) of ground stratum.
Height of ground stratum	height in metres of ground stratum.

Analytical Procedures

Three quantitative measures of species biomass were assessed, the visually assessed combined cover-abundance value over the 30 x 30m quadrat (+, 1, 2, 3, 4, 5), the local shoot frequency data (0, 1, 2, ... 20) from the 20 sub-plots, and the visually assessed cover-abundance values from the 20 sub-plots. Percentage cover was calculated by the conversion of the visually assessed cover-abundance value to the mid-points of the particular cover classes as follows:

Field score	Cover Class	Mid-point
+	<5%	1.0
1	<5%	2.5
2	5-25%	15.0
3	26-50%	37.5
4	51-75%	62.5
5	76-100%	87.5

To calculate the mean percentage cover scores, the individual mid-point values were then expressed as a mean percentage across the 20 sub-plot observations. The visually assessed cover-abundance values were subject to ordinal transformation to increase their reliability when subject to multivariate analysis (Kantvilas and Minchin 1989, Leps and Hadincová 1992). Species with a local frequency score of 0 (ie. present in the 30 x 30 m quadrat but not observed in the 20 2 x 2 sub-plots) were allocated a nominal value of 0.01.

Following the studies of Faith *et al.* (1987) and Minchin (1987) Non-metric Multi--dimensional Scaling (NMDS) was chosen as the method of ordination. All ordinations were performed in the "global" form (Kruskal 1964). The compositional dissimilarities between each pair of quadrats were computed using the Bray-Curtis dissimilarity coefficient (Bray and Curtis 1957). All ordinations used the mean percentage cover scores. No transformation was applied to the data before standardisation. The data were standardised to adjust species to unit maxima (ie. each species adjusted to a maximum value of 1.0 over all quadrats, following Faith *et al.* 1987). The NMDS was performed using ten random starting configurations. Random starting co-ordinates were used and there were 50 runs with real data and 50 with randomised data. The stability criterion chosen (standard deviations in stress over the last 10 iterations) was 0.0001. Since the number of dimensions of the data set is not known in advance, the analyses were performed in either 1-3 or 1-6 dimensions. The two and three dimensional ordinations were compared and interpreted in terms of the fitting of vectors of maximum correlation for exploratory variables and their minimum stress solutions. Stress is "a measure of departure from monotonicity in the relationship between the dissimilarity (distance) in the original p-dimensional space and distance in the reduced k-dimensional ordination space" (McCune and Mefford 1999). The plot of stress versus number of dimensions was used to choose the appropriate minimum stress solution following Bowman and Minchin (1987). In some instances hybrid scaling (Faith, Minchin and Belbin 1987) was applied to an initial configuration of objects derived from an input ordination to produce a re-configured inter-object association matrix. In these cases the starting configuration used for hybrid scaling was a two dimensional non-metric multi-dimensional solution. For multivariate analyses, both the DECODA ecological database package (Minchin 1989 version 2.04) and PC-ORD Version 4.25 (McCune and Metford 1999) were used. Unless otherwise stated above, the default analysis options available in these programs were chosen.

To aid with the interpretation of the environmental and survey design variables with the ordination patterns, the analytical technique of vector fitting (Kantvilas and Minchin 1989) was used. This technique has been used elsewhere (Bowman and Minchin 1987, Kantvilas and Minchin 1989) to examine correlations between compositional dissimilarity derived from ordinations and environmental variables. A Monte-Carlo procedure was used to derive probabilities for the maximum correlations with 1000 random permutations. The probabilities were computed as the number of correlations which were greater than or equal to the values computed from the actual data. Only those vectors representing a variable with a significance level of <0.05 are presented. The results of the vector fitting are displayed following Kantvilas and Minchin (1989). Vector diagrams are plotted to display the position of each of the significant fitted vectors relative to the ordination space.

The relationship between age since logging and the scores of the quadrats along the fitted vector for age is illustrated by the scatter diagrams, with age as the y-axis and the score on the fitted vector for age as the x-axis. This plot displays the strength of the relationship between quadrat age and the vegetation trend represented by the stand age vector and identifies those quadrats which diverge from this trend.

In order to display compositional trends along the fitted vectors, condensed ordered tables were produced. The quadrats are ordered according to their scores along the vector fitted into the ordination space for stand age. The species are also sorted by their score along the vector, the score for each species being computed as the average of the scores of the quadrats in which it occurs, weighting each quadrat by the mean percentage cover abundance score of the species in that quadrat (Kantvilas and Minchin 1989). For ease of presentation, the visually assessed cover-abundance values are presented in the ordered, condensed data matrix, instead of the actual values for the mean percentage cover scores based on the twenty replicate sub-plot observations. This avoids the need to set arbitrary class limits to the mean percentage cover scores whilst converting them to integers.

Two classification methods have been used in this report, a divisive hierarchical approach (TWINSPAN, Hill 1979) and unweighted pair group arithmetic averaging, (UPGMA) an agglomerative hierarchical approach. For TWINSPAN, the default options were chosen including the recommended pseudospecies cut levels for visually assessed cover-abundance values as follows; 0, 2, 5, 10 and 20. The same mean percentage cover score data used in the ordination was used in the classification. The eigenvalues stated with the analysis results refer to the first axis of the reciprocal averaging ordination performed by TWINSPAN. The DECODA program was used to prepare the data matrix for TWINSPAN. The version of TWINSPAN used is the one supplied with version 2.04 of DECODA, it has been significantly re-configured from the original version according to the DECODA documentation notes. Unweighted pair group arithmetic averaging cluster analysis was performed using the FUSE algorithm in PATN (Belbin 1992). The option chosen was flexible UPGMA, a hierarchical agglomerative polythetic clustering method with beta = 0.1. It utilised the same associated matrix described for the TWINSPAN procedure.

The univariate analysis strategy involved a regression approach where vegetation parameters such as cover and richness are considered the response variables and stand age since logging the explanatory variable. A regression approach is generally more powerful than an ANOVA based approach as ANOVA does not consider the ordering of the factor levels, only the differences between their means (Crawley, 2001). Logging type is here considered a factor with two levels, cable or ground-based. Linear regressions were fitted for the various response variables against stand age for the two chronosequences. Regressions were fitted only to the logged quadrats, and the comparisons of the linear models restricted to regrowth aged 5-30 years, where both the cable and ground-based quadrats were surveyed. Analysis of covariance (ANCOVA, Crawley 2001) was applied to test whether the functional relationships described by these regression equations differed. The comparisons tested for concurrent or coincident regressions by testing for equality of slopes and intercepts. Approaches such as these have several limitations, the error associated with the regression only being estimated in terms of within quadrat variation and not between quadrat variations, as both degrees of freedom were taken up with the model comparing the two treatments (cable versus ground-based logging). For the simultaneous comparison of homogeneity of slopes and intercepts for response variables, the independent variable is stand age. The null hypothesis is that there is no difference between the regression equations for the two logging methods. The graphical presentation of the regression models uses all of the chronosequence data points for regrowth aged 5-54 years, although the ANCOVA approach above is restricted to the matched 5-30 year chronosequence.

A number of floristic variables and structural variables were derived from the quadrat data to assist in the interpretation of the ordination analyses (Table 2.6). The linear relationships between these variables were examined using Pearson product-moment correlations.

Table 2.6 *Derived floristic and structural variables.*

Variable	Description
Richness	alpha diversity at the 0.09 ha scale
Richness of Trees > 15m	number of tree species greater than 15 m at maturity.
Richness of Tall shrubs 8-15 m	number of shrub species 8-15 m at maturity.
Richness of Shrubs 1-8 m	number of shrub species 1-8 m in height at maturity.
Richness of Shrubs < 1 m	number of shrub species less than 1 m in height at maturity.
Richness of Graminoids > 0.3 m	number of species with a graminoid growth habit less than 0.3 m in height at maturity.
Richness of Graminoids and Grasses < 0.3 m	number of species with a graminoid growth habit greater than 0.3 m in height at maturity.
Richness of Ground ferns	number of ground fern species.
Richness of Forbs and Herbs (non-graminoid)	number of forbs and herbaceous species
Richness of Climbers	number of climber species
Cover of Trees > 15 m at maturity	cumulative projective cover of trees greater than 15 m in height at maturity
Cover of Tall shrubs 8-15 m	cumulative projective cover of shrub species 8-15 m in height at maturity.
Cover of Shrubs 1-8 m	cumulative projective cover of shrub species 1-8 m in height at maturity.
Cover of Shrubs < 1 m	cumulative projective cover of shrub species less than 1 m in height at maturity.
Cover of Graminoids > 0.3 m	cumulative projective cover of graminoids greater than 0.3 m in height at maturity.
Cover of Graminoids/Grasses < 0.3 m	cumulative projective cover of graminoids/grasses less than 0.3m in height at maturity.
Cover of Ground ferns	cumulative projective cover of ground fern species.
Cover of Forbs and herbs (non-graminoid)	cumulative projective cover of non-graminoid forb and herb species.
Cover of Climbers	projective cover of species with a climbing habit

Experiment 2: Impacts of cable and ground-based logging on the frequency and cover of woody and herbaceous vascular in wet and dry forests

Long term monitoring studies using permanent quadrats were established which encompassed the characteristic environments, forest types and cable logging methods used in Tasmania. The two studies are;

(1) the impacts of cable logging on woody and herbaceous species in wet forests at Wayatinah (Pearce and Blue logging units) in central Tasmania, and

(2) the impacts of cable and ground-based logging on woody and herbaceous species in dry forest at Wielangta in eastern Tasmania.

The study therefore uses three locations (Pearce 04, Blue 09, Wielangta 42a), two treatments (cable and ground-based logging) and unlogged controls (Table 2.7). The experimental factors are forest type (with two levels, wet and dry) and forest successional status for the wet forest study (with two levels, mature forest and late mature forest). The design is not orthogonal (Table 2.7), the ground-based logging treatment could not be applied at Wayatinah due to the steep slopes present and late mature dry forest environments were not present due to the recurrent nature of wildfires in eastern Tasmania.

Table 2.7 *Experimental design of long term monitoring studies.*

location	treatment		forest type		successional status	
	cable	ground-based	wet	dry	mature	late mature
Pearce 04	✓		✓		✓	
Blue 09	✓		✓			✓
Wielangta 42a	✓	✓		✓	✓	

Study Areas

Blue 09 (30 ha, Figure 2.7) consisted of a mosaic of *E. regnans* dominated wet and mixed (Gilbert 1959) forest. The wet forest understoreys were predominantly *Olearia argophylla*, *Dicksonia* and *Polystichum*, and the mixed forest understoreys were dominated by *Atherosperma*. (REG 1001 and REG 101 in Kirkpatrick *et al.* 1988). The overstorey is in the late mature stage. On the streamside reserve along the Florentine River, *Nothofagus* dominated callidendrous rainforest was present. The response of the streamside reserve to cable harvesting is presented in Chapter 4 for both Pearce 04 and Blue 09 (Table 2.8).

Pearce 04 (22 ha, Figure 2.8) is located on the western side of the Derwent River upstream of the Wayatinah Road crossing and consists of *Eucalyptus regnans - Acacia dealbata - Pomaderris apetala* wet forest (REG 1001 in Kirkpatrick *et al.* 1988), where tree dominance is shared by *E. regnans* and *E. obliqua* (Table 2.8). The overstorey is in the pole to mature stage (Ashton 1976). More sheltered aspects, particularly at the southern end of the coupe near Robinsons Creek consist of *Eucalyptus regnans -Atherosperma moschatum - Acacia dealbata - Olearia argophylla* wet sclerophyll forest/mixed forest (REG 101 in Kirkpatrick *et al.* 1988). The streamside reserve area itself consisted of a scrubby forest with an overstorey of *E. obliqua* and *E. amygdalina*, which was distributed on a gravelly river terrace. *E. regnans* dominated wet forest is widespread in Tasmania, with some 74 400 ha extant, 18% reserved and 20% as mature or old growth forest (Tasmanian Public Land Use Commission 1996).

Control quadrats were surveyed in December 1992 and permanently marked in an area south of Blue 09 (Figure 2.9) known as the Wayatinah control area. The quadrats were surveyed at the same time the treatment quadrats were established. The controls quadrats were surveyed in an area selected on the basis of having similar forest types, geology, growth stage and stream classes as the those present in the treatment areas of Blue 09 and Pearce 04. The Wayatinah control area was then dedicated in Forestry Tasmania's GIS forest zoning system as an area of permanent research activity.

Wielangta 42a (30 ha, Figure 2.10) is located in a drier and warmer near coastal environment compared to the cool, wet environment near Wayatinah. While the topography at Blue and Pearce are uniform and the coupes roughly rectangular in shape, the Wielangta coupe is centred on a class 4 (Forest Practices Board 2000) internal drainage line with several ridge spurs and different aspects. The vegetation is dry and scrubby, and exhibits the effects of past wildfires. It consists of a mosaic of *E. obliqua - E. delegatensis - E. pulchella* dominated scrubby dry forest with *E. delegatensis* dominating the higher more exposed aspects, and *E. pulchella* dominating the steepest boulder fields. *E. obliqua* is the most widespread species in the coupe, dominating the broader slopes where soils are better developed. *E. obliqua* dominated dry forest is widespread in Tasmania, with some 160 700 ha extant, 23% reserved and 28% as mature or old growth forest (Tasmanian Public Land Use Commission 1996).

Table 2.8 *Locational details and logging and silvicultural systems at the three cable logging operations subject to long-term monitoring.*

coupe	Pearce 04	Blue 09	Wielangta 42a
size (hectares)	22	30	30
vegetation pre-logging sampling date	March 1992	February-March 1992	April - May 1992
number of vegetation quadrats in coupe area[1]	34	30	27
number of control vegetation quadrats[2]	7	6	7
date logged	September – December 1992	February - September 1992	April - August 1992
ground-based logging system	limited ground snigging and shovel logging (on an excavator) on the roadside	none	ground snigging (on flatter sections) with a rubber tyred snigger
cable logging system	running skyline with mobile tail spar	running skyline with mobile tail spar	standing skyline with north bend
cable yarder	Madill 122 Interlock Swing Yarder 60' Gantry and Live Boom	Madill 122 Interlock Swing Yarder 60' Gantry and Live Boom	Madill 071
carriage	grapple	grapple	block carriage with chokers
stream classes present	class 1 (Derwent River) class 2 (Robinsons Creek)	class 1 (Florentine River) class 3 (feeder stream to class 1)	class 4 internal drainage line
silvicultural system	clearfall, slash burn, aerial sowing	clearfall, slash burn, aerial sowing	clearfall, natural seedfall from logging residue and stump coppice
date burnt	25-3-93	19-4-93	not burnt
date sown	19-4-93 & 11-5-93	12-5-93	not sown
majority forest type of logged area	E1 with scattered eucalypt regrowth	E1 and E2 with wet forest and ST	fire damaged E3 and E4
vegetation types in logged area	mosaic of pole and mature stage *E. regnans* and *E. obliqua* - *Pomaderris apetala* dominated wet forest	mosaic of mature stage *E. regnans* - *Atherosperma* - *Olearia argophylla* dominated wet and mixed forest	mosaic of *E. obliqua* - *E. delegatensis* - *E. pulchella* dominated scrubby dry and damp forest
associated studies established on quadrat	*Dicksonia* logging impact study, streamside logging impacts study *Atherosperma* coppice regeneration	streamside vegetation logging impacts study	dragline recovery study

[1] not necessarily all logged, includes quadrats in streamside reserves
[2] in areas designated as long term controls, away from the coupes.

Survey Design

The experimental design adopted was the approach commonly used in impact studies: the Before-After-Control-Impact-Pairs design (BACIP, Green 1979, Green 1993, Humphrey *et al.* 1995, Faith *et al.* 1995). The BACIP design surveyed paired vegetation quadrats before the impact (logging treatment), after the impact and repeated this procedure for control quadrats paired to the treatment quadrats based on their environmental and floristic attributes. By sampling treatment quadrats and control quadrats at similar intervals, the magnitude of the difference between the treatment and control quadrats becomes the size of the treatment effect. The hypothesis examined is that there is no difference in species composition and cover measured prior to and after treatment in the paired treatment and control quadrats. The selection of the sampling points at both the treatment and adjacent controls was based on a pooled analysis of all sampling points with the aim of matching floristically pairs of quadrats in both the treatment and distant control areas. Cluster and ordination analyses were used to identify pairs of quadrats based on Bray-Curtis similarity scores. Sampling points were then selected in the treatment areas with the greatest statistical similarity to the adjacent control quadrats. These quadrats were re-established following cable logging and re-surveyed.

Sampling Methods

A systematic arrangement of quadrat points throughout the coupes was used. Permanently surveyed take off points on the primary cable logging access roads were established. From these points fixed bearings and distances were taken to locate quadrat points. At each study area, treatment and control quadrats were established prior to logging (Table 2.9, 2.10 and 2.11) and to date one post-treatment measure has been completed at each study area.

Table 2.9 *The treatment program for the Wayatinah Blue 09 study area.*

Event	Date	time period (months)
floristic survey of treatment and control quadrats	February-March 1992	-9
cable logging commences	March 1992	0
cable logging completed	September 1992	7
treatment quadrats burnt	April 1993	14
treatment quadrats aerially sown	May 1993	15
first post treatment floristic survey	June 1994	28

Table 2.10 *The treatment program for the Wayatinah Pearce 04 study area.*

Event	Date	time period (months)
floristic survey of treatment and control quadrats	April-May 1992	-9
cable logging commences	September 1992	0
cable logging completed	December 1992	3
treatment quadrats burnt	March 1993	6
treatment quadrats aerially sown	April 1993	7
first post treatment floristic survey	May 1994	20

Table 2.11 *The treatment program for the Wielangta 42a study area.*

Event	Date	time period (months)
floristic survey of treatment and control quadrats	March-May 1992	-1
cable and ground-based logging commences	April 1992-	0
cable and ground-based logging completed	August 1992	
Eucalyptus regeneration stocking survey	June 1993	15
first post treatment floristic survey	May 1994	25
second post treatment floristic survey	April 1995	36

The number of quadrats in the logging and control treatments at Blue 09, Pearce 04 and Wielangta 42a are given in Table 2.8. The location of each quadrat are presented in Tables 2.12, 2.13, 2.14, and 2.15. The measurement schedule and allocation of quadrats to treatments for Wielangta 42a are detailed in Table 2.16. Vegetation surveys with 30 x 30 m quadrats and their nested 20 2 x 2 m sub-plots follows the procedures outlined previously for the chronosequence study. The extent of soil disturbance within each quadrat was visually assessed after logging but prior to burning. The quadrats at which either a control, cable or ground-based logging treatment was applied at Wielangta differed significantly with respect to their slope (Figure 2.11).

Table 2.12 *Quadrat names, geocodes, pre-logging forest types (after Stone 1998), and slope for the long term monitoring study at Blue 09, Wayatinah. The geology was all Jurassic Dolerite. The stream class classification is from the Forest Practices Code (1993).*

quadrat	stream class	easting	northing	alt. (m)	forest type	aspect	slope degrees
Blue 09_1	-	458160	5300100	280	E2cST	80	15
Blue 09_2	-	458240	5300200	260	E2cST	62	17
Blue 09_3	1	458330	5300050	240	M3T	92	23
Blue 09_4	1	458310	5300000	220	M3T	340	18
Blue 09_5	-	458040	5300800	340	M3T	90	25
Blue 09_6	-	457970	5301000	260	E2cST	90	22
Blue 09_7	-	458100	5299800	300	E2cST	70	23
Blue 09_8	-	458200	5299810	280	E2cST	72	34
Blue 09_9	-	458260	5299830	260	E2cST	70	23
Blue 09_10	1	458350	5299850	220	M3T	20	9
Blue 09_11	4	458180	5299740	260	T	50	33
Blue 09_12	4	458020	5299700	320	T	110	25
Blue 09_13	-	458000	5299800	340	E2&1b	240	23
Blue 09_14	-	457910	5299810	360	E2&1b	105	27
Blue 09_15	-	458010	5299630	320	E2&1b	65	23
Blue 09_16	-	457920	5299620	360	E2&1b	60	20
Blue 09_17	4	457990	5299500	320	E2dST	65	24
Blue 09_18	-	458100	5299640	280	E2&1b	80	27
Blue 09_19	-	458200	5299650	240	E2&1b	80	35
Blue 09_20	4	458140	5299530	240	E2&1b	60	30
Blue 09_21	-	458110	5299450	300	E2&1b	60	29
Blue 09_22	-	458190	5299460	280	E2&1b	60	29
Blue 09_23	1	458300	5299500	230	M3T	70	30
Blue 09_24	-	458140	5299230	300	E2&1b	80	19
Blue 09_25	1	458250	5299240	280	M3T	80	32
Blue 09_26	-	458040	5299230	340	E2&1b	62	29
Blue 09_27	-	457950	5299200	360	E1&2b	45	19
Blue 09_28	-	457900	5299390	360	E1&2b	58	20
Blue 09_29	-	458020	5299430	340	E1&2b	60	18
Blue 09_30	-	458090	5299190	320	E2&1b	70	25

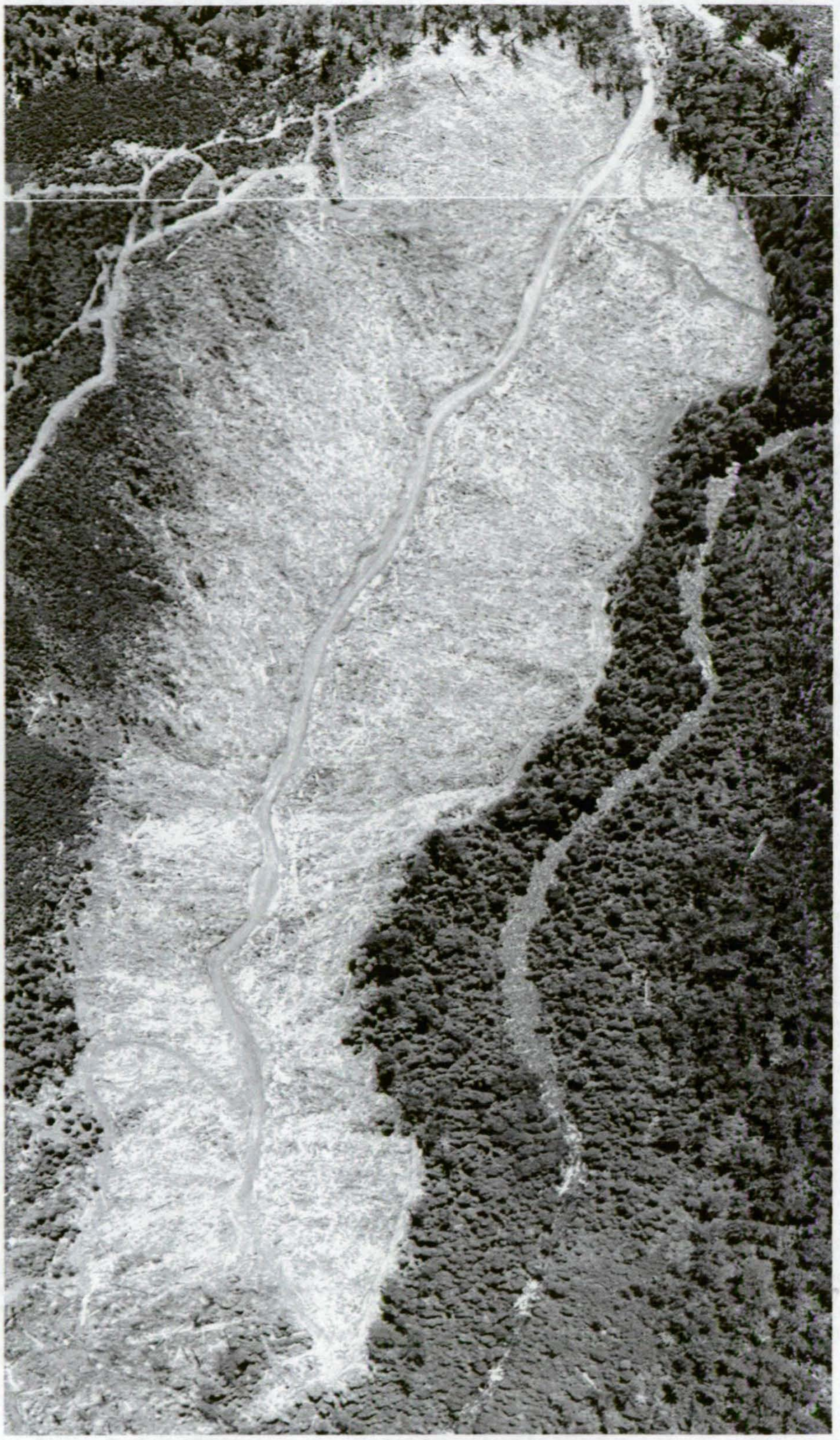


Figure 2.7 *Aerial view of Blue 09 long term monitoring area shortly after logging was completed in December 1992. The Florentine River on the right and the strip of vegetation retained as a streamside reserve is shown. The draglines are barely just visible. (Forestry Tasmania Florentine River film no. 9203, frames 7-8, run 1, 1:5000, 5650 feet, 50 mm lens enlarged x 6).*

Table 2.13 *Quadrat names, geocodes, pre-logging forest types (after Stone 1998), slope and geology for the long term monitoring study at Pearce 04, Wayatinah. Geological codes; JD= Jurassic Dolerite, _A = Quaternary Alluvium (Riverine). The stream class classification is from the Forest Practices Code (1993).*

quadrat	stream class	easting	northing	alt. (m)	forest type	aspect	slope degrees	geology
Pearce 04_1	2	457600	5303450	250	$M_3T/1$	50	8	JD
Pearce 04_2	-	457700	5303530	250	$M_3T/1$	155	11	JD
Pearce 04_3	-	457780	5303550	250	E1dTS	320	25	JD
Pearce 04_4	-	457750	5303630	280	E1bSER	150	15	JD
Pearce 04_5	-	457870	5303620	260	E1dTS	140	25	JD
Pearce 04_6	-	457850	5303670	280	E1bSER	130	18	JD
Pearce 04_7	-	457950	5303650	260	E1dTS	105	15	JD
Pearce 04_8	-	457900	5303700	280	E1dTS	110	20	JD
Pearce 04_9	-	458030	5303720	250	E1bSER	125	15	JD
Pearce 04_10	-	457980	5303740	270	E1dTS	90	15	JD
Pearce 04_11	-	458000	5303690	250	E1bSER	110	19	JD
Pearce 04_12	-	457980	5303560	230	$M_3T/1$	315	15	JD
Pearce 04_13	-	457920	5303520	230	$M_3T/1$	140	10	JD
Pearce 04_14	-	458050	5303780	250	E1bSER	70	12	JD
Pearce 04_15	-	457980	5303790	280	E1bSER	80	16	JD
Pearce 04_16	-	458020	5303880	260	E1bSER	45	18	JD
Pearce 04_17	-	457960	5303870	280	fdE1aER3d	50	11	JD
Pearce 04_18	-	457950	5303960	260	fdE1aER3d	75	14	JD
Pearce 04_19	-	457930	5304040	260	fdE1aER3d	30	19	JD
Pearce 04_20	-	457890	5304120	260	fdE1aER3d	40	7	JD
Pearce 04_21	-	458120	5303670	220	E1bSER	85	21	JD
Pearce 04_22	-	458200	5303620	210	E1bSER	100	10	JD
Pearce 04_23	-	458290	5303580	200	E2bERS	80	5	_A
Pearce 04_24	1	458290	5303580	200	MTE1f	80	5	_A
Pearce 04_25	1	458030	5304180	210	fdE1cER5dS	50	5	_A
Pearce 04_26	1	458080	5304200	210	fdE1cER5dS	50	5	_A
Pearce 04_27	-	458030	5304020	220	E1bSER	50	32	JD
Pearce 04_28	1	458140	5304070	210	fdE1cER5dS	40	13	_A
Pearce 04_29	-	458140	5303810	220	E1bSER	95	22	JD
Pearce 04_30	1	458200	5303830	210	E1bSER	80	25	JD
Pearce 04_31	-	458100	5303930	220	E1bSER	60	23	JD
Pearce 04_32	-	458020	5304090	220	E1bSER	50	33	JD
Pearce 04_33	-	457560	5303590	290	E1dTS	145	29	JD
Pearce 04_34	-	457730	5303630	280	E1dTS	150	19	JD

Figure 2.8 *Aerial view of Pearce 04 long term monitoring area shortly after logging was completed in December 1992. The Derwent River on the right and the strip of vegetation retained as a streamside reserve is shown. The draglines are just visible and exhibit a pattern typical of yarding perpendicular to the central access road (Wayatinah Rd.) (Forestry Tasmania Wayatinah film no. 9203, frame 15, run 1, 1:5000, 5650 feet, 50 mm lens enlarged x 6)*

Table 2.14 *Quadrat names, geocodes, pre-logging forest types (after Stone 1998) and slope for the control quadrats at Wayatinah. The geology was all Jurassic Dolerite. The stream class classification is from the Forest Practices Code (1993).*

quadrat	stream class	easting	northing	alt. (m)	forest type	aspect	slope degrees
Wayatinah A	-	458400	5298300	330	E2&1dT	100	29
Wayatinah B	-	458500	5298300	315	E2&1dT	100	20
Wayatinah C	1	458600	5298200	305	M3T	100	12
Wayatinah D	1	458600	5298200	300	M3T	100	20
Wayatinah E	4	458700	5298500	290	S/2	0	3
Wayatinah F	4	456400	5257500	435	E1&2dTSM3	20	19
Wayatinah G	-	458400	5298300	320	E2&1dT	65	32

Figure 2.9 *Aerial view of Wayatinah control area in December 1992 on the Florentine River south of Blue 09. The control quadrats are in the centre of the photograph and adjacent to the river. (Forestry Tasmania Florentine River film no. 9203, frame 10, run 1, 1:5000, 5650 feet, 50 mm lens enlarged x 6)*

Table 2.15 *Quadrat names, geocodes, pre-logging forest types (after Stone 1998), and slope for the long term monitoring study at Wielangta 42a. eastern Tasmania. The stream class classification is from the Forest Practices Code (1993). The geology was all Jurassic Dolerite.*

quadrat	stream class	easting	northing	alt. (m)	forest type	aspect	slope degrees
Wielangta_1	4	637122	5270620	440	fdE4b.ER.	30	13
Wielangta_2	-	637127	5270500	400	fdE4b.ER.	300	17
Wielangta_3	-	637136	5270460	430	fdE4b.ER.	305	15
Wielangta_4	-	637133	5270620	390	fdE4b.ER.	328	23
Wielangta_5	-	637135	5270540	420	fdE4b.ER.	345	14
Wielangta_6	-	637143	5270490	430	fdE4b.ER.	330	4
Wielangta_7	-	637128	5270720	360	fdE4b.ER.	358	19
Wielangta_8	-	637142	5270720	350	fdE3b.ER.	358	22
Wielangta_9	4	637128	5270860	300	fdE3b.ER.	290	15
Wielangta_10	-	637144	5270830	340	fdE3b.ER.	228	16
Wielangta_11	-	637138	5270900	350	fdE3b.ER.	228	21
Wielangta_12	3	637121	5270910	290	fdE3b.ER.	310	12
Wielangta_13	-	637131	5270940	320	fdE3b.ER.	230	33
Wielangta_14	-	637101	5270700	375	fdE4b.ER.	320	10
Wielangta_15	-	637106	5270610	385	fdE4b.ER.	320	12
Wielangta_16	-	637113	5270660	380	fdE4b.ER.	35	16
Wielangta_17	-	637108	5270760	360	fdE4b.ER.	360	23
Wielangta_18	-	637103	5270850	340	fdE4b.ER.	35	14
Wielangta_19	-	637095	5270780	355	fdE4b.ER.	45	8
Wielangta_20	-	637122	5270510	400	fdE4b.ER.	180	2
Wielangta_21	-	637124	5270430	400	fdE4b.ER.	275	4
Wielangta_22	-	637150	5270620	400	fdE4b.ER.	3	18
Wielangta_23	-	637150	5270550	420	fdE4b.ER.	3	10
Wielangta_24	-	637161	5270780	400	fdE3b.ER.	230	23
Wielangta_25	-	637159	5270880	420	fdE4b.ER.	230	21
Wielangta_26	-	637145	5270730	360	fdE3b.ER.	0	16
Wielangta_27	-	637161	5270750	390	fdE3b.ER.	220	21
Wielangta_28	-	637160	5270900	440	fdE4b.ER.	230	25
Wielangta_29	-	637130	5270900	740	fdE4b.ER.	240	35
control quadrats							
Wielangta_30	3	637120	5270900	280	fdE4b.ER.	315	30
Wielangta_31	-	637150	5270500	435	fdE4b.ER.	355	9
Wielangta_32	-	637090	5270900	320	fdE4b.ER.	220	22
Wielangta_33	-	637100	5270500	370	fdE4b.ER.	22	28
Wielangta_34	-	637100	5270400	385	fdE4b.ER.	295	12

Table 2.16 *Quadrat treatments and sampling frequency at the long term monitoring study at Wielangta 42a, eastern Tasmania.*

quadrat	logging treatment	sampling year		
		1992 (pre-treatment)	1994	1995*
Wielangta_1	cable	y		y
Wielangta_2	ground-based	y		y
Wielangta_3	ground-based	y		y
Wielangta_4	ground-based	y		y
Wielangta_5	ground-based	y		y
Wielangta_6	ground-based	y		y
Wielangta_7	cable	y	y	y
Wielangta_8	cable	y		y
Wielangta_9	cable	y		*
Wielangta_10	cable	y		y
Wielangta_11	cable	y		y
Wielangta_12	cable	y	y	y
Wielangta_13	cable	y		y
Wielangta_14	ground-based	y		y
Wielangta_15	ground-based	y		y
Wielangta_16	ground-based	y		y
Wielangta_17	ground-based	y		y
Wielangta_18	ground-based	y		y
Wielangta_19	ground-based	y		y
Wielangta_20	ground-based	y		y
Wielangta_21	ground-based	y		*
Wielangta_22	ground-based	y		y
Wielangta_23	ground-based	y		*
Wielangta_24	cable	y		y
Wielangta_25	cable	y		y
Wielangta_26	cable	y		*
Wielangta_27	cable	y		y
Wielangta_28	control	y		y
Wielangta_29	control	y		y
Wielangta_30	control	y		y
Wielangta_31	control	y		y
Wielangta_32	control	y		y
Wielangta_33	control	y		
Wielangta_34	control	y		y
	n	34	2	29

*quadrats 9, 21, 23, 26 and 33 were not re-scored in 1995 due to logistical problems

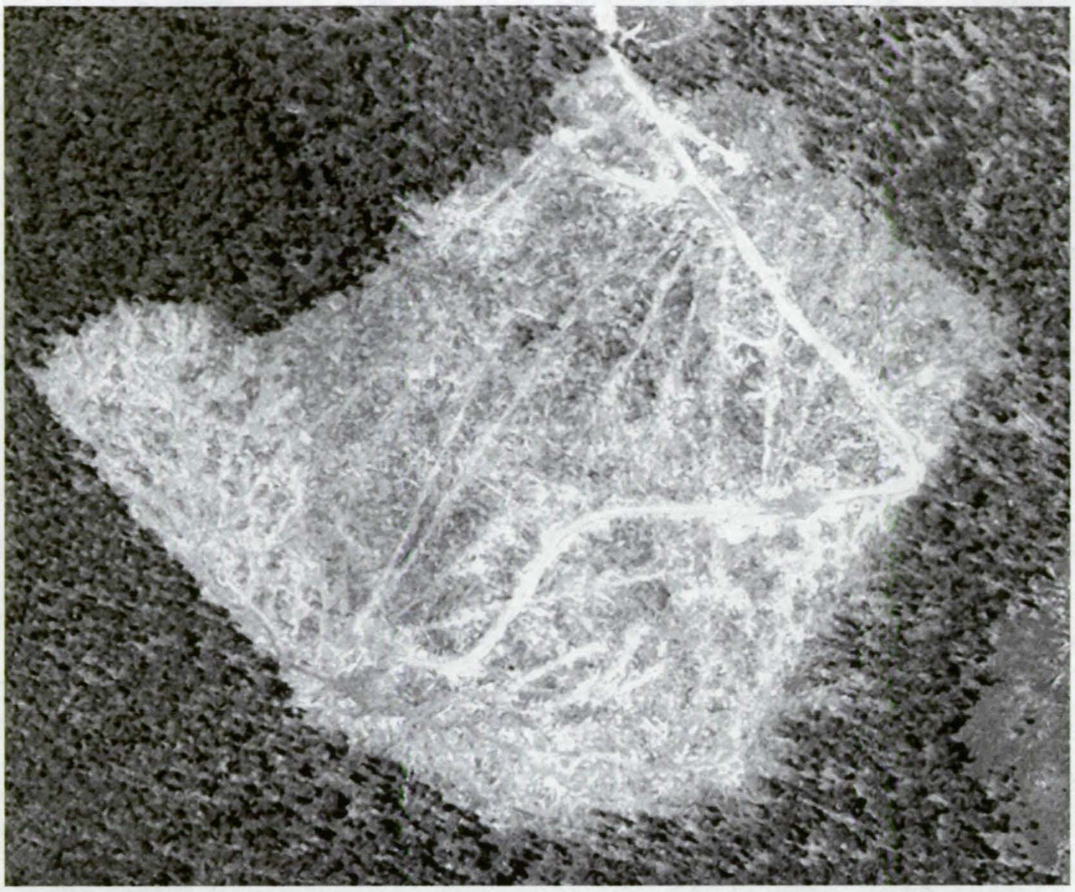


Figure 2.10 *Aerial view of Wielangta 42a long term monitoring area four months after cable and ground-based logging was completed. The vegetation sampling points, landings and logging system are shown in Figure 3.1. Draglines from cable logging are evident as longitudinal strips while ground-based logging in the bottom left and right appears as a more concentrated network of meandering snig tracks. (Forestry Tasmania WT42a film no. 9204, frame 22, run 1, 1:4000, 11,150 feet, 50 mm lens enlarged x 6, dated 18.12.92)*

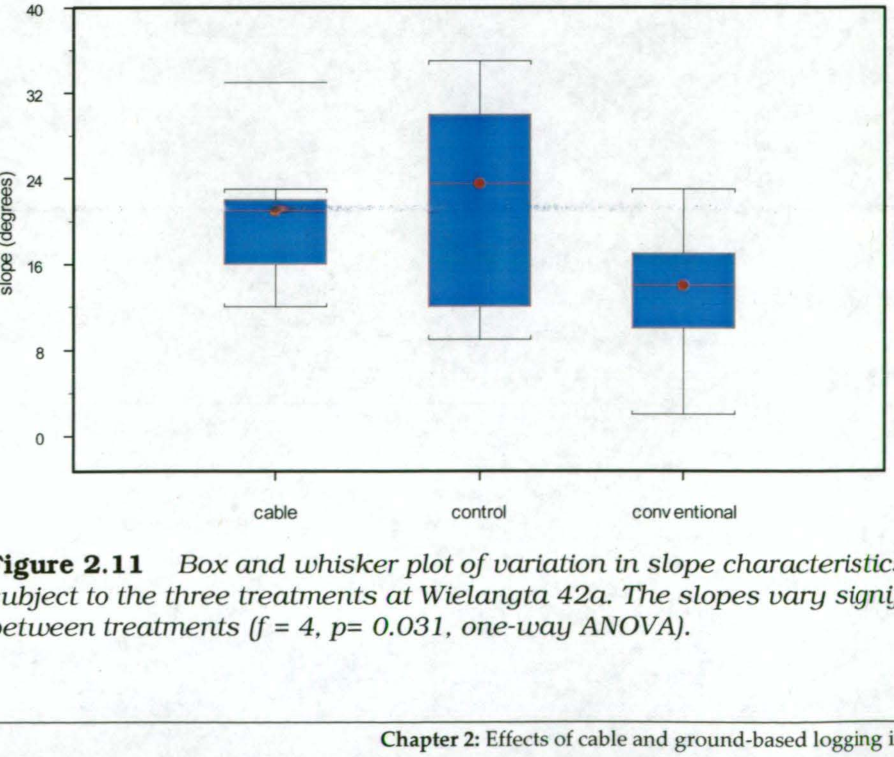


Figure 2.11 *Box and whisker plot of variation in slope characteristics for quadrats subject to the three treatments at Wielangta 42a. The slopes vary significantly between treatments (f = 4, p= 0.031, one-way ANOVA).*

Analytical Procedures

Cluster analysis was performed on all pairs of re-measured treatment quadrats. Hierarchical classification was performed using flexible pair group arithmetic averaging (UPGMA) with a beta parameter of –0.13 (see Belbin & McDonald 1993). Dendograms were generated from the cluster analysis as a visual representation of the binary tree structure. The association matrices were calculated using the Bray & Curtis measure following Faith, Minchin and Belbin (1987). Cluster analysis was performed using PC-ORD, Version 4.25 (McCune and Metford 1999).

The degree of dispersion of each quadrat within the treatment groups throughout the multivariate space was explored using the Multi-Response Permutation Procedures (MRPP) of PC-ORD, Version 4.25 (McCune and Metford 1999). The distance measures were calculated using the Bray and Curtis formula and the weighting option was n/sum(n) as recommended by Mielke (1984). Six groups were defined by a factorial combination of the three treatments (cable logging, ground-based logging and control) by the two observation periods, pre-logging (1992) and post-logging (1995).

Compositional or multivariate differences between paired (before and after logging) quadrats were examined using one-way ANOVA. The compositional differences (species and their cover) between the paired quadrats was first summarised in a univariate form through the construction of a dissimilarity matrix between the paired quadrats. The Bray and Curtis dissimilarity score varies from 0 to 1 with higher scores indicating greater compositional differences between the paired quadrats. The influence of the treatment (cable or ground-based logging and control) on the difference between the paired quadrats was then explored with a one-way ANOVA. Plots of the square root of the absolute values of the residuals versus the fitted values were examined to check for the constant variance assumption of the ANOVA model. Multiple comparisons between the three treatments were made using the Bonferroni method of central point calculation. The Bonferroni method is a more conservative approach (compared to Tukey's) where homogeneity of variances cannot be assured. The simultaneous confidence levels for the multiple comparison was 0.95. These analyses were performed using S-PLUS 6.1 for Windows Professional Edition (Lucent Technologies, Inc. Insightful Corp. 2002)

Indicator species analysis using Dufrene and Legendre's (1997) method was applied to groups of permanent quadrats defined by the combinations of treatment (cable logging, ground-based logging, and control) and measurement period (before and after treatment). Calculations were performed using PC-ORD, Version 4.17 (McCune and Metford 1999). The method produces statistics for the relative frequency and relative abundance of each species in each group and a composite indicator value. Relative frequency in a floristic group is defined as the percentage of quadrats that contain a particular species within a given group, where the species occurrence is >0. Relative abundance is defined as the average abundance of a given species in a given group divided by the average abundance of that species across all groups, expressed as a percentage. Finally, the indicator values are defined by combining the above values for relative abundance and relative frequency and expressing them as a percentage of perfect indication. The observed maximum indicator values for species were tested for statistical significance using a Monte Carlo technique with 10 000 randomised group starts.

Box and whisker plots (Tukey 1977; Chambers *et al*, 1983) were used to graphically summarise the statistical distribution of quadrat slope and treatment dissimilarity scores. The central tendency of the data is displayed as the median, the spread of the data as the interquartile range that correspond to the 25th and 75th percentiles. Box-plots were prepared using the quartiles method in S-PLUS 6.1 for Windows Professional Edition (Lucent Technologies, Inc. Insightful Corp. 2002).

2.3 Results

Experiment 1: Chronosequence analysis comparing the response of woody and herbaceous species to cable and ground-based logging.

Adequacy of the Experimental Design

The relationship between quadrat age and their score on a fitted vector for quadrat age through the two dimensional ordination space was generally a linear one, with the exception of the 120 year old Islet control quadrat. (Figure 2.12).

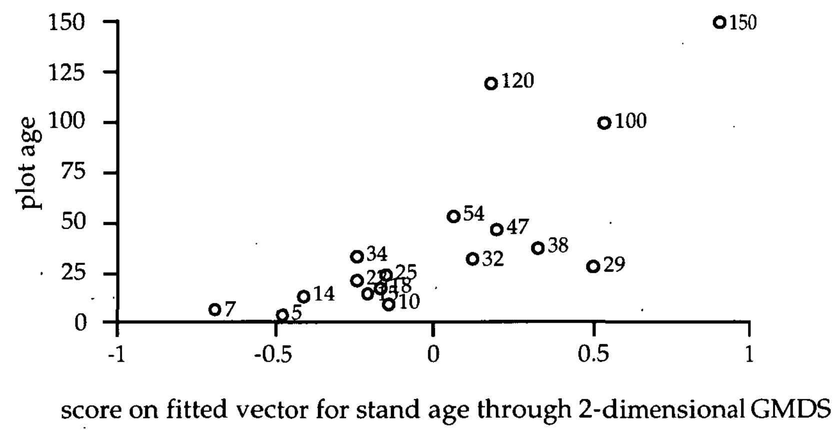


Figure 2.12 *Quadrat ages in years since logging and scores on fitted vector for stand age (extracted from the 2-dimensional global MDS ordination space) for the wet forest cable logged chronosequence.*

In addition, the vector fitting analysis (Table 2.17, Figure 2.13) indicated that none of the environmental variables tested had obscured the experimental response. Altitude, climate, radiation, rainfall, slope, geology or rock cover were not correlated with the recovered pattern of compositional dissimilarity. Only aspect and the cover of bare ground were significantly correlated with the pattern of compositional dissimilarity recovered by the ordination (Figure 2.14). However neither were correlated with the direction of the stand age vector, and are therefore not expected to confound the analysis. Quadrat age had a correlation coefficient of 0.78 ($p=0.003$), indicating the strength of the temporal sequence recovered from the chronosequence. From this analysis, the experimental design followed is considered satisfactory, in terms of elucidating the experimental response.

Table 2.17 *Maximum correlations and significance levels for variables derived from fitting vectors into the two dimensional ordination for the cable logged and unlogged wet forest chronosequence. *p< 0.05, ** p< 0.01*

Variable	Correlation coefficient
stand age	0.78**
cover of ground ferns	0.76**
Dicksonia cover	0.75**
richness of obligate fern epiphytic ferns	0.71**
richness of epiphyte-substrate (phorophyte) associations	0.71**
richness of ferns growing as epiphytes	0.70**
richness of epiphytic shrub species	0.68**
cumulative cover of ground stratum	0.66*
dominance	0.65*
cover of obligate epiphytic ferns	0.65*
abundance	0.57*

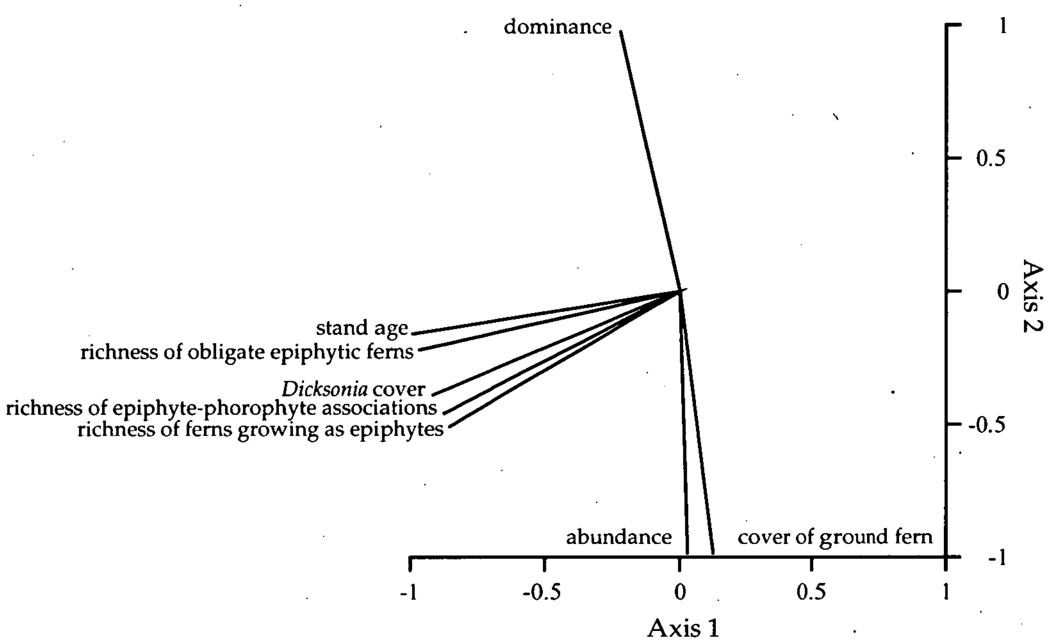


Figure 2.13 *Significant vectors (p<0.05) of maximum correlation fitted into the two dimensional global MDS ordination of cable logged and unlogged wet forest floristic. See Table 2.17 for correlation coefficients.*

Patterns of Secondary Succession in Cable Logged Wet Forest

Wet forest logged using cable technology formed a continuum of floristic change directly related to the age of the stand from the advanced ages of regrowth through to the mature unlogged forest (Figure 2.12, 2.14). Extracting the compositional trend along the fitted vector for stand age through the two dimensional ordination indicated the young more recently logged quadrats were characterised by species such as *Cyathodes glauca* and *Gahnia grandis*, through species characteristic of the first decades of regrowth (eg. *Phebalium squameum*, *Zieria arborescens*) and those of older regrowth and unlogged forest such as the epiphytic fern species (Table 2.18). Some species, while present in young regrowth, do not become abundant until more advanced regrowth or unlogged forest (eg. *Olearia argophylla*, *Dicksonia*) others show a reverse trend of decreasing abundance with age (eg. *Zieria*). Other species such as the herb *Galium australe* and the climber *Clematis aristata* had cover scores independent of stand age. A few species were only present in the unlogged stands (eg. *Polyphlebium venosum*).

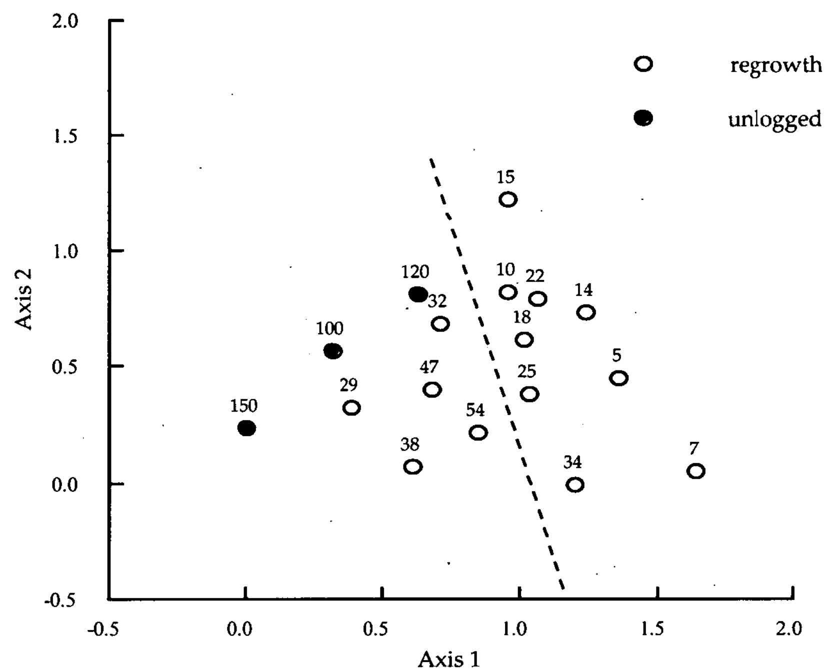


Figure 2.14 *Two dimensional MDS ordination of cable logged and unlogged wet forest, minimum stress in two dimensions is 0.1579. The dotted line indicates the principal division of the TWINSPAN classification. Logged quadrats are indicated as open circles, unlogged quadrats as closed circles, and quadrat age as symbol labels.*

Early successional colonist species including shrubs, graminoids, grasses, forbs and herbs are uncommon or absent from unlogged forest (Table 2.18). All of the ground ferns in unlogged forest are present in regrowth forest, although their covers differed. All of the obligate epiphytic ferns present in unlogged forest were present in regrowth forest, several of them only becoming apparent in regrowth forest over 41 years of age. Composition, alone, is not an effective way of discriminating between the logged and unlogged quadrats. Most of the differences are in terms of relative species cover (Table 2.18). The primary division (TWINSPAN eigenvalue = 0.194) in the age related floristic continuum recovered by the ordination occurs at age 29 years (Figure 2.14). Quadrats greater than 29 years of age are more similar to unlogged quadrats than the younger aged cable logged quadrats.

The vector for stand age is close to the first axis and has a correlation coefficient of 0.78, suggesting much of the floristic compositional variation along the first axis is associated with the age of the stand (Figure 2.13 and Table 2.17). Vectors associated with the vector for stand-age collectively describe aspects of the epiphytic fern flora (Table 2.17). Several are highly correlated, for example the angle between stand age and richness of obligate epiphytic ferns is only 2.9^{o}, with increasing age, the forest has a greater number of obligate epiphytic ferns present. Trends in species dominance and abundance in the regenerating stands were unrelated to age since cable logging. Where the dominance was elevated, the cover of ground ferns was reduced ($r=-0.70$, $p=<0.01$, Table 2.19), the cover of forbs and herbs was reduced ($r=-0.61$, $p=<0.05$) and total vegetation cover was reduced ($r=-0.74$, $p=<0.01$).

Species richness following cable logging increases with stand age, although the linear relationship is not strong (Figure 2.15, Table 2.19). Species richness reached a peak at an intermediate stage of stand development, declining to 16-26 species in unlogged stands. The reduced richness of the unlogged quadrats is due to the absence of species that characterise young regrowth, while the richness of the intermediate aged stands was due to the simultaneous expression of species of tall shrubs, forbs and herbs and epiphytic ferns (Table 2.19).

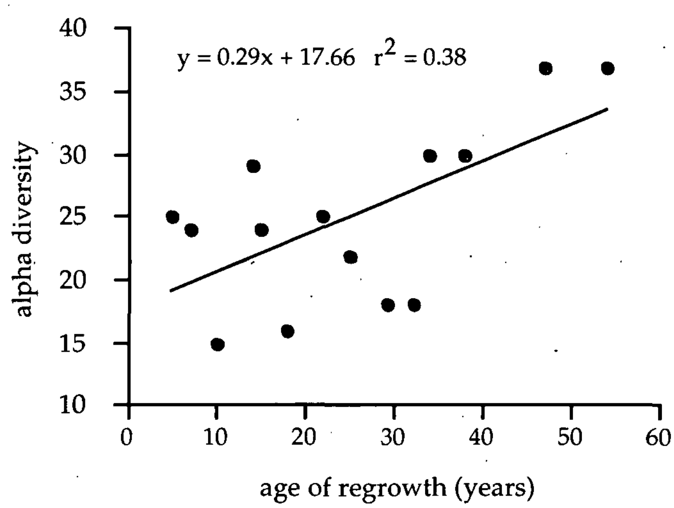


Figure 2.15 *Species richness for cable logged wet forest. The richness of the unlogged control quadrats between 16 and 26 species. F=7.48 p=0.012*

Table 2.18 *Compositional trends in the cable logged wet forest chronosequence. The quadrats are ordered according to their scores along the vector fitted into ordination space for stand age and species are ordered according to their weighted average scores on that vector. Braun-Blanquet cover abundance scores are displayed.*

Acacia mucronata	+-+-------------
Juncus pauciflorus	2--+---+--------
Hypochoeris radicata	++--------------
Cyathodes glauca	2+1-1----+----+--
Agrostis spp.	+---1+----------
Eucalyptus delegatensis	1212-+-+--------
Prostanthera lasianthos	-3---------1----
Dianella tasmanica	-+1-------------
Cassinia aculeata	-++---+---------
Histiopteris incisa	5-++5-+2-1++12111
Blechnum wattsii	+-1--------+----
Leptospermum lanigerum	--+--+----------
Gahnia grandis	1111+-++1+-1+--1+
Phebalium squameum	3-3-4-5-----+----
Pimelea drupacea	111-2+---+-121--+
Acaena novae-zelandiae	-1-+---+-+-------
Galium australe	-1+--+-+-+--+1-+-
Zieria arborescens	-312412-2---11-+2
Gnaphalium collinum	--+-----+-------
Luzula meridionalis	---++-----------
Chiloglottis cornuta	----1-----------
Hypolepis rugosula	2---3--3-++--+1-2
Gnaphalium spp.	---+-+----------
Uncinia tenella	---+2--------+---
Acacia dealbata	33223+51221222211
Echinopogon ovatus	--+1--++----+----
Atherosperma moschatum	11--1-+---++1++1+
Asplenium flabellifolium	-----+---1-------
Nothofagus cunninghamii	1---1-------+----
Billardiera longiflora	-++--+--1--------
Senecio spp.	++-++--+-+---++++
Carex appressa	----+-----------+
Drymophila cyanocarpa	----1-------+----
Hydrocotyle hirta	111-2++++2---1-1+
Viola hederacea	-1-+----++---+-+-
Eucalyptus regnans	23333+33334323323
Pomaderris apetala	-5541553542451341
Pteridium esculentum	-3++1+-12++-1+11-
Bedfordia salicina	--------3--3-----
Urtica incisa	-------+-+------+
Pterostylis spp.	---------+-------
Poa tenera	--1+1+--+1----1+--
Acacia melanoxylon	+--------3---3---
Oxalis corniculata	----+----2---1++-
Ctenopteris heterophylla	---------+--+-+--
Grammitis magellanica ssp. noth.	---------+--+----
Monotoca glauca	---------1--+----
Hymenophyllum peltatum	-----------+----+-
Corybas diemenicus	---------1---1---
Deyeuxia spp.	---+----++---+---
Acianthus exsertus	-----------++----
Pittosporum bicolor	--+---------1++--
Aristotelia peduncularis	------------+----
Pterostylis longifolia	------------+----
Hymenophyllum rarum	---------1+11+-1-
Coprosma quadrifida	-+1+1+---1+122-1+
Australina pusilla	------++-----1--+
Clematis aristata	-+-+1+-+-1+-+1+1+
Cardamine gunnii	-------------+---
Uncinia riparia	-------------+---
Olearia argophylla	-+++1+11-242+1423
Hymenophyllum cupressiforme	---------1+12---1
Polystichum proliferum	+21+2+1+-21+34424
Tmesipteris obliqua	------------+--+-
Blechnum nudum	--+---+--------2-
Eucalyptus obliqua	---------------1-
Hydrocotyle sibthorpioides	---------------+-
Polyphlebium venosum	---------------1-
Dicksonia antarctica	1-1+2+1++1++22344
Microsorum diversifolium	--1+++-+-+1-+---2
Grammitis billardieri	-----+---11+1++12
Asplenium bulbiferum	---------++-----2
Hymenophyllum flabellatum	---------+--+--12
Rumohra adiantiformis	---------+------1

Table 2.19 *Pearson product-moment correlation analysis of the wet forest cable logged quadrat data. (n=14) The top figure is the correlation coefficient, the bottom figure its significance level. A positive correlation indicates the variables vary in the same direction and a negative correlation indicates they vary in the opposite direction.*

	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
1 Age	.92 <.001	.76 <.01	n.s.	.80 <.001	.85 <.001	.92 <.001	.62 <.05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.82 <.001	.61 <.05	n.s.
2 No. of ferns growing as epiphytes	-	.76 <.01	n.s.	.86 <.001	.82 <.001	.96 <.001	.64 <.05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.57 <.05	n.s.	n.s.	n.s.	.91 <.001	.60 <.05	n.s.
3 No. of epiphytic shrubs		-	.57 <.05	.55 <.05	.86 <.001	.74 <.001	.78 <.001	-.55 <.05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.67 <.01	.74 <.01	.59 <.05	n.s.	n.s.
4 No. of epiphytic herbs			-	n.s.	.61 <.05	n.s.	.59 <.05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.68 <.01	n.s.	n.s.	n.s.	n.s.
5 No. of substrates colonised by epiphytes				-	.58 <.05	.91 <.001	.64 <.05	n.s.	n.s.	.54 <.05	n.s.	n.s.	n.s.	n.s.	.63 <.05	n.s.	n.s.	n.s.	.82 <.001	.7 <.001	n.s.
6 Mean height of tallest *Dicksonias*					-	.79 <.001	.70 <.01	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.68 <.01	.75 <.01	.69 <.01	n.s.	n.s.
7 No. of epiphyte-substrate associations						-	.73 <.01	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.93 <.001	.76 <.01	n.s.
8 Richness							-	n.s.	n.s.	n.s.	.68 <.01	n.s.	n.s.	n.s.	n.s.	n.s.	.71 <.01	.58 <.05	.59 <.05	n.s.	n.s.
9 Dominance								-	-.58 <.05	n.s.	n.s.	n.s.	-.67 <.01	n.s.	n.s.	-.70 <.01	n.s.	-.61 <.05	n.s.	n.s.	-.74 <.01
10 Trees >15m (Cover)									-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.57 <.05
11 Tall shrubs 8-15 m (Richness)										-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
12 Shrubs 1-8 m (Richness)											-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
13 Graminoids < 0.3 m (Richness)												-	.58 <.05	n.s.	n.s.	.63 <.05	n.s.	n.s.	n.s.	n.s.	n.s.
14 Graminoids < 0.3 m (Cover)													-	.57 <.05	n.s.	.81 <.001	n.s.	n.s.	n.s.	n.s.	.84 <.001
15 *Dicksonia* cover														-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
16 Ground fern richness															-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
17 Ground fern cover																-	n.s.	n.s.	n.s.	n.s.	.72 <.01
18 Forbs and herbs (Richness)																	-	.76 <.01	n.s.	n.s.	n.s.
19 Forbs and herbs (Cover)																		-	n.s.	n.s.	n.s.
20 No. of obligate fern epiphytes																			-	.76 <.01	n.s.
21 Cover of obligate fern epiphytes																				-	n.s.
22 Total cover																					-

Chronosequence Analysis of Ground-Based Logging in Wet Forest

Adequacy of the Experimental Design

The relationship between quadrat age and their score on a fitted vector for quadrat age through the two dimensional ordination space indicated a linear trend was evident (Figure 2.16).

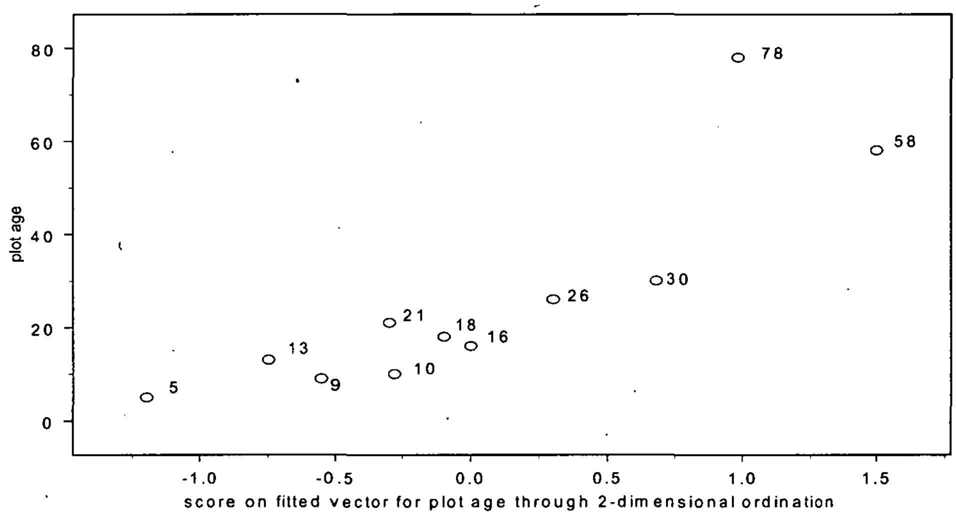


Figure 2.16 *Quadrat ages in years since logging and scores on fitted vector for quadrat age (extracted from the 2-dimensional hybrid MDS ordination space) for the wet forest ground-based logged chronosequence.*

Several climatic variables were correlated with the compositional pattern recovered by the ordination (Table 2.20 and Figure 2.17). Their effect is assumed to be negligible as the quadrats were selected from a limited geographic area which supports a community only present in relatively cool, wet environments. Quadrat age had a correlation coefficient 0.85 ($p<0.001$), indicating the strength of the temporal sequence recovered from the chronosequence. None of the environmental factors such as altitude, radiation, slope, geology or rockiness were correlated with the pattern of composition recovered. From this analysis, as well as several preceding analyses, it appears the experimental design used was satisfactory in terms of elucidating the experimental response, despite the possible influence of the climatic variables on the patterns recovered.

Table 2.20 *Maximum correlations and significance levels for variables derived from fitting vectors into the two dimensional ordination for the ground-based logged and unlogged wet forest chronosequence. *p< 0.05, ** p< 0.01, *** p<0.001.*

Variable	Correlation coefficient
stand age	0.85***
no. of substrates colonised by epiphytes	0.95***
fern epiphyte-substrate (phorophyte) richness	0.95***
richness of obligate epiphytic fern species	0.91***
richness of ferns growing as epiphytes	0.87***
annual rainfall range	0.86***
rainfall seasonality	0.78*
Acacia cover	0.78*
cover of litter	0.78*
rainfall of driest quarter	0.77*
richness of epiphytic shrubs	0.77*
annual temperature range	0.76*
Dicksonia cover	0.73*
dominance	

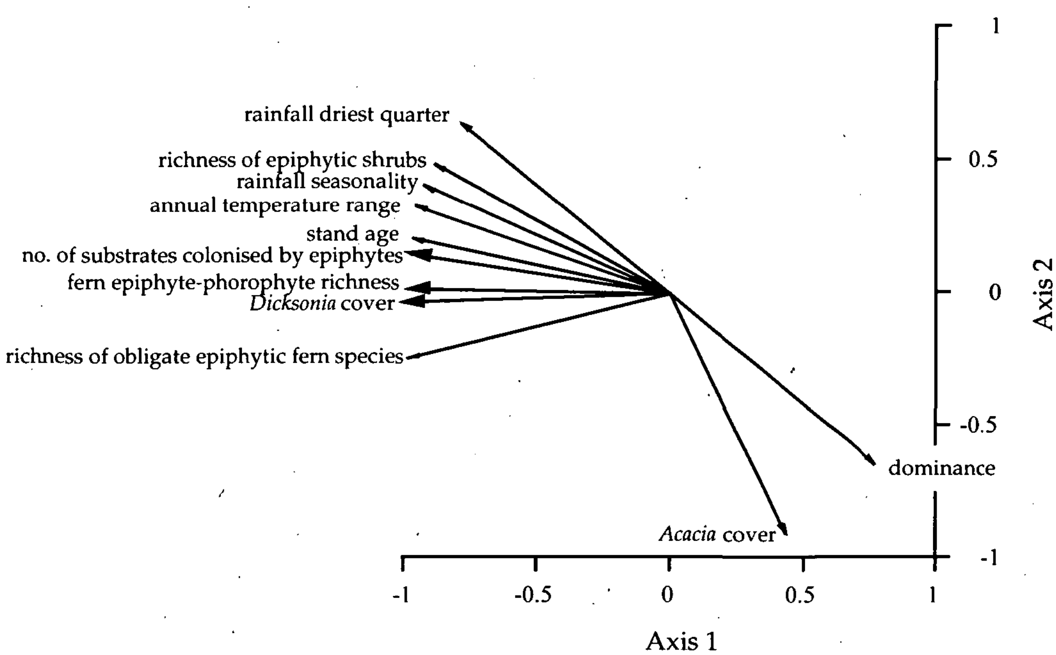


Figure 2.17 *Significant vectors (p<0.05) of maximum correlation fitted into the two dimensional hybrid MDS ordination of ground-based logged and unlogged wet forest floristics. See Table 2.20 for the correlation coefficients.*

Patterns of Secondary Succession in Ground-Based Logged in Wet Forest

Floristic compositional change in the nine ground-based logged and two unlogged quadrats was continuous (Figure 2.18). Vectors correlated with stand age were predominantly those describing the epiphytic community and the cover of *Dicksonia* (Figure 2.17). The principal division (from the first division of the classification) in this composition continuum was between logged and unlogged quadrats (Figure 2.18). With increasing stand age, litter cover and the number of fern species with an epiphytic habit increased

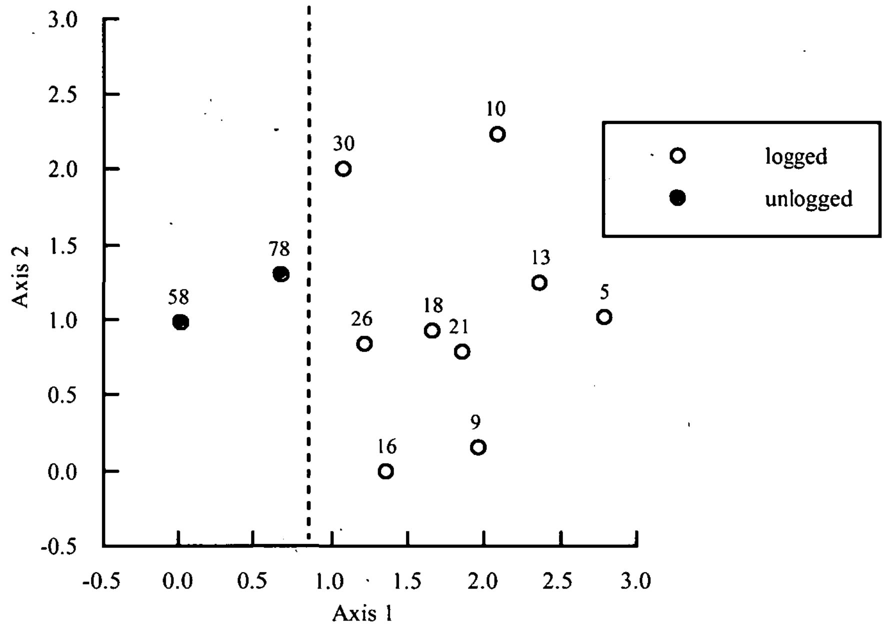


Figure 2.18 *Ground-based logged wet and mature forest hybrid multi-dimensional scaling ordination, stress = 0.1058 in two dimensions The dotted line indicates the principal division by the TWINSPAN classification. Logged quadrats are indicated as open circles, unlogged quadrats as closed circles, and stand age as symbol labels.*

Young ground-based logged stands were characterised by shrub species including *Cassinia aculeata* and *Helichrysum dendroideum*, through to species characteristic of the first decades of regrowth (e.g. *Acacia verticillata, Gahnia grandis, Monotoca glauca*) and those of older regrowth (e.g. *Olearia argophylla, Phebalium squameum*) (Table 2.21). Some of the species present in young regrowth in low abundance increased with time (e.g. *Polystichum proliferum, Dicksonia antarctica*), others decreased in abundance with increasing stand age (e.g. *Acacia dealbata, Gahnia grandis*). The presence of some species were unaffected by stand age (e.g. *Clematis aristata, Coprosma quadrifida*). A small number of species were only recorded in the unlogged quadrats (e.g. *Tmesipteris obliqua, Polyphlebium venosum, Ctenopteris heterophylla*).

The most abundant trees and shrubs in 5-30 year old ground-based logged regrowth were *Pomaderris apetala, Acacia dealbata, E. regnans* and *Phebalium squameum* (Table 2.21). With increasing stand age, the frequency and cover of the various understorey shrubs continuously changed, some of the shade tolerant species increased in cover (*Pimelea drupacea* and *Coprosma quadrifida*), while several species preferring more open understoreys decreased in cover (e.g. *Acacia verticillata, Helichrysum dendroideum* and *Cassinia aculeata*).

Graminoids, grasses, herbs and forbs exhibited a variable response to ground-based logging, with the overall trend being a decline in their frequency and abundance with increasing stand age (Table 2.22). Ground ferns, with the exception of *Blechnum nudum* and *B. wattsii*, were present in all of the age classes of regrowth surveyed. *Histiopteris, Hypolepis* and *Polystichum* became more abundant with increasing stand age, *Pteridium* became less abundant. The rhizomatous ferns were the most abundant, more shade tolerant non-rhizomatous species such as *Polystichum proliferum* and *B. wattsii* were less abundant. Species richness was un-related to stand age (Figure 2.19). Richness was elevated following ground-based logging in the early stages of secondary succession where low shrubs, grasses, herbs and forbs, species which are relatively poorly developed in unlogged forest, reached their peak. The correlation coefficients in Table 2.22 indicate that species richness was depressed where the tall shrub cover was high ($r=-0.80$, $p<0.01$ $n=9$). Species richness was elevated however where ground fern cover was high ($r=0.70$ $p<0.05$ $n=9$). The only plant group to be directly correlated with the trend in species richness was the richness of shrubs 1-8m tall ($r=0.83$, $p<0.01$, $n=9$).

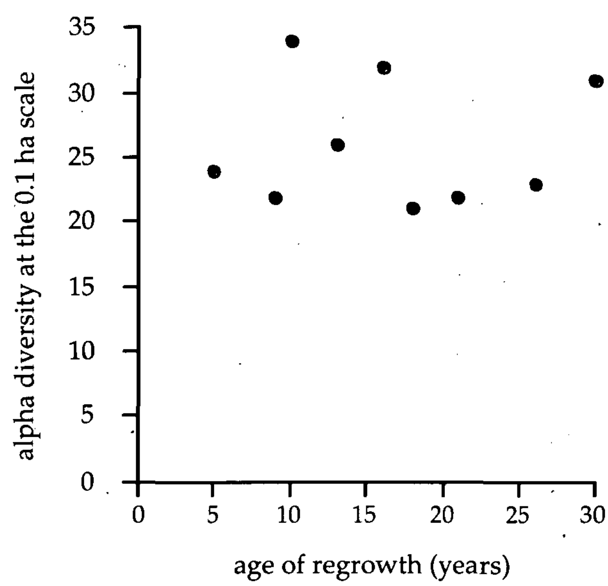


Figure 2.19 *Species richness for ground-based logged wet forest. The richness of the two unlogged control quadrats were 28 and 30 species.*

Table 2.21 *Compositional trend in species abundance of species recorded in the Arve Valley ground-based harvested steep country wet forest chronosequence. The quadrats are ordered according to their scores along the vector fitted into the ordination space for stand age and species are ordered according to their weighted average scores on that vector. The species are weighted by their cover scores. Braun-Blanquet cover abundance scores are displayed.*

Centaurium erythraea	+----------
Helichrysum dendroideum	2----------
Senecio linearifolius	+----------
Sonchus oleraceus	+----------
Deyeuxia spp.	+----------
Helichrysum spp.	+----------
Cassinia aculeata	2-2+--+--+-
Epilobium billardierianum	++---------
Juncus pauciflorus	++---------
Trochocarpa cunninghamii	++---------
Acaena novae-zelandiae	++----+----
Parsonsia brownii	-+---------
Hypochoeris radicata	+---+------
Senecio spp.	+++-+-+----
Urtica incisa	+----+-----
Lepidosperma elatius	--2---2----
Acacia melanoxylon	-1-1-+--+--
Gahnia grandis	12212211++1
Billardiera longiflora	+-2-+-1----
Hydrocotyle hirta	1++-1--+1--
Acacia verticillata	--3-2-3----
Pteridium esculentum	-31232222-2
Pimelea drupacea	-+11121211+
Cyathea australis	----1------
Caladenia spp.	----+------
Microtis spp.	----+------
Pterostylis pedunculata	-+++++1-++-
Acacia dealbata	3323+2133++
Monotoca glauca	---+2-+1-+-
Pomaderris apetala	5555-555+43
Chiloglottis cornuta	-+--1--+++-
Asplenium bulbiferum	-+-+-1+----
Correa lawrenciana	--2---2----
Corybas diemenicus	--1+++1+--+
Eucalyptus regnans	33332333333
Leptospermum lanigerum	----+-+----
Oxalis corniculata	------+----
Zieria arborescens	-+2-----1--
Clematis aristata	-+-+1+-1+++
Prostanthera lasianthos	------+---+
Leptospermum scoparium	----+-1-2--
Eucalyptus obliqua	----2---+--
Histiopteris incisa	-2-+321142-
Cyathodes glauca	----2---2+-
Hypolepis rugosula	-2-+32113+-
Nothofagus cunninghamii	----+---1+-
Rumohra adiantiformis	---+-1+-1++
Coprosma quadrifida	++1+11111+1
Thelymitra spp.	--------1+-
Phebalium squameum	----3-1-43-
Polystichum proliferum	+++++212231
Drymophila cyanocarpa	--------++-
Hymenophyllum cupressiforme	------++-++
Microsorum diversifolium	-++++1+++-+
Olearia argophylla	++++-2++-45
Dicksonia antarctica	-+++-2+2345
Pittosporum bicolor	---------++
Grammitis billardieri	-------+-12
Hymenophyllum flabellatum	-------+-12
Tmesipteris obliqua	---------+2
Hymenophyllum rarum	-------+-12
Atherosperma moschatum	+--+-+--+-4
Blechnum wattsii	-------+++3
Tasmannia lanceolata	---------++
Ctenopteris heterophylla	---------+1
Polyphlebium venosum	----------+

Table 2.22 *Pearson product-moment correlation analysis of the wet forest ground-based logged quadrat data. (n=9) The top figure is the correlation coefficient, the bottom figure its significance level. A positive correlation indicates the variables vary in the same direction and a negative correlation indicates the opposite direction.*

		2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23
1	Age	.80 .05	.68 .05	n.s.	.87 .01	.81 .01	.93 .001	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	-.69 .05	.72 .05	.82 .01	n.s.	n.s.	n.s.	.67 .05	.70 .05	n.s.	n.s.
2	No. of ferns growing as epiphytes	-	n.s.	n.s.	.73 .05	n.s.	.91 .001	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	-.76 .01	n.s.	.73 .05	n.s.	-.73 .05	n.s.	.93 .001	n.s.	n.s.	n.s.
3	No. of epiphytic shrubs		-	.70 .05	.87 .01	n.s.	n.s.	n.s.	-.77 .05	n.s.	.73 .05	n.s.	n.s.	n.s.	.90 .001	n.s.	n.s.	n.s.	n.s.	n.s.	.93 .001	n.s.	-.70 .05
4	No. of epiphytic herbs			-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
5	No. of substrates colonised by epiphytes				-	.76 .05	.92 .001	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	-.73 .05	.80 .01	n.s.	n.s.	n.s.	n.s.	n.s.	.82 .01	n.s.	n.s.
6	Mean height of tallest Dicksonias					-	.76 .05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	.78 .05	n.s.	n.s.	n.s.	n.s.	n.s.	.76 .05	n.s.	n.s.
7	No. of epiphyte-substrate associations						-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	-.79 .05	n.s.	.77 .05	n.s.	n.s.	n.s.	.80 .05	n.s.	n.s.	-.68 .05
8	Richness							-	n.s.	n.s.	n.s.	-.80 .01	.83 .01	n.s.	n.s.	n.s.	.70 .05	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
9	Dominance								-	n.s.	-.80 .05	n.s.	n.s.	n.s.	-.70 .05	n.s.	-.79 .05	n.s.	n.s.	n.s.	-.73 .05	n.s.	n.s.
10	Trees >15 m (Richness)									-	.82 .01	n.s.	n.s.	n.s.	n.s.	n.s.	.84 .01	n.s.	.74 .05	n.s.	n.s.	n.s.	n.s.
11	Trees >15m (Cover)										-	n.s.	n.s.	n.s.	.74 .05	n.s.	.88 .01	n.s.	.66 .052	n.s.	.74 .05	n.s.	-.77 .05
12	Tall shrubs 8-15 m (Cover)											-	-.68 .05	n.s.	n.s.	n.s.	-.79 .05	n.s.	-.68 .05	n.s.	n.s.	-.72 .05	n.s.
13	Shrubs 1-8 m (Richness)												-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
14	Graminoids > 0.3 m (Cover)													-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
15	Dicksonia cover														-	n.s.	n.s.	n.s.	n.s.	n.s.	1.00 .001	n.s.	-.86 .01
16	Ground fern richness															-	n.s.	n.s.	n.s.	.70 .05	n.s.	n.s.	n.s.
17	Ground fern cover																-	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
18	Forbs and herbs (Richness)																	-	n.s.	-.73 .05	n.s.	n.s.	n.s.
19	Forbs and herbs (Cover)																		-	n.s.	n.s.	n.s.	n.s.
20	No. of obligate fern epiphytes																			-	n.s.	n.s.	n.s.
21	Cover of obligate fern epiphytes																				-	n.s.	.86 .01
22	Climber richness																					-	n.s.
23	Total cover																						-

Chronosequence Analysis Comparing Cable and Ground-Based Logging in Wet Forest

The recovery of the wet forest floristic composition towards mature forest was broadly similar following either cable or ground-based logging. A weak trend was evident where a more rapid recovery of floristic composition towards mature forest occurred following ground-based logging for a given age of regrowth than following cable logging, using the fitted vector for stand age through a combined ordination of both the cable and ground-based wet forest chronosequences as a model of temporal change (Figure 2.20). The sensitivity of this weak trend was explored with the addition of the 32-54 year old cable logged quadrats to a further series of analyses (results not presented). The magnitude of this trend, in terms of ecological distance between their respective logged and unlogged quadrats, dissipated with increasing stand age.

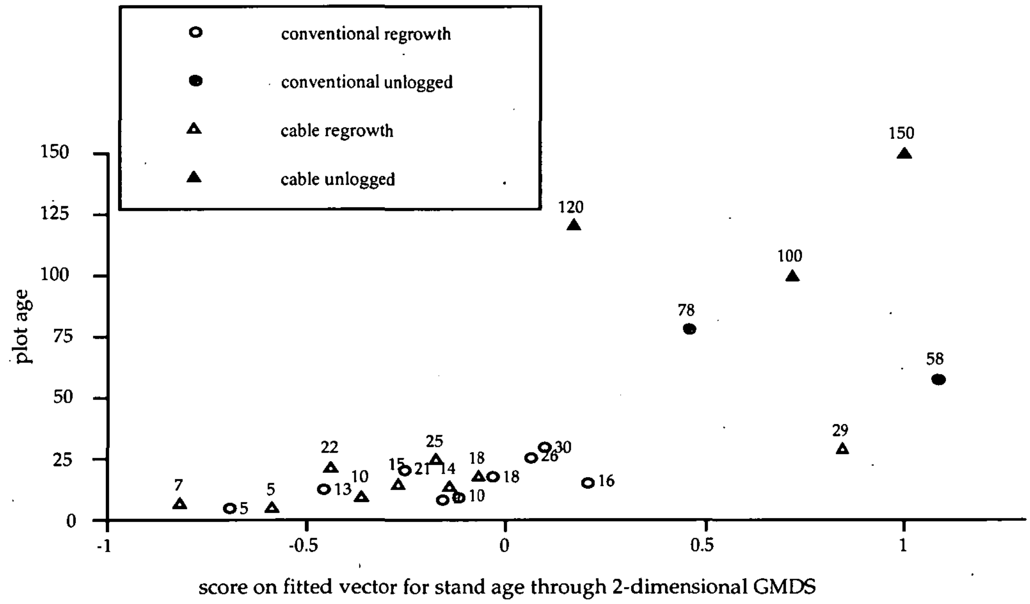


Figure 2.20 *Quadrat ages and scores on the fitted vector for stand age for the wet forest cable and ground-based (conventional) logged chronosequences.*

An pooled ordination of both chronosequences (Figure 2.21) indicated that compositional patterns of variation along axis 1 were associated with increasing stand age ($r = 0.71$ $p < 0.001$), and those floristic variables described previously for the individual chronosequences, for example the epiphytic flora (Figure 2.22). None of the variation in floristic composition was related to any of the measured environmental variables such as altitude, slope, aspect, or the temperature, radiation and rainfall parameters (Figure 2.22). The assumptions, inherent in comparing two chronosequences, have been met in terms of the control of environmental variation.

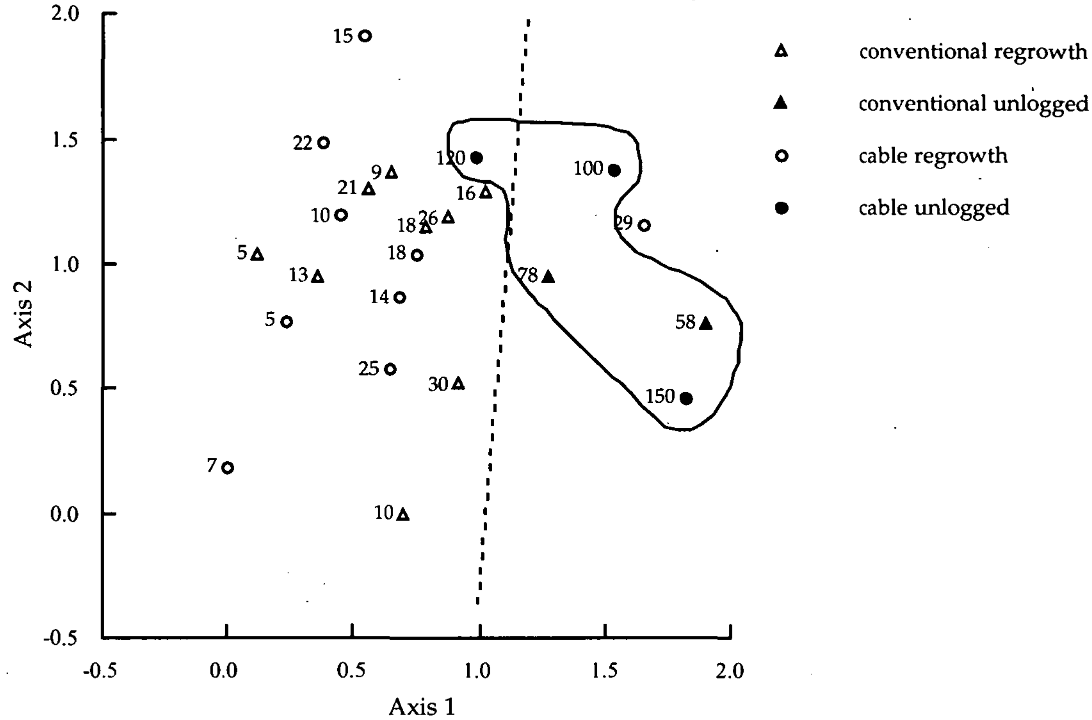


Figure 2.21 *Ordination of cable and ground-based (conventional) logged wet forest regrowth (5-30 years) and unlogged forest. Minimum stress = 0.1721. The unlogged quadrats are enlosed by the continuous line. The principal division of the classification of the quadrats is indicated by the dotted line. Quadrat age is in years.*

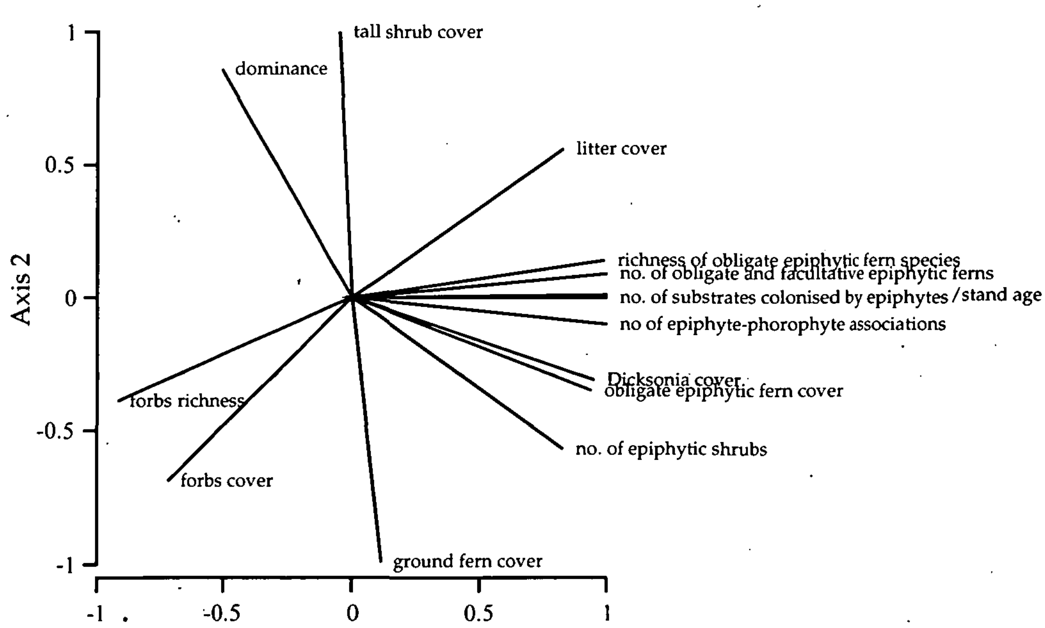


Figure 2.22 *Significant vectors (p<0.05) of maximum correlation fitted into the two dimensional ordination of cable and ground-based logged (5-30 year old) and unlogged wet forest floristics.*

While the response of wet forest to cable or ground-based logging was broadly similar, it differed in the response of particular species groups, in the abundance of individual species, and in the rates of secondary succession. The compositional differences between the two chronosequences are tabulated in Table 2.23, with the treatment effects extracted using indicator species analyses in Table 2.24.

Ground ferns responded differently to the logging treatments, *Hypolepis*, *Blechnum* and *Polystichum* increased their biomass more rapidly following cable logging, *Pteridium* did so following ground-based logging. *Histiopteris* however decreased its cover in cable logged regrowth more rapidly with age than in ground-based regrowth. Grasses, forbs and herbs were more abundant following cable logging than ground-based logging, especially in the early stages of secondary succession (Table 2.23). Orchid species are more abundant however in the ground-based logged quadrats (Table 2.23). Tall shrub species were better developed following ground-based logging, *Monotoca*, *Acacia melanoxylon*, *A. verticillata*, *Leptospermum*, and *Cassinia* were all more abundant in the first two decades.

In the 11-20 year old group *Gahnia grandis* reached its peak abundance in the ground-based regrowth quadrats, its cover was significantly less in cable regrowth when the two chronosequence regression equations were compared (Figure 2.23).

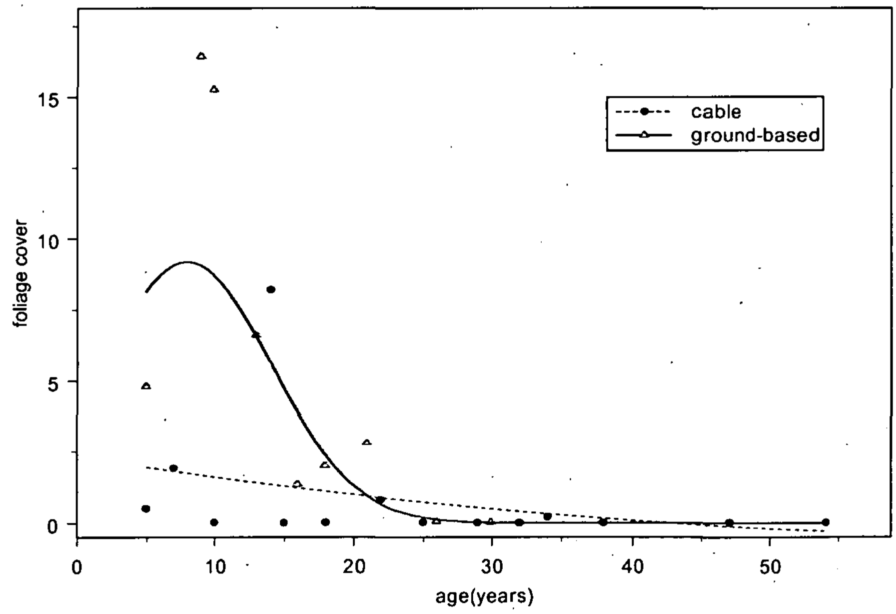


Figure 2.23 Cover of tall (> 0.3m) graminoids in the cable and ground-based logged wet forest chronosequences. The two regression equations were significantly different (ANCOVA comparing linear regressions, F = 7.187, p = 0.004, df = 2,19). In this figure an exponential curve if fitted to the ground-based data, and a linear least squares curve was fitted to the cable data.

The ground-based quadrats were particularly depauperate, in terms of grass and small graminoid species, compared to the cable quadrats in this age group.

When the temporal trends are ignored, and only the logging method is examined in the two chronosequences restricted to regrowth quadrats (5-30 years), the treatment differences between species groups are less apparent (Table 2.24). Both cable and ground-based logged quadrats include a range of small and large shrub and ground fern species as indicator taxa, with the only clear treatment difference being the prevalence of herbaceous species in the ground-based treatment. Clearly the treatment differences in floristic composition between the two chronosequences are largely related to the age group examined.

Table 2.23 *The floristic composition of the cable (n=9) and ground-based (n=9) logged wet forest and unlogged (n=5) chronosequences. n = the number of quadrats surveyed, the frequency (freq.)and mean % cover (cov.) scores were derived from 20 contiguous sub-plots within each quadrat.*

	cable logged						ground-based logged						unlogged	
	5-10 yr		11-20 yr		21-30yr		5-10 yr		11-20 yr		21-30yr			
	freq.	cov.	freq.	cov.	freq.	cov.	freq	cov.	freq.	cov.	freq.	cov.	freq.	cov.
Trees >15m at maturity														
Acacia dealbata	87	22	38	3	42	4	47	9	33	8	68	14	20	1
Acacia melanoxylon	8	<1	-	-	-	-	-	-	8	1	7	<1	-	-
Atherosperma moschatum	10	<1	<1	<1	<1	<1	5	<1	2	<1	<1	<1	16	8
Eucalyptus delegatensis	17	2	5	<1	2	<1	-	-	-	-	-	-	-	-
Eucalyptus obliqua	-	-	-	-	-	-	<1	<1	-	-	<1	<1	2	<1
Eucalyptus regnans	62	15	57	16	92	18	72	22	78	22	72	18	70	10
Eucryphia lucida	-	-	-	-	-	-	<1	<1	-	-	-	-	-	-
Nothofagus cunninghamii	5	<1	-	-	-	-	<1	<1	-	-	<1	<1	<1	<1
Phyllocladus aspleniifolius	-	-	-	-	-	-	3	<1	-	-	-	-	1	<1
Tall shrubs 8-15 m														
Bedfordia salicina	20	6	-	-	-	-	-	-	17	4	-	-	19	3
Olearia argophylla	<1	<1	12	<1	40	18	<1	<1	15	6	5	<1	54	19
Phebalium squameum	23	7	52	22	-	-	5	<1	<1	<1	30	15	6	2
Pittosporum bicolor	-	-	<1	<1	<1	<1	-	-	-	-	-	-	3	<1
Pomaderris apetala	67	45	95	48	73	29	65	47	100	59	67	46	65	35
Shrubs 1-8 m														
Acacia mucronata	3	<1	<1	<1	-	-	-	-	-	-	-	-	-	-
Acacia verticillata	-	-	-	-	-	-	25	5	23	4	-	-	-	-
Anopterus glandulosus	-	-	-	-	-	-	<1	<1	-	-	-	-	-	-
Aristotelia peduncularis	-	-	-	-	-	-	<1	<1	-	-	-	-	<1	<1
Cassinia aculeata	5	<1	2	<1	-	-	18	1	<1	<1	<1	<1	-	-
Coprosma quadrifida	<1	<1	5	<1	3	<1	10	<1	3	<1	15	<1	19	<1
Correa lawrenciana	-	-	-	-	-	-	5	<1	10	2	-	-	-	-
Cyathodes glauca	13	1	<1	<1	<1	<1	5	<1	-	-	5	1	<1	<1
Cyathodes juniperina	-	-	2	<1	-	-	-	-	-	-	-	-	-	-
Helichrysum dendroideum	-	-	-	-	-	-	10	1	-	-	-	-	-	-
Leptospermum lanigerum	-	-	13	2	-	-	<1	<1	2	<1	-	-	-	-
Leptospermum scoparium	2	<1	-	-	-	-	2	<1	7	1	3	<1	-	-
Melaleuca squarrosa	-	-	-	-	-	-	3	<1	-	-	-	-	-	-
Monotoca glauca	-	-	-	-	-	-	2	<1	<1	<1	2	<1	<1	<1
Pimelea drupacea	17	<1	3	<1	-	-	7	<1	12	<1	25	<1	3	<1
Prostanthera lasianthos	13	2	-	-	-	-	-	-	2	<1	-	-	<1	<1
Tasmannia lanceolata	-	-	-	-	-	-	-	-	-	-	-	-	1	<1
Zieria arborescens	52	10	7	1	10	<1	5	<1	3	<1	5	<1	7	<1
Shrubs < 1 m														
Trochocarpa cunninghamii	-	-	-	-	-	-	<1	<1	<1	<1	-	-	-	
Graminoids > 30 cm														
Carex appressa	-	-	-	-	-	-	-	-	-	-	-	-	<1	<1
Dianella tasmanica	2	<1	<1	<1	-	-	-	-	-	-	<1	<1	-	-
Gahnia grandis	17	<1	20	3	10	<1	45	12	22	3	8	<1	6	<1
Juncus pauciflorus	5	<1	-	-	7	<1	2	<1	2	<1	-	-	-	-
Lepidosperma elatius	-	-	-	-	-	-	7	<1	2	<1	-	-	-	-
Graminoids/grasses <30cm														
Agrostis spp.	5	<1	2	<1	-	-		-	-	-	-	-	-	-
Deyeuxia spp.	<1	<1	-	-	<1	<1		<1	-	-	-	-	-	-

Table 2.23 (cont.) *The floristic composition of the cable (n=9) and ground-based (n=9) logged wet forest and unlogged (n=5) chronosequences. N = the number of quadrats surveyed, the frequency and mean % cover scores were derived from 20 contiguous sub-plots within each quadrat.*

	cable logged						ground-based logged						unlogged	
	5-10 yr		11-20 yr		21-30yr		5-10 yr		11-20 yr		21-30yr			
	freq.	cov.	freq.	cov.	freq.	cov.	freq	cov.	freq.	cov.	freq.	cov.	freq.	cov.
Echinopogon ovatus	-	-	5	<1	3	<1	-	-	-	-	-	-	-	-
Isolepis spp.	2	<1	-	-	-	-	-	-	-	-	-	-	-	-
Juncus spp.	3	<1	-	-	-	-	-	-	-	-	-	-	<1	<1
Luzula meridionalis	-	-	2	<1	3	<1	-	-	-	-	-	-	-	-
Poa spp.	<1	<1	-	-	-	-	-	-	-	-	-	-	-	-
Poa tenera	<1	<1	2	<1	3	<1	-	-	-	-	-	-	-	-
Uncinia tenella	-	-	-	-	<1	<1	-	-	-	-	-	-	-	-
Tree ferns														
Cyathea australis	-	-	-	-	-	-	-	<1	-	-	-	-	-	-
Dicksonia antarctica	3	<1	3	<1	20	4		<1	17	<1	20	6	57	30
Ground ferns														
Blechnum nudum	-	-	<1	<1	-	-		<1	-	-	-	-	7	1
Blechnum wattsii	2	<1	13	<1	-	-	-	-	-	-	3	<1	13	3
Histiopteris incisa	30	20	2	<1	20	<1		4	13	3	27	8	13	<1
Hypolepis rugosula	13	2	-	-	27	8		<1	12	1	23	5	3	<1
Polystichum proliferum	12	<1	8	<1	32	6		<1	13	<1	18	2	32	12
Pteridium esculentum	20	2	2	<1	17	<1		9	47	8	28	2	4	<1
Forbs/herbs (non-gram.)														
Acaena novae-zelandiae	2	<1	-	-	3	<1	3	<1	5	<1	-	-	-	-
Acianthus exsertus	-	-	-	-	-	-	-	-	-	-	-	-	4	<1
Australina pusilla ssp. *muell.*	-	-	<1	<1	2	<1	-	-	-	-	-	-	<1	<1
Caladenia spp.	-	-	-	-	-	-	3	<1	-	-	-	-	-	-
Centaurium erythraea	-	-	-	-	-	-	<1	<1	-	-	-	-	-	-
Chiloglottis cornuta	-	-	-	-	-	-	5	<1	3	<1	<1	<1	1	<1
Corybas diemenicus	-	-	-	-	-	-	<1	<1	3	<1	7	<1	<1	<1
Drymophila cyanocarpa	-	-	-	-	-	-	-	-	-	-	<1	<1	<1	<1
Epilobium billardierianum	-	-	-	-	-	-	<1	<1	<1	<1	-	-	-	-
Galium australe	3	<1	<1	<1	<1	<1	-	-	-	-	2	<1	<1	<1
Gnaphalium collinum	<1	<1	<1	<1	-	-	-	-	-	-	-	-	-	-
Gnaphalium spp.	-	-	<1	<1	2	<1	-	-	-	-	-	-	-	-
Helichrysum spp.	-	-	-	-	-	-	3	<1	-	-	-	-	-	-
Hydrocotyle spp.	32	<1	7	<1	13	<1	12	<1	<1	<1	10	<1	6	<1
Hypochoeris radicata	3	<1	-	-	-	-	<1	<1	-	-	-	-	-	-
Microtis spp.	-	-	-	-	-	-	<1	<1	-	-	-	-	-	-
Oxalis corniculata	-	-	-	-	<1	<1	-	-	<1	<1	-	-	<1	<1
Pterostylis pedunculata	-	-	-	-	-	-	12	<1	13	<1	18	<1	<1	<1
Senecio linearifolius	-	-	-	-	-	-	2	<1	-	-	-	-	-	-
Senecio spp.	2	<1	-	-	5	<1	7	<1	2	<1	-	-	<1	<1
Sonchus oleraceus	-	-	-	-	-	-	<1	<1	-	-	-	-	-	-
Thelymitra spp.	-	-	-	-	-	-	-	-	-	-	12	<1	<1	<1
Urtica incisa	-	-	-	-	2	<1	<1	<1	<1	<1	-	-	<1	<1
Viola hederaceae	8	<1	-	-	<1	<1	-	-	-	-	-	-	<1	<1

Table 2.23 (cont.) *The floristic composition of the cable (n=9) and ground-based (n=9) logged wet forest and unlogged (n=5) chronosequences. N = the number of quadrats surveyed, the frequency and mean % cover scores were derived from 20 contiguous sub-plots within each quadrat.*

	cable logged						ground-based logged						unlogged	
	5-10 yr		11-20 yr		21-30yr		5-10 yr		11-20 yr		21-30yr			
	freq.	cov.	freq.	cov.	freq.	cov.	freq	cov.	freq.	cov.	freq.	cov.	freq.	cov.
Epiphytic ferns														
Asplenium bulbiferum	-	-	-	-	-	-	-	-	2	<1	<1	<1	4	<1
Asplenium flabellifolium	-	-	2	<1	-	-	-	-	-	-	-	-	-	-
Ctenopteris heterophylla	-	-	-	-	<1	<1	-	-	-	-	-	-	7	<1
Grammitis billardieri	-	-	2	<1	<1	<1	-	-	-	-	2	<1	33	1
Hymenophyllum cupressiforme	-	-	-	-	-	-	-	-	<1	<1	2	<1	20	<1
Hymenophyllum flabellatum	-	-	-	-	-	-	-	-	-	-	<1	<1	24	2
Hymenophyllum peltatum	-	-	-	-	-	-	-	-	-	-	-	-	<1	<1
Hymenophyllum rarum	-	-	-	-	-	-	-	-	-	-	<1	<1	25	1
Microsorum diversifolium	-	-	2	<1	2	<1	3	<1	2	<1	5	<1	7	<1
Polyphlebium venosum	-	-	-	-	-	-	-	-	-	-	-	-	<1	<1
Rumohra adiantiformis	-	-	-	-	-	-	-	-	2	<1	5	<1	7	<1
Tmesipteris obliqua	-	-	-	-	-	-	-	-	-	-	-	-	11	<1
Climbers														
Billardiera longiflora	5	<1	<1	<1	-	-	8	1	2	<1	-	-	-	-
Clematis aristata	3	<1	<1	<1	3	<1	7	<1	<1	<1	13	<1	21	<1
Parsonsia brownii	-	-	-	-	-	-		-	<1	<1	-	-	-	-

Table 2.24 *Indicator species associated with either the cable or ground-based logging treatments for regrowth forest quadrats aged 5-30 years. Species are ordered by their maximum indicator value with indicator values ≥30 shown. Monte carlo randomisation tests for indicator values are shown only for species with p values ≤0.05.*

species	treatment with maximum indicator value	cable logged indicator value	ground-based logged indicator value	*p*
Eucalyptus delegatensis	cable	67	0	0.009
Poa tenera	cable	67	0	0.008
Polystichum proliferum	cable	65	27	
Hydroctyle hirta	cable	60	15	
Olearia argophylla	cable	59	19	
Zieria arborescens	cable	58	4	
Echinopogon ovatus	cable	44	0	
Histiopteris incisa	cable	39	33	
Galium australe	cable	39	1	
Clematis aristata	cable	38	22	
Juncus pauciflorus	cable	32	1	
Senecio spp.	cable	30	25	
Atherosperma moschatum	cable	30	14	
Coprosma quadrifida	ground-based	4	90	0.005
Gahnia grandis	ground-based	13	84	0.031
Pterostylis pedunculata	ground-based	0	78	0.002
Pteridium esculentum	ground-based	11	76	
Corybas diemenicus	ground-based	0	67	0.009
Pimelea drupacea	ground-based	16	58	
Eucalyptus regnans	ground-based	44	56	
Acacia dealbata	ground-based	49	51	
Pomaderris apetala	ground-based	40	49	
Dicksonia antarctica	ground-based	38	45	
Monotoca glauca	ground-based	0	44	
Rumohra adiantiformis	ground-based	0	44	
Chiloglottis cornuta	ground-based	0	44	
Acacia melanoxylon	ground-based	1	41	
Billardiera longiflora	ground-based	9	36	
Leptospermum scoparium	ground-based	0	33	
Cassinia longifolia	ground-based	0	33	
Acacia verticillata	ground-based	0	33	
Hypolepis rugosula	ground-based	20	31	

The 21-30 year old year old ground-based quadrats were particularly noticeable for their poor development of *Olearia argophylla* (Figure 2.24), a species favoured by epiphytes due to its bark characteristics. After cable logging, *Olearia* reached a maximum cover value in the 5-30 year age group in excess of 40% (Figure 2.25), in ground-based regrowth of the same age it was less than 1% (Figure 2.24). The correspondingly low frequency figures suggests the species is also uncommon in ground-based regrowth.

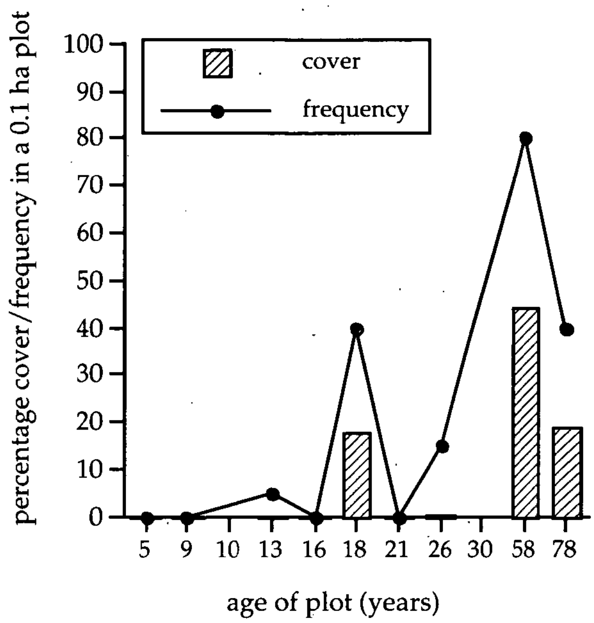


Figure 2.24 *The percentage frequency and cover of Olearia argophylla in the ground-based logged (5-30 years) and unlogged (58 and 78 years) wet forest quadrats.*

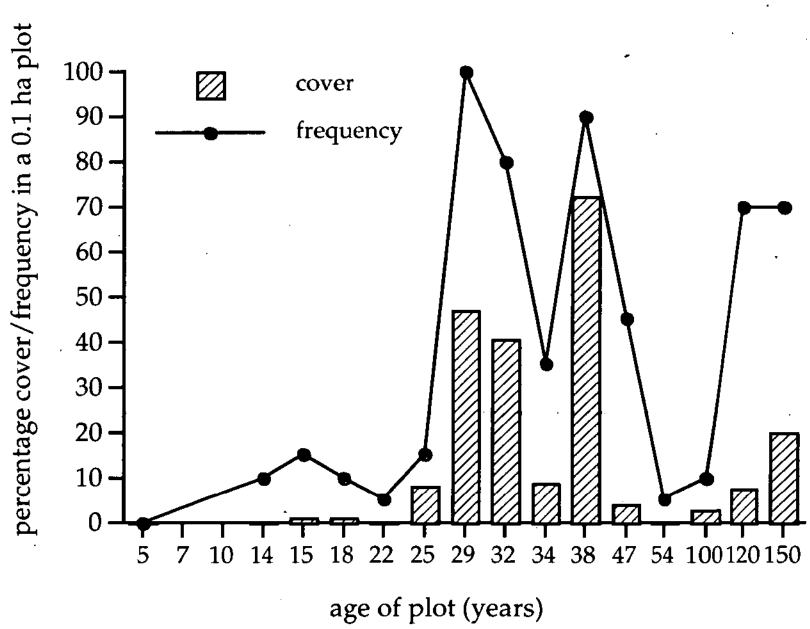


Figure 2.25 *The percentage frequency and cover of Olearia argophylla in the cable logged (5—54 years) and unlogged (100, 120, 150 years) wet forest quadrats.*

Experiment 2: Impacts of cable and ground-based logging on the frequency and cover of woody and herbaceous vascular in wet and dry forests

Short Term Effects of Cable Logging in Wet Forest

The short terms effects of clearfelling with a cable system, followed by a high intensity burn on woody and herbaceous species in four permanent established quadrats are summarised in Table 2.25. Cable logging led to the near complete removal of the existing vegetation cover, followed by an intensive slash burn. Soil disturbance from cable yarding accounted for 5-20% of the logged area (Figure 2.26). It was localised in extent where soil plowing occurred when inadequate lift was available to logs, especially where their yarding was interrupted by stumps and boulders.

Figure 2.26 *Localised soil plowing and rutting (see Figure 1.14) occurred where inadequate lift was available to logs, especially where their yarding is interrupted by stumps and boulders. Blue 09 experimental area, Wayatinah, 1992.*

Approximately one-year following logging and burning, a substantial change in species composition occurred, associated with a 32% increase in species richness on the two quadrats (late mature growth stage) with *Atherosperma* and *Nothofagus* in their small tree stratum, and a 10-30% decrease in species richness in the two quadrats (mature growth stage) with an understorey of tall shrubs and ferns. Species which were indicative of the pre-treatment composition of the vegetation include a range of herb, shrub and tree species of mature or late successional wet forest (Table 2.25). Following cable logging and burning, these species either decreased in importance or were absent from the re-measure of the paired quadrats (Table 2.25). The suite of woody and herbaceous species indicative of the post-logging and burning environment (Table 2.26) included relatively short lived ruderal species such as *Rorippa gigantea*, *Senecio bisseratus*, and *Solanum lacinatum*. Numerous species, primarily herbaceous such as *Rorippa gigantea* (Figure 2.27) were absent from the experimental area prior to treatment (Table 2.25). Ruderal species were conspicuous, covering up to one-third of the ground layer in some quadrats (Table 2.25). Monte carlo tests of significance of the observed maximum indicator values were significant ($p<0.05$) only for *Acacia dealbata*, *Rorippa gigantea*, *Senecio bisseratus* and *Solanum lacinatum* after cable logging and burning (Table 2.26), and for *Clematis aristata* prior to treatment (Table 2.25). Weed species increased in number and cover. In total, there were twenty-nine woody and herbaceous species prior to treatment, and forty-five after it. Twenty-five species were new to the quadrats after treatment, and nine woody and herbaceous species were absent (Table 2.25).

Figure 2.27 *Rorippa gigantea* (in flower) was prolific one year following cable logging and burning at Pearce 04, Wayatinah and was absent prior to treatment.

Table 2.25 *Short term effects (< 2 years) of cable logging on woody and herbaceous species at four long term monitoring quadrats at Wayatinah. The mean percent cover score is based on 20 sub-plots per quadrat.*

	Blue09 q. 2		Blue09 q. 24		Pearce04 q. 5		Pearce04 q.12	
Species	pre	post	pre	post	pre	post	pre	post
Acacia dealbata	-	11.88	0.75	7.09	0.28	37.25	-	39.25
Acacia melanoxylon	-	-	0.01	-	-	-	-	-
Acacia mucronata	-	0.01	-	-	-	-	-	-
Atherosperma moschatum	32.13	1.75	10.63	0.01	0.98	-	52.68	1
Australina pusilla subsp. *muell.*	-	2.5	-	-	-	-	-	-
Blechnum nudum	-	-	-	-	-	0.01	-	12.56
Cassinia aculeata	-	1	-	-	-	0.01	-	1
Chiloglottis spp.	-	-	-	-	0.05	-	-	-
Cirsium arvense	-	0.01	-	-	-	-	-	-
Cirsium vulgare	-	1.6	-	10.5	-	-	0.01	1
Clematis aristata	0.05	-	0.05	-	0.53	-	0.05	-
Coprosma hirtella	-	2.5	-	0.01	-	-	-	1
Coprosma quadrifida	0.1	0.01	0.05	-	0.15	-	0.33	1
Cyathodes glauca	-	-	-	-	-	1	-	0.01
Deyeuxia spp.	-	-	-	-	-	-	-	0.01
Dianella tasmanica	-	0.01	-	-	-	1	-	-
Drymophila cyanocarpa	14.3	9.44	29.5	2.5	1.8	1.75	8.63	5.25
Eucalyptus obliqua	-	-	-	-	-	-	-	1.21
Eucalyptus regnans	1.63	1	1.75	1.17	6.18	6.27	13.88	-
Gahnia grandis	-	1	-	-	-	1	0.01	1
Galium australe	-	-	-	-	0.01	-	-	-
Gnaphalium collinum	-	0.01	-	0.01	-	-	-	-
Gonocarpus teucriodes	-	-	-	-	-	0.01	-	1.33
Histiopteris incisa	0.01	0.01	0.05	1	0.05	-	0.75	1
Hydrocotyle hirta	0.01	-	0.01	-	0.43	19.4	-	-
Hydrocotyle sibthorpiodes	0.05	1.75	-	1	0.1	-	0.05	15.89
Hypochoeris radicata	-	-	-	-	-	-	-	1
Hypolepis rugosula	-	0.01	-	-	0.01	-	-	0.01
Luzula meridionalis	-	1	-	-	-	-	-	-
Monotoca glauca	-	-	-	-	-	-	-	0.01
Notelaea ligustrina	-	-	-	-	0.01	-	-	-
Nothofagus cunninghamii	1.55	-	-	-	-	-	34.8	0.01
Olearia argophylla	6	14	12.13	2.5	1.8	-	-	-
Oxalis corniculata	-	0.01	-	-	-	-	0.05	-
Pimelea drupacea	-	1.5	0.01	-	0.23	-	0.01	1
Pittosporum bicolor	-	-	-	-	0.05	-	-	-
Poa tenera	-	-	-	-	-	0.01	-	2.5
Polystichum proliferum	8.98	15.69	6.38	1	5.35	0.01	0.01	2
Pomaderris apetala	3.68	24.36	11.25	15.97	32.75	43.5	4.5	67.63
Pseudanthus luteus	-	-	-	1	-	-	-	-
Pteridium esculentum	-	-	-	-	0.28	1	0.13	-
Rorippa gigantea	-	2.75	-	7.62	-	2.42	-	1
Sonchus maschinus	-	-	-	-	-	-	-	1
Senecio biseratus	-	41.75	0.05	44.66	-	9.65	-	22.29

Table 2.25 (cont.) *Short term effects (< 2 years) of cable logging on woody and herbaceous species at four long term monitoring quadrats at Wayatinah. The mean percent cover score is based on 20 sub-plots per quadrat.*

	Blue09 q. 2		Blue09 quadrat 24		Pearce04 quadrat 5		Pearce04 quadrat 12	
Species	pre	post	pre	post	pre	post	pre	post
Senecio linearifolius	-	0.01	-	1	-	-	-	0.01
Senecio spp.	0.05	-	-	-	0.28	-	-	-
Senecio velliodes	-	0.01	-	-	-	-	-	0.01
Solanum lacinatum	-	32.5	-	1.3	-	1	-	1
Sonchus oleraceus	-	0.01	-	-	-	-	-	0.01
Taraxicum officianale	-	0.01	-	-	-	-	-	-
Uncinia tenella	-	-	-	-	0.01	-	-	-
Urtica incisa	-	1.25	-	13.17	-	-	-	-
Viola hederaceae	-	0.01	-	2.5	0.1	1	0.01	1.45
Zieria arborescens	-	-	-	-	0.05	-	-	-
total vascular species richness	23	34	21	19	27	18	21	31

Table 2.26 *Indicator species for quadrats subject to cable logging at the two Wayatinah experimental areas (Blue 09, Pearce 04) measured prior to and two years following cable logging and burning. Species are ordered by their maximum indicator value with indicator values ≥40 shown.*

species	pre-logging indicator value	post-logging indicator value
Clematis aristata	**100**	0
Atherosperma moschatum	**97**	2
Eucalyptus regnans	**74**	20
Dicksonia antarctica	**74**	26
Polystichum proliferum	**53**	47
Nothofagus cunninghamii	**50**	0
Olearia argophylla	**41**	23
Rorippa gigantea	0	**100**
Senecio biseratus	0	**100**
Solanum lasi	0	**100**
Acacia dealbata	1	**99**
Viola hederaceae	1	**98**
Cassinia aculeata	0	**75**
Cirsium vulgare	0	**75**
Coprosma hirtella	0	**75**
Gahnia grandis	0	**75**
Senecio linearifolius	0	**75**
Hydrocotyle sibthorpiodes	1	**74**
Blechnum nudum	0	**50**
Cyathodes glauca	0	**50**
Dianella tasmanica	0	**50**
Gnaphalium collinum	0	**50**
Gonocarpus teucriodes	0	**50**
Poa tenera	0	**50**
Senecio velliodes	0	**50**
Sonchus oleraceus	0	**50**
Urtica incisa	0	**50**
Pimelea drupacea	7	**45**

A two-dimensional ordination (Figure 2.28) of the paired treatment quadrats indicated a significant degree of floristic change after treatment, with a similar direction of change indicated by the vectors linking quadrats in the multi-dimensional space. The two quadrats from the late mature growth stage (Blue 09 quadrat 2, Pearce 04 quadrat 12) underwent a greater degree of floristic change after cable logging and burning than the quadrats from the mature growth stage forest.

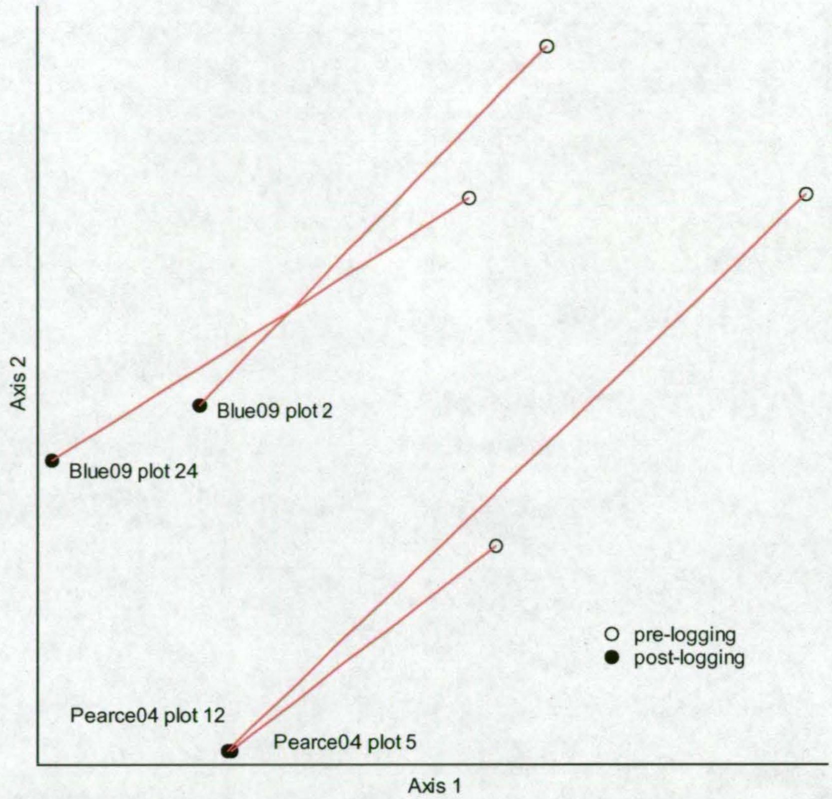


Figure 2.28 *Two-dimensional ordination of 8 paired quadrats using mean percent cover data for woody and herbaceous species. The quadrats were from two Wayatinah experimental areas (Blue 09, Pearce 04) and were measured prior to and two years following cable logging and burning. Minimum stress for the two-dimensional solution was 0.1670*

A further treatment effect was evident when the degree of dispersion of quadrats within treatment groups in the multi-dimensional space was examined (Table 2.27). Prior to treatment, the weighted within group average distance was 0.63, following cable logging and burning it was 0.55. Cable logging and burning reduced the floristic heterogeneity of the vegetation.

Table 2.27 *Average within group weighted distance scores (Multi-Response Permutation Procedures, or MRPP McCune and Metford 1999) for groups defined by before or after cable logging and burning at Wayatinah.*

treatment group	n	weighted average within group distance	difference
cable, pre-logging	4	0.6336	
cable, post-logging	4	0.5537	-0.0798

Comparitive Short Term Effects of Cable and Ground-Based Logging in Dry Forest

Two years after the cable and ground-based logging at Wielangta was completed, floristic variation within the experimental area lacked any significant dichotomy related to the treatment groups; cable logging, ground-based logging and control (Figure 2.30). Cluster analysis of all paired (before and after treatment) quadrats indicated that most paired quadrats classified at or near the lowest point in the dendogram with their pre-treatment pair, they had a greater degree of similarity with their pre-treatment composition than with other quadrats. Cable and ground-based logging therefore had not masked the initial pre-logging variability in vegetation composition across the experimental unit as a whole.

The association measure scores, which summarised the difference in floristic composition between quadrats into a single score, differed significantly ($p < 0.001$) between the three treatment group means (Table 2.28, Figure 2.29).

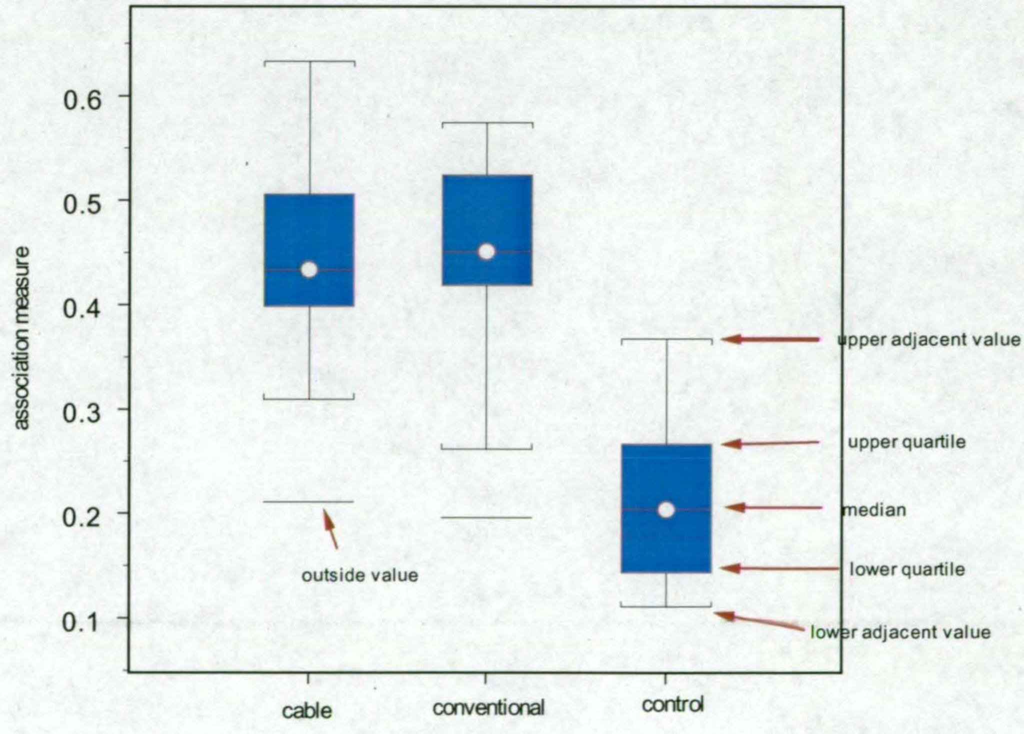


Figure 2.29 *Box-whisker plot of within treatment association measures for quadrats in the cable, ground-based and control portions of Wielangta 42a, measured prior to logging and 2 years post-logging. The association matrix was calculated using the Bray and Curtis measure by pairing the individual pre and post logging quadrats. The Bray and Curtis dissimilarity score varies from 0 to 1 with higher scores indicating greater compositional differences between the paired quadrats*

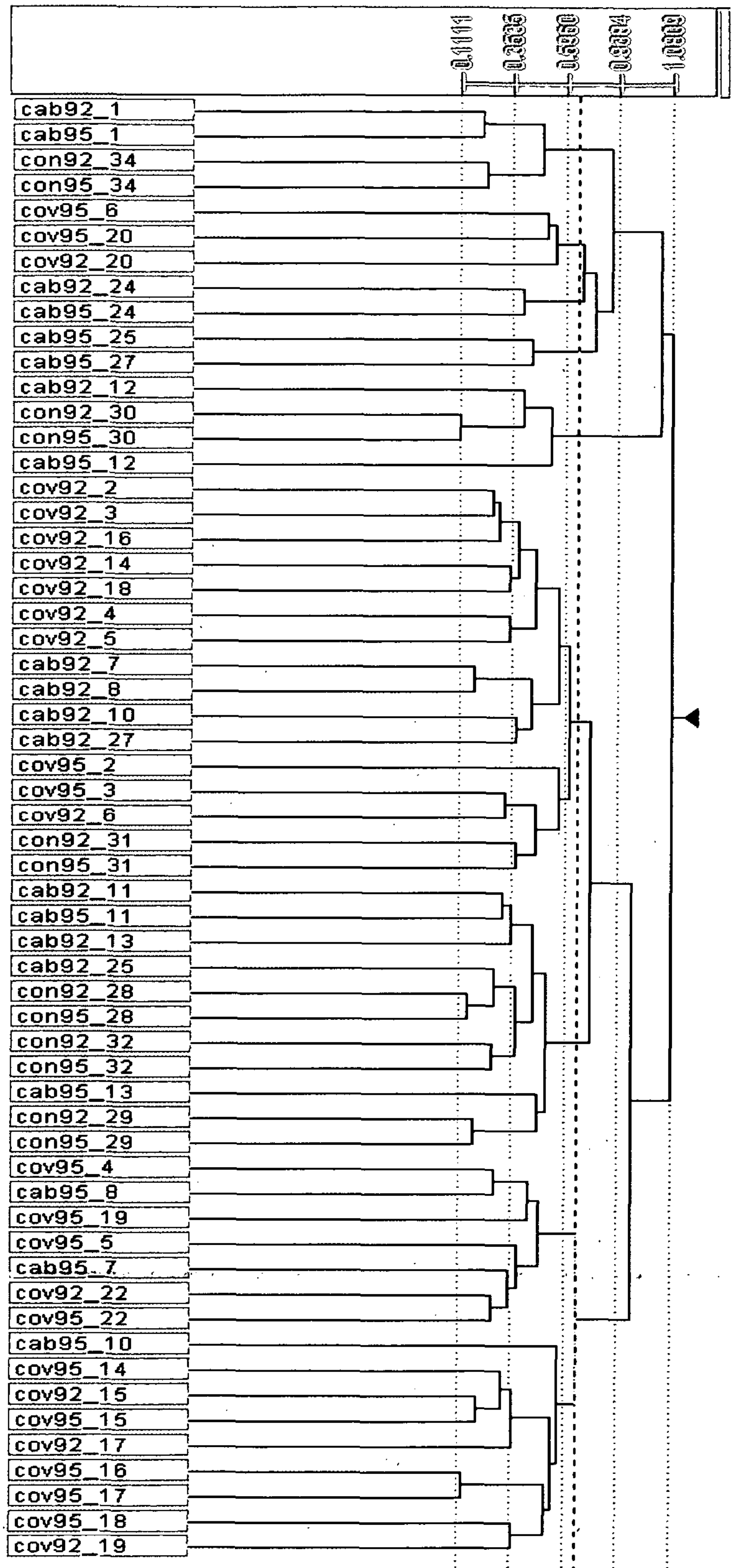


Figure 2.30 Cluster analysis (flexible UPGMA Beta, = 0.13) of paired treatment (cable = cab, ground-based =conv and control = con) quadrats in 1992/95 at Wielangta.

Table 2.28 *A one-way ANOVA of the Bray-Curtis association measures between paired quadrats and the treatments (cable, ground-based logging, and control)*

	Df	sum of squares	mean square	F value	probability of F
treatment	2	0.2388	0.1194	9.49	0.0008
residuals	26	0.3270	0.0125		

A multiple comparison procedure which adjusted for unequal variances between the treatments indicated that while the cable and ground-based logging treatments do not differ significantly based on their association measures, they both did from the controls (Table 2.29). While the treatment effect, here summarised by the association score, was of a greater magnitude in the ground-based portions of the Wielangta 42a experimental area than in the cable logged or control areas, this effect was not statistically significant.

Table 2.29 *A one-way ANOVA with multiple comparisons for the association distances between permanently established paired quadrats (before and after treatment) and the treatments (cable logging, ground-based logging, and control). The grand mean = 0.3936 and the critical value of t with 95% simultaneous confidence intervals adjusted for the sample sizes between treatments is 2.5589.*

treatment comparison	mean association distance	n	estimate	s.e.	lower bound	upper bound
cable v. ground-based	0.438	10	-0.0044	0.0472	-0.1250	0.116
cable v. control	0.442	13	0.2220	0.0579	0.0733	0.370 ****
ground-based v. control	0.216	6	0.2260	0.0554	0.0843	0.368 ****

A further treatment effect was evident when the degree of dispersion of quadrats within treatment groups in the multi-dimensional space was examined through MRPP (Table 2.30). In both logging treatments the weighted within group average distances increased post-logging, with a slightly greater magnitude of increase occurring in ground-based logged quadrats (0.07), compared to cable logged quadrats (0.05). Within group homogeneity is increasing after logging, suggesting the logging treatments are reducing floristic compositional variation. Floristic variation in the quadrats subjected to ground-based logging was reduced to a greater extent than those quadrats subject to cable logging. The least amount of change occurred in the control quadrats (0.04, Table 2.30).

Table 2.30 *Average within group weighted distance scores (Multi-Response Permutation Procedures, or MRPP McCune and Metford 1999) for the logging treatment groups at Wielangta 42a.*

treatment group	n	average group distance	difference
cable, pre-logging	10	0.5774	
cable, post-logging (1995)	10	0.6300	0.0526
ground-based, pre-logging	13	0.4711	
ground-based, post-logging (1995)	13	0.5438	0.0727
control, pre-logging	6	0.5758	
control, post-logging (1995)	6	0.6158	0.0400

The degree of floristic change resulting from the three treatments (cable, ground-based and control) was further evaluated with a two-dimensional ordination (Figure 2.31) of all paired quadrats. From the distances indicated by the vectors joining paired quadrats, ground-based logging imposed a greater degree of compositional change, and the control treatment the least. The magnitude of the cable logging treatment on composition was intermediate between the ground-based and control treatments. The treatment effect was variable however, with some of the cable logged quadrats also undergoing large changes in composition. There is a broadly similar direction of floristic change in vegetation subject to both logging treatments, where the direction of change is correlated with axis 1.

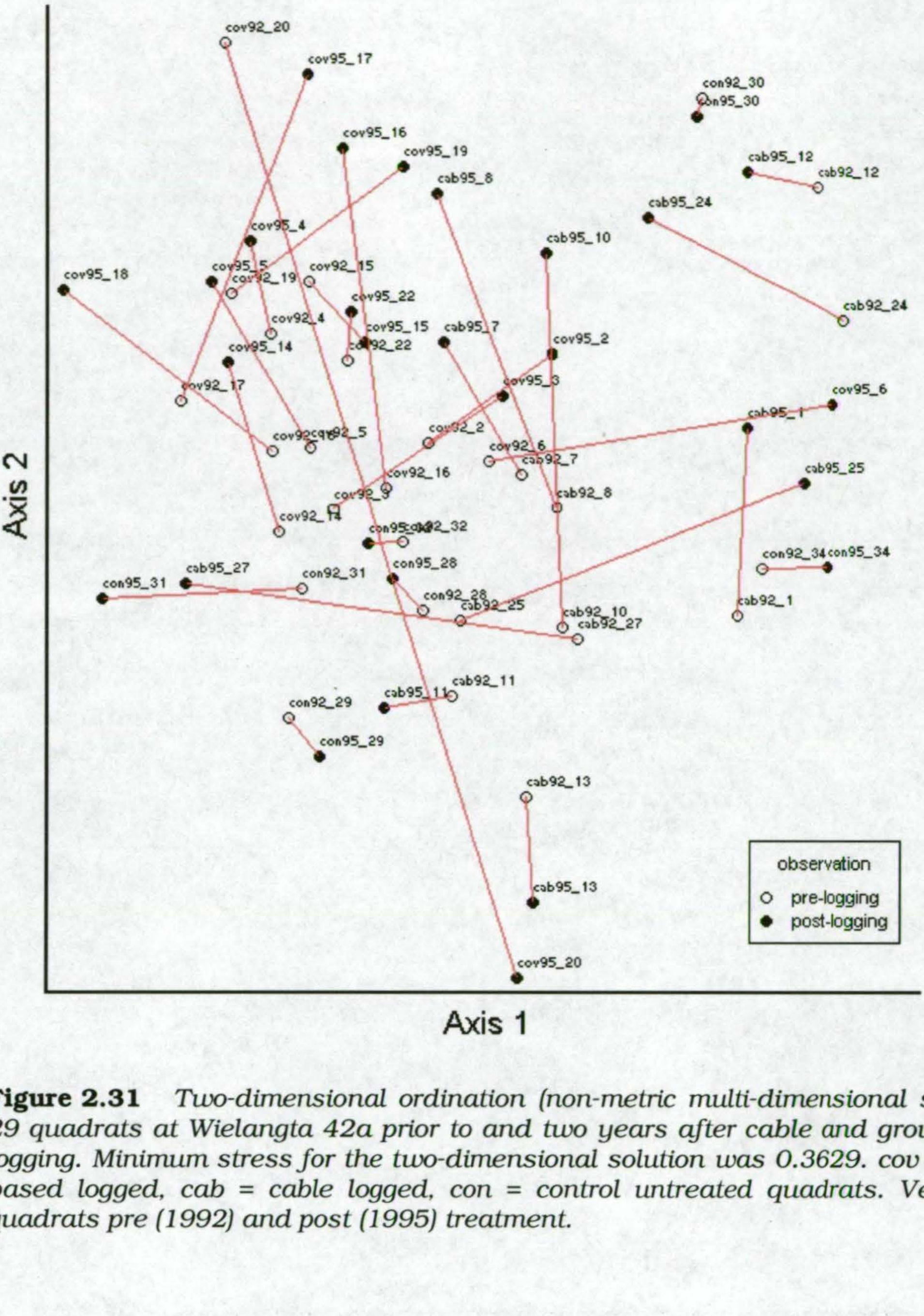


Figure 2.31 *Two-dimensional ordination (non-metric multi-dimensional scaling) of 29 quadrats at Wielangta 42a prior to and two years after cable and ground-based logging. Minimum stress for the two-dimensional solution was 0.3629. cov = ground-based logged, cab = cable logged, con = control untreated quadrats. Vectors join quadrats pre (1992) and post (1995) treatment.*

A further exploration of this trend was undertaken for a sub-sample of the Wielangta study quadrats which had been re-scored twice, ie. one year and two years after cable logging (Figure 2.32). Included was the control quadrats matched to those treatment quadrats on the basis of an analysis of their composition prior to logging. The results of this vector analysis, which can be viewed as portraying the length and direction of the successional change, varies between the dry scrubby quadrat (7) and the damp drainage line quadrat (12). The vector for the dry scrubby quadrat is multi-directional while the drainage line quadrat is closer to uni-directional. In both cases change in the control quadrats are of a lesser magnitude than change in the treatment quadrats, however change in the two control quadrats differs in their magnitude. The cable logging effect was of a greater magnitude in the drainage line environment than in dry scrubby forest above it.

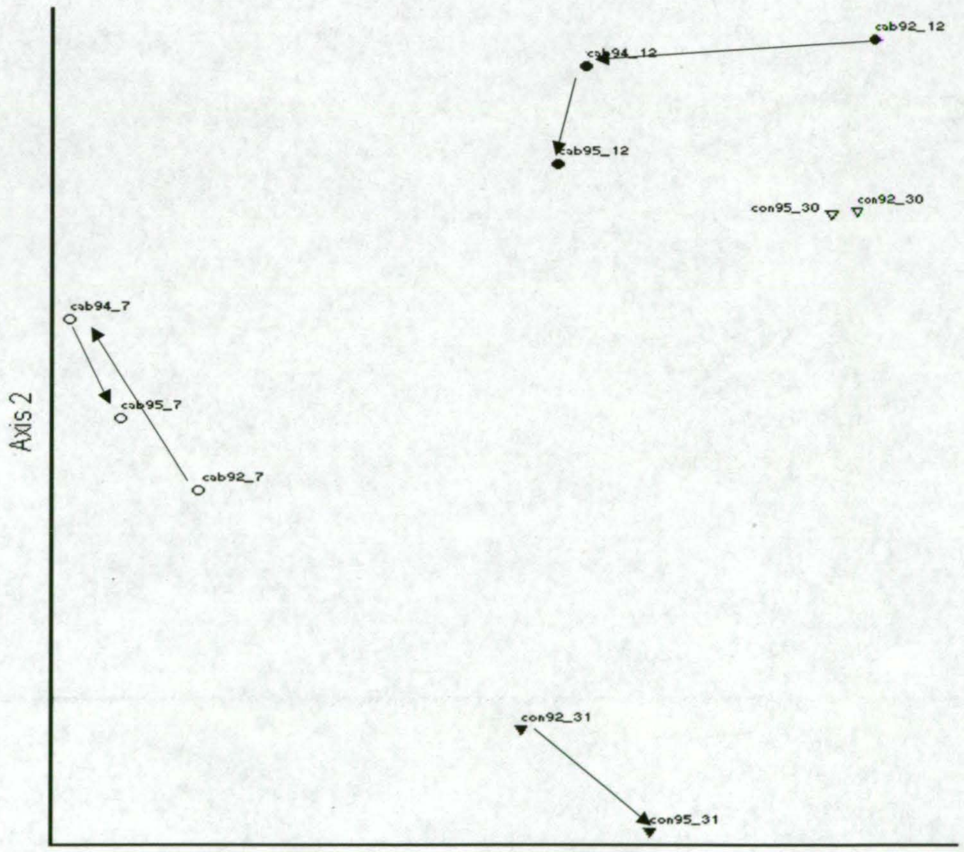


Figure 2.32 *Two-dimensional ordination (non-metric multi-dimensional scaling) of 10 quadrats at Wielangta 42a prior to, one year and two years post cable logging. Minimum stress for the two-dimensional solution was 0.06. cab = cable logged, con = control quadrats. Quadrat 7 was logged and matched with control quadrat 31* (Eucalyptus obliqua - E. delegatensis *dry forest group) and quadrat 12 (also logged) is matched with control quadrat 30 (*Beyeria viscosa - Olearia argophylla *damp closed scrub group on minor drainage line). Vectors join quadrats pre (1992) and post (1994, 1995) treatment.*

Species richness varied between observation periods for the three treatments (Figure 2.33). Considering that the three treatments varied with respect to species richness at the outset (treatments in operational studies of logging cannot practically be allocated at random across the experimental unit), the cable logged quadrats decreased in mean richness, the ground-based logged quadrats increased, and the control quadrats decreased. The degree of variability was greatest in the cable logged quadrats, both before and after treatment. None of the observed difference in species richness were statistically significant.

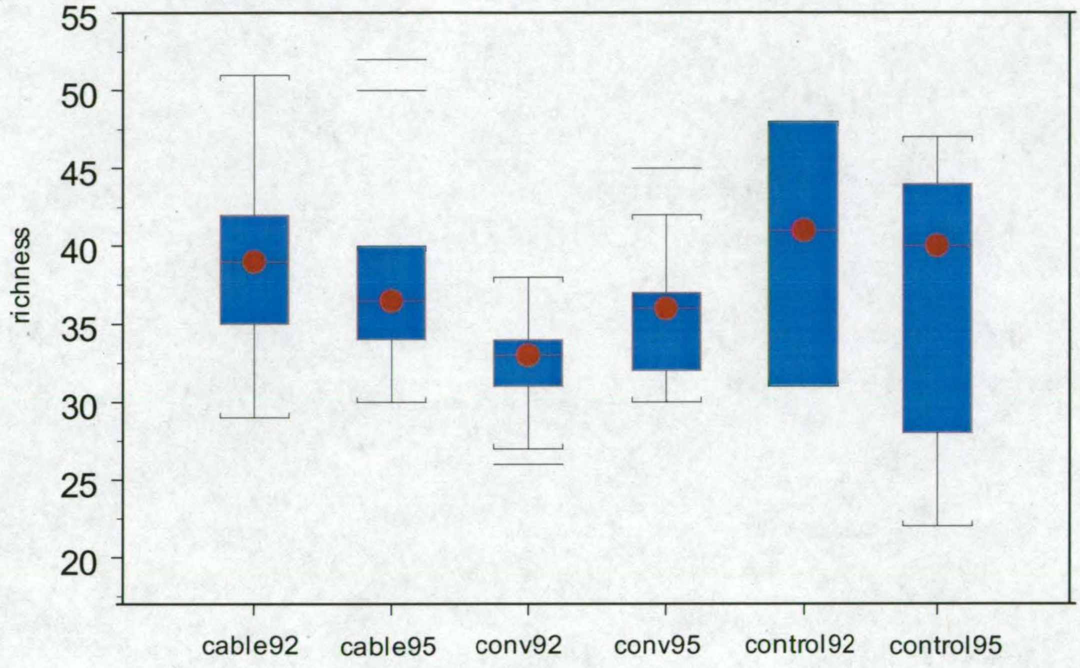


Figure 2.33 *Box-whisker plot of mean species richness at Wielangta 42a for the three treatments (cable, ground-based or conventional and control) and two observation periods (pre and two years post logging). Species richness at the 0.09 ha scale did not differ significantly (p=0.08, one-way ANOVA) between the observation periods.*

Table 2.31 *List of species which reach their maximum indicator value (>10) prior to cable logging in 1992 (cab92 in bold) compared with other treatments post cable logging (cab95) or before and after ground-based logging and the controls. Significant differences (where p<0.05) are given for particular indicator species.*

		treatment group indicator values						
species	average indicator value for all treatments	cab92	cab95	ground -based 92	ground- based 95	control92	control95	p
	n	10	10	13	13	6	6	
Pterostylis spp.	14	49	3	13	5	10	7	0.001
Drymophila cyanocarpa	7	40	1	0	0	1	2	0.010
Eucalyptus delegatensis	12	39	8	1	1	16	9	0.004
Galium australe	15	34	11	6	2	27	12	0.010
Blechnum wattsii	8	31	12	0	0	3	2	
Dianella tasmanica	15	31	15	20	7	13	3	
Pimelea cinerea	5	30	0	0	0	0	0	0.090
Poa spp.	6	30	0	1	1	1	1	0.047
Deyeuxia spp.	16	28	9	12	3	27	15	
Cyathodes glauca	17	26	14	16	3	22	19	
Acianthus exsertus	7	24	1	2	3	0	15	
Aristotelia peduncularis	5	24	2	0	0	1	1	
Leucopogon collinus	8	24	11	1	0	9	1	
Hydrocotyle hirta	14	22	12	21	9	16	4	
Gahnia grandis	15	19	13	19	14	14	12	
Coprosma hirtella	12	18	10	16	3	18	11	
Leptospermum scoparium	4	18	8	1	0	0	0	
Hypericum gramineum	3	15	1	2	0	0	0	
Lagenifera stipitata	3	14	0	2	0	0	0	
Olearia lirata	3	13	2	3	1	0	0	
Dicksonia antarctica	4	11	1	0	0	8	1	
Asplenium bulbiferum	2	10	0	0	0	0	0	
Cyathea australis	4	10	7	0	0	4	4	
Goodenia lanata	2	10	0	0	0	0	0	
Lobelia gibbosa	2	10	0	0	0	0	0	
Lycopodium varium	2	10	0	0	0	0	0	
Pittosporum bicolor	2	10	0	0	0	0	0	

Quadrats classified according to treatment type were assessed for species considered indicative of a particular treatment outcome. Species which reached their maximum indicator value prior to cable logging included shrubs (*Pimelea cinerea, Cyathodes glauca, Aristotelia peduncularis*), the ground fern (*Blechnum wattsii*) and herbaceous species (*Dianella tasmanica, Galium australe, Drymophila cyanocarpa, Acianthus excertus*, Table 2.31).

After cable logging many of these species were considerably reduced in importance. The composition of the cable logging quadrats after treatment saw species such as *Viola hederaceae, Helichrysum dendroideum, Pteridium esculentum, Cassinia aculata and Stackhousia monogyna* increasing in relative importance (Table 2.32). Weed species were present but were infrequent.

Table 2.32 *List of species which reach their maximum indicator value (>10) after cable logging in 1995 (cab95 in bold) compared with other treatments pre cable logging (cab92) or before and after ground-based logging and the controls. Significant differences (where p<0.05) are given for particular indicator species.*

		treatment group indicator values							
species	average indicator value for all treatments	cab92	cab95	ground -based 92	ground-based 95	control92	control95	p *
Viola hederacea ssp. *hederacea*	12	16	20	11	15	6	3	
Helichrysum dendroideum	4	0	20	1	1	0	0	
Pteridium esculentum	11	14	16	8	14	7	7	
Cassinia aculeata	3	2	14	1	4	0	0	
Stackhousia monogyna	4	0	14	0	0	4	4	
Cirsium arvense	3	3	13	0	0	0	0	
Blechnum nudum	5	8	11	0	1	4	4	
Acacia mucronata var. mucronata	2	0	10	0	0	0	0	
Bauera rubiodes	2	0	10	0	0	0	0	

Low shrub species reached their maximum indicator value in areas subjected to ground-based logging prior to treatment (Table 2.33). After ground-based logging there was a significant reduction in these species (Table 2.34). Species such as *Opercularia varia, Epacris impressa* and *Billardiera longiflora* which reached their maximum indicator value prior to ground-based logging, were surpassed by ruderals such as *Senecio* species, *Acaena nova-zelandie, Agrostis spp.* and *Isolepis* spp. Weed species including *Cirsium vulgare, Hypochaeris radicata,* and *Gnaphalium collinum* reached their maximum indicator values on the ground-based logged portion of the experimental area.

Table 2.33 *List of species which reach their maximum indicator value (>10) before ground-based logging in 1992 (ground-based* **92** *in bold) compared with other treatments post cable logging (cab95) or before and after cable logging and the controls. Significant differences (where p<0.05) are given for particular indicator species.*

	treatment group indicator values							
species	average indicator value for all treatments	cab92	cab95	**ground-based 92**	ground-based 95	control92	control95	p *
Poranthera microphylla	12	20	6	**39**	4	2	2	0.008
Epacris impressa	14	25	9	**32**	7	4	6	
Opercularia varia	7	8	1	**30**	2	1	0	
Gonocarpus teucrioides	16	19	16	**25**	16	14	9	
Eucalyptus obliqua	17	18	9	**23**	10	20	21	
Lagenifera stipitata	16	15	11	**23**	17	20	13	0.045
Monotoca glauca	4	0	0	**23**	0	0	0	0.050
Correa reflexa	13	13	12	**22**	15	9	9	
Eucalyptus globulus ssp. globulus	14	13	0	**22**	11	20	17	
Comesperma volubile	9	4	4	**22**	10	12	3	
Lomatia tinctoria	14	16	16	**16**	11	16	10	
Lobelia rhombifolia	3	0	0	**15**	0	0	0	
Billardiera longiflora	6	0	6	**14**	0	7	7	
Pimelea nivea	7	2	13	**13**	4	6	6	
Olearia ramulosa	4	1	3	**11**	7	2	2	

Table 2.34 *List of species which reach their maximum indicator value (>10) after ground-based logging in 1995 (ground-based* **95** *in bold) compared with other treatments pre ground-based (ground-based 92) or before and after cable logging and the controls. Significant differences (where p<0.05) are given for particular indicator species.*

	treatment group indicator values							
species	average indicator value for all treatments	cab92	cab95	ground-based 92	**ground-based 95**	control92	control95	p
Senecio spp.	13	7	16	1	**54**	0	0	0.001
Cirsium vulgare	9	2	9	0	**40**	2	2	0.001
Acaena novae-zelandiae	14	21	18	3	**31**	7	1	
Isolepis spp.	8	0	20	0	**31**	0	0	0.045
Hypochoeris radicata	11	8	12	16	**28**	1	1	0.013
Gnaphalium collinum	9	0	17	0	**28**	5	5	0.048
Acianthus exsertus	4	0	0	0	**23**	0	0	
Callistemon pallidus	7	4	0	14	**21**	2	2	
Senecio linearifolius	4	1	1	1	**15**	3	3	

2.4 DISCUSSION

Cable and ground-based logging differ principally in the way they disturb the physical environmental of the coupe, in particular the soil surface, the distribution and mass of woody debris, and species which rely on regenerative shoots or rootstock's to recolonise the site. Differences in the rate of secondary succession, the species composition of this succession and in the regenerative traits possessed by the species in the regenerating stand can be interpreted in this context.

The immediate response to cable logging and burning was an increase in species richness (compared to their pre-logging composition) within sites characterised by mature growth stage understoreys of wet forest shrubs and trees, and a reduction in species richness within sites characerised by late mature growth stage understoreys of rainforest elements (ie mixed forest, Gilbert 1959). This contrasting species richness response is interpreted as reflecting initial differences in composition and structure prior to cable logging, in accordance with the the initial floristic composition model of Bastow-Wilson *et al.* (1992). Early successional patterns observed on permanent quadrats showed dominance by wind dispersed annuals such as *Senecio* and *Rorripa* where the intense fire created abundant germination sites. Several of these species were not recorded in the vicinity of the permanent quadrats prior to logging. Typically these species form a transitory element to the regional flora, similar to that described following wildfire (Ashton and Martin 1996). Weed species increased in cover, however they were a minor component of the total vegetation cover, consistent with the observations of Appleby (1998). Patterns such as these are almost universal following intensive logging and burning (West and Chilcote 1968, Halpern *et al.* 1997), where the conditions required for the dominance of short-lived ruderal species (Parrotta *et al.* 2002) are created. Typically this phase of stand development persists for two to five years (Conde *et al.* 1983, Crow et al. 1991, Crowell and Freedman 1994). At this stage of early stand development the overall direction of floristic change described by successional vectors was similar across the replicate permanently monitored quadrats, floristic heterogeneity was reduced in the first two years compared to their pre-logging composition. Where the regenerating stand was dominated by a dense thicket of *Eucalyptus* and *Acacia* saplings (between 5 and 15 years after logging and burning), species richness, particularly for small shrubs and herbs was depressed. This is a typical response where dominance in regenerating forest stands is pronounced (Alaback 1982, Wohlgemuth *et al.* 2002). The initial or pioneer stage typically lasts around 15 years (Fuhr *et al.* 2001) until overstorey mortality and self thinning commences (Katagiri *et al.* 1985, Taylor *et al.* 1988, Fuhr *et al.* 2001).

Species richness following cable logging peaked at 40-50 years, at a time when elements of the early and late successional flora co-existed (sensu Denslow 1985) and composition appeared to be based on non-equilibrium mechanisms (Aubert *et al.* 2003). Species richness at these intermediate stages of stand development exceeded richness during the early or late stages, a trend consistent with the patterns of secondary succession described by Horn (1974) and with studies of forest regeneration following logging in south eastern Australia (Griffiths and Muir 1991, Binns 1993), in north American (Elliot *et al.* 1997, Schoonmaker and McKee 1988) and south America (Saldarriaga *et al.* 1988).

After cable logging in wet forest, where approximately 5-20% of the soil and litter was disturbed, compositional trends in the first fifty years were generally consistent with those described following ground-based logging and burning in south eastern Australian wet forests (Cunningham and Cremer 1965, Gilbert 1965, Jackson 1968, Ashton 1976, Ashton 1981, Hickey and Savva 1992, Mueck and Peacock 1992, Chesterfield 1980, Chesterfield 1996, Murphy and Ough 1997, Ough 2001). Floristic differences between the oldest regrowth sampled and unlogged forest were primarily in relative species cover, not composition. Most understorey species for example occurred in all stages of stand development, but differed in cover, a pattern similar to that described for softwood forests in western Oregon and Washington following clearfelling and burning (Halpern and Spies 1995). When Halpern (1989) also described an analogous scenario for clearcut and burned Oregon forests, early successional change was interpreted as a process of generally persistent species changing relative abundance with time. Where differences in composition persisted after cable logging in the wet forests studied, they were confined to a small number of epiphytic ferns which did not colonise regrowth stands, or the short lived post-fire colonists that were absent from unlogged stands.

After ground-based logging in wet forest, where soil disturbance and mixing was more prevalent, compositional trends in the first thirty years were generally similar to those following cable logging. The dynamics of secondary succession were however more pronounced, late successional specialists appeared at an earlier age following ground-based logging. Unlike cable logging, species richness following ground-based logging was unrelated to stand age.

The similarities between the effects of cable and ground-based logging in wet forest were greater than the differences between them. Wet forest logged with either system exhibited strong successional dynamics, and where measured in terms of the rate of species return and turnover, was greater following ground-based logging. Both chronosequences formed a clear floristic continuum with age, from the youngest (5 years) to the oldest regrowth (54 years) surveyed. Importantly, the differences between the two treatments described by their chronosequences became less evident with time. Patterns of species regeneration and recolonisation for many species converged (ie the compositional convergence of Chistensen and Peet 1984) at 30-50 years to the extent that differences for woody and herbaceous species existing primarily in relative species cover. While floristic composition at the community level and species richness did not identify differences between the two logging treatments, analyses conducted at the functional attribute or species group level and at the individual species level did. As a group for example, ground ferns recolonised after cable logging more quickly than after ground-based logging, with the exception of *Pteridium esculentum* which recolonised more rapidly following ground-based logging. Grasses, forbs and herbs were more abundant following cable logging than ground-based logging. Species which relied in part on resprouting from lignotuberous rootstock's such as *Olearia argophylla* were slower to recover following ground-based logging. Its relatively slow recovery in ground-based logged sites is of concern as it is one of the most commonly colonised bark substrates by vascular epiphytes in unlogged forest. The slow growth rates exhibited in the first thirty years after logging by *Trochocarpa laurina*, a tall shrub preferentially colonised by vascular epiphytes in wet forests of northern NSW, was similarly highlighted by Binns (1991) as being a possible hindrance for epiphytic recolonisation following logging.

Compared to wet forests, the differences in cover and species richness following cable or ground-based logging in dry forests were directly attributable to the logging treatment as both could be investigated within the one coupe and the additional imposition of a slash burn on the experiment could be avoided. When the magnitude of floristic change resulting from cable or ground-based logging was compared to floristic variation existing prior to logging in the dry forest experimental area, ground-based logging resulted in a greater degree of pre to post-logging change. Cable logged sites also underwent a reduction in floristic heterogeneity, and importantly so did the controls, although it was of a lesser degree. Both logging treatments however resulted in a broadly similar direction of floristic change, although the treatment effect varied within the type of vegetation present prior to logging, with drainage line vegetation within the dry forest coupe undergoing the greatest degree of change. Ground-based sites, with their direct soil and habitat disturbance, saw a marked increase in disturbance adapted ruderals and other wind dispersed colonisers, including weeds. Species richness increased accordingly following ground-based logging, but was reduced following cable logging, and was also reduced to a lesser extent in the controls. While trends in total species richness per quadrat were not particularly informative, trends in particular species functional groups were, a trend described elsewhere for studies comparing logging treatments (Gondard *et al.* 2003).

Species regenerating vegetatively following cable or ground-based logging and burning were in the minority. The vast majority relied upon either soil stored or wind dispersed seeds and spores. Those species which did regenerate vegetatively were infrequent and suffered high initial mortality. Only forty-three percent of *D. antarctica* stems survived the immediate effects of cable logging and burning and regenerated vegetatively. For *Atherosperma moschatum*, a common rainforest tree in the long term monitoring study area, 75% of the stumps cut during logging regenerated vegetatively after cable logging, yet three months after slash burning, only 3% of the monitored individuals had survived. An ability to perennate from subterranean structures was considered central to the ability of *Pseudotsuga menziesii* forests in Oregon to maintain resilience to disturbances such as logging (Halpern 1988). Some authors have hypothesised that plant resiliency traits which provide an adaptation to fire also confer resiliency to logging disturbance (Greenberg *et al.* 1995). The utility of this hypothesis is perhaps limited to environments where recurrent fires occur, the wet forests in south eastern Australia are infrequently burnt and the few studies which contrast the logging and wildfire response of wet forest vegetation have argued important differences in species composition and structure remain (Hickey 1994, Lindenmayer *et al.* 1990, Franklin and Lindenmayer 1997, Ough 2001, Lindenmayer and McCarthy 2002). Deterministic disturbances such as logging are also likely to have very different impacts compared to natural stochastic processes (Halpern 1988, McCarthy and Burgman 1995). Of greater relevance than the vital attributes (Noble and Slatyer 1980) of the recolonising or regenerating plant species is their interaction with the nature of the disturbance process, the post-logging environment and the biological legacies which remain from the logged stand.

While cable and ground-based logging differed in their site impacts, this impact did not appear to be expressed solely as an intensity or gap size effect in the vegetation response, a more common trend observed in logging studies in environments ranging from tropical (Salick *et al.* 1995) to temperate (northern European spruce forests, Hannerz and Hannell 1997) forests. In a comparison of woody vegetation responses to wildfire and different logging techniques, Carelton and MacLellan (1994) argued that the vegetation changed in response to both the qualitative nature and the intensity of the disturbance event. In this study, the patterns of vegetation change following either cable or ground-based logging were most divergent in the first two decades following logging, and a result of both the qualitative and quantitative differences between cable and ground-based logging systems. Using the interpretation of secondary succession proposed by Gilliam *et al.* (1995), a temporal shift existed in the processes influencing species composition following cable and ground-based logging from allogenic factors (e.g. severity of soil disturbance) to autogenic factors (e.g. stand structural characteristics). Gilliam *et al.* (1995) proposed that the degree to which logging alters vegetation was a function of how it alters the linkages between vegetation strata during secondary succession. Allogenic factors were of greater significance in the first decade following cable or ground-based logging. Autogenic factors were of more significance in the second and third decades following either cable or ground-based logging, where stand characteristics were being mediated by the tall shrub and small tree stratum, which following cable logging included a greater proportion of resprouting species, and following ground-based logging, a greater proportion of species regenerating from soil stored seed. These differences in vegetation strata in turn determined the ability of late successional specialists to occupy preferred substrates, such as the bark or caudexes of *Olearia argophylla* and *D. antarctica*.

Chapter 3 Effects of soil disturbance arising from cable and ground-based logging and thinning on vegetation

3.1 Introduction

The recovery of native forest after cable and ground-based logging varies in both the direction and rate of secondary succession (Chapter 2), in the diversity and cover of plant groups with different regenerative traits (Chapter 2), in the degree to which species susceptible to mechanical disturbance regenerate successfully (Chapter 4), and in frequency with which late successionary specialist species recover (Chapter 6). One of the principal ways in which the two logging methods differ is in the form of soil disturbance they create. Soil disturbance from cable or ground-based logging was examined by comparing a logging operation in mature vegetation which utilised both cable and ground-based yarding, and a regrowth thinning operation which utilised both cable and ground-based approaches.

Soil disturbance is a natural consequence of logging, disturbing soil organic matter, compacting, exposing or displacing the soil horizons and resulting in changed processes of soil nutrient, water and air availability, and at larger scales, mass movement and sediment transport (Halpern and McKenzie 2001). Soil disturbance will effect vegetation composition by its direct effects on site disturbance (Wass and Smith 1997b) and indirectly by its effects on site productivity (Krag *et al.* 1991, Senyk and Craigdallie 1997a). The process of yarding timber will effect the vegetation response by removing and damaging existing individual plants and uprooting or burying the above-ground stems of understorey species (Halpern and McKenzie 2001). While the impacts of cable logging on soil properties has been examined by numerous studies (see Chapter 1), very few (Smith and Wass 1976, Borhan *et al.* 1990, Fransen 1998) have examined the recovery of vegetation following the specialised types of soil disturbance resulting from cable logging. Highlead cable logging, in comparison to ground-based logging, will result in greater seedling mortality, tree damage and slower recolonisation of bare soil (Borhan *et al.* 1990). Where draglines form, in the short term they are characterised by reduced vegetation cover, compared to adjacent logged vegetation (Fransen 1998). Importantly, differences in seedling growth and vegetation composition between draglines and adjacent vegetation decrease over time (Smith and Wass 1976). While longer term vegetation responses are likely to be mediated by the nature of the overstorey development, shorter term responses are more likely to be mediated by levels of ground disturbance resulting from logging (Halpern and McKenzie 2001).

Classically, thinning aims to redistribute growth potential to fewer trees, leaving a stand with the desired structure and composition (Nyland 1996, Smith *et al.* 1997, Cochran and Barrett 1998). Most of our knowledge of the effects of thinning on vegetation is based on studies in north American and Canadian softwoods. These studies typically correlate changes in understorey vegetation after thinning to the changed light regimes following overstorey thinning (Young *et al.* 1967, Hill 1978, Crawford 1976, Alaback and Herman 1988, Metzger and Schultz 1984), and occassionally this is tested through experimentation (e.g. Bauhus *et al.* 1999, Riegel *et al.* 1995). Fewer studies (e.g. Aude and Lawesson 1998) interpret the thinning effect as a function of soil disturbance. Where they do, they have considered the increased heterogeneity of soil conditions post-thinning to lead to an increase in species richness, and even to be necessary for the maintenance of species diversity (Brunet *et al.* 1996). When soil disturbance in the context of ground-based thinning has been investigated, Reader (1987) reported that ground-based thinning led to a loss of woody species with increasing thinning intensity. In contrast to woody species, the loss of herbaceous species did not increase significantly with an increase in thinning intensity (Reader 1987). However, Young *et al.* (1967) observed that heavily disturbed areas in mixed coniferous forests following ground-based thinning in Oregon were easily invaded by pest species such as the Canadian thistle (*Cirsium arvense*).

Thinning is a method of intensive management of regrowth native forests that is increasingly employed in Australia, where a reduction in the area of public forest available for timber harvesting is focussing managers efforts to increase productivity on the remaining land. However, the ecological effects of native forest regrowth thinning are poorly known (O'Connell 1997) and the complexities of multi-species interactions and responses in the understorey provide challenges to experimenters (Bailey and Tappeiner 1998). Far more numerous are the studies focussing on thinning the overstorey (e.g. Brown 1997, Connell and Raison 1996), an approach described by Reader (1987) as focusing on the remaining trees and not the component lost after treatment.

There is a wide divergence of opinion on the effects of thinning on understorey vegetation. Some consider it has minimal effect on understorey vegetation (Sullivan *et al.* 2002), the effect is short lived, and the longer term pattern of understorey succession is not altered (Crawford 1976, Metzger and Schultz 1984, Alaback and Herman 1988). Others consider thinning will alter the relationship between overstorey species (Ball and Walker 1997), stimulate understorey development (Crouch 1986, Tappeiner and Zasada 1993) and alter species relative covers (Ward

1992). Species which responded to thinning in one study (Falkengren-Grerup and Tyler 1991) were found to be un-response in another study (Graae and Heskjaer 1997). More recent studies suggest, however, that ground-based thinning will increase species richness, increase the cover of herbaceous species, and promote site colonisation by weeds (Bailey *et al.* 1998, Schacht *et al.* 1988). Thinning has also been implicated in crushing large logs in advanced decay stages (McCarthy and Bailey 1994) and limiting the contribution of large coarse woody debris to the forest floor (Hayes *et al.* 1997, Fleming and Freedman 1998, Prescott 1997). The considerable volumes of logging debris generated by thinning can also kill existing vegetation and prevent its re-establishment (Metzger and Schultz 1984). Thinning has also been viewed as a means of increasing the structural heterogeneity (Hunter 1993), floristic diversity (Cole 1996), old-growth characteristics (Bailey and Tappeiner 1998) or proportion of mid to late successional understorey species (Chan *et al.* 1999) in young or intensively managed forest stands. Halpern *et al.* (1999) argued however that thinning will contribute little to enhancing understorey diversity in managed forests if it relies on the species response to occur from soil seed banks. Thinning will also increase levels of photosynthetically active radiation, decrease relative humidity, and increase air temperatures (Riegel *et al* 1992).

Cable thinning of regrowth forests is a relatively new practice, and is still being evaluated globally (McConcachie 1993, Verani 1995, Visser and Stampfer 1998) and in Tasmania (Cunningham 1997). Favourable results, especially on sensitive sites (Visser and Stampfer 1998) have led to predictions the practice will increase (McNeel and Dodd 1996, Cunningham 1997). Compared to ground-based systems, cable thinning can either result in less damage (Fairweather 1991, Youngblood 1990, Minato *et al.* 1997, Han *et al.* 2000), more damage (Nichols *et al.* 1994) or the same amount of damage to retained stems (Thompson 1996). In addition, cable thinning retains more advanced tree regeneration and understorey vegetation, and results in less site invasion by disturbance adapted woody and herbaceous species than ground-based thinning (Pominville 1993). While most disturbance during cable thinning occurs during the yarding of regrowth logs along the outrows to the log dump or landing (Han *et al.* 2000), it effectively confines soil disturbance and tree damage (Howard 1996) into these log extractions routes. Ground-based thinning however leads to increased soil compaction and mineralisation, surface disturbance and root damage, and reduces organic matter content (Wronski 1984, McIver 1988, Terlinden and Andre 1988, Han *et al.* 2000).

Cable thinning of regrowth forests is a relatively new practice and the experimentation reported in this chapter formed part of the first trials of the technique in Tasmania. While other trials have been conducted in pine plantations in the ACT (Melmoth 1979) and in New Zealand (McConachie 1993, Murphy 1982) and in oak (*Quercus cerris, Q. pubcescens*) coppice forests in central Italy (Verani 1995), none of have examined the how native vegetation composition and cover respond. Additionally the incidence of thinning using cable yarders has been increasing (McNeel and Dodd 1996), in Tasmania it has gone from virtually nil in 1993 to 1600 h/year in 1998 (Forestry Tasmania 1998).

The object of this chapter is to compare how native vegetation composition and cover varies in response to soil disturbance resulting from either cable or ground-based logging. The impacts on vegetation of two specialised types of soil disturbance associated with steep country logging operations are investigated, soil disturbance created during cable thinning of a regrowth forest, and soil disturbance created during cable log yarding from log butt gouging creating draglines.The null hypothesis for this chapter is that no significant differences in the vegetation composition and biomass of regenerating understoreys exist among the soil disturbance types resulting from the two logging methods, ground-based and cable thinning of regrowth eucalypt forest stands.

3.2 Methods

Two studies were conducted to evaluate the effects of soil disturbance arising from cable and ground-based logging on vegetation. The first study examines soil disturbance associated with draglines, the second study examines soil disturbance associated with cable and ground-based thinning.

The first study used two experimental approaches, chronosequence and experimental. The chronosequence study was conducted in clearfelled native forest coupes by examining the effect of cable logging draglines on vegetation composition and cover by comparing vegetation quadrats on and adjacent to draglines in northern Tasmania at Loatta, five years after cable logging. The experimental dragline study used pre-and post-logging observations in eastern Tasmania at Wielangta 42.

The second study examines the effects of soil disturbance resulting from cable or ground-based thinning on vegetation composition and cover in twenty-nine year old regrowth eucalypt forest. It uses a chronosequence approach and is conducted at Jungle 3 in the Florentine Valley of southern Tasmania.

Experiment 1: Dragline experimental study

Study area

Wielangta 42a is a dry sclerophyll forest coupe in eastern Tasmania which is being studied as part of the long term monitoring experiment to determine the response of vegetation to cable and ground-based logging (Chapter 2). It was considered characteristic of much of the eastern Tasmania dry sclerophyll forests cable logged since the 1980's. Wielangta 42a was logged between April and August 1992 with a North Bend standing skyline rigging system (Studier and Binkley 1974). Logs were yarded from the stump to the landing (a central area for receiving, sorting, de-barking and loading logs) using choker lines (wire ropes which attach the log(s) to the mainline or carriage) and depending on log size, up to three logs were yarded simultaneously. Generally logs were only partially suspended and dragged along the surface during yarding. The location of the quadrats and their environmental characteristics are provided in Table 3.1 and the logging system is further described in Table 2.8.

Aerial photographs were taken of the study area in order to map the extent of landings, spur and main access roads, snig tracks and draglines resulting from cable and ground-based logging. Aerial photography was undertaken at 11,150 feet at a scale of 1:4000 approximately six months after logging was completed, and these features were mapped onto clear overlays and the areas calculated and calibrated using known distances on the ground. For snig tracks, their total length was mapped and the area occupied calculated by multiplying the total length by Williamson's (1990) constant of 3.91m (track width). Similarly, the total length of draglines were mapped and the area occupied calculated by using a constant estimated width of 4 m, based on the detailed field dragline observations.

Table 3.1 *Quadrat number, treatment, location (easting and northing), altitude, pre-logging forest types, aspect and slope for the long term monitoring study of draglines at Wielangta 42a. The forest types are as per Forestry Tasmania (Stone 1998). The whole of Wielangta 42a is underlain by Jurassic Dolerite.*

Quadrat no.	easting	northing	altitude (m)	forest type	aspect	slope
Wielangta_26	6371450	5270730	360	fdE3b.ER.	360⁰	16⁰
Wielangta_27	6371610	5270750	390	fdE3b.ER.	220⁰	21⁰

Experimental design

The design of the dragline experiment was nested within the broader design of the logging coupe vegetation monitoring program at Wielangta 42a which incorporates pre-logging quadrat establishment, matched controls and post-logging re-survey of these permanently marked quadrats (see Chapter 2). Dragline quadrats were established prior to cable logging in two areas indicated on the harvesting plan as likely to be disturbed by a dragline (quadrats 26 and 27, Figure 3.1).

Vegetation sampling

The vegetation surveying regime described in Chapter 2 using 30x30 m quadrats was followed with the exception of the sub-plot sampling. The alternate 2 x 2m sub-plots (Figure 2.2) were replaced with a single 40 x 2m belt transect with continuous sub-plots of 2 x 2m. A continuous transect was used to ensure the land where the transect was positioned would be intersected by log movement. The transects were arranged perpendicular to the contour so that the dragline was be intercepted. Within each 2 x 2 m sub-plot, all vascular plant species were identified and allocated a Braun-Blanquet cover score (Chapter 2) if they were rooted in or overhanging the 2 x 2 m sub-plots. The cover of bare soil or rocky ground was estimated along the transect. After cable logging the dragline width and depth was measured, dragline width was considered to include the area where vegetation was flattened. Width was measured from the highest point of the side wall or berm (as in skid trails, Dykstra *et al.* 2000) of the dragline, and depth was measured as the distance below the adjacent undisturbed soil surface including the organic matter.

The slope and aspect of the belt transect was measured with a clinometer and compass. The transects (marked as 26 and 27 in Figure 3.1) were measured prior to logging in May 1992, were logged by August 1992, and were re-scored 1 and 2 years after cable logging.

Analytical procedure

Species presence data for each observation period for the dragline effected portion of the belt transect is summarised in a tabular form.

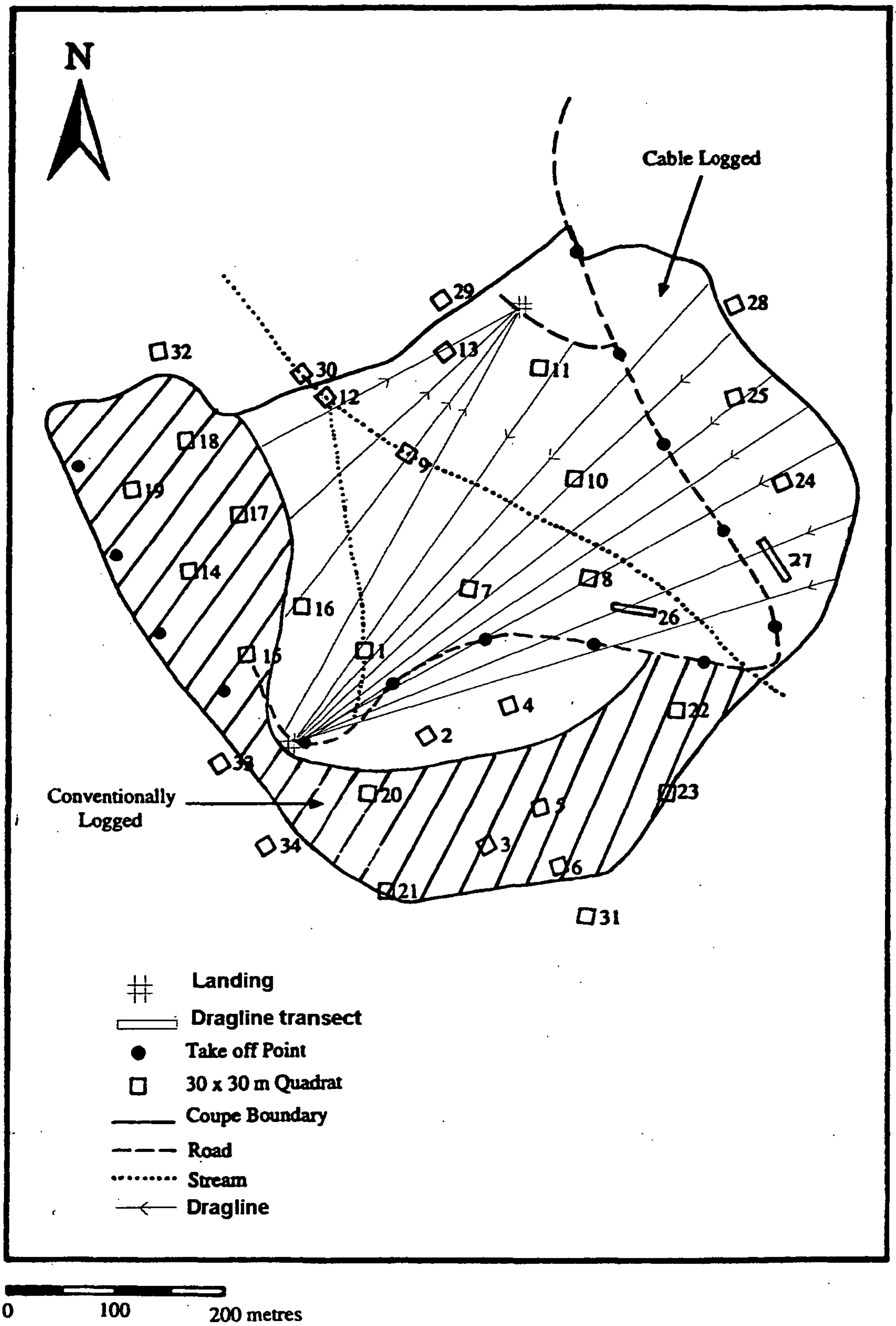


Figure 3.1 *The location of vegetation quadrats at Wielangta 42a (30 ha), eastern Tasmania based on the 1992 harvesting plan, indicating areas of ground-based ground-based logging and cable logging. The oblong shaped quadrats with numbers 26-27 are the dragline assessments. Dragline positions are approximated from the harvesting plan, the actual draglines are mapped in Figure 3.6.*

Experiment 2: Dragline chronosequence study

Study area

Loatta lies in the Martha Creek catchment of the Mersey Valley in a landscape of long, steep slopes typical of cable logging environments in northern Tasmania. Two logged coupes 1.4 km apart in the Martha Creek catchment (Loatta 209 & 217d) were surveyed (Table 3.2). These coupes were characterised by similar pre-logging vegetation type, slope, aspect and geology/soil type (Dove group mudstone and shales) and silvicultural treatment. Both were high altitude *E. delegatensis* forest. Their silvicultural history was the same, being logged, burnt and aerially sown and regenerated to *E. delegatensis*. Both were considered adequately (50-55%) stocked in a 1990 regeneration assessment according to the prescribed stocking standard (Forestry Commission of Tasmania 1991b). The logging system was also common, both were logged in 1986/87 using a Madill 071 skyline cable system rigged with intermediate supports. The resulting draglines form a fan shaped arrangement with logs being yarded uphill towards the landing (Figure 3.2).

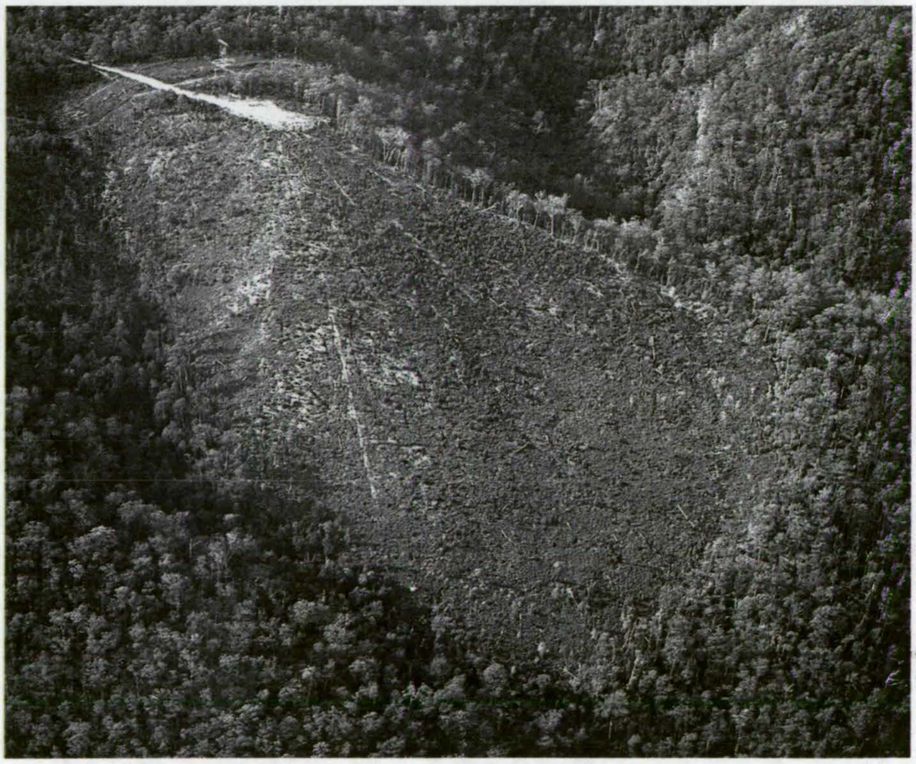


Figure 3.2 *Draglines were most pronounced on mid-slope spurs and convex slopes at Loatta 209, cable logged in 1986/87, photographed April 1991.*

Table 3.2 *Quadrat number, location (easting and northing), altitude, pre-logging forest types (after Stone1998) aspect and slope at Loatta 209.*

Quadrat no.	easting	northing	altitude (m)	forest type	aspect	slope
Loatta 209a	435100	5392200	730	E3b.ER.	225-240^0	25-26^0
Loatta 209a	436000	5391200	720	E3b.ER.	280-285^0	24-25^0

Experimental design

At the replicate study areas Loatta 209a and 217d, the cable logged areas were stratified into dragline and non-dragline portions from aerial photography (Figure 3.2). Using the aerial photographic mapping, one dragline from each of Loatta 209 and 217 was then selected using stratified random sampling. A random sampling regime was not appropriate because the spatial arrangement of draglines is deterministic and the draglines themselves are subject to localised variation in width, depth and other characteristics according to the variables described in the introduction. The selected dragline was then located in the field for quadrat survey. Replication was achieved at the coupe level.

Vegetation sampling

At each dragline, one 30 m long belt transect was established along its length. The belt transect was then partitioned into six contiguous 5 m long by 1 m wide quadrats along its longitudinal axis. Within each 5 x 1 m quadrat, all vascular plant species were identified and allocated a Braun-Blanquet cover score (Section 2.2) if they were rooted in or overhanging the quadrats. A visual estimate of the cover of bare or rocky ground was also made for each quadrat. For each 5 x 1 m quadrat, dragline width and depth at the central point along the longitudinal axis was measured with a tape measure, and the slope and aspect measured with a clinometer and compass. Width was measured from the highest point of the side wall or berm (as in skid trails, Dykstra *et al.* 2000) of the dragline, and depth was measured as the distance below the adjacent undisturbed soil surface including the organic matter. A second 30 m long belt transect was then established parallel to the dragline transect in immediately adjacent vegetation (within 10 m) subject to the same silvicultural treatment but not to the additional dragline disturbance.

Analytical procedure

The Braun-Blanquet cover scores were converted to the mid-points of the particular cover class using the methods described in Section 2.2. These cover scores were then averaged across the six quadrats per dragline and presented as a mean percentage foliage cover score. The cumulative cover was also calculated using the sum of these scores across the six quadrats. Quadrat variables (cover of bare or rocky ground, dragline width, depth, aspect and slope) measured on each quadrat were similarly averaged across quadrats within a dragline.

Experiment 3: Comparison of cable and ground-based thinning effects on soil and vegetation

Study area

The trial was located on Jungle Rd, Repulse area, Florentine Valley, southern Tasmania (grid reference E460500, N5297200). The management unit is known as Jungle 3 and the study was conducted across an area of approximately 18 ha. The vegetation consists of *Eucalyptus regnans* silvicultural regrowth originating from logging in 1961 and regeneration treatments (burning and sowing) between 1961 and 1964. The study area was underlain by Jurassic Dolerite and the soil A horizon is a gritty brown clay loam. Boulders comprised 5-20% of the surface cover. Average slope was 10 degrees. The experimental area had a north easterly aspect and an altitude of 460-490 m ASL. The (photo-interpreted) forest type (as per Forestry Tasmania, Stone 1998) is E2cST. The silvicultural regrowth was 29 years old when subject to the experimental thinning, which is considered beyond its optimum age (in terms of growth response) for a first thinning in a production forest of this type.

Cable thinning was carried out with a purpose built skyline cable system using a gravity return carriage mounted on a Prentice excavator (Figure 3.3). The harvesting pattern consisted of 4 metre wide outrows at 15 m centres placed at right angles to the contour (Figure 3.4). Trees were felled manually and logs were secured to the choker lines and pulled laterally into the outrow and then upslope along the outrow to the cable yarder (Figure 3.4). Logs were partially suspended, allowing the load to clear obstacles on the ground and minimise soil and litter disturbance. The cable yarder remained on the roadside and logs were yarded along the outrows and stockpiled on the roadside. In the flatter ground-based treatment, thinning was undertaken using a Bell tracked feller-buncher mounted with a hydraulically operated chainsaw. Felled logs were placed in bundles within the treatment area and snigged to the roadside landing with a CAT rubber tired grapple-skidder for stockpiling. The thinning prescription for both the ground-based and cable regions was form, vigour and spacing. The experiment aimed to limit damage to retained trees in both treatments to less than 10%.

Figure 3.3 *Cable thinning was carried out at Jungle 3 in 1991/92 with a purpose built skyline cable system using a gravity return carriage mounted on a Prentice excavator.*

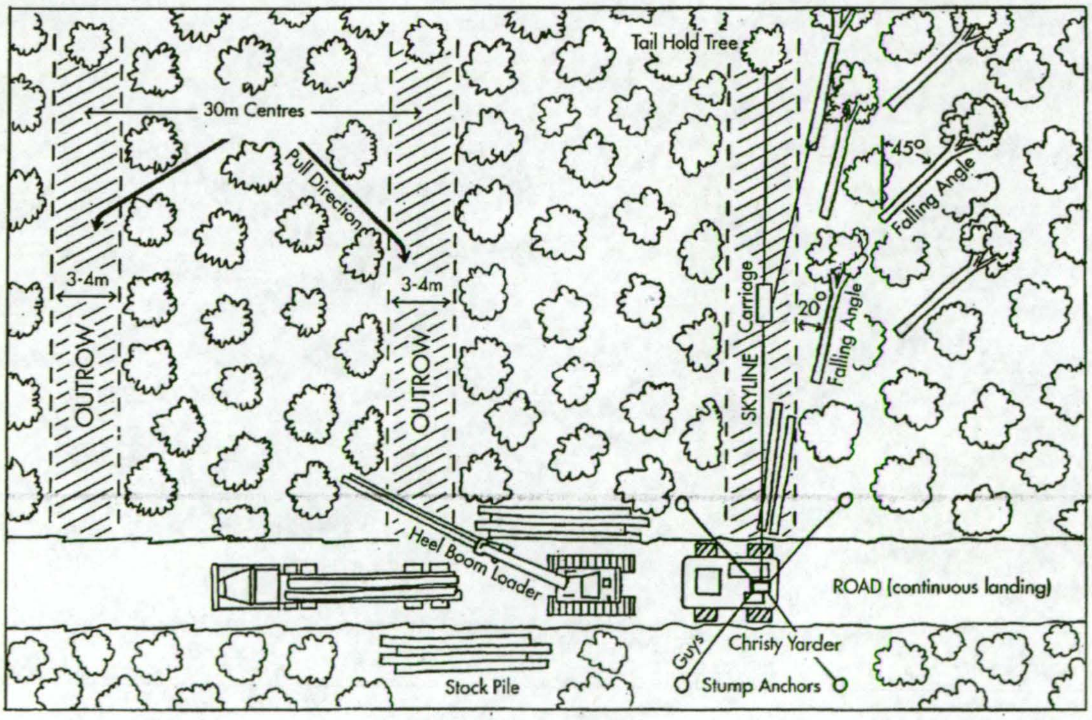


Figure 3.4 *Diagram of the cable thinning operation at Jungle 3, vegetation surveys were undertaken in the sections between outrows (diagram from Cunningham 1997).*

Experimental design

The Jungle 3 study area was divided into four treatment areas determined by ground slope. The steeper area (8-10^0) was allocated to cable thinning and the less steep area (4-6^0) to ground-based thinning (Figure 3.5). The cable thinning treatment area portion was further subdivided into two equal areas, one to be thinned to 150 stems/hectare, the other to 270 stems/hectare. The ground-based area was to be thinned to 230 stems/hectare. The treatments could not be randomly assigned due to the constraints of providing adequate machinery access for the cable yarder on the central access road and the requirement that the cable carriage have a down slope profile from the road to the forest to permit the gravity release system on the carriage to function. As thinning had already been implemented prior to quadrat assessment, adjacent control quadrats were used to indicate vegetation characteristics prior to thinning.

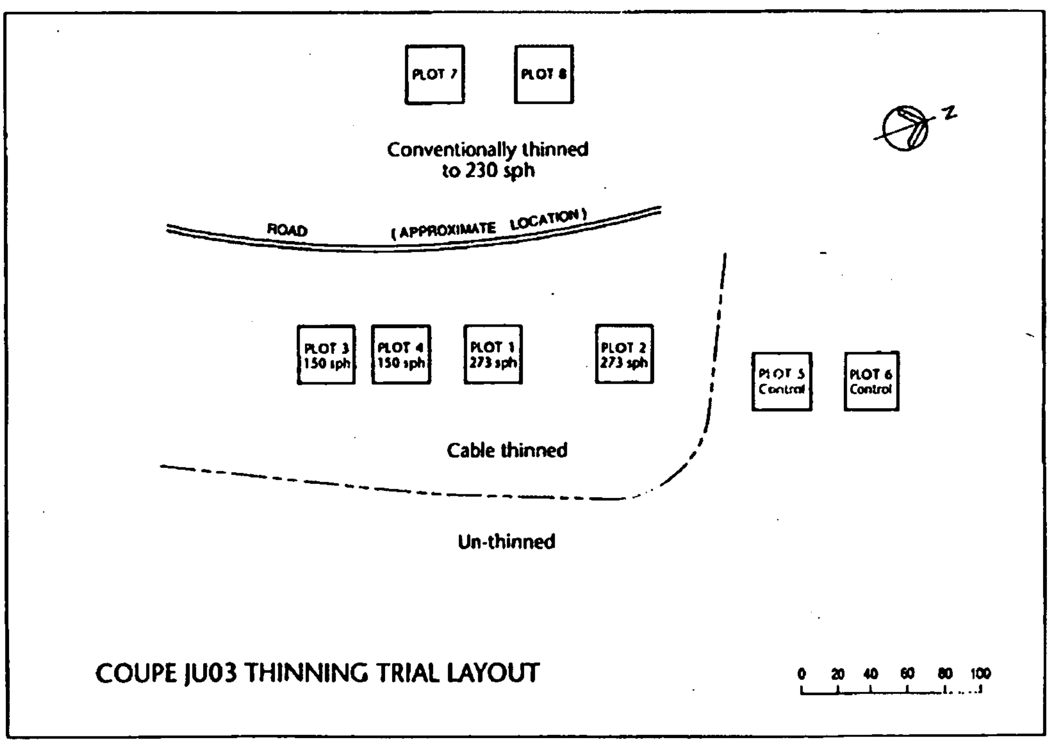


Figure 3.5 *Distribution of vegetation quadrats (squares) within the three treatment areas at the Jungle 3; (conventional thinning, cable thinning and unthinned control). Surveys in the cable thinned treatment occurs between outrows (Figure 3.4), and in the ground-based ground-based thinning treatment, between snig tracks. sph = stems per hectare*

Soil disturbance and canopy opening each have an effect on forest regeneration and these two factors probably interact. However, they were not considered separately in this experiment. It was not practical to mechanically thin the overstorey (and therefore change the light and temperate regime) without causing some soil disturbance.

Vegetation sampling

For each of the four treatments (Table 3.3) two replicate 30 x 30m vegetation quadrats each with 20 2x2 m sub-plots were established. Within each 30 x 30m quadrat, all vascular plant species growing within or projecting into or above the quadrat were recorded and assigned a visually assessed cover-abundance value modified from the Braun-Blanquet (Poore 1955) scale as described in section 2.2. Cover was defined as foliage projective cover following Walker and Tunstall (1981). Ten sub-plots (2 x 2m) were spaced at 7.5m intervals along a transect established either side of the quadrat's mid-point (Figure 2.6). The sub-plots were 1m apart and alternated from side to side along the transect. All vascular plant species present in the 20 sub-plots were recorded and assigned a visually assessed cover-abundance value which was then converted to a presence score. The presence score formed a local shoot frequency score of between 0 and 20 based on the number of times it was recorded within the twenty sub-plots. Within each 30 x 30m vegetation quadrat vascular epiphytes were recorded and classified into three groups based on their life forms; ferns, shrubs or trees and herbs. Both obligate, facultative, accident or hemi-epiphytic species were recorded using the procedures described in Chapter 6.

Table 3.3 *Quadrat and treatment attributes for the Jungle 3 cable and ground-based thinning study. sph = stems per hectare*

quadrat	alt. (m)	aspect (degrees)	slope (degrees)	treatment
quadrat 1	470	60°	10°	cable thinning to 270 sph
quadrat 2	470	62°	8°	cable thinning to 270 sph
quadrat 3	470	65°	10°	cable thinning to 150 sph
quadrat 4	470	78°	9°	cable thinning to 150 sph
quadrat 5	470	48°	13°	unthinned control
quadrat 6	470	6°	9°	unthinned control
quadrat 7	490	162°	4°	ground-based thinning to 230 sph
quadrat 8	490	170°	6°	ground-based thinning to 230 sph

The two replicate quadrats within the treatment were positioned at the same point on the contour and were approximately 40 m apart at their centroids (Figure 3.5). Vegetation surveying in the cable thinning experiment was undertaken in the sections between outrows (Figure 3.4) and in the ground-based thinning experiment in areas between the snig tracks created by the skidder. Vegetation surveys were undertaken between April-May 1992, approximately 6 months after the thinning treatments were completed, and took account of different levels of disturbance; litter removal, gouging, and distribution of thinning debris.

Litter displacement was defined as the removal of leaf litter and other fine woody debris sufficient to expose mineral soil. Soil gouging was defined as the displacement of the organic layer and A horizon to a depth >10 cm. for > 1 m in length. Residue was defined as an area with accumulated residual of tree heads, branches and bark from the thinning operation. The proportion of the treatment areas subject to litter displacement, soil gouging and thinning residue was estimated with a series of random traverses using 1 m wide belt transects. For each treatment, a minimum of 10% of the treatment area was surveyed.

Analytical procedure

Mean percentage cover scores and percent frequency scores were derived from 20, 2 x 2m sub-plots within each 30x30 m quadrat using the procedures previously described (see Chapter 2). Species identified in each quadrat and later grouped into life form categories of trees, shrubs, herbs, graminoids and ground ferns had a mean percentage foliage cover score derived for these groups by averaging across 40 sub-plots for each treatment. Hemi-epiphytic species richness was summed for the two replicate 30 x 30 m quadrats. Results are presented graphically using histograms. Compositional differences between the four treatment groups were examined using the ANOSIM algorithm of Clarke and Green (1988). This method is effectively a distribution free one-way ANOVA (Clarke 1993). In this procedure, the test statistic was based on 100 calculations, groups were defined by the four treatments, with two replicates per treatment. The mean percentage cover and percent frequency data was first standardized to equal species maxima prior to constructing the input association matrix. The dissimilarity measure used was Bray-Curtis. Cluster analysis was performed using the association matrix described above following the FUSE algorithm in PATN (Belbin 1992). The option chosen was flexible UPGMA (unweighted pair group arithmetic averaging), a hierarchical agglomerative polythetic clustering method with beta = 0.1. The compositional similarity between treatments are examined graphically using the MST (Minimum Spanning Tree) algorithm in PATN. The spatial configuration of each quadrat is displayed by the network diagram, with the distance between objects scaled to reflect the degree of similarity or association. The diagram can be considered as a set of lines representing the pair-wise associations that interconnect all the objects (vegetation quadrats) The ultrametric-distances from MST are constrained to be along the branches of the tree, where 'neighbours' are accurately represented while the increase in the uncertainty is proportional to the increase in separation (Belbin

1992). The ANOSIM, MST and FUSE routines were performed using PATN (Belbin 1992).

The multivariate relationship between treatments was further examined using ordination, based again on the previous association matrix. The ordination technique used was non-metric multi-dimensional scaling in the global form. The ordination was performed in 1-6 dimensions, and the minimum stress solutions for each dimension examined. The ordination methods used are described in more detail Chapter 2. Species richness is presented for the 0.09 ha. scale, data on hemi-epiphytic species richness is pooled however for the two replicate quadrats within each treatment.

3.3 Results

Experiment 1: Dragline experimental study

Dragline immediate effects

Cable and ground-based logging, through the development of landings (Figure 3.1), spur and main access roads, snig tracks and draglines (Figure 3.6) at Wielangta disturbed 15.4% of the coupe area (Table 3.4). Approximately the same proportion of the ground-based logged area was disturbed by snig tracks (8.8%) as the cable logged area was disturbed by draglines (7.9%). Dragline disturbance was manifest as flattening or covering the vegetation with bark and debris stripped away from logs during yarding. In contrast, of the vegetation disturbed on snig tracks was generally displaced by the repeated machine passes, dragging of logs and soil scraping.

Dragline disturbance varied throughout the cable logged area depending on the slope profile and the position of the landing areas. Disturbance was concentrated near landings where multiple draglines converged (Figure 3.6). On the vegetation transect (quadrat 26) closest to the landing the dragline was 4 m wide (see Figure 3.1), on the transect (quadrat 27) further from the landing it was only 1.2 m wide. In the dragline effected area, there was initially an abundance of flattened and decaying *Gahnia grandis* tussocks, small woody debris deposited during log yarding and vegetatively regenerating *Lepidosperma laterale* and *Lomatia tinctoria* (Table 3.5, Figure 3.7). A small portion of the area (5%) consisted of exposed soil, providing a substrate for several *E. obliqua* seedlings. Herbaceous species regenerated by sending shoots from surviving stems and roots through the logging residue (Table 3.5). Where woody debris is present in large amounts vegetation cover was lowest.

Table 3.4 *Percentage of land affected by logging operations at Wielangta 42, six months after completion of logging, The total coupe area is 27 ha., 38.8% of which is ground-based, 65.2% is cable.*

Type of disturbance	%
landings	2.2
spur roads	0.9
main access all weather road	4.1
ground-based snig tracks[1]	8.8
draglines[2]	7.9
Total coupe area disturbed[3]	15.4

[1]of ground-based ground-based area, [2]of cable area (17.6 ha), [3]sum of affected areas as a percentage of total coupe area (27 ha)

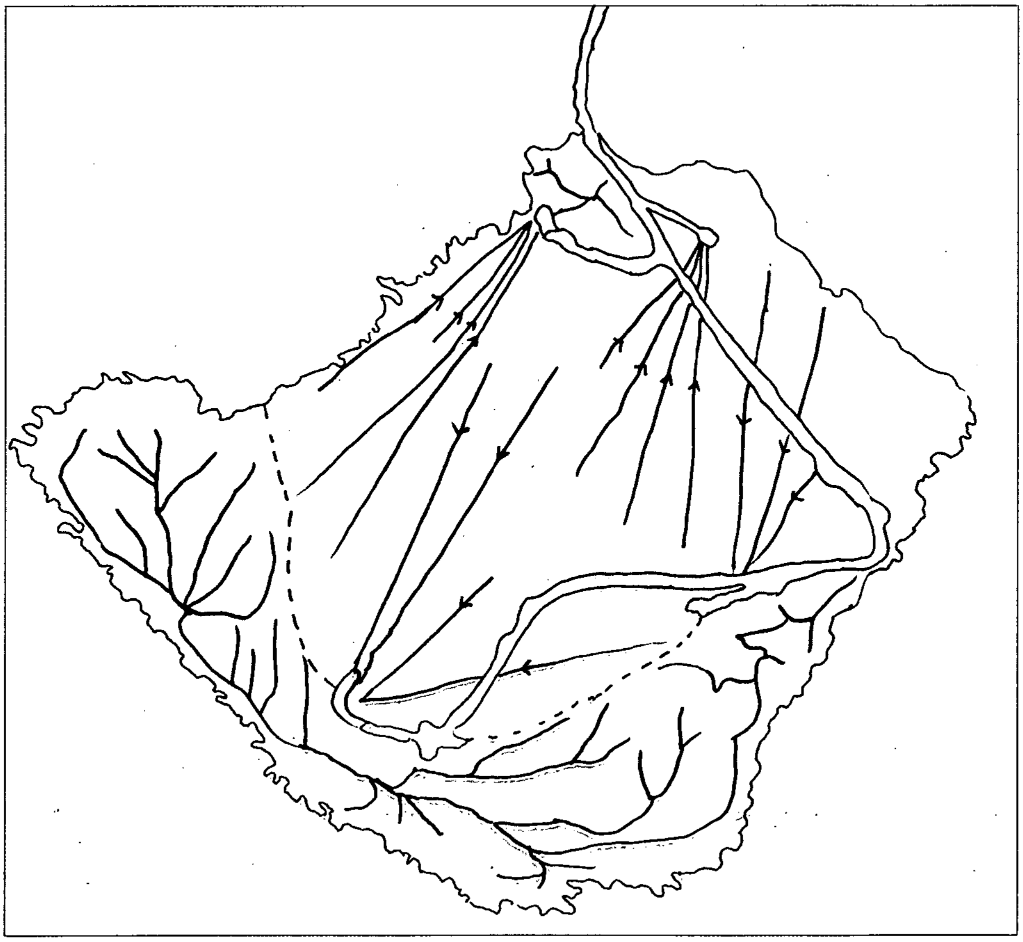


Figure 3.6 *The location of cable draglines (with arrows) and ground-based logging snig tracks (without arrows) at Wielangta 42. The arrows indicate the direction of log yarding to the landings. Drawn from spot aerial photography taken in 1992 (Figure 2.10). The dotted line indicates the boundary of the cable and ground-based logged portions.*

Figure 3.7 *A dragline transect at Wielangta (quadrat 26) one year after logging in July 1993. The dragline is characterised by flattened vegetation and deposits of logging debris.*

No species presence data is presented for the replicate transect further upslope (quadrat 27, Figure 3.1) for two reasons. Firstly, there was little observable dragline effect this distance from the landing because of the small amount of wood yarded over this section of the coupe. Secondly, the transect re-sampling was compromised by windthrow activity after cable logging, where a large volume of woody material was deposited within the quadrat unrelated to the logging treatment (Figure 3.8).

Figure 3.8 *Dragline quadrat 27 at Wielangta in July 1993 one year after cable logging. The transect re-sampling was compromised by windthrow activity after cable logging and was abandoned.*

Dragline induced changes in vegetation composition were variable (Table 3.5), several species did not re-establish within the quadrat areas within two years of being disturbed, several other species persisted, and others were recorded after 1 and 2 years which were absent prior to disturbance. In the short term, species richness was increased by dragline disturbance.

Table 3.5 *Species present at Wielangta on quadrat 26 (a region destined to become a dragline), before logging (1992) and then 1 year (1993) and 2 years (1994) after logging and dragline disturbance.*

Wielangta quadrat_26	before logging	1 year	2 years
Comesperma volubile	+		
Epacris impressa	+		
Dianella tasmanica	+		
Pteridium esculentum	+		
Eucalyptus obliqua	+		+
Cathodes divaricata	+		+
Deyeuxia rodwayi	+		+
Gonocarpus teucriodes	+	+	+
Gahnia grandis	+	+	+
Lepidosperma laterale	+	+	+
Lomatia tinctoria	+	+	+
Galium australe		+	
unidentified orchid species		+	
Goodenia ovata		+	+
Hydrocotyle hirta		+	+
Viola hederaceae		+	+
Callistemon viridens			+
Leptospermum scoparium			+
Poranthera microphylla			+
Senecio spp.			+
no. of species recorded	10	9	14
no. of introduced species recorded	0	0	0
% bare and rocky ground	<5	5	5
mean dragline depth (m)	-	0.05-0.08	0.05-0.08
mean dragline width (m)	-	4.0	4.0

Experiment 2: Dragline chronosequence study

Dragline five year effects

Draglines were a non-continuous feature of the ground surface five years after cable logging at Loatta. Draglines exhibited a pattern of radiating lines upslope to the centralised landing areas in the direction of log yarding (Figure 3.2). Dragline depth and width varied from 0.05 to 0.40m and 0.90 to 3.0m respectively across the twelve sub-plots measured. There was no relationship detected between the width of dragline and distance from the landing area, except at the furthest point from the landing. The area of bare and rocky ground recorded on draglines was twice that of adjacent logged areas (Table 3.6). Draglines could also be recognised by accumulations of bark and other small woody debris where soil disturbance was insignificant.

Few differences were evident in vegetation composition between the dragline and adjacent regrowth (Table 3.6). Species richness was slightly less on draglines than adjacent regrowth at both quadrats examined. Two of the three introduced weed species observed, *Cirsium vulgare* and *Senecio jacobaea* were only recorded on the draglines. The third species, *Hypochaeris radicata* was ubiquitous on both coupes, although it was more abundant on the draglines. When weed species were present, they were more abundant on the draglines.

Total vegetation cover on draglines was approximately 60% of the cover on adjacent regrowth at both quadrats (Table 3.6). Species relying on vegetative reproduction (e.g. from tussocks, rhizomes and persistent root stocks) to recover from logging and burning were less abundant on the draglines; examples include *Pultenaea juniperina, Dianella tasmanica Diplarrena moraea, Amperea xiphoclada* and *Poa labillardieri. Pteridium esculentum,* which commonly reproduces via its rhizomatous stipe was also less abundant on the draglines. Of the three *Eucalyptus* species present, regrowth saplings were more abundant in adjacent regrowth than on draglines.

Table 3.6 *Mean percentage cover and frequency of plant species recorded on draglines and adjacent five year old regrowth at Loatta, north western Tasmania. * introduced species*

coupe	Loatta 209				Loatta 217d			
transect	dragline		adjacent regrowth		dragline		adjacent regrowth	
species	cover	frequency	cover	frequency	cover	frequency	cover	frequency
Acaena novae-zelandiae	-	-	0.2	16.7	-	-	-	-
Amperea xiphoclada	-	-	-	-	0.3	33.3	0.7	50
Blechnum nudum	0.2	16.7	-	-	-	-	-	-
Caladenia lyallii	-	-	-	-	-	-	-	-
Chiloglottis gunnii	1.5	100	1.8	83.3	-	-	-	-
Cirsium vulgare*	0.3	33.3	-	-	0.2	16.7	-	-
Coprosma hirtella	-	-	-	-	-	-	0.2	16.7
Cotula filiculata	-	-	0.3	33.3	0.2	16.7	-	-
Cyathodes glauca	0.6	33.3	1.6	83.3	0.2	16.7	-	-
Deyeuxia contracta	0.5	50	0.4	16.7	-	-	-	-
Deyeuxia monticola	8.3	83.3	5.0	33.3	-	-	-	-
Deyeuxia quadriseta	0.3	33.3	0.8	33.3	-	-	-	-
Dianella tasmanica	13.9	83.3	32.5	83.3	18.7	66.7	53.7	100
Diplarrena morae	-	-	-	-	0.2	16.7	0.6	33.3
Drymophila cyanocarpa	0.5	50	2.9	33.3	0.3	33.3	0.2	16.7
Ehrharta stipoides	-	-	-	-	0.2	16.7	-	-
Epacris impressa	0.2	16.7	-	-	-	-	-	-
Eucalyptus amygdalina	1.2	50	6.7	33.3	2.8	50	5.9	100
Eucalyptus dalrympleana	1.2	50	2.8	50	0.4	16.7	0.4	16.7
Eucalyptus delegatensis	2.5	100	10.4	83.3	29.2	100	14.6	100
Galium australe	-	-	-	-	0.2	16.7	0.2	16.7
Gnaphalium collinum	0.2	16.7	-	-	0.7	50	0.3	33.3
Gonocarpus tetragynus	-	-	-	-	-	-	0.2	16.7
Gonocarpus teucriodes	5.0	33.3	0.7	50	-	-	3.1	50
Helichrysum scorpiodes	-	-	-	-	0.6	33.3	-	-
Hypochaeris radicata*	0.7	66.7	0.3	33.3	1.6	83.3	0.7	66.7
Leucopogon collinus	1.0	50	0.8	33.3	0.2	16.7	0.2	16.7
Lomatia tinctoria	8.9	50	3.2	66.7	13.5	83.3	14.6	100
Luzula meridionalis	-	-	0.6	33.3	-	-	0.2	16.7
Marianthus procumbens	-	-	0.3	33.3	-	-	-	-
Olearia phlogopappa	0.7	66.7	0.3	33.3	-	-	-	-
Olearia ramulosa	-	-	-	-	0.3	33.3	-	-
Pimelea ligustrina	-	-	0.2	16.7	-	-	-	-
Poa labillardieri	0.4	16.7	-	-	18.3	100	60.4	100
Pteridium esculentum	43.3	100	64.2	100	46.2	83.3	72.5	100
Pultenaea juniperina	5.8	66.7	27.5	100	1.8	83.3	12.5	100
Senecio biserratus	0.2	16.7	0.6	33.3	0.9	66.7	0.8	83.3
Senecio jacobaea*	-	-	-	-	1.0	100	-	-
Senecio linearifolius	0.3	33.3	1.0	50	2.9	33.3	0.2	16.7
Stackhousia monogyna	-	-	-	-	0.2	16.7	-	-
Stylidium graminifolium	0.2	16.7	-	-	-	-	-	-
Viola hederaceae	-	-	-	-	0.6	33.3	0.2	16.7
Wahlenbergia spp.	-	-	-	-	0.8	83.3	0.3	33.3
% cover	98		165		142		242	
no. of species recorded	26		24		27		23	
no. of introduced species recorded	2		1		3		1	
% cover of introduced species	1		0.3		2.7		0.7	
% bare and rocky ground	48		22		50		21	
mean dragline depth (m)	0.21		-		0.28		-	
mean dragline width (m)	1.65		-		1.20		-	

Experiment 3: Comparison of cable and ground-based thinning effects on soil and vegetation

Vegetation responses to soil disturbance from thinning

Prior to thinning at Jungle 3, *E. regnans* formed a 35 m tall overstorey over a discontinuous 18-20 m tall small tree stratum of *Acacia dealbata*. The dense broad-leaved 8-10 m tall shrub stratum was dominated by *Pomaderris apetala*, *Phebalium squameum* and *Bedfordia salicina*. The heavily shaded ground stratum consisted of a discontinuous fern layer of *Pteridium esculentum*, *Histiopteris incisa* and *Polystichum proliferum*. Herbaceous species were well developed in canopy or shrub stratum openings and sparse beneath areas of continuos ground fern cover.

The degree of soil disturbance from gouging and from litter removal differed markedly between the cable and ground-based thinned areas, with 38% of the ground-based treatment area receiving ground disturbance from soil gouging, compared to <1% in the cable thinning treatments (Table 3.7). The proportion of the treatment areas covered with thinning residue also varied with thinning method, with nearly twice the area in the ground-based treatment covered with residue compared to the cable thinning area (Table 3.7).

Table 3.7 *Pre and post-thinning stand characteristics at the Jungle 3 trial area. Unpublished data from Forestry Tasmania. sph = stems per hectare. na = data not available. The data represents the mean values for the two replicate quadrats per treatment, with 20 trees per quadrat measured.*

thinning method	ground-based		cable		cable		control
thinning intensity (sph)	230		150		270		-
	pre	post	pre	post	pre	post	
E. regnans stand density (sph)	600	230	433	150	416	273	483
E. regnans ba (m2/ha.)	34.4	na	32.8	26.1	36.0	28.9	35.4
E. regnans mean dom. ht. (m.)	35.5	na	35.2	36.2	35.8	32.8	35.4
area with soil gouging (%)	-	38	-	0.45	-	0.20	-
area with litter displacement (%)		na	-	0.49	-	0.10	-
area with thinning residue (%)	-	70-80	-	<50	-	<50	-
%) of retained trees damage		25.0	-	10[1]	-	5	-

[1] the lower stand density results in a proportionally higher value for damage to retained trees

Comparison of thinned to un-thinned quadrats

The analysis of compositional data from all treatments at Jungle 3, using both clustering (Figure 3.9-3.10) and ordination techniques (Figure 3.11) indicated that using either cover or frequency species data, the primary division or partition in the whole data set was between thinned and un-thinned quadrats (dissimilarity value = 0.91, Figure 3.10), and then between cable and ground-based thinned quadrats (dissimilarity value = 0.81, Figure 3.10). Further, when the floristic compositional differences are examined via a network diagram (Figure 3.9), it is clear that the control replicate quadrats are the most similar to each other, and the conventional ground-based quadrats the least similar to the controls and from each other. When the floristic differences between quadrats are examined in terms of the a priori treatment groups, and a test for statistical significance applied via a procedure which randomly allocates treatments to quadrats, it was also clear (ANOSIM $p<0.05$) that the thinning treatments resulted in a statistically significant change to the floristic composition of the vegetation quadrats.

Thinning, despite resulting in an obvious reduction in tree crown cover, also resulted in the tall shrub foliage cover being reduced from 46% to 0.1-2.1% (Table 3.8, Figure 3.12). Apart from several graminoid species whose distributions were probably due more to micro-topographic variation than to treatment effects, all of the species absent from the thinned areas were shrubs (*Bedfordia salicina, Phebalium squameum, Pittosporum bicolor* and *Zieria arborescens*). A number of herbaceous species were recorded only in the thinned quadrats, these included several species of *Agrostis, Geranium potentilloides, Gnaphalium collinum, Hypericum gramineum* and *Urtica incisa*.

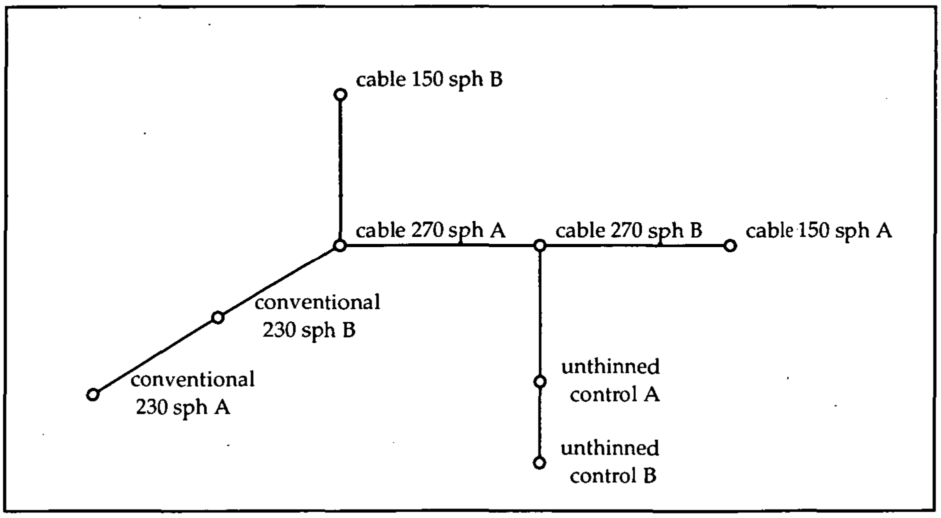


Figure 3.9 *Minimum spanning tree network diagram displaying the spatial configuration of each quadrat in the Jungle 3 thinning study, the distance between quadrats is scaled to reflect the degree of floristic similarity.*

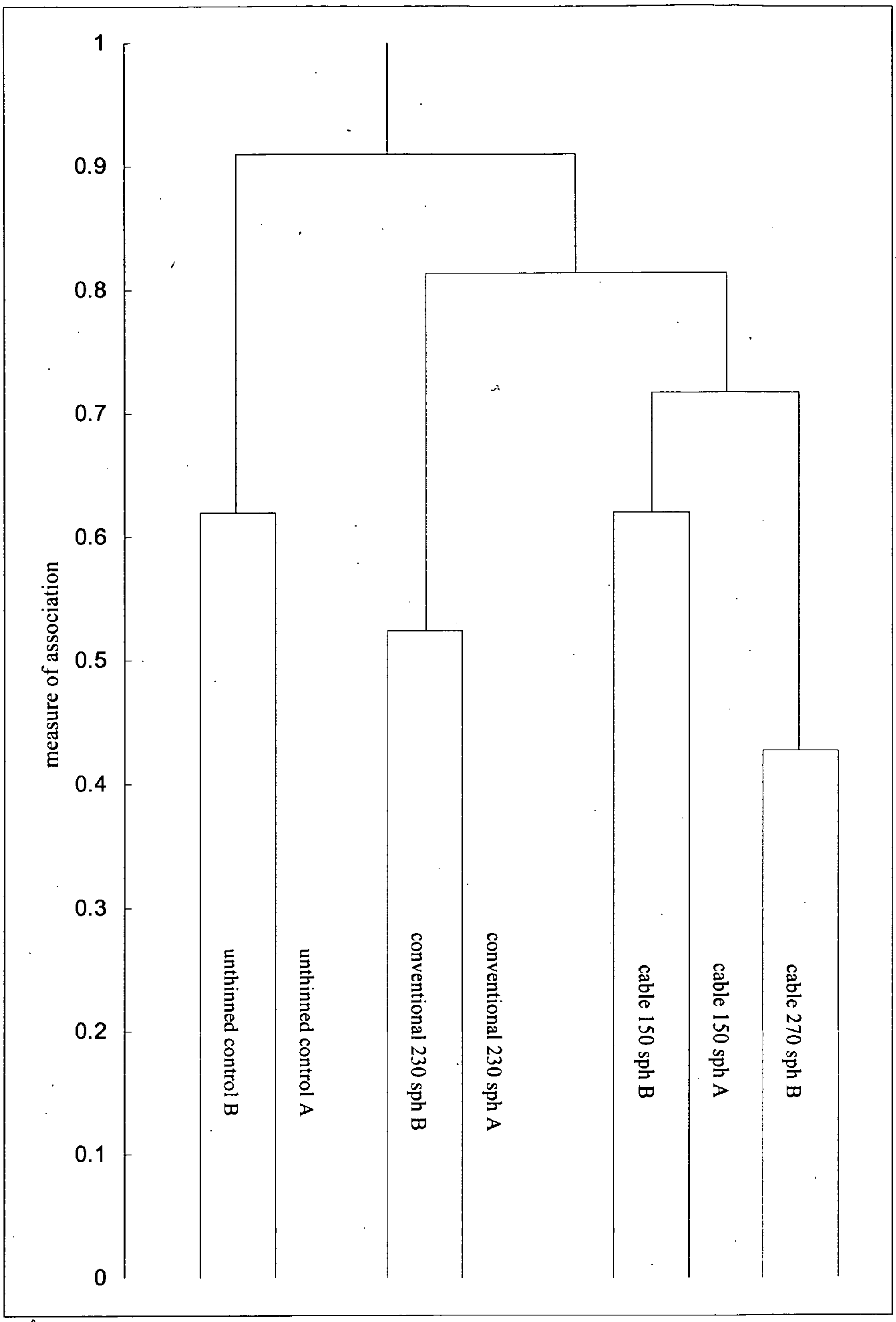


Figure 3.10 *Classification dendogram of the eight quadrats based on the mean percentage cover scores for each species. The cluster analysis was performed using the flexible UPGMA option within PATN (Belbin 1992). sph = stems per hectare, A and B are the replicate quadrats per treatment*

Table 3.8 *Floristic composition of the three thinning treatment areas and unthinned regrowth forest control based on the mean percentage cover (first column) and frequency scores (second column). # are mean percentage cover scores of <0.05. + are species were present in the 0.09 ha. quadrat but not in the sub-plots. *introduced species*

species / treatment	cable 150 sph		cable 270 sph		ground-based 230 sph		control 483 sph	
	cover	frequency	cover	frequency	cover	frequency	cover	frequency
Trees								
Acacia dealbata	-	-	-	-	0.2	10.0	6.1	60.0
Atherosperma moschatum	<0.05	+	-	-	<0.05	+	-	-
Eucalyptus regnans	7.1	75.0	19.2	90.0	13.7	97.5	22.8	100.0
Shrubs								
Bedfordia salicina	-	-	-	-	-	-	2.8	27.5
Coprosma quadrifida	-	-	0.3	17.5	0.2	10.0	0.4	20.0
Cyathodes glauca	<0.05	+	-	-	-	-	-	-
Olearia argophylla	-	-	-	-	1.5	10.0	2.0	30.0
Phebalium squameum	-	-	-	-	-	-	2.8	32.5
Pimelea cinerea	-	-	-	-	<0.05	5.0	-	-
Pimelea drupacea	0.1	7.6	0.1	5.1	<0.05	5.1	0.4	25.0
Pittosporum bicolor	-	-	-	-	-	-	<0.05	+
Pomaderris apetala	-	-	<0.05	+	0.4	7.5	37.4	85.0
Zieria arborescens	-	-	-	-	-	-	0.2	12.5
Herbs								
Acaena novae-zelandiae	0.1	10.1	0.7	15.0	2.9	42.5	0.4	2.0
Agrostis avenacea	0.3	20.0	0.3	15.0	-	-	-	-
Agrostis rudis	-	-	-	-	6.3	35.1	<0.05	+
Agrostis venusta	-	-	-	-	0.1	5.1	-	-
Australina pusilla	2.0	30.0	7.2	67.5	0.2	1.0	0.3	20.0
*Cirsium vulgare**	-	-	0.1	10.1	0.6	37.5	0.1	10.0
Clematis aristata	0.2	12.5	0.2	17.5	0.2	20.0	0.6	20.0
Deyeuxia contracta	<0.05	+	-	-	0.2	10.0	<0.05	10.0
Drymophila cyanocarpa	-	-	<0.05	+	0.1	5.0	-	-
Echinopogon ovatus	0.7	30.0	0.4	17.5	2.6	42.5	0.5	25.0
Ehrharta stipoides	-	-	1.2	25.0	-	-	0.7	35.0
Galium australe	<0.05	+	0.1	5.1	0.4	7.5	0.6	30.0
Geranium potentilloides	-	-	-	-	0.1	5.0	-	-
Gnaphalium collinum	-	-	-	-	0.2	10.1	-	-
Hydrocotyle hirta	2.4	5.0	5.2	2.6	11.7	7.5	0.9	10.0
Hydrocotyle sibthorpiodes	0.1	60.0	<0.05	77.5	1.4	87.5	0.1	27.5
Hypericum gramineum	<0.05	+	-	-	<0.05	+	-	-
*Hypochoeris radicata**	-	-	-	-	0.1	5.0	0.1	5.0
Oxalis corniculata	0.1	10.1	5.9	52.5	0.2	12.5	0.6	20.0
Senecio biserratus	0.3	25.0	0.6	42.5	-	-	0.3	20.0
Senecio linearis	-	-	-	-	0.2	10.0	-	-
Senecio minimus	-	-	-	-	1.4	57.5	-	-
Senecio spp.	-	-	-	-	-	-	0.1	5.0
*Sonchus oleraceus**	-	-	-	-	-	-	<0.05	+
Uncinnia tenella	0.3	15.0	0.8	5.1	0.1	5.0	0.6	40.0
Urtica incisa	0.1	5.1	<0.05	+	0.1	+	-	-
Viola hederacea	<0.05	+	0.5	10.0	0.1	2.6	0.1	7.6
Graminoids								
Carex appressa	-	-	-	-	-	-	0.3	25.0
Dianella tasmanica	0.8	5.0	-	-	-	-	-	-
Gahnia grandis	0.4	2.6	0.1	2.6	<0.05	2.6	0.1	5.0
Juncus australis	-	-	-	-	-	-	0.1	5.0
Juncus bufonius	-	-	-	-	0.1	7.5	-	-
Juncus pauciflorus	<0.05	+	-	-	<0.05	+	0.1	10.0
Luzula meridionalis	-	-	-	-	<0.05	-	-	-
Ground Ferns								
Blechnum nudum	<0.05	+	-	-	0.1	5.0	-	-
Dicksonia antarctica	2.3	17.5	<0.05	+	0.9	10.0	-	-
Histiopteris incisa	15.8	95.0	4.6	37.5	1.4	22.5	1.0	20.1
Hypolepis rugosula	15.1	50.0	20.3	100.0	0.4	5.1	4.5	55.0
Microsorium diversifolium	<0.05	+	-	-	-	-	-	-
Polystichum proliferum	8.7	47.5	7.3	65.0	7.5	42.5	3.7	25.1
Pteridium esculentum	12.2	47.5	15.9	82.5	16.4	67.5	5.6	60.0

Comparison of cable and ground-based thinning

Ground-based thinning resulted in a proportionally greater degree of change to the vegetation composition and biomass than the cable thinning treatment. The distribution of quadrats in the ordination space indicates a broadly similar response to thinning treatment using either vegetation cover or frequency as the measure of change. However, there was a greater degree of differentiation using vegetation cover between the ground-based thinned quadrats and the cable and unthinned quadrats (Figure 3.11).

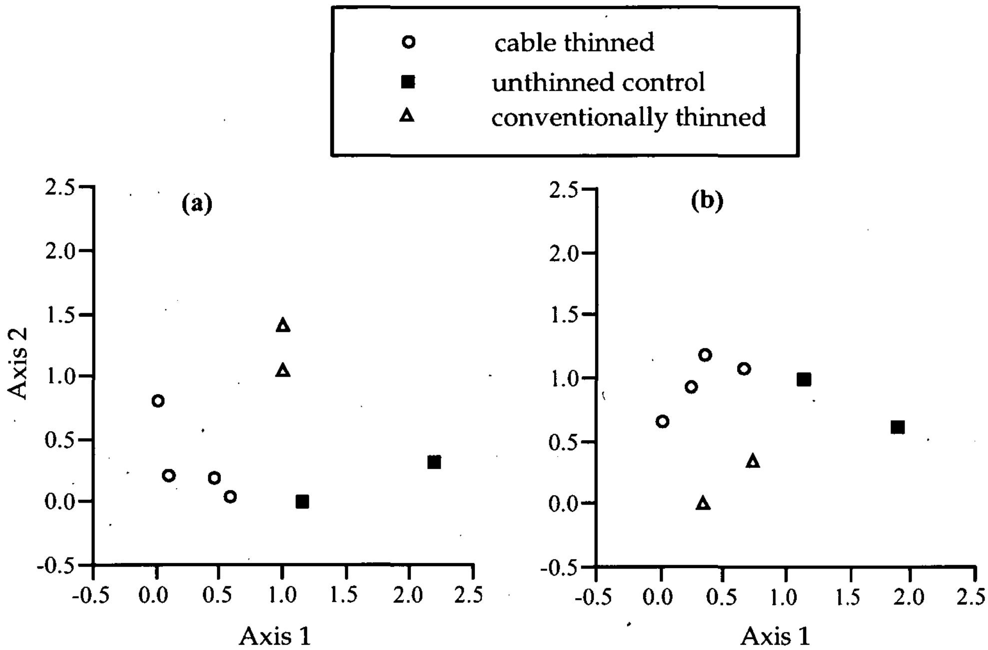


Figure 3.11 *Ordination of (a) mean percentage cover and (b) mean frequency of species subject to the two thinning treatments at Jungle 3. Minimum stress in two dimensions was 0.032 and 0.028 respectively.*

The immediate effect of ground-based thinning was a greater proportion of retained shrub cover (Figure 3.12) compared to the cable thinned areas (primarily of *Acacia dealbata, Pomaderris apetala* and *Olearia argophylla*), especially in areas free of direct disturbance from the logging machine (a feller buncher). Ground-based thinning was associated with an increase in the cover of herbaceous species relative to the cable thinned quadrats (28.9%, compared to 6.5-23.1% in the cable thinned quadrats) and the unthinned control. The increased cover of herbaceous species following ground-based thinning was attributed predominantly to increases in *Acaena novae-zelandiae, Agrostis rudis, Echinopogon ovatus* and *Hydrocotyle hirta* (Table 3.8). Following ground-based thinning, many of the herbaceous species, apart from colonising disturbed soil, adopted an hemi-epiphytic (a plant that maintains a vascular connection with the soil over part of its life, Benzing 1990, 1995) or accidental epiphytic (a typically terrestrial species which occasionally grows to maturity as an epiphyte, Benzing 1990, 1995) habit (Figure 3.13), and colonised logs (Figure 3.13) which had been on the ground since the original 1961 harvest. Cable thinning saw the richness of herbaceous species decreased by between 11 and 17% (Figure 3.14). By comparison after ground-based thinning, it increased by 18%. Ground ferns increased their cover following ground-based thinning (26.7%) relative to the control quadrats (15%), although the increase was considerably less than that observed in cable thinned quadrats (48-54%). Rhizomatous ground ferns including *Histiopteris incisa*, *Hypolepis rugosula* and *Polystichum proliferum* (Table 3.8) were primarily responsible for this trend.

Comparison of cable thinning intensity

While increasing the intensity of cable thinning provided a more open forest canopy (Table 3.7), it made no effective difference to soil or litter disturbance (Table 3.7) or species richness (Figure 3.14). The response of the understorey vegetation to increasing thinning intensity appeared to be an increase in ground fern cover and a decrease in herbaceous cover (Figure 3.12). Any other floristic changes attributable to thinning intensity are probably undetectable at such an early stage in the recovery of the understorey vegetation. Total richness of understorey species was also affected by thinning. The number of species present in the control quadrats (36) was similar to the range of values in the thinned quadrats (26-38).

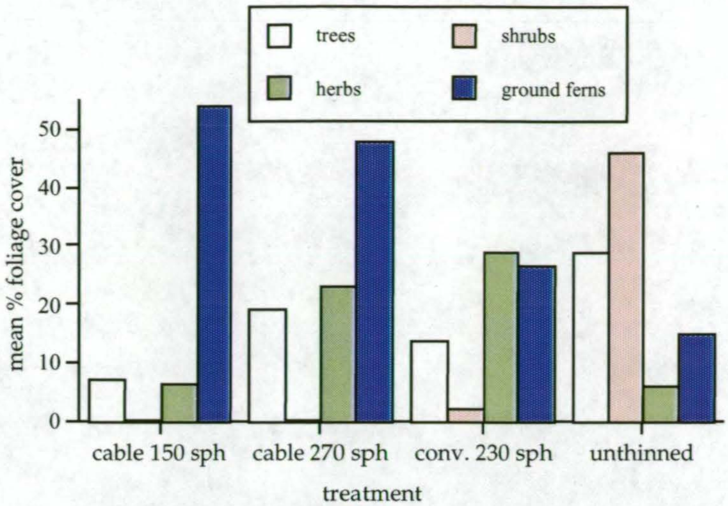


Figure 3.12 *% cover of various life forms in the four thinning treatments; cable thinning to 150 stems per hectare, cable thinning to 270 stems per hectare, conventional (ground-based) thinning to 230 stems per hectare, unthinned control.*

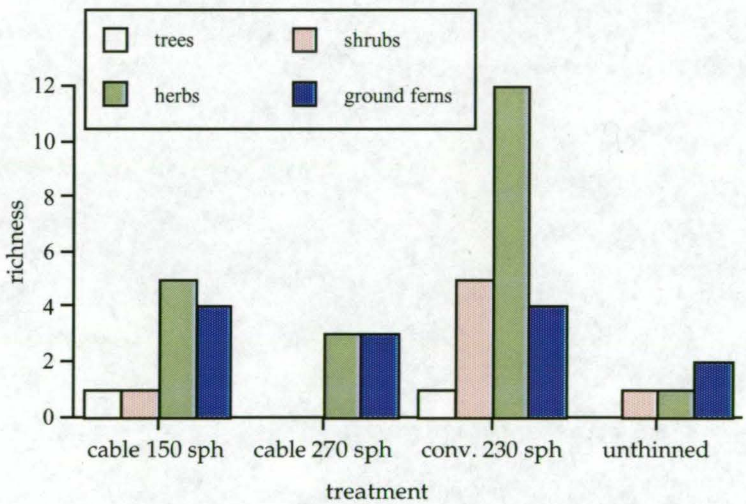


Figure 3.13 *Richness of hemi-epiphytic and accidental epiphytic species summed across the two replicate 0.09 ha quadrats in each of the four thinning treatments.*

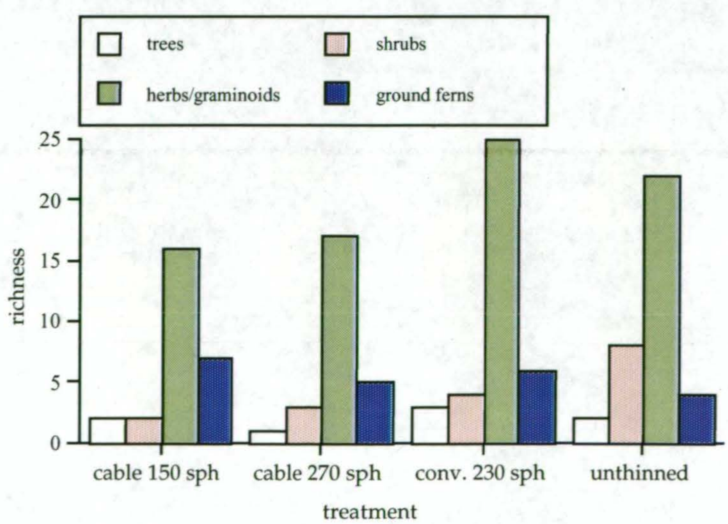


Figure 3.14 *Species richness of various life forms in the four thinning treatments*

Figure 3.15a,b *Unthinned control area (left) and cable thinned area (right) approximately six months after treatment.*

Figure 3.16a,b *Ground disturbance along extraction routes associated with ground-based (left) and cable thinning (right). Large residual logs felled in the original 1961 harvest and not extracted in cable thinned area were partially or fully crosscut to assist cable extraction along the outrows and minimise damage to retained trees (see diagram in Figure 3.4).*

3.4 Discussion

The pattern, extent and severity of soil surface disturbance resulting from cable logging closely resembled the patterns observed elsewhere in Tasmania (Laffan *et al.* 2001) and overseas (Miller and Sirois 1986). Draglines were particularly visible following cable logging, forming radiating lines from the centralised landing areas. Dragline width varied from 1-4 m, and extended for hundreds of metres where the topography was suitable. Variation in dragline length and depth was related to the slope configuration (e.g. convex slopes) more than local soil attributes. Butt gouging of the soil surface in the manner described by Smith and Wass (1976) was infrequent. Litter displacement on dragline edges, associated with a compacted central path with a reduced organic content (e.g. Laffan *et al* .2001) was commonly observed in the experimental area.

At Wielangta, at least 90% of the area mapped as draglines, were on closer inspection found to be lines of flattened vegetation and barked stripped away from logs during cable hauling. Less than 5% of the total dragline area was bare or exposed soil. It is relatively uncommon to not burning post-logging residue in dry forest in Tasmania (McCormick and Cunningham 1989), and this may account for the lower area of exposed soil. Draglines and soil disturbance, when present, were minimal in area and concentrated at points where the logs were in contact with the ground (Lucci 1987) and near the landing (Laffan *et al.* 2001). Where shallow troughs of soil formed as a result of logs skidding along the surface, they were, at least partially, covered by scattered litter and slash as described by Dryness (1967). Draglines provided a favourable seed bed for species preferring mineral soil for establishment, contrasting with other recruitment surfaces after logging which often had accumulated logging residue. The soils of draglines differ from adjacent soils by their reduced hydraulic conductivity and increased bulk density (Smith *et al.* 1988, Purser and Cundy 1992), reduced organic matter (Guo *et al.* 1997) and a propensity to actively erode in the short term (Fransen 1998). With their compacted and nutrient poor microsite conditions, draglines offer an unfavourable seedling environment immediately after logging (Pinard *et al.* 1998). The accumulated bark and flattened *Gahnia* tussocks which characterised the dragline surfaces at Wielangta appeared to suppress recolonisation by herbaceous species, similar to the effect of logging residue suppressing graminoid regeneration described by Olsson and Staaf (1995). With a relatively small proportion of the Wielangta cable logged area affected by draglines, the overall affect on regeneration (mainly a suppressive one) was considered minimal compared with disturbance from ground-based logging.

The regenerating vegetation on draglines differed from adjacent regrowth at both Loatta and Wielangta by its reduced vegetation cover, especially of those species regenerating from tussocks, rhizomes and persistent rootstocks. The reduced cover of these species may be a direct result of the soil disturbance along the dragline, removing or damaging the buried rootstocks, or removing part of the soil cover, enhancing vulnerability to the regeneration burn at Loatta. While a silvicultural burn may hinder regeneration via vegetative means, it may advantage vegetation re-establishment for species germinating from seed. Weed species, especially those with wind dispersed propagules, were more common on draglines, although their overfall cover in the cable logged area was low compared to ground-based logging.

In the short term (two years), at Wielangta vegetative recolonisation of draglines was inhibited by the accumulation of logging debris and flattened vegetation. Five years after logging and burning at the Loatta the differences between dragline and adjacent vegetation still existed. The number of weed species was greater, and the proportion species regenerating vegetatively was lower. Ruderal species and herbaceous species in general were more frequent on draglines, they are advantaged by soil disturbance resulting from cable or ground-based logging (Brumelis and Carleton 1988) and may dominate the site and preclude other species from establishing (Gordienko and Gordienko 1988). Vegetation cover after five years was still 40% less than adjacent regrowth, and is within the magnitude of change for draglines described by Fransen (1998). The reduced vegetation cover probably resulted from increased runoff, a reduction in soil stored propagules and lack of safe sites for germination. The compositional differences between the draglines and adjacent logged areas are likely to persist and lead to a retarded successional process on the most severely disturbed portions of the logged area. Similar conclusions have been reached from comparative observations on snig track and adjacent areas in both wet and dry forest communities in Australia (Williamson and Neilson 2002, Loyn *et al.* 1983) and softwood forests in Canada (Smith *et al.* 1988).

The strongest influence on the composition of the ground flora in young even-aged forests and plantations is the nature and biomass of the overstorey (Young *et al.* 1967, Crawford 1976, Hill 1978, Metzger and Schultz 1984, Alaback and Herman 1988, Taylor *et al.* 1988, Fuhr *et al.* 2001). The processes operating include light penetration (Hill 1978), soil moisture, litter fall, root competition and soil temperatures (Dryness *et al.* 1988). *Eucalyptus regnans* forests exhibit rapid early growth in the first three decades following stand initiation (Ashton 1975b), corresponding with a ruthlessly episodic and spatially clustered pattern of individual tree self-thinning mortality (Burgman *et al.* 1994). The thinning regimes implemented in this study effectively reduced the tree stocking to a level that, according to stand tables used by local forest managers, would have occurred in the following two decades through natural self-thinning when the less vigorous trees in the stand are becoming sub-dominant and suppressed (Opie *et al.* 1994).

Differences in the response of the understorey vegetation in this study to the two cable thinning intensities implemented at Jungle 3 were of less significance floristically than the differences due to thinning method and between thinned and un-thinned vegetation. An increase in thinning intensity can lead to a wide variety of vegetation changes, including an increase in ground fern cover (Alaback and Herman 1988), a reduction in woody species (Reader 1987), or an increase in woody species (Crawford 1976, Wetzel and Burgess 2001), a reduced cover of herbs, an increase in vines and broadleaved species (Shelton and Murphy 1994) or ground layer species (Ducrey and Boisserie 1992, Alaback and Herman 1988) or alternately it may not effect composition and richness (Locasio *et al.* 1990), or produce a highly variable response (Clement and Keeping 1996, Alaback and Herman 1988). Where compositional differences due to different thinning treatment intensities are reported, the responsible mechanisms have included differential shade tolerances (Deal 1999), the competitive interaction of woody and herbaceous species promoting shade tolerant species (Crawford 1976), and surface disturbance (Reader 1987). Brunet *et al.* (1996) suggested floristic change following thinning is minimal in the absence of soil disturbance.

A consistent theme from differing forest types (Horsley and Marquis 1983, Bailey and Tappeiner 1998, Alaback and Herman 1988) and both mechanical and chemical thinning (Pitt *et al.* 2000) in the understorey species response is a rapid increase in rhizomatous ground fern species (Crow *et al.* 1991). Rhizomatous species have the potential to store resources or acquire new ones through lateral spread in dense forests (Lezberg *et al.* 1999) An increase in the cover of ground ferns can also lead to a decrease in those species requiring germination spaces to reproduce (Horsley and Marquis 1983). Canopy reduction by cable thinning was directly correlated with an increase in the cover of ground ferns, with the most intensive cable thinning treatment being associated with a ground fern cover greater than 50%, compared to approximately 15% in the unthinned control. This increase was associated with minimal ground disturbance in the cable thinned treatment disturbing their rhizomes, the effective removal of the shrub layer, the thinning of the eucalypt canopy and increased light availability.

The increased cover of ground ferns post-thinning at Jungle may have retarded the development of other understorey species, similar to the trends observed after woodland thinning (Kirby 1990) or landslide revegetation (Guariguata 1990). Ground ferns will increase in frond cover, length and number following small gap formation in the absence of soil disturbance (Collins and Pickett 1987). Hughes and Fahey (1991) interpreted this type of post-thinning response in ground ferns as being the result of the survival of individuals and the expansion of existing patches. In other examples of clonal (O'Dea *et al.* 1995, Huffman *et al.* 1994), vegetative (Messier and Kimmins 1991, Messier and Mitchell 1994) or horizontal (Mallik *et al.* 1997) expansion following thinning, it was interpreted as a result of either increased light availability or thinning slash encouraging stem layering. Short term changes (after thinning with minimal soil disturbance) in species cover such as these (e.g. Riegel *et al.* 1995, Hughes and Fahey 1991) have been considered by some authors as insufficient to disrupt long term vegetation recolonisation patterns or reduce herbaceous species diversity (Reader and Bricker 1992), and by other authors, they have been implicated as potentially out-competing other ground flora over the longer term (Rooney and Dress 1997) and preventing the natural establishment of tree and shrub species in the ground layer (Huffman *et al.* 1994).

Trends in herbaceous species richness and cover were opposite to those described for ground ferns, herbaceous species increased in cover and richness where ground-based thinning was used and the canopy was less intensively thinned. Increases in herbaceous species are commonly reported following ground-based thinning (e.g. Young *et al.* 1967, Metzger and Schultz 1984, Brunet *et al.* 1996, Uresk and Severson 1998,) although other authors have reported more variable responses such as increased frequency, no change, or a decrease (Reader 1988), a short term increase (Reader and Bricker 1992) or an eventual decrease in herbaceous cover as other life forms, such as woody species increase in cover (Crawford 1976, Bailey and Tappeiner 1998). While changes in the herbaceous flora can be rapid following soil disturbance from thinning, the effects of the canopy removal on herbaceous species may be more subtle (Frego 2000). Some of the variation between these studies in the response of the vegetation to thinning may be due to variability in stand age at treatment (Deal and Farr 1994).

Soil disturbance associated with thinning was an important mechanism for the response of understorey species, particularly herbs and other invasive species. During thinning, the severity of soil disturbance is a function of machine type, soil conditions and the number of load passes (Murphy 1982). The actual area affected is then dependent on the thinning intensity, the planning and location of skid routes, and the log bunching system used (Murphy 1982). Soil disturbance from gouging and litter removal in the cable logged quadrats differed from soil disturbance caused by tracked machinery during ground-based thinning (0.2-0.45 and 38% respectively). Although compaction was not measured in this study, other authors comparing cable and ground-based thinning concluded ground-based techniques cause significantly more (23% in Wronski 1984) soil compaction than cable systems (McIver 1998). On the ground-based thinned area, rutting was observed where the thinning machinery followed snig tracks created 30 years ago during harvesting of the original old growth stand. These snig tracks were colonised by ruderal species while openings in the cable logged portion were colonised by herbaceous genera relying on vegetative spread (e.g. *Oxalis, Viola*). Most of the herbaceous species common on the ground-based thinned quadrats were able to expand their cover on the more disturbed areas where the soil and little layers were disturbed by the tracked machine. An interactive effect between disturbance on the one hand and a lack of tree cover is recognised, luxuriant herbaceous ground cover reported by Zoysa *et al.* (1986) on a skid trails was attributed to the lack of tree cover on these more open areas, not necessarily to the skid trail disturbance itself. The weed species *Cirsium vulgare* spread throughout the ground-based thinned area presumably because of the larger area of disturbed soil and open ground, or the absence of the previous shrub and herbaceous layer (Clement and Keeping 1996). Herbaceous species in the ground-based thinned area were also more frequently observed to adopt an epiphytic habit, the preferred substrate being large downed logs. These trends in increasing cover of weedy or invasive species following soil disturbance associated with ground-based thinning have been reported elsewhere (Young *et al.* 1967, Metzger and Schultz 1984). With increasing thinning intensity, the number of woody species lost can increase (Reader 1987), while changes in herbaceous species are not necessarily related to thinning intensity. In this study ground-based thinning caused proportionally more ground disturbance and this was correlated with an increase in the weed species *Cirsium vulgare.*

One of the legacies of cable and ground-based thinning is the volume of slash residue. Following cable thinning, up to half of the ground was covered by slash, rising to 70-80% in the ground-based quadrats, which is high compared to results reported elsewhere (e.g. 10-15% in Rosen *et al.* 1987). Additionally, the volume of woody residue persisting from logging in 1961 was approximately 120 tonnes/ha (A.N.M Forest Forestry Commission of Tasmania 1992.). The type, depth and distribution of thinning slash also appeared to differ between the cable and ground-based areas, being distributed more evenly in the cable area, a trend recorded elsewhere (Hall 1991) and of greater volume in the cable area, probably due to its less impacted state (Woodward and Henry 1985). Slash can alter the micro-climate, degree of plant browsing protection, suppress (Hill 1978) or kill established vegetation cover and prevent its re-establishment (Metzger and Schultz 1984), alter soil chemistry (Rosen *et al.* 1987) and suppress suckering (Bella 1986). It can favour species using clonal expansion (O'Dea et al. 1995) and reduce those using suckering (Bella 1986).Slash covering of extraction routes will also reduce soil compaction (McDonald and Seixas 1997, Jakobsen and Moore 1981). Similarly, a comparison of the pre-thinning and post-thinning (4 months later) data of Moorees (1988) and Griffiths and Muir (1990) from East Gippsland suggests thinning slash also appeared to be suppressing the short term recovery of much of the understorey except for sprawling species such as *Tetrarrhena juncea* and *Cassytha phaeolasia.* Thinning slash is likely to restrict the ability of the understorey to respond vegetatively to increased light levels and may favour species with a sprawling or trailing habit which can reproduce vegetatively amongst thinning slash i.e. rhizomatous ground ferns. These trends may be short lived however, as thinning residue can rapidly loose its volume and depth (Christiansen and Pickford 1991) with much of the smaller diameter material decaying within two to four years (Christiansen and Pickford 1991, O'Connell 1997).

It is unclear from these initial results comparing cable and ground-based thinning if the patterns of secondary succession described for regrowth forests in Chapter 2 will differ in regrowth subject additionally to thinning disturbance. The practice of cable thinning regrowth native forests overseas (McNeel and Dodd 1996) and in Australia is an area of evolving forest practices. Where researchers have monitored thinning conducted 50 years ago (Metzger and Schultz 1981) it has provided new insights into the persistence of invasive species and of the effects of repeated thinning on the vegetation composition. The trends described from the current study may not persist with time and, in the long term, treatment differences in species composition and cover may be small. The dense shrub understorey stratum is likely to recover and strongly influence again the ground layer vegetation. One of the difficulties of attempting to extrapolate from short term results is that different species may respond to environmental change (e.g. thinning) at different rates. Reader (1988) suggested that variation in the response of herbaceous species to thinning was, at least partly due to differing response times, and that by lengthening the study, species that showed no apparent short term response to thinning may do so. Short-term vegetation responses to thinning intensity are likely to reflect the intensity of the logging disturbance, whereas longer-term trends are expected to reflect the effects of variation in overstorey retention (Halpern *et al.* 1999).

The ability of the mid stratum of tall shrubs (e.g. *Pomaderris, Phebalium, Acacia dealbata*) to re-establish may be crucial if the structure of the understorey vegetation on the thinned quadrats is to recover. Patterns of secondary succession in understorey vegetation (Chapter 2) for stands 21-30 years after cable logging appeared to be mediated by micro environmental changes associated with the self-thinning of the dense tall shrub stratum, and micro-habitat features (Chapter 6). Thinning may limit the contribution of large coarse woody debris to the forest floor (Hayes *et al.* 1997, McCarthy and Bailey 1994, Fleming and Freedman 1998) for a period of approximately 20 years (Prescott 1997). Thinning machinery may also crush large logs in advanced decay stages (McCarthy and Bailey 1994). Large diameter fallen logs are often a legacy of earlier stands (Hayes *et al.* 1997, Lindenmayer *et al.* 1999), and at Jungle they were abundant from the 1961 harvest event. While an ample decaying log substrate was present to facilitate the re-establishment of vascular epiphytes and rainforest tree seedlings, other specialised substrates such as *Olearia argophylla* bark and *D. antarctica* caudexes were lacking. Vascular epiphytic species such as *Microsorium pustulatum* were too infrequent in the thinning study to attribute any treatment effect to their presence. The rapid removal of the understorey stratum during thinning, though an unintentional aspect of machinery movement, may therefore disrupt the normal pattern of secondary succession in these forests, removing future resources such as bark substrates for epiphytic establishment. The rapid response of ground ferns to canopy opening through cable thinning may also result in the competitive exclusion of herbaceous and shrub understorey species.

In conclusion, soil disturbance from either dragline formation during cable logging or from thinning operations within regrowth forest have a detectable impact on the process of vegetation recovery from logging disturbance. It is suggested that much of the localised impact of draglines is due to vegetation suppression by heavy logging slash, and where soil was exposed, it provides a seed bed for species not regenerating vegetatively. Soil disturbance during ground-based thinning operations is potentially more significant, as it is providing a regenerating niche for ruderal and weed species which had otherwise been competitively displaced shrub stratum closure in the first decade following post-logging regrowth establishment.

Chapter 4 Effects of cable logging on streamside vegetation

4.1 Introduction

Retaining streamside reserves, filter strips, corridors or buffers has been a common approach to protecting stream environments in production forests since the 1960's and 1970's in North America (Gregory 1997, Hibbs and Bower 2001) and Australia. The use of streamside reserves gradually evolved from concern over minimising sediment input to watercourses (e.g. Clinnick 1985) to a broader view that they should aim to protect streamside aquatic and riparian habitat values (Dignan and Bren 2003) and allow natural interactions between the riparian and aquatic systems to be maintained (Ried and Hilton 1998). For streamside reserves in production forests to function effectively, they must not only withstand the direct effects of logging and burning on their margins, but remain as intact, windfirm and healthy forest stands.

Despite the functional importance of riparian vegetation in the landscape (Gregory 1991, Gregory *et al.* 1991, Naiman *et al.* 1993, Naiman and Decamps 1997) and within steep logging areas in particular (Coker 1992) few studies have directly addressed the response of riparian vegetation to logging disturbance. No published studies of the impacts of logging on streamside reserve vegetation in Australia are known. Most of the existing studies use water or light as the response variable, with the description of vegetation change simply being a means of describing how the site was disturbed (e.g. Clinnick 1985, Erman *et al.* 1977, van Groenewoud 1977, Davies and Nelson 1993, Doeg and Koehn 1990, Campbell and Doeg 1984, 1989, Dignan and Bren 2003).

Streamside vegetation exhibits variable responses to ground-based clearcut or partial logging, from nil effect (Hibbs and Bower 2001), to minor short (Messina *et al.* 1997) or medium (Hibbs and Giordao 1996) term effects, to rapid short term changes in composition and dominance (France 1997, Timoney *et al.* 1997). The most pronounced responses are in the herbaceous layer (Sagers and Lyon 1997, Pabst and Spies 1998). Cable and ground-based logging results in a similar increase in pioneer and shrub species abundances (Andrus and Froehlich 1988), differences in the degree of soil disturbance from either treatment are rapidly negated by the herbaceous colonising species (Keim and Schoenholtz 1999).

The diverse nature of riparian zones, both in terms of species composition and structure, complicates their disturbance response to the extent that Agee (1988) considered them both sensitive to disturbance and at the same time a product of disturbance. Lamb *et al.* (2003) effectively validated this concept by arguing that the adaptation of riparian vegetation to high-light environments and flooding disturbances made the floristic response to adjacent logging or wildfire a relatively insignificant one.

Factors effecting the integrity of streamside reserves within production forests include the arrangement of the cable logging system (Sauder and Wellburn 1987, Krag *et al.* 1987), direct logging damage (Gratowski 1956, Palmer *et al.* 1996), log spearing (Sauder and Wellburn 1987), edge effects (Chen *et al.* 1993), windthrow (Alexander 1964, Moore 1977, Steinblums *et al.* 1984, Garland 1987, Chatwin 2001), escaped regeneration fires weakening trees and pre-disposing them to further windthrow (Moore 1977), myrtle wilt (Packham 1991, Cameron and Turner 1996) and increased fire risk (Neyland 1991). Responses by the streamside vegetation may not be immediately obvious, the onset of visible symptoms can be delayed by several years (Hartman *et al.* 1987). Methods such as cable corridors or full log suspension (Garland 1987) can only attempt to minimise such damage on steep streamside reserves, full suspension logging over streamside reserves is almost impossible to successfully implement (Adams *et al.* 1988) unless long distance skyline rigging or more elaborate lateral yarding systems or even helicopter logging is used (Moore 1991, Palmer *et al.* 1996, Boswell 1999). Various operational cable logging trials in Tasmania since the 1980's have effectively confirmed this experience to the extent that the current Forest Practices Code (January 2000) specifies minimum horizontal reserve widths on class 1, 2 and 3 streamside reserves and log yarding across these reserves is not permitted, although cables may be pulled through them (but not be dragged laterally across) if minimal damage occurs.

No accounts of floristic change following cable logging within streamside or riparian vegetation are known. Further, no published information of the impacts of logging on streamside reserve vegetation in Australia is available.

Three hypotheses are tested in this chapter;

(1) the method and intensity of cable logging directly effects the degree and nature of post-logging floristic response,

(2) cable logging adjacent to small strips of retained streamside vegetation will have both immediate and on-going effects on health and survival, and

(3) that the detrimental effects of cable logging adjacent to retained streamside vegetation is the result of a range of forest practices, only a portion of which are directly attributable to the process of cable logging.

4.2 Methods

Three studies of the response of streamside reserve vegetation to cable logging were conducted;

1. the effect of method and intensity of cable logging adjacent to and through the streamside reserve on vegetation species composition and cover,
2. an examination of the immediate and on-going effects on tree health and survival of cable logging adjacent to small patches of retained streamside vegetation, and
3. an examination of the relative contribution of different forest practices to damage and ill-health of vegetation retained in streamside reserves

In this chapter the results of these experiments are presented for woody and herbaceous species only, data on *Dicksonia antarctica* and vascular epiphytes are presented in Chapters 5 and 6 respectively.

Experiment 1: Effect of method and intensity of cable logging adjacent to and through the streamside reserve on vegetation

Study Areas

Three study areas in northern Tasmania were sampled to examine the response of the vegetation in streamside reserves to cable logging treatments of different types and intensities carried out adjacent to or through the retained vegetation. The study areas, all logged 2-3 years prior to sampling with the same cable yarder, (a Skagitt Bu 739 live skyline slackline yarder, Figure 1.9) were Roses Tier 126a, Roses Tier 137f, Roses Tier 137g. The study areas were selected on the basis of similar pre-logging vegetation (from forest type mapping, Stone 1998, Table 4.1), stream size (class 2 in Forest Practices Code 1993), stream slope (5^o), geology (granodiorite) and soils, and were within an area of $3km^2$ (Figure 4.1).

The soils of the study area consist of a 40cm+ dark (organic rich) friable sandy loam surface layer overlying a brown very friable sub-soil of course sandy loam, except on the north west facing slopes where the soils were thin and rocky. The soil is well drained, highly permeable and all layers have high aggregate stability. The erodibility rating was assessed as low. In areas burnt post-logging the course organic component was reduced (Laffan *et al.* 1998). The geology is Upper Devonian granodiorite and the landform is moderate to steep (40^o) slopes of mainly northerly and easterly aspects often resembling V shaped gullies down to the streamsides, or occasionally flattening out at the base of the slope to form streamside flats. The rainfall of the Roses Tier area is estimated to be 1250-1500 mm per year.

Figure 4.1 *The location if the three coupes studied at Roses Tiers, northeastern Tasmania, April 1992. The coupes Roses Tier 126a, 137f, 137g are shaded and the sampled portion indicated with a rectangle. From Ben Nevis 5441 1:25,000 sheet. The grid spacing is 1 km.*

Table 4.1 *Quadrat locations at Roses Tier, northern Tasmania.*

Cable logging coupe and quadrat number	Year logged	Easting	Northing	Altitude (m)	Forest type (after Stone 1998)	Stream side slope
Roses Tier 126a						
quadrat 1	1989	457400	5314400	650	E. regnans and	30^0
quadrat 2		457600	5314600	630	Callidendrous Rainforest	
quadrat 3		457700	5314700	600	T(W)M3	
quadrat 4		457700	5315000	550		
Roses Tier 137f						
quadrat 1	1989	458600	5316600	530	E. regnans/E. obliqua and	45^0
quadrat 2		458400	5316700	450	Callidendrous Rainforest	
quadrat 3		458300	5316700	450	M2.T.	
quadrat 4		457900	5316900	500		
Roses Tier 137g						
quadrat 1	1990	458200	5315500	470	E. regnans and	40^0
quadrat 2		458300	5315500	460	Callidendrous Rainforest	
quadrat 3		458300	5315600	450	T(W)M3	
quadrat 4		458400	5315700	450		

Experimental design

The experimental design followed is post-logging monitoring with controls, a relatively common design used in studies of disturbed environments (e.g. Davies and Nelson 1994). This method eliminates pre-logging surveys and replaces them with control sites, a reasonable surrogate for comparing pre-harvest sampling levels to post-harvest levels (van Rees *et al.* 2001). The post-logging parameters of the vegetation are compared with the values on unlogged sites. Stratified random sampling of vegetation was conducted with the survey stratified in relation to three categories or levels of cable logging intensity; high (RS137g), moderate (RS126a) and low (RS137f), which were based on cable logging categories in management records (Table 4.2, 4.3). Control areas were identified on both the upstream and downstream portions of the logged area (ie. outside of the coupe) to minimise vegetation variability due to changes in environmental gradients along the streamside. Four areas were identified for stratified random sampling along the streamside, with the total distance between the four quadrats being 100-200m, a compromise between introducing extraneous variation from sampling over a wider geographic area and auto-correlation from too confined an area (Ash and Pollock 1999, Petranka 1999). Temporal and spatial variation in the experiment is assumed to be minimal due to the close proximity of the coupes (an area of $3km^2$, Figure 4.1), similar pre-logging vegetation type and because they were all logged and burnt in 1989 or 1990.

Table 4.2 *Vegetation characteristics and cable logging practices at Roses Tier, north eastern Tasmania.*

	Roses Tier 137g Memory Creek (class 2)	Roses Tier 126a tributary of Memory Creek (class 2)	Roses Tier 137f Tombstone Creek (class 2)
intensity of cable logging	high	Moderate	low
pre-logging vegetation surrounding streamside	*E. regnans* and *E. obliqua* wet forest	*E. regnans* wet forest with a broad-leaved shrub and ferny understorey and a variable component of *Nothofagus* and *Atherosperma*	*E. regnans* wet forest with an understorey of broad leaved shrubs (e.g. *Olearia argophylla*) and rainforest trees (*Nothofagus* and *Atherosperma*) towards the drainage lines
type of cable logging disturbance	A motorized dropline carriage was used initially to laterally yard logs to the skyline. When deflection is poor, as was the case here, pulling the skyline laterally with a motorised carriage is going to be of greater consequence to the streamside vegetation with the swinging of the skyline and rolling of the logs. When trees were felled, they tended to spear downslope into the streamside reserve. Damage to the reserve also resulted from directional falling difficulties and logs moving downslope towards the streamside when tension was taken up on the skyline before the inhaul.	Skyline tracks or cable corridors were cut through the streamside reserve to allow the eastern slope above Memory Creek was to be logged. The plan stipulated that a minimum number of narrow skyline tracks be cut, that the cables not drag laterally across vegetation between these tracks, that debris in the main stream course be removed and trees felled in a manner to minimise potential damage to retained vegetation.	A small number of narrow skyline tracks were cut through the streamside reserve to tie off the cable on the opposite bank within of Tombstone Creek Forest Reserve
forest practices on streamside	Full log suspension over Memory Creek was attempted. A post-logging "cleaning up" of the logged gap across Memory Creek was ordered with the felling of all damaged or broken trees to cut stumps and removal of logging debris from stream channel.	Damage from logs and cables passing through the streamside reserve was minimised with the deliberate cutting of cable corridors through it. 30m streamside reserve	Minimal disturbance from pulling cables through to opposite bank and flexing of cable during it use 40m streamside reserve.

Table 4.3 *Survey design and quadrat treatment attributes at Roses Tier, north eastern Tasmania.*

Quadrat No.	Burnt/ Unburnt	Treatment attributes	Reserve Width
Roses Tier 137g			
1	unburnt	upstream undisturbed control	60m
2	burnt	severe, logged portion with no canopy on steep slope	0
3	burnt	severe, logged portion with no canopy on flat area	0
4	unburnt	downstream with control little disturbance	30m
Roses Tier 126a			
1	unburnt	Upstream control, unlogged on both stream edges, sampled 10 m above creek bank	>25m
2	unburnt	in cable corridor cut 10-20 m wide through streamside reserve, sampled 15 m above creek	corridor
3	unburnt	normal portion of streamside reserve between cable corridors, sampled 15 m above creek	25m
4	unburnt	downstream control, unlogged on both stream edges, sampled 10 m above creek	>25m
Roses Tier 137f			
1	unburnt	undisturbed downstream control	within Tombstone Creek Forest Reserve
2	unburnt	in cable corridor within streamside reserve	10-20m wide and 70-60m deep gap
3	unburnt	in bulging portion of streamside reserve	bulge is up to 80m wide
4	unburnt	undisturbed upstream control	45m

Sampling methods

Each coupe (RS137g, RS126a and RS137f), was divided into four portions; (1) disturbed by cable logging, (2) reserved vegetation area, (3) upstream control and (4) downstream control and within each of these portions data were collected within five 2 x 2m contiguous sub-plots (arranged into a segmented 10 x 2m continous quadrat). Within each 2 x 2m sub-plot a full vascular species list was compiled with Braun-Blanquet abundance data recorded for each species within or overhanging the sub-plot. These sub-plots were regarded as pseudo-replicates within the 10 x 2m area and used to generate a mean percentage foliage cover score.

Analytical procedure

Data from each of the 2 x 2m sub-plots was converted from Braun-Blanquet cover scores to their mid-points in the respective ranges (see section 2.2). These mid-point scores were averaged over the five sub-plots to derive a mean percentage foliage cover score for each species for each floristic quadrat. The analysis methods adopted (NMDS ordination, multi-response permuation procedures, vector fitting, UPGMA clustering, and indicator species analysis) follow the descriptions in section 2.2.

Experiment 2: The immediate and on-going effects on tree health and survival of cable logging adjacent to small patches of retained streamside vegetation

Study Area

The study area was a minor feeder stream of Wallaby Creek, part of the Branchs Creek catchment within the Dazzler Ranges, northern Tasmania (Table 4.4). The topography of the area was steep (slopes of 24-32^{o}) with streams situated between deeply incised narrow interfluve ridges on sedimentary soils (Figure 4.2). The stream channel had a slope of approximately 8^{o}. The underlying geology was un-metamorphosed Precambrian quartzwackes and phyllites (McClenaghan and Baillie 1975). Infertile phyllite material occurred in the vicinity of the transect. Sedimentary soils were shallow on hill slopes surrounding the stream, but at the base of the slope, along the stream course, deeper alluvial soils occurred.

Table 4.4 *The location of the vegetation quadrats at Branchs Creek 80a, northern Tasmania, 1993*

Quadrat	Year logged	Easting	Northing	Altitude
quadrat 1	1988	473600	5333500	200
quadrat 2		473600	5333500	190
quadrat 3		473600	5333600	190
quadrat 4		473600	5333500	180

Figure 4.2 *The Branchs Creek study area in north eastern Tasmania with its steeply incised narrow drainage lines and narrow ridges.*

Prior to logging, the vegetation on the streamside studied was photo-typed as S.T(W)E2fc/o, indicating that the eucalypts were present only as remnants of an old stand with a crown cover of <5% over a scrub (S), secondary species (T) and wattle (W) understorey (after Stone 1998). The vegetation of the stream prior to logging was classified floristically by Neyland (1991) as Callidendrous Myrtle Rainforest (Riverine).

Branchs Creek 80a was cable logged between August and December 1988 using was a Madill 071 with a skyline rigging. The cable was dragged laterally across the streamside vegetation during movements between settings. The regeneration burn around the streamside reserve in September 1989 was described in the management records as low intensity.

Survey design

Two surveys were undertaken. One re-measured a permanent monitoring transect established in 1988 (Neyland 1991) along a tributary of Branchs Creek. This transect followed the watercourse for appoximately 100 m and provided a continuous measure of the level of disturbance in the streamside reserve. It was established immediately after cable logging was completed and was measured three times over five years (Table 4.5). The second survey used floristic quadrats stratified in relation to the degree of disturbance from cable logging (Table 4.6). These floristic quadrats measured the same tributary of Branch's Creek, approximately 4m upslope from the stream channel. The 100 m. transect was stratified into four types portions on the level of cable logging and burning disturbance received (Table 4.6).

Table 4.5 *The treatments and sampling at Branchs Creek 80a.*

Event	Date	period since logging (months)
logging of coupe commences	August 1988	0
logging and yarding completed	December 1988	5
establishment of permanent transect	December 1988	5
first transect re-measurement	14 September 1989	14
coupe burnt	late September 1989	14
second transect re-measurement	3 August 1990	25
third transect re-measurement	20 April 1993	57
establishment and measurement of floristic quadrats	20 April 1993	57

Table 4.6 *Sample stratification for floristic quadrats established along the permanent monitoring transect at Branchs Creek 80a.*

Quadrat	Intensity of cable logging treatment	Treatment attributes	Reserve width	Distance along transect
1	high	burnt, severe, windthrow common	none	0-10 m
2	moderate	unburnt, open to light infiltration, windthrow evident	10-20 m	35-45 m
3	low	unburnt, little light infiltration, closed canopy	>20 m	87-97 m
4	control	unburnt and undisturbed confluence of sampled feeder stream and main branch of Wallaby Creek	>30 m	outside transect

Sampling methods

The variables in Table 4.7 were scored for every woody shrub or tree intersecting the permanent monitoring transect. For each of the four floristic quadrats in Table 4.6 data were collected within five 2 x 2m contiguous sub-plots (arranged into a segmented 10 x 2m continous quadrat). Within each 2 x 2m sub-plot a full vascular species list was compiled with Braun-Blanquet abundance data recorded for each species within or overhanging the sub-plot. These sub-plots were regarded as pseudo-replicates within the 10 x 2m area and used to generate a mean percentage foliage cover score.

Table 4.7 *Variables sampled for individual woody plants on the permanent monitoring transect at Branchs Creek 80a.*

Variable	scoring system
Species identification	Genus and species
Logging damage assessment	Absent (0), moderate (1), severe (2)
Coppice regrowth assessment	Absent (0), moderate (1), abundant (2)
Symptoms of ill-health or dieback	Absent (0), moderate (1), severe (2)
Fire damage assessment	Absent (0), moderate (1), severe (2)
Windthrow assessment	Absent (0), partial, e.g. crown only (1), severe, e.g. whole tree (2)

Analytical procedure

Variables scored for individual woody species in Table 4.7 were summed by species in 10 m segments. Data for the canopy dominants *Nothofagus cunninghamii* and *Atherosperma moschatum* is presented by species, while data for the remaining non-rainforest woody species (*Coprosma quadrifida, Eucalyptus obliqua, Monotoca glauca, Notelaea ligustrina, Phebalium squameum, Pomaderris apetala* and *Prostanthera lasianthos*) is pooled and presented in 10 m segments.

For each of the four floristic quadrats, data from the 2 x 2 m sub-plots was converted from Braun-Blanquet cover scores to their mid-points in the respective ranges (see section 2.2). These mid-point scores were averaged over the five sub-plots to derive a mean percentage foliage cover score for each species for each floristic quadrat. This data is presented in a tabular form.

Experiment 3: The relative contribution of different forest practices to damage and ill-health of retained streamside reserve vegetation

Study Area

The study area is near Wayatinah in southern Tasmania. Two replicate logging areas (Pearce 04 and Blue 09, Table 4.8) were selected for study because; (1) their streamside environments were considered representative of cable logged forests in southern Tasmania, (2) they were to be logged using the same type of cable systems, (3) they included both large and small watercourses, (4) neither had been logged previously, (5) buffers would be retained on both sides of their major streams and (6) they both had relatively long (>600 m) and continuous streamside reserves within the same logging area. The study areas were logged with a Madill 122 Swing Yarder cable machine using a mobile tail spar in 1992. A D7 dozer cleared a perimeter track adjacent to the streamside reserves to provide access for the mobile tail spar and provide a fire break between the streamside reserve and the area to be logged and burnt. The perimeter track was constructed either on the outer edge of the streamside reserve or further upslope on the steepest or rockiest sections where the mobile tail spar used cross slope uphill yarding. The streamside reserve width was determined by the stream classification in the Forest Practices Code (1993). Blue 09 encompassed a class 1 stream (Florentine River, minimum reserve of 40m) a class 2 stream (stream feeding into Florentine River, minimum reserve of 30m), and several drainage lines categorised as class 4 streams (no reserve applied). Pearce 04 encompassed a class 1 stream (Derwent River, minimum reserve of 40m) and a class 2 stream (Robinsons Creek, no adjacent logging or reservation required)

Table 4.8 *Sampling locations for the streamside reserve perimeter study at Wayatinah.*

coupe	locality	grid reference	altitude (m)
Pearce 04	Derwent River upstream of Wayatinah Rd crossing.	E458000 N5303800 (various)	210 on transect
Blue 09	Florentine River above the Derwent River	E458100 N5299800 (various)	220-260 on transect

Pearce 04 is located on the western side of the Derwent River and prior to logging consisted of 22 ha of *E. regnans* dominated wet sclerophyll forest in a mature of pole growth stage. The streamside reserve vegetation consisted of a scrubby forest with an overstorey of *E. obliqua* and *E. amygdalina* on a gravelly river terrace.

Blue 09 is located on the immediate slopes above the Florentine River and prior to logging consisted of 47 ha of wet eucalypt forest and rainforest. The streamside reserve was Callidendrous 1c Rainforest - *Atherosperma moschatum* over *Olearia argophylla - Dicksonia antarctica - Polystichum proliferum* (Jarman *et al.* 1984). *Atherosperma moschatum* is the rainforest canopy dominant upslope from the river, and *Nothofagus cunninghamii* is restricted to the riverine environment and the larger drainage lines. Some portions of the streamside reserve are occupied by small boggy drainage depressions and seepage areas where a tangled lower stature rainforest is present (Thamnic Rainforest of *Anodopetalum-Anopteris*, Jarman *et al.* 1984). The upper edge of the rainforest-wet eucalypt forest boundary formed the streamside reserve boundary.

Survey design

The experiment aims to describe post-logging and burning changes in tree health to species occupying the perimeter of the streamside reserve. The survey established and scored permanent monitoring transects along the streamside reserve perimenters at the completion of cable logging in order to facilitate repeated measurement of vegetation change in 30 m segments of the perimeter. The perimenter segments were then re-scored after silvicultural burning (Table 4.9). Pre-logging sampling if this perimeter could not be undertaken as its precise location could not be predicted prior to the completion of logging and firebreak construction. The potential for an interaction of logging and burning treatments on the streamside reserve perimenter is recognised but could not be examined seperately without independent treatments.

Table 4.9 *The treatment and sampling program for the study of streamside reserve perimeter change at Wayatinah, southern Tasmania.*

	Date	time since logging (months)	
		Blue 09	Pearce 04
Blue 09			
cable logging	3-9/1992	0	
perimeter post-logging transect established and scored	10/1992	7	
silvicultural burn	4/1993	13	
perimeter post-burning transect scoring	7/1993	16	
Pearce 04			
cable logging	9-12/1992		0
perimeter post-logging transect established and scored	12/1992		3
silvicultural burn	3/1993		6
perimeter post-burning transect scoring	7/1993		10

Sampling methods

The unit of sampling on the permanently established streamside perimeters transect were 30 x 20 m segments of the transect. The attributes in Table 4.10 were scored in each segment to a distance of 20m inside the reserve from the logged edge. At Blue 09 the perimeter assessment measured 1120 m of class 1 streamside and 285 m of class 2 streamside. At Pearce 04 the perimeter assessment measured 668 m of class 1 streamside and 120 m of class 2 streamside. Within each 30 m segment the variables in Table 4.10 were scored.

Analytical procedure

The relationships between the variables scored using ordinal categories in Table 4.10 were examined using Pearson product-moment correlations. Categorical data from the 30 m transect segments were summed by species, and the proportion of species in the various health or damage categories presented in tabular form. Slash and logging gap areas within the streamside reserves were approximated by multiplying their depth and width to provide an area in m^2.

Table 4.10 *Variables scored for 30 m segments of streamside reserve perimeter at Wayatinah, southern Tasmania.*

Variable	Post-logging	Post-burning
Reserve depth from logged edge to watercourse in metres	✓	
Tree logging damaged type (0) pushed, (1) felled, (2) buried, (3) butt damaged	✓	
Proportion of rainforest trees (Atherosperma and Nothofagus) physically damaged in each 30m segment; none visible, 0-10% of stems, 11-25% of stems, 26-50% of stems and >50% of stems	✓	
Presence of log spears, the species of tree and its length inside reserve	✓	
Logging gap depth inside reserve in metres	✓	
Logging gap width inside reserve in metres	✓	
Slash incursion depth inside reserve in metres	✓	
Slash incursion width inside reserve in metres	✓	
Tree regenerative coppice response +/-	✓	✓
Tree health symptoms (0) healthy (1) slight of ill health, (2) with yellowing leaves in *Atherosperma* or wilt like symptoms (Packham 1991, Cameron and Turner 1996) in *Nothofagus*.	✓	✓
Tree windthrow (0) Absent (1), partial, e.g. crown only (2), severe, e.g. whole tree (2)	✓	✓
Tree health symptoms (0) healthy (1) slight of ill health, (2) with yellowing leaves (*Atherosperma*) or wilt like symptoms (*Nothofagus*)	✓	✓
Tree windthrow (0) Absent (1), partial, e.g. crown only (2), severe, e.g. whole tree (2).	✓	✓
Reserve edge (0) burnt, (1) unburnt		✓
Reserve edge (0) formed fire break (1) no formed fire break		✓
Tree butt fire damage; (0) absent, (1) moderate, (2) severe		✓
Tree trunk fire damage; (0) absent, (1) moderate, (2) severe and its height (m.)		✓
Tree trunk fire damage height in metres		✓
Tree crown fire damage; (0) absent, (1) moderate, (2) severe		✓

4.2 Results

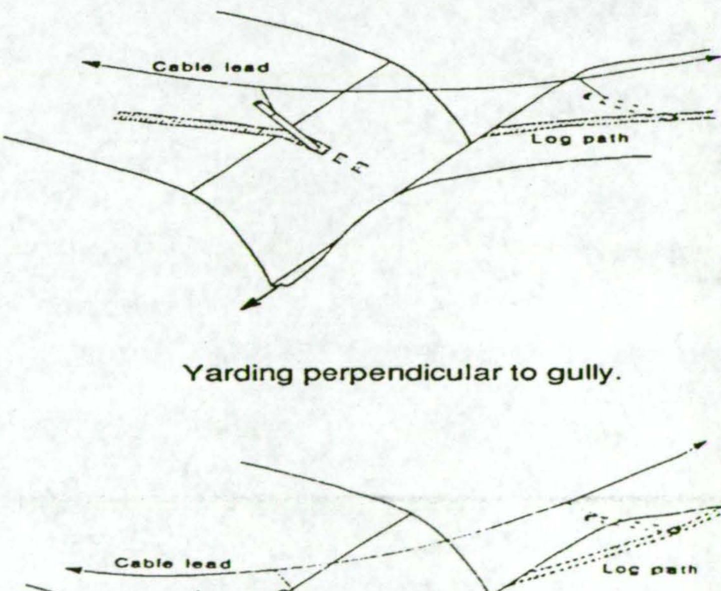
Experiment 1: Effect of method and intensity of cable logging adjacent to and through the streamside reserve on vegetation

The nature of the cable logging method at Roses Tier directly effected the type of physical disturbance to the vegetation and the subsequent short term (two-three year) floristic response. Minimal vegetation disturbance resulted from cutting skyline corridors through streamside reserve vegetation (Figure 4.3) in order to tie off the skyline cable. Direct disturbance from the corridors included a gap 10-20m wide and 60 m long and the felling of one mature *Nothofagus*. Disturbance from the flexing of the skyline cable tied off through the corridor included broken upper limbs and branches on *Atherosperma* and *Nothofagus* trees which subsequently responded with coppice growth.

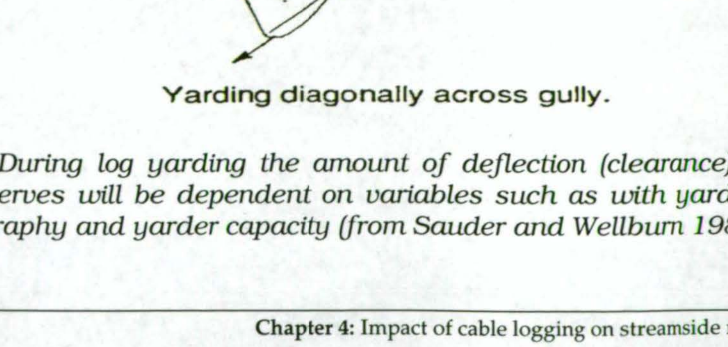
Figure 4.3 *Roses Tier 137f, logged 1987-90, photographed 1992. Logs were yarded upslope to one of the landings with the skyline cables tied back across Tombstone Creek within the reserved forest to the right.*

Skyline corridors 10-20m wide cut through the streamside reserve vegetation at Roses Tier 126a (Figure 4.4-4.5) resulted in a moderate degree of physical disturbance. Within the corridors much of the ground layer consisted of downed trunks and limbs from woody material stripped away from the moving logs.

Figure 4.4 *Cable yarding logs in a perpendicular and diagonal manner (Figure 4.5) across the streamside reserve at Roses Tier 126a (logged 1989, photographed 1992) damaged streamside vegetation, but the damage was limited by cutting corridors across the streamside to facilitate log movement.*



Yarding perpendicular to gully.



Yarding diagonally across gully.

Figure 4.5 *During log yarding the amount of deflection (clearance) gained over streamside reserves will be dependent on variables such as with yarding direction, payload, topography and yarder capacity (from Sauder and Wellburn 1987 p.30)*

Twenty metre tall *Nothofagus* and *Atherosperma* trees suffered crown damage but responded with coppice regrowth. Shrub and herbaceous species were infrequent in the skyline track, most of the vegetative regeneration response consisted of ground ferns such as *Blechnum wattsii*, *B. nudum* and *Hypolepis rugosula* increasing in cover.

The failed attempt at full log suspension at Roses Tier 137g led to the complete loss of the original streamside reserve vegetation, compounded by the salvage logging 'clean up' and burning of logging debris on the streamside area (Figure 4.6, 4.7, 4.8). After three years a 1-2 m tall dense cover of *Prostanthera lasianthos*, *Acacia verniciflua* and *Zieria arborescens*, with occasional *Pomaderris*, *Acacia dealbata* and eucalypt saplings over the ground fern *Blechnum wattsii* had colonised the streamside. After three years, ruderal species such as *Senecio linearifolius* were senescing. Differing intensities of logging treatment formed a floristic continuum in two dimensions that was independent of the spatial and temporal scale (Figure 4.9). The heavily disturbed treatment, where log suspension was attempted, formed one end to a disturbance continuum along axis 1 (Figure 4.9). Vegetation subject to the cutting of corridors for log movement, where partial disturbance occurred, were positioned mid-way between the most disturbed and the least disturbed vegetation.

Figure 4.6 *Vegetation sampling within the severely disturbed streamside at Roses Tier 137g (see aerial view in Figure 4.7) two years after cable logging was completed. Herbaceous and shrub species are regenerating vigorously from seed and coppice.*

Vegetation retained along the streamside according to the forest practices guidelines were at approximately the mid-point of the disturbance continuum along axis 1. At the other extremity on this axis was the vegetation of the undisturbed controls. Vector fitting into the multi-dimensional space indicated a weak relationship exists where species richness increases with the degree of cable logging disturbance (r=0.65, p=0.07, based on 10 000 randomisations).

Figure 4.7 *Cable logging which attempted to fully suspend logs across Memory Creek in 1990 (Roses Tier 137g, photographed 1991) failed to do so, caused significant disturbance, and led to the procedure being abandoned Logging debris then entered the drainage line as per the diagram below.*

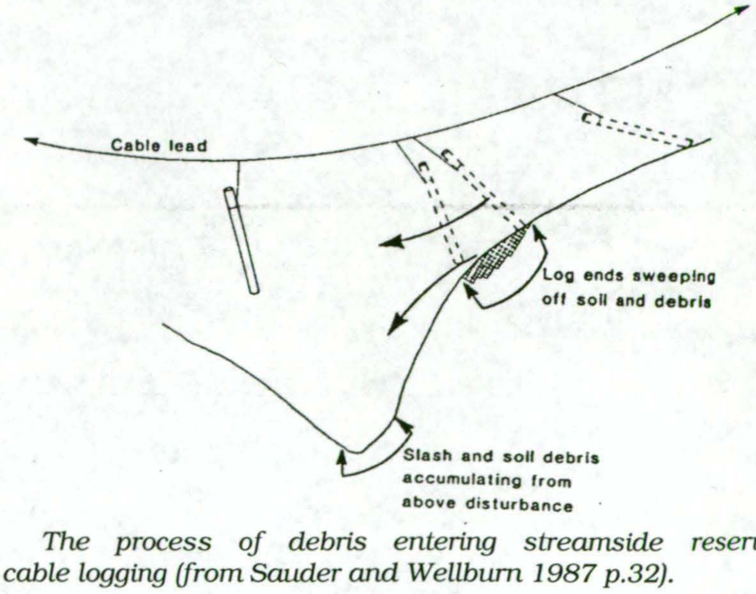


Figure 4.8 *The process of debris entering streamside reserves from full suspension cable logging (from Sauder and Wellburn 1987 p.32).*

Cluster analysis identified three groups (using approximately 80% of the information available in the agglomerative fusion process, Figure 4.10) which corresponded directly to the three *a priori* cable logging intensity treatment groups at Roses Tier.

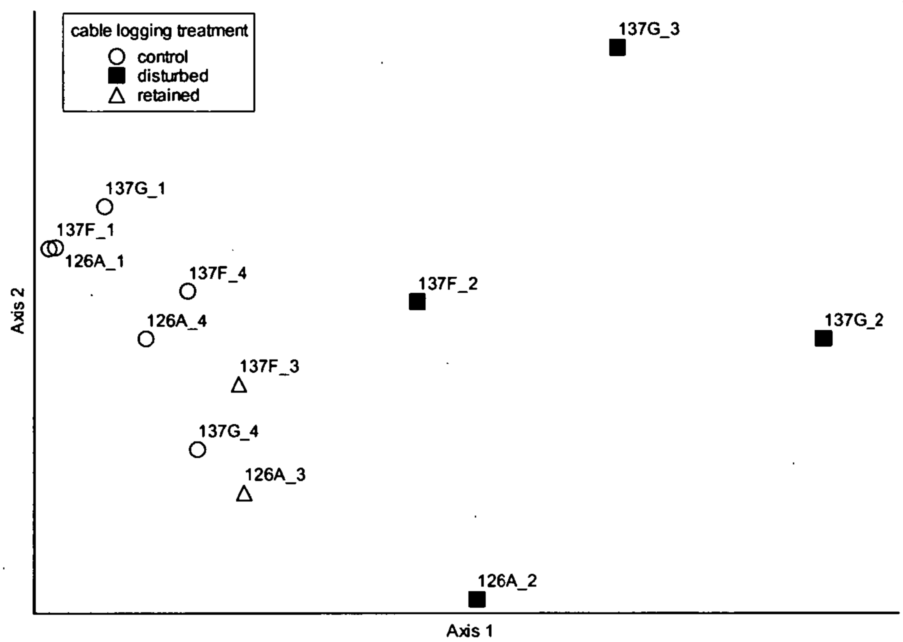


Figure 4.9 *Results of an ordination the twelve samples from three coupes at Roses Tier coupes sampled two years after cable logging. The point labels indicate the disturbance stratification followed. minimum stress =0.0907*

The weighted average within group distance scores indicated that compositional variability was reduced (ie lower distance scores) in the vegetation most severely disturbed by cable logging and burning. Heterogeneity is approximately the same in the control and retained reserve vegetation. The treatment group distances (*a prior* groups) were significantly different (p=0.0014, Table 4.10).

Table 4.10 *Average within group weighted distance scores (Multi-Response Permutation Procedures, McCune and Metford 1999) for groups defined by the streamside reserve logging treatments at Roses Tier.*

treatment group	n	weighted average within group (treatment) distance
control	6	0.3482
disturbed	4	0.8545
retained	2	0.3436
test statistic	-3.9153	
observed delta	0.5162	
p	0.0014	

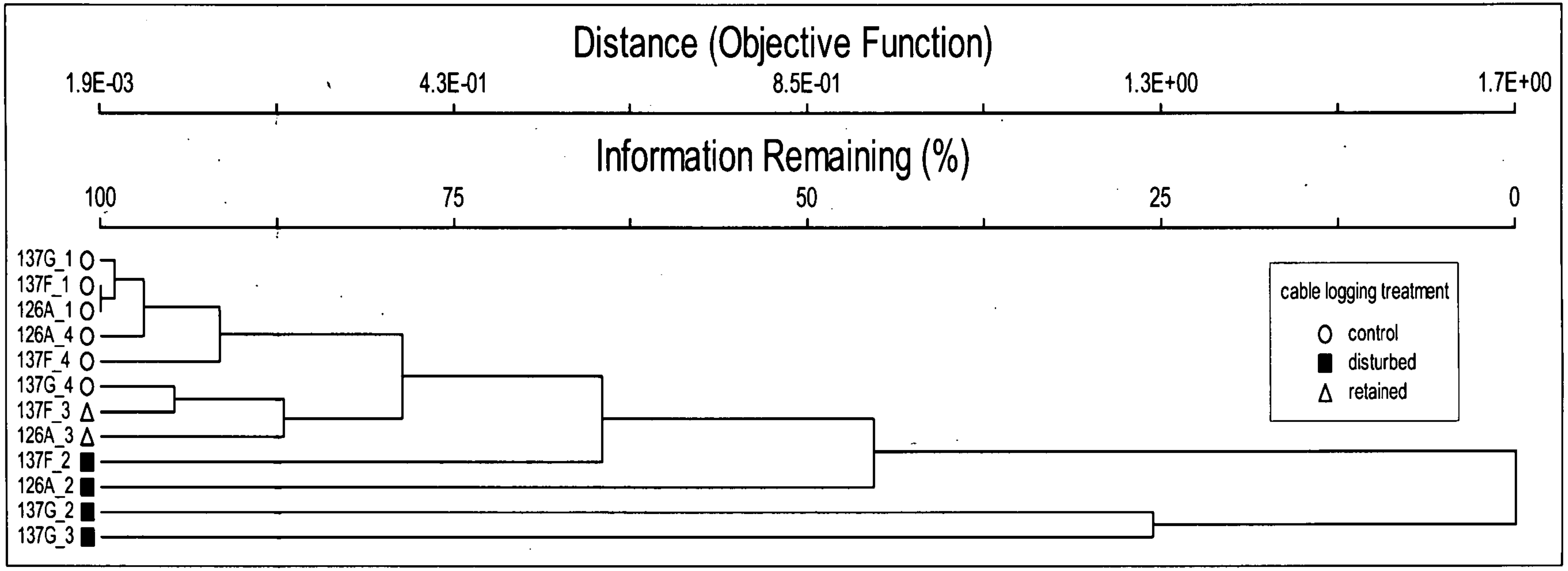


Figure 4.10 *Cluster analysis of twelve quadrats stratified by cable logging treatment at Roses Tier two-three years after cable logging. The cluster analysis was performed using the group average UPGMA method within PC-ORD (McCune and Metford 1999). The distance measure was Bray-Curtis.*

Indicator species for the control and retained treatments at Roses Tier were broadly similar, being primarily rainforest tree, shrub and fern species (Table 4.11), including epiphytic ferns (Chapter 6). Indicator species for the cable logged (disturbed) treatment included *Acacia dealbata*, *A. verticillata*, *Billardiera longiflora*, *Epilobium billardierum*, *Gnaphalium spp.*, *Senecio linearifolius*, *Zieria arborescens* and the weed species *Hypochaeris radicata* and *Sonchus oleraceus* (Table 4.11). Of the thirty five species which reached their maximum indicator value in the disturbed treatment, 77% were absent from the control and retained treatments.
Indicator species for the control and retained treatments at Roses Tier were broadly similar, being primarily rainforest tree, shrub and fern species (Table 4.11), including epiphytic ferns (Chapter 6). Indicator species for the cable logged (disturbed) treatment included *Acacia dealbata*, *A. verticillata*, *Billardiera longiflora*, *Epilobium billardierum*, *Gnaphalium spp.*, *Senecio linearifolius*, *Zieria arborescens* and the weed species *Hypochaeris radicata* and *Sonchus oleraceus* (Table 4.11). Of the thirty five species which reached their maximum indicator value in the disturbed treatment, 77% were absent from the control and retained treatments.

Table 4.11 *Indicator species analysis for sites subject to the three streamside reserve treatments (control, disturbed and retained) at Roses Tier. Species are displayed for the treatment in which they reached their maximum indicator value (iv) of >30.*

species	treatment with maximum iv	control iv (n=6)	retained iv (n=2)	disturbed iv (n=4)	p
Dicksonia antarctica	control	70	23	7	0.0033
Coprosma quadrifida	control	64	4	0	
Atherosperma moschatum	control	60	39	0	0.0431
Pittosporum bicolor	control	33	0	0	
Uncinnia riparia	control	33	0	0	
Uncinnia tenella	control	33	0	8	
Pimelea drupacea	control	32	2	0	
Olearia argophylla	retained	2	92	3	
Blechnum wattsii	retained	11	54	21	
Acacia melanoxylon	retained	0	50	0	
Gahnia grandis	retained	0	50	0	
Aristotelia peduncularis	retained	0	49	0	
Eucalyptus regnans	retained	1	46	1	
Nothofagus cunninghamii	retained	39	43	8	
Clematis aristata	retained	13	30	0	
Acacia dealbata	disturbed	0	0	100	0.0021
Senecio linearifolius	disturbed	0	0	75	0.0475
Hypolepis rugosula	disturbed	0	0	73	
Acacia verniciflua	disturbed	0	0	50	
Billardiera longiflora	disturbed	0	0	50	
Dianella tasmanica	disturbed	0	0	50	
Epilobium billardierum	disturbed	0	0	50	
Gnaphalium sp. 1	disturbed	0	0	50	
Gonocarpus teucriodes	disturbed	0	0	50	
Hypochoeris radicata	disturbed	0	0	50	
Sonchus olearaceus	disturbed	0	0	50	
Ziera arborescens	disturbed	0	0	50	
Pomaderris apetala	disturbed	0	2	48	
Histiopteris incisa	disturbed	0	2	47	
Acaena nova-zealandiae	disturbed	3	0	41	
Senecio spp.	disturbed	6	7	38	

Experiment 2 The immediate and on-going effects on tree health and survival of cable logging adjacent to small patches of retained streamside vegetation

Mortality of non-rainforest woody species and of *Nothofagus* and *Atherosperma* five years following cable logging and burning occurred primarily in the most exposed upper 0-40 m section of streamside reserve (Figure 4.11, 4.12). In non-rainforest woody species mortality was less severe of over the five year period (Figure 4.11). Further, all *Nothofagus* and *Atherosperma* trees in the most exposed 20 m portion were dead in five years. Mortality followed the regeneration burn which entered the uppermost 25m of the transect in late September 1989 (Figure 4.2). Most individual trees in the uppermost 30 m of the transect exhibited evidence of ill-health when first assessed five months after completion of cable logging. Ill-health took the form variously of broken limbs and branches, uprooting and whole tree crowns broken away. Nine months later in September 1989, only 2 weeks before the coupe was burnt the majority of these trees were in poor health. In August 1990 many of these trees had died, with the coppice regrowth following either the cable logging disturbance or the silvicultural burning succumbing to the tree's poor health or lack or vigour. Further mortality and windthrow followed. Windthrow of large individuals compounded the canopy tree damage already present as they damaged further rainforest trees during their fall. Windthrow was more observed to be more common in *Atherosperma* than *Nothofagus*.

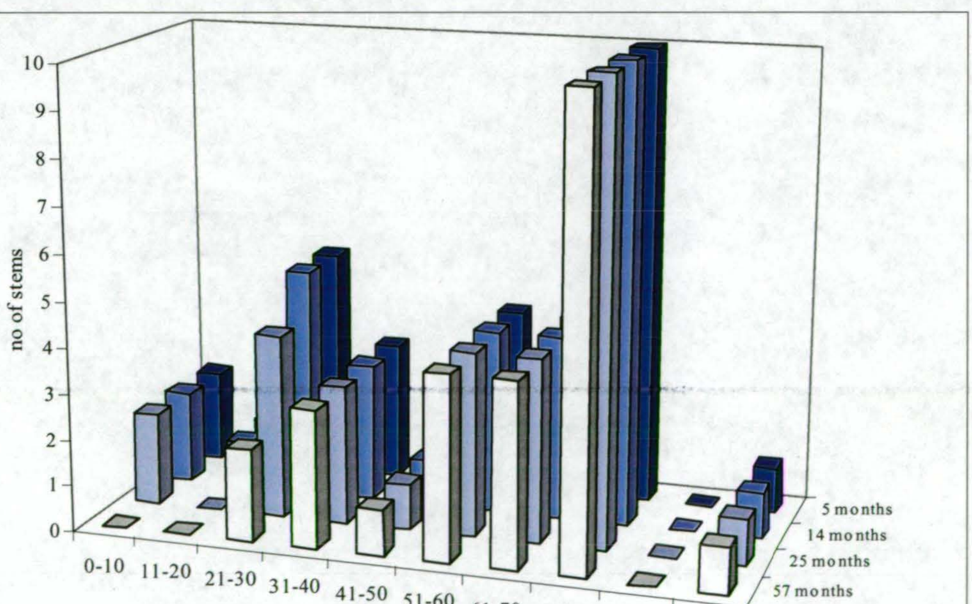


Figure 4.11 *Number of living stems of non-rainforest woody species within a streamside reserve over a five year period following cable logging on both margins, Branchs Creek BC80a, northern Tasmania.*

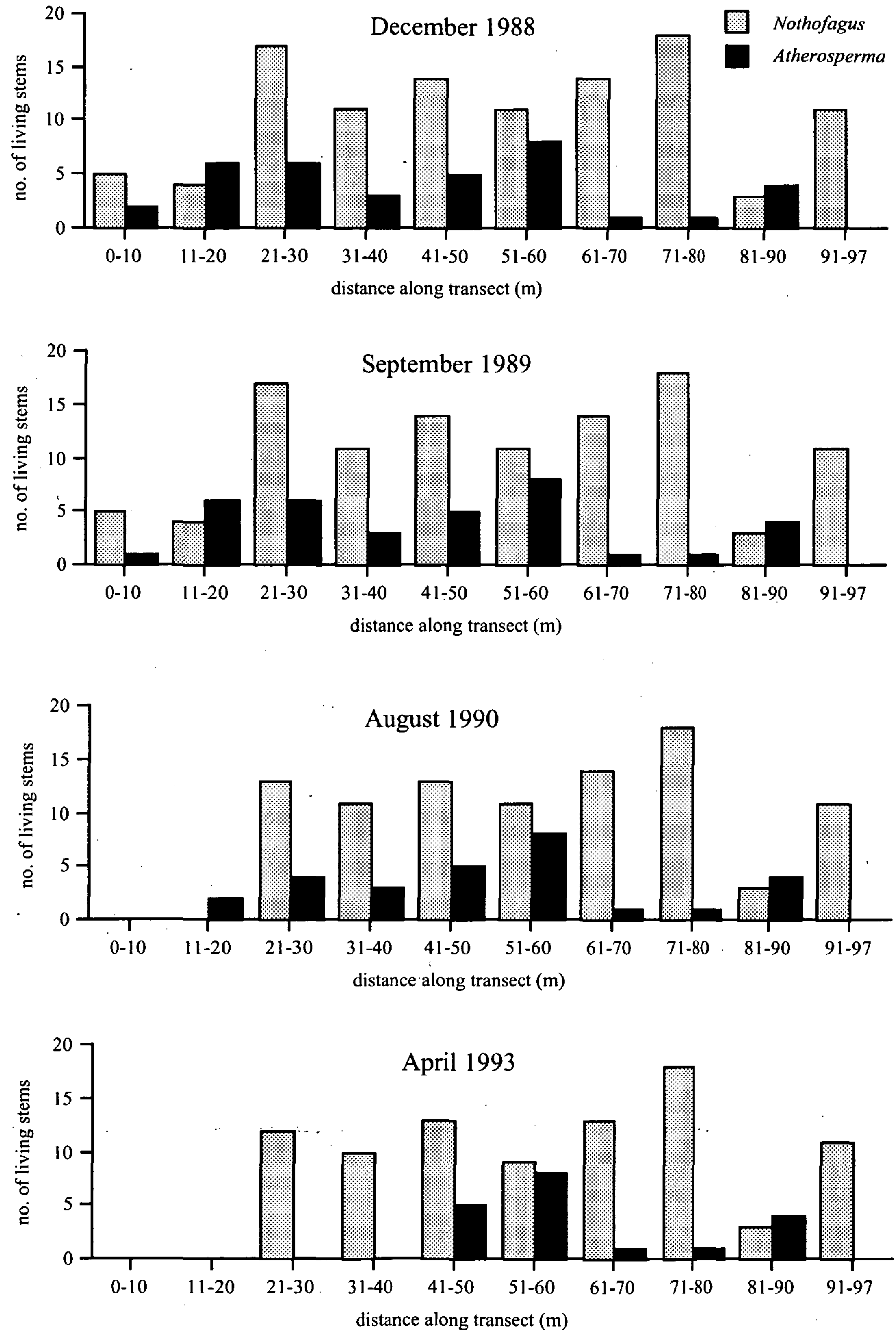


Figure 4.12 *Number of living stems of* Nothofagus cunninghamii *and* Atherosperma moschatum *within a streamside reserve over a five year period following cable logging on both margins at Branchs Creek BC80a, northern Tasmania*

Floristic change

The portion of the streamside reserve undergoing tree canopy decline and mortality was also the portion where the understorey vegetation was subject to the most change. *Zieria arborescens, Histiopteris incisa* and *Hypolepis rugosula* dominated the exposed portions, forming a dense swathe of regeneration. Beneath a dense low ground cover of ferns infrequent seedling regeneration of *Atherosperma* and *Nothofagus* was recorded. The most disturbed portions of the streamside reserve were also the twice as diverse, made up of low shrub and herbaceous species absent from the less disturbed portions (Table 4.12)

Table 4.12 *Floristic composition of the four quadrats measured on the Wallaby Creek tributary (Branchs Creek BC80a), April 1993. The mean percentage foliage cover scores from each species are given.*

	quadrat 1	quadrat 2	quadrat 3	quadrat 4
	intensity of logging disturbance			
species	high	moderate	low	control
Acacia dealbata	0	0	0.06	0.02
Acacia melanoxylon	0	0	0	0.5
Acacia mucronata	3	0	0	0
Atherosperma moschatum	0.02	20	26.02	50
Billardiera longiflora	0.02	0	0	0
Blechnum wattsii	0.04	57.5	77.5	10.54
Coprosma quadrifida	0.02	0.02	0	7.5
Dianella tasmanica	0.04	0	0	0
Dicksonia antarctica	7.52	3	33.5	30
Eucalyptus obliqua	0.5	0	0	0
Eucalyptus viminalis	0.5	0	0	0
Gahnia grandis	0	0	3.5	0
Gonocarpus humilis	0.52	0	0	0
Histiopteris incisa	33.5	0	0	0
Hypolepis rugosula	8.52	0	0	0
Monotoca glauca	0.54	0	0	0
Nothofagus cunninghamii	0.1	38.02	67.5	48.5
Pimelea drupaceae	0.04	0	0	0.04
Pittosporum bicolor	0.02	0.52	0.5	0
Polystichum proliferum	0	0	0.02	0.02
Pomaderris apetala	1.06	0	0	7.52
Pteridium esculentum	0.04	0	0	0
Senecio spp.	0.06	0	0	0
Viola hederaceae	0.02	0	0	0
Ziera arborescens	82.5	0	0	0.04
richness	21	6	8	11

Experiment 3: The relative contribution of different forest practices to damage and ill-health of retained streamside reserve vegetation

Numerous causes of physical damage and symptoms of ill-health in trees retained on the perimeter of streamside reserves at Wayatinah were observed. Log spearing was relatively infrequent and resulted in minimal damage to the retained streamside vegetation (Table 4.13, Figures 4.13, 4.14). Proportionally more log spears were observed within the narrower class of streamside reserve with steeper side slopes (1 per 50m of reserve perimeter in the class 2) compared to 1 per 140m in the class 1 perimeter. Gaps in the streamside reserve at Blue 09 were infrequent, being observed at a rate of 1 per 370 m on the larger streamside reserve and 1 per 70m on the smaller reserve. Most were associated with log spears. At Pearce 04 they were more frequent, at an average rate of 1 per 60 m on the larger streamside reserve.

Table 4.13 *Log spears recorded within the streamside reserve or adjacent to it in retained vegetation, Blue 09, Wayatinah 1992. 'Old' logs spears resulted from road verge logging or a coupe further upslope.*

streamside reserve	Reserve width	no. of log spears new	old	position in relation to streamside reserve
Class 1	40 m	7	1	all logs outside reserve except 1 'old' log
Class 2	30 m	1	4	all within reserve

Figure 4.13 *Log spears and debris occasionally entered streamsides which in the longer term may be beneficial where course woody debris inputs are limited. Loatta 217, northern Tasmania, 1993.*

Figure 4.14 *Log spears occasionally became embedded in the road surface or behind other logging debris at Blue 09 in 1992, therefore avoiding damage to the streamside reserve further downslope.*

The accumulation of logging slash and woody debris from cable logging (Figure 4.15) and the associated clearing for tail spar track was the most common disturbance to streamside reserve perimeters on both the class 1 and 2 streams. Debris on the tail hold track which followed the contour break was pushed aside, usually downslope, against or into the retained vegetation (Figure 4.15, 4.17)

Figure 4.15 *View of the upslope perimeter of the streamside reserve at Blue 09 shortly after logging in 1992. Logging slash accumulated along the edge of the retained streamside vegetation.*

Trees dislodged by slash debris commonly opened up larger gaps by falling further into the retained vegetation and damaging additional vegetation. The distance logging slash penetrated the retained vegetation varied from 3 to 15 m and the width from 4 to 32 m. A proportionally greater area of the smaller class 2 streamside reserve was affected by slash accumulation than the larger class 1 reserve (Table 4.14). In general the proportion of the perimeter reserve area effected by cable logging (ie canopy gaps) was small, except for slash accumulation (Table 4.14).

Table 4.14 *Percentage of streamside reserve vegetation affected by canopy gaps and logging slash from cable logging at Blue 09, Florentine Valley, 1992-93.*

	Streamside Reserve	
disturbance	Class 1	Class 2
canopy gaps	0.6%	3.2%
slash accumulation	0.2%	19.2%

The exception was Pearce 04, where a breach of the forest practices guidelines relating to streamside reserves occurred and 21% of the class 1 streamside perimeter was disturbed by canopy gaps (Table 4.15). Compared to disturbance from cable logging (0.6%), 40% of the 1120m of perimeter of the class 1 streamside reserve at Blue 09 was disturbed by fire (Table 4.16). The burnt perimeters were generally those lacking a formed fire break, although a small section (90m) burnt that had a formed fire break in place. Of the perimeter length with a formed fire break, 88% remained unburnt.

Table 4.15 *Percentage of streamside reserve vegetation affected by canopy gaps and logging slash from cable logging at Pearce 04, Florentine Valley, 1992-93.*

	Streamside Reserve	
disturbance	Class 1	Class 2
canopy gaps	21.7%	-
slash accumulation	0.1%	-

Table 4.16 *Percentage of streamside reserve perimeter vegetation affected by slash burning at Blue 09, Florentine Valley, 1992-93.*

	Streamside Reserve	
disturbance	Class 1	Class 2
perimeter burning	40.1%	0%

The degree of fire damage at Pearce 04 (Figure 4.17) varied between tree species on the streamside reserve perimeter, more severe fire damage occurred on the species with stringybark, and the least on the gum barked species (Table 4.17).

Table 4.17 *The proportion of eucalypts (excluding Eucalyptus amygdalina) at Pearce 04 exhibiting fire damage four months later.*

Species	Unburnt	Crown scorched only	butt, trunk and crown burnt
Eucalyptus obliqua (n=87)	9.3%	59.7%	31%
Eucalyptus viminalis (n=11)	27.3%	54.5%	18.2%
Eucalyptus regnans (n=7)	14.3%	85.7%	0

Figure 4.16 *Streamside reserve perimeter at Pearce 04 two days after slash burning, March 1993. The formed firebreak along its perimeter was partly successful in retarding the fire but it did little to minimise scorch damage to retained trees.*

Approximately equal proportions of the rainforest trees *Nothofagus* and *Atherosperma* were damaged on the streamside reserve perimeter (Table 4.18). Most damage was due to the heaping of debris during firebreak construction (r=0.37, Table 4.20, Figure 4.17), or root disturbance and windthrow of parts of their crown limbs. Physical cable logging damage to *Nothofagus* and *Atherosperma* was infrequent (0-10% of stems, Table 4.18).

Table 4.18 *The proportion (%) of Nothofagus and Atherosperma trees damaged within the streamside reserve areas, July 1993, three months after coupe burning, Blue 09, Florentine Valley. n = number of 30m segments where the species was observed.*

Physical damage to remaining rainforest trees	*Nothofagus* n = 19	*Atherosperma* n = 43
none visible	26	28
0-10% of stems	47	40
11-25% of stems	5	16
26-50% of stems	11	4
>50% of stems	11	2

Figure 4.17 *View of the cable logged perimeter of the streamside reserve at Blue 09 shortly after logging and prior to slash burning, 1992. The clearing represents the mobile tail-hold track upgraded to a firebreak at the completion of cable logging.*

Atherosperma tree health was generally poorer than *Nothofagus* three months after slash burning (Table 4.19). Smaller *Nothofagus* trees near the firebreak which received direct physical disturbance from slash heaping exhibited wilt like symptoms. Where larger *Nothofagus* trees bordered the streamside reserve area perimeter, 84% exhibited wilt like symptoms.

Table 4.19 *The proportion (%) of* Nothofagus *and* Atherosperma *trees within 30 m segments of the streamside reserve areas showing signs of ill-health, three months after coupe burning, Blue 09, Florentine Valley.*

Tree Health of Remaining Rainforest Trees	*Nothofagus* n = 19	*Atherosperma* n = 43
no signs of ill health	16	2
early signs or slight evidence of ill health	21	37
most trees with yellowing leaves (Atherosperma) or wilt like symptoms (Nothofagus)	63	61

The relationships between the cable logging events and forest practices applied at Wayatinah are not particularly strong when averaged across the whole perimeter length, except for the obvious relationship between burnt vegetation being uncommon where a firebreak was formed (Table 4.20).

Table 4.20 *Pearson product-moment correlation analysis of the Blue 09 streamside reserve vegetation post-burn assessment data, collected July 1993. The top figure is the correlation coefficient, the bottom figure its significance level. n.s. = no significant association n=63. A positive correlation indicates that the variables vary in the same direction and a negative correlation indicates they vary in the opposite direction.*

	1	2	3	4	5	6	7
1. stream class	-	-0.29 <.05	n.s.	n.s.	-0.39 <.001	0.50 <.0001	n.s.
2. presence of burnt vegetation		-	n.s.	0.44 <.001	0.42 <.001	n.s.	-0.78 <.0001
3. tree species			-	n.s.	n.s.	n.s.	0.25 <.05
4. degree of tree damage				-	n.s.	n.s.	0.37 <.01
5. degree of tree ill-health					-	-0.37 <.01	-0.31 <.05
6. no. of eucalypts retained						-	n.s.
7. presence of formed firebreak							-

4.3 Discussion

Few previous studies have examined the structural and floristic response of streamside reserve vegetation retained along the margins of watercourses adjacent to clearfell logging operations. The focus of previous work has been to examine changes in the soil environment following logging and the degree to which the retained vegetation maintains water quality or its light environment (eg Dignan and Bren 2003). This lack of information became acute in Tasmania in the 1980's when community interest in cable logging, especially adjacent to streamside reserves, was highlighted by several unsuccessful full log suspension trials. As a result of the requirement to examine the effects of existing forest practices associated with cable logging in northern Tasmania, a range of chronosequence and experimental studies were initiated by the author.

The method and intensity of cable logging disturbance adjacent or across streamside reserve vegetation was directly related to the nature of the resulting disturbance and the floristic response. The practice of cutting skyline corridors through streamside reserve vegetation to either drag cables through to tailhold trees, or to facilitate log yarding through the streamside reserve itself, was introduced to minimise vegetation disturbance. While methods exist to minimse further damage to streamside vegetation from full suspension cable logging, or logging adjacent to retained vegetation (Garland 1987, Moore 1991, Palmer *et al.* 1996) in most cases the costs would be prohibitive. Most descriptions of cable yarding over streamsides with either partial or full log suspension consider that some disturbance to the vegetation is inevitable regardless of deflection, with disturbance being more severe where deflection is poor (Krag *et al.* 1987). While the overall area damaged by cable yarding was reduced by cutting corridors of vegetation, localised damage to canopy trees occurred, probably due to flexing of the cable during log yarding. Skyline corridors through the streamside reserve environment disrupted the continuity of vegetation cover over the streamside, fragmenting the interior habitat required by vascular epiphytes (see Chapter 6). Species capable of vegetative regeneration in the absence of a fire, for example rhizomatous ferns, increased in abundance.

Disturbance to streamside vegetation was more pronounced where both streamside reserve boundaries were cable logged. Damage due to log spearing, directional falling difficulties, silvicultural burn escapes, and windthrow increased. These diffculties were compounded where salvage logging of damaged streamside reserve vegetation was undertaken, with post-logging burning leading to the complete loss of the streamside tree and shrub cover. Logging debris dragged downslope into the streamside area accumulated in the streamside and was only partially consumed in the fire. After three years at Roses Tier the debris had consolidated in situ similar to the manner described 2-3 years after cable logging by Sauder and Wellburn (1987). Logging debris entering the streamside reserve may not necessarily be harmful, large woody debris plays a important structural role in streams (Bisson *et al.* 1987) and is an important substrate for epiphytes (Jonsson 1997) and rainforest tree seedlings (Harmon *et al.* 1986, Harmon 1989, Harmon and Franklin 1989), especially those of fragmented stands (Kruys and Jonsson 1997). While a short term increase in woody debris in the streamside from cable logging and associated windthrow occurred at Roses Tier, in the longer term a shortage of input sources for large woody debris is usually predicted (Bisson *et al.* 1987, Hartman *et al.* 1987 Nakamoto 1998,) as approximately 30% of tree falls within riparian areas are triggered by upslope treefalls (Reid and Hilton 1998). As large woody debris loadings in streamside environments change with forest successional status (Hedman *et al.* 1995), and are less abundant in streamside forests within regrowth or partially logged stands (Hayes *et al.* 1996), the short term slash and woody debris accumulations observed may in the medium to long term help balance any predicted deficits.

Cable logging residue in the stream channel and the associated attempt to burn it at Roses Tier further disadvantaged the regenerative capacity of the rainforest trees present. Rainforest trees were largely displaced by rapidly growing ruderal species such as *Senecio linearifolius* and *Zieria arborescens* which formed a depauperate wet forest shrub thicket. The re-establishment of riparian community patterns following severe disturbance can be slow (Gecy and Wilson 1990).

As noted elsewhere (e.g. Andrus and Froehlich 1988), the floristic response of streamside vegetation to logging disturbance was closely related to the intensity and type of forest practices applied. The shift in abundance to post-fire ruderal species, especially shrubs, was extenuated by the removal of the canopy cover and imposition of burning treatments. Intensive disturbance led to an increase in vegetation biomass, especially of herbaceous species, a trend also observed following silvicultural manipulation of streamside vegetation (Aust *et al.* 1997, Keim and Schoenholtz 1999). Three years after cable logging retained vegetation in strips approximately 25 m wide on either stream bank have a composition resembling the undisturbed sites. Retained vegetation in these areas may be important for the dispersal of rainforest species into the disturbed habitat once the necessary micro-climatic conditions (for example a closed canopy) have re-established.

The experimentation at the Blue and Pearce experimental sites, combined with the repeated measurements of rainforest tree health at Branches Creek in northern Tasmania, illustrate the complexity of the interactions present between cable logging method and intensity, landscape features, and chance effects such as log spearing and silvicultural burn escapes, have on the maintenance of intact streamside reserve vegetation. It is also clear from these studies that exposed remnant stands of streamside vegetation will continue to decline in health at least five years after the initial logging disturbance, a point rarely made in the literature. Many of the forest practices resulting in detrimental effects on streamside vegetation were inter-dependent. Where a slash burn entered a reserve or scorched the edge tree species, tree ill-health was more pronounced. Clearing a firebreak between the streamside reserve and the clearfelled area increased the proportion of streamside reserve with a disturbed perimeter. Reserve perimeters disturbed in this manner exhibited an increased incidence of physical tree damage, such as broken stems and over-turned trees. It also contributed to the incidence of ill-health in *Atherosperma*, and the wilt like symptoms of *Nothofagus* trees. The sharply defined clearfelled edge is also likely to lead to significant changes in light environment to the retained vegetation to a distance varying with topographic and vegetation characteristics (Dignan and Bren 2003). Direct disturbance to the streamside reserve perimeter caused by fire break clearing is considered acceptable to forest managers within the context of the risk associated with the regeneration burn escaping into the streamside reserve. It is somewhat contradictory however that much of forest practices guidelines developed to maintain streamside reserves in an intact condition during logging are rendered partly ineffective by damage resulting from the fire protection mechanisms.

Slash burning within coupes was frequently observed to enter streamside reserves, especially into internal reserves with both side slopes logged. Where the streamside reserve had already been damaged by tree falling or yarding, with gaps in the canopy and additional fuel sources on the ground, it may have been more susceptible to ignition than a relatively intact reserve which maintained a reasonable moisture differential with the slash material. Excluding regeneration burns from retained class 4 watercourse vegetation in dry coupes is seen as being central to the recovery of the watercourse vegetation from disturbance. Fires provided a niche for pioneer species to invade the watercourse and significantly alter its floristic composition, at least in the short term. Fire damage was also correlated with increased instances of windthrow and ill health of the remnant rainforest canopy. In other coupes in which the cable portions were not burnt for top disposal or regeneration purposes, the remnant watercourse vegetation, which was often heavily disturbed by yarding or falling, was able to vegetatively recover and maintain a relatively moist micro-climate with the remaining foliage cover and phorophytes. In some cases where special efforts had been made to protect the streamside reserve vegetation from tree falling and yarding damage, those efforts were partly wasted when the reserve vegetation was burnt anyway. Forest practices designed to minimise logging disturbance to streamside reserves were compromised when this vegetation was subsequently burnt or scorched by the regeneration burn.

Windthrow following cable logging adjacent to streamside reserves, of both living and dead trees, contributed significantly to mortality, especially for *Atherosperma* trees in the most exposed upslope portions of streamside reserves. Windthrow of retained streamside trees can be a common occurrence in the first years after logging (Chatwin 2001). Windthrow can impair streamside reserve functions including the maintenance of shade and a stable stream channel (Chatwin 2001), and woody debris dynamics (Grizzel and Wolff 1998). Opinions vary on whether windthrow into streamsides is more frequent on steep slopes (Alexander 1964, Moore 1977, Hairston-Strang and Adams 1998). The greater susceptibility of *Atherosperma* compared to *Nothofagus* to windthrow at Branches Creek is attributed to crown shape and size characteristics, one of the mechanisms identified by Ford (1978) as influencing a trees susceptibility to windthrow. A similar scenario of windthrow of retained trees (predominantly *Eucalyptus regnans*) exposed by logging has been described in the montane ash forests of Victoria by Lindenmayer (1990b), with many of the retained trees falling over and dying within two years after logging and burning. In that study the author predicted the majority of the retained trees would not survive twenty-five years.

In addition to windthrow, damage from felling and yarding adjacent trees, log spearing, fire and scorching and possibly myrtle wilt-like disease symptoms were all concentrated in the upslope portions of streamside reserves where the reserve width was at its narrowest point. Five years of monitoring of a small exposed remnant stand of callidendrous rainforest in northern Tasmania following cable logging on both margins indicated that streamside reserves are particularly susceptible to edge effects which can lead to ongoing ill health and death of the canopy trees, and these edge effects and the subsequent tree death and windthrow are hastened when fuel reduction/regeneration burns are conducted on the edge or through the streamside reserve vegetation. Faced with the same scenario at Caspar Creek in Oregon, Reid and Hilton (1998) suggested that buffer strips may require widdening (ie. 'buffering the buffer') to protect the core buffer from accelerated mortality due to factors operating on its edge. Streamside reserve rainforest canopy trees within small exposed remnant stands of callidendrous rainforest were regressing with a continuous loss of the remaining canopy cover due to windthrow, ill-health and myrtle-wilt.

A further forest practice resulting in detrimental effects on streamside vegetation is the construction of mobile tail spar tracks along their perimeter. At Blue and Pearce, both experimental areas had perimeter track established prior to yarding, a requirement of the highlead cable logging system employed. Some pushing of debris against the reserve perimeter occurred during their construction but the more significant damage was done when these tracks were upgraded to a firebreak standard. Firebreaks were largely successful in keeping the slash burn out of the reserve, but they had no effect on the amount of scorch damage to trees on the streamside reserve perimeter. More damage to the reserve perimeter vegetation resulted from firebreak construction than due to log falling, log spears and other disturbance while cable logging was underway. Less physical damage to the reserve perimeter would therefore have occurred if a mobile tailspar track and firebreak were not constructed. The cable logging system at the Blue and Pearce experimental sites required the construction of an access track for the mobile tailspar. Similar wet eucalypt forest vegetation on steep coupes in Tasmania has been burnt without a formed firebreak where moisture differentials have been sufficient to stop the fire spreading into the reserve vegetation. Other cable logging systems, such as running skylines, could have been applied to the Blue area which don't required mobile tail spar tracks.

While the experiments reported on the condition of streamside reserves up to five years after treatment can be considered of a preliminary nature, there are sufficiently consistent trends across the six locations examined in detail for a re-assessment of the streamside forest practices operating within cable logging coupes. Changes in streamside reserve vegetation following cable logging and burning were highly correlated with the intensity and nature of the disturbance. They range from full canopy loss and burning to narrow skyline tracks on one edge of the streamside reserve only, and minor slash and fire incursions on the streamside reserve edge. Longer term edge effects from the reserve perimeter into the interior habitat of the reserve are also likely to manifest themselves over forthcoming years, with the magnitude of the effect changing with the age of the adjacent regrowth (Bradshaw 1992). While it is acknowledged that streamside vegetation within or adjacent to cable logging operations represents a significant challenge for harvest planners attempting to ameliorate cable logging impacts, the planning process would benefit from a statement of the nature of the flora conservation objectives for the retained streamside areas beyond the existing assumption that they function as a filtering mechanism for water quality maintenance. The absence of an explicit description of the reference conditions that represent the goals of forest management within streamside areas is not unique to the Forest Practices Code used in Tasmania, it was also lacking in state and federal forest riparian guidelines reviewed by Gregory (1997). With such a statement more objective forest practices could be developed for streamside reserve vegetation management than the existing broad assumption that their primary function is to maintain stream water quality.

Chapter 5 Effects of cable and ground-based logging on *Dicksonia antarctica*

5.1 Introduction

Dicksonia antarctica Labill. (DICKSONIACEAE) (Manfern or Soft tree-fern) is a widespread and conspicuous understorey species inhabiting a range of forest environments in south eastern Australia including fern gullies (Patton 1933), wet sclerophyll forest (Beadle and Costin 1952), mixed forest (Gilbert 1959, Peel 1999) and rainforest (Jarman *et al.* 1984, Cameron 1992 and Floyd 1990). It occupies an altitudinal range from sea level to 1000 m ASL (Duncan and Isaac 1986) in areas typically of high rainfall (e.g. wet sclerophyll forest at 1000 - 1500 mm per annum, Ashton and Attiwill 1994). Outside the species optimum moist forest environments *D. antarctica* is restricted to wetter micro-environments such as streamside areas and relict rainforests in Tasmania (Neyland 1991).

Dicksonia antarctica is conspicuous by its large crown of fronds and its thick, erect caudex [a thick erect stock or trunk, especially of tree ferns (Flora of Australia Volume 48 1998) generally of long duration and slow increment] which is covered in matted adventitious roots and often supports a species rich cargo of vascular and non-vascular epiphytes. *Dicksonia antarctica* can be regarded as a functionally important (*sensu* Hulbert 1974) because it provides a habitat for obligate and facultative epiphytes (Ford and Gibson 2000, Ough and Murphy 1996), particularly those using its trunk as a safe microsite for seedling recruitment (Peacock and Duncan 1994, Ashton 2000). *Dicksonia antarctica* may also play other important ecological roles in wet forest understoreys, for example, as an ecological filter (George and Bazzaz 1999 a,b) for the establishment and growth of rainforest tree seedlings (see chapter 6), the provision of fauna habitat (Lindenmayer *et al.* 1994, Schwarz 1988) and possibly the amelioration of the light and microclimate environment on the forest floor (Ashton and Turner 1979). Additionally, the reported slow growth rates (e.g. Neyland 1986) and the species' strong stomatal sensitivity to reduced humidity (Unwin and Hunt 1996, Hunt *et al.* 2002), suggest it's physiological characteristics make it inherently susceptible to adverse microclimatic changes associated with logging (e.g. Ashton 1975, Adams *et al.* 1991).

Ground-based and cable logging systems (see descriptions in Chapter 1) differ in three principal respects that are likely to affect the survival, regeneration and recruitment of *D. antarctica*. The first is the nature of the two logging systems. A ground-based clearfell system will concentrate soil and vegetation disturbance along the defined extraction route (skid trails or snig tracks). Some variants of ground-based systems, for example used in Victorian wet forest (Department of Conservation, Forests and Lands 1989) can result in additional understorey disturbance when logging debris (and therefore remnant *D. antarctica*) is heaped into windrows for more efficient burning. By comparison, a cable clearfell system results in significantly less soil disturbance in terms of its intensity and the proportion of the logged area effected (Chapter 1). Secondly, while a ground-based clearfell system will result in more intensive soil and vegetation disturbance where the logging machinery passes, other areas of vegetation will remain relatively undisturbed where no logs are removed or a machine passes. A cable clearfell system however will flatten almost all vegetation during log yarding either directly as the logs as yarded or indirectly as the cable setting is re-positioned across the logging area. Thirdly, the two logging systems tend to leave the remaining vegetation and logging residue distributed differently over the logged area. Following cable logging, logging residue is more evenly distributed over the logged area and less compacted. However, following ground-based logging, the residue is concentrated in particular locations and may also be mixed through the soil profile. These differences in logging residue distribution become relevant in the context of the after logging slash burn, as cable logged coupes will tend to burn more consistently and evenly throughout, while ground-based logged coupes may burn more patchily, with localised areas not burning, or other areas burning intensely, due to the heaping of residue or the mixing of residue in the soil profile respectively. These three differences are significant for *D. antarctica* populations both in terms of direct mechanical disturbance and burial of individuals during logging, and to their ability to withstand fires of different intensities.

Dicksonia populations exhibit a marked decline in abundance following ground-based logging (Cremer and Mount 1965, Harris 1986, Harris 1989, Neyland 1986, Barker 1988, Ough and Ross 1992, Mueck and Peacock 1992, Murphy and Ough 1997, Cook and Drinnan 1983, Tanjung 1992, Hickey and Savva 1992, Chesterfield 1996). These studies utilised a variety of methods including after logging observations (Harris 1986), paired plot comparisons (Murphy and Ough 1997) or chronosequence (Mueck and Peacock 1992) methods, but experimental studies were rarely attempted. In all cases *D. antarctica* was significantly less abundant in logged than in unlogged stands, trends in abundance related to stand age were often not examined. After logging changes in abundance across these studies were variable and the extent to which they were related to time since logging (ie. old forest versus young regenerated forest), the method of logging and regeneration, or to variation in forest type was often unclear due to the lack of experimentation or the use of control sites. None of these studies have examined cable logging, or compared the response of *D. antarctica* to different logging methods.

The reliance of these studies on limited data sets, often lacking spatial replication, may partly explain the variable results and the difficulty with their interpretation. A variety of mechanisms have been attributed to the observed declines in abundance of *D. antarctica* following logging. Mechanical disturbance associated with ground-based logging is the most common reason (Ough and Ross 1992, Ough and Murphy 1997, 1998), although no direct evidence was provided. The greater frequency of *D. antarctica* in regrowth forest originating from wildfire compared to logging led to Hickey and Savva (1992) reaching the same conclusion. Similarly, the increased survival of *D. antarctica* within clumps of retained understorey vegetation within logging areas (69%) compared to adjacent logged vegetation (15%) led Ough and Murphy (1998) to again attribute the effect to physical disturbance. Barker (1988) suggested that mortality in regrowth forests originating from 1980's logging operations was attributable to the more intensive wood utilisation standards applying at that time compared to those applied in the 1960-70's. Neither of these studies provided any direct data or experimentation to test these conclusions. A further difficulty with these conclusions lies in the inherent limitation of chronosequence approaches (Chapter 1), the unknown level of sampling error, the lack of experimentation using permanently marked individuals (e.g. Ough and Murphy 1997, 1998), and the absence of any consideration of population dynamics in undisturbed forests using control or reference plots.

Wildfire is considered a significant factor in the regeneration biology of *D. antarctica* (Ough and Ross 1992, Hickey and Savva 1992, Neyland 1986), yet its ability to constrain the species regenerative response to logging is untested experimentally, and is of considerable relevance as nearly all silvicultural systems applied in southern Australia where *D. antarctica* is present make use of after logging residue burning. The morphology of *D. antarctica* provides both an incentive to burn [the skirt of dry fronds is slow to decompose (Enright and Ogden 1987)] and a protective sheath around the central vascular bundle (the caudex) during burning. Neyland (1986) suggested that after logging burns unbalanced *D. antarctica* caudexes leading to further mortality, and (Ough and Ross 1992) suggested *D. antarctica* was capable of surviving wildfire based on their abundance in 50 year old wildfire regrowth forests. In some circumstances survivorship following controlled burning of *D. antarctica* populations can be 100% (Chesterfield 1996).

In the drier environments of eastern and north eastern Tasmania where cable logging is conducted, *D. antarctica* is effectively restricted to climatic and disturbance refuges such as the larger drainage lines or relict rainforest stands. *Dicksonia antarctica* forms a conspicuous and potentially functionally important role in the understorey vegetation of streamside reserves, and due to its presumed susceptibility to mechanical disturbance (Ough and Murphy 1996), may suffer population decline following logging. These environments are disturbed to different degrees by cable logging, depending on the logging system used and topography present, but not by ground-based logging which excludes logging machinery within streamside reserves. While cable logging disturbance to these riparian environments represents only a small proportion of the net logging area, it may be particularly significant for the available habitat of *D. antarctica* and its dependent epiphytic flora. No published data is available on the response of *D. antarctica* to cable logging within riparian habitats.

Present information of the response of *D. antarctica* to logging and subsequent regeneration is therefore largely anecdotal or based on the extrapolation of very limited data sets. Consequently, these studies have yielded apparently conflicting results and are difficult to interpret. Only one study (Chesterfield 1996) has published data with an adequate level of experimental rigour and pre-treatment data to confidently assess ground-based logging treatments effects on *D. antarctica*. No studies have been published comparing the effects of different logging treatments on *D. antarctica*, none have examined cable logging, and none have used permanently tagged individuals to monitor individual recovery and recruitment patterns.

5.2 Methods

Four experiments examining of the response of *D. antarctica* to cable or ground-based logging were conducted;

1. a chronosequence analysis comparing the response of *D. antarctica* to cable and ground-based logging in wet forest in southern and central Tasmania
2. a long term replicated monitoring study using permanent plots examining the impacts of cable logging and burning on the frequency and cover of *D. antarctica* in wet forests,
3. a study of tagged *D. antarctica* individuals whose location, health and dimensions were monitored before and after cable logging and before and after the slash burn. Results were compared with individuals monitored in undisturbed controls,
4. a study of the effects of method and intensity of cable logging adjacent to and through the streamside reserves on *D. antarctica* within those reserves.

Experiment 1: Chronosequence analysis comparing the response of *D. antarctica* to cable and ground-based logging

Study Areas

The study areas were cable logged and unlogged sites in the Florentine, Styx and Tyenna Valleys of southern central Tasmania and ground-based logged and unlogged sites in the Southern Forests, southern Tasmania. For these study areas, the survey design for site selection and the analysis of inter-site environmental heterogeneity were described in detail in Chapter 2 as Experiment 1.

Experimental design

The chronosequence design sampled regrowth forest aged 5-54 years originating from cable or ground-based logging of steep slopes and unlogged forest. Regrowth forest age was grouped into the following classes 5-10, 11-20, 21-30, 31-40, 41-54 and unlogged, with three replicates within each class, e.g. 5-10 years was sampled from regrowth aged 5, 7 and 10 years. The chronosequence survey design and stratification is described in detail in Chapter 2.

Sampling methods

All *D. antarctica* individuals present within a 30 x 30 m plot were allocated to one of the five mortality, development, and orientation categories in Table 5.1. For the 30 x 30 m plot an overall estimate of cover-abundance (+, 1, 2, 3, 4, 5) modified from the Braun-Blanquet (Poore 1955) scale was made. Cover was defined as foliage projective cover following Walker and Tunstall (1981). Within the 30 x 30m quadrat, ten sub-plots (2 x 2m) were spaced at 7.5m intervals along two transect lines either side of the quadrat's mid-point (Figure 2.6). The sub-plots were 1m apart and alternated from side to side along the transect. In each of the 20 sub-plots *D. antarctica* was again assigned a visually assessed cover-abundance value (+, 1, 2, 3, 4, 5). The mean caudex height for the 30 x 30 m plot of the dominant cohort of *D. antarctica* present was estimated. The substrate preferences of all *D. antarctica* new recruits were recorded and an estimate of the total relative abundance of new recruits on each category of substrate (e.g. rocks, fallen logs) made on a simple ordinal scale as either rare, moderate and abundant.

Table 5.1 *Terminology for mortality, development, and orientation categories scored for individual* D. antarctica.

Category	Description
1. Dead prostrate	These *D. antarctica* individuals were lying prostrate on the ground. As there were no green photosynthetically active fronds or developing croziers present it was assumed that they were dead (apparent death, sensu Curtis 1998)
2. Dead upright	These *D. antarctica* individuals remained upright throughout logging and regeneration burning and were dead at the time of sampling. Most exhibited signs of physical damage to their caudex from logging, such as the loss of their terminal bud.
3.Alive prostrate	These *D. antarctica* individuals were positioned on the ground or at a shallow angle with the ground at the time of sampling. The terminal bud was intact following logging disturbance and had resprouted following the slash burn. Most *D. antarctica* in this category developed new upright caudexes from their terminal bud after falling over, with this height increment scored at 10 years after logging only.
4. Alive upright	These *D. antarctica* individuals remained upright throughout the logging and burning events and regenerated a crown of fronds after the slash burn.
5. New recruits	New recruits are the first recognisable spore producing diploid generation in the life of a *D. antarctica*. New recruits were identified as soon as they could be recognised as being that of *D. antarctica*, i.e. distinguishable from other fern species. While most of the new recruits were small (<10cm tall), the term is used here to also define *D. antarctica* individuals recruited after the logging and regeneration events, for example *D. antarctica* individuals observed growing on cut tree stumps in the chronosequence study.

Analytical procedures

Counts of *D. antarctica* individuals in 30 x 30 m plots in each of the five mortality, development, and orientation categories above were averaged within their respective plot age categories (5-10, 11-20, 21-30, 31-40, 41+ years and unlogged). The mean values were presented graphically. No statistical tests are applied to examine differences in mean values between age categories due to the small number of replicates (three) per class and the large standard deviations. Data on relative substrate preferences of the *D. antarctica* new recruits within the 30 x 30 m plots is presented in tabular form. Data on the mean dominant height of *D. antarctica* in each sample is presented graphically for each logging treatment in a bi-variate plot of age by height. The visually assessed cover-abundance values (+, 1, 2, 3, 4, 5) recorded from the 20 2x2 m sub-plots were converted to a mean percentage cover scores following the procedures described in Chapter 2. If *D. antarctica* was present in the in 30 x 30 m plot only, and absent from the 20 2x2 m sub-plots, it was allocated a nominal mean percentage cover score of 0.01.

Regression analysis was used to assess the linear relationship between mean percentage cover and height of *D. antarctica* (dependent variable) and plot age (independent variable). Where a simple linear regression model was not appropriate, exponential or 2^{nd} order polynomial models were fitted. Due to the large variation in mean percentage cover scores across ages, they were converted to a logarithmic scale. Differences between logging treatments in *D. antarctica* cover were examined using analysis of covariance (ANCOVA, Crawley 2001). This analysis compares the two regressions by testing for equality of slopes and intercepts (Chapter 2).

Experiment 2: Impacts of cable logging and burning on the frequency and cover of *D. antarctica* in wet forests

Data on the frequency and cover of *D. antarctica* from the long term floristic monitoring of two steep country cable logging areas at Wayatinah (Blue 09 and Pearce 04) is presented. The data is restricted to those four permanent plots which have been re-measured following after logging and burning. The aim of the experiment is to provide a replicated estimate of the change in *D. antarctica* frequency of occurrence and foliage biomass with time following cable logging and burning that was directly comparable to the estimates derived from the chronosequence analyses (Experiment 1 in this chapter). Comparative data such as this permits the chronosequence analyses to be validated. The description of the study area, experimental design, and sampling methods followed are presented in further detail Chapter 2 as Experiment 2.

Study Area

The experiment was conducted at the Pearce 04 and Blue 09 logging units (Table 5.2) at Wayatinah in central Tasmania, and are approximately 5 km apart. Both Pearce 04 (22 ha) and Blue 09 (30 ha) consisted of a mosaic of *E. regnans* dominated wet (Kirkpatrick *et al.* 1988) and mixed (Gilbert 1959) forests. *D. antarctica* was a conspicuous feature of the understorey stratum is association with tall shrub and ground layer species such as *Atherosperma, Acacia dealbata Pomaderris apetala, Olearia argophylla,* and *Polystichum.* Both Pearce and Blue have moderate to steep slopes and are underlain by Jurassic Dolerite. Pearce and Blue were cable logged using the same machinery in 1992, and burnt and regenerated in 1993 (Table 2.7).

Table 5.2 *Location of* D. antarctica *long term monitoring study plots at Blue 09 and Pearce 04, Wayatinah.*

Sampling Unit	Easting	Northing	Altitude (m)	Forest type (after Stone 1998)	Slope	Aspect
Blue 09 quadrat 2	458240	5300200	260	E2cST	17	62
Blue 09 quadrat 24	458140	5299230	300	E2&1b	19	80
Pearce 04 quadrat 5	457870	5303620	260	E1dTS	25	140
Pearce 04 quadrat 12	457980	5303560	230	M3T/1	15	315

Experimental design

The design adopted in the study involved the establishment of permanent plots prior to disturbance and the repeated sampling of those same plots after disturbance (Table 5.3).The pre-disturbance state are the controls.

Table 5.3 *The sampling program for the* D. antarctica *long term monitoring study at Blue 09 and Pearce 04, Wayatinah*

Event	Date	time (months)
Blue 09		
pre-treatment sampling	February-March 1992	-9
cable logging commences	March 1992	0
cable logging completed	September 1992	7
treatment plots burnt	April 1993	14
treatment plots aerially sown	May 1993	15
first treatment plot re-measure	June 1994	28
Pearce 04		
pre-treatment sampling	April-May 1992	-9
cable logging commences	September 1992	0
cable logging and	December 1992	3
treatment plots burnt	March 1993	6
treatment plots aerially sown	April 1993	7
first treatment plot re-measure	May 1994	20

Sampling methods

The unit of vegetation sampling adopted was a square quadrat of 30 x 30m or 900m^2. Within the 30 x 30m plot, ten sub-plots (2 x 2m) were spaced at 7.5m intervals along a transect either side of the quadrat's mid-point (Figure 2.6). The sub-plots were 1m apart and alternated from side to side along the transect. Within each of the 20 sub-plots, *D. antarctica* was allocated a visually assessed cover-abundance value modified from the Braun-Blanquet (Poore 1955) described in Chapter 2. *D. antarctica* was allocated a value if it was within, above or over-hanging the 2 x 2 m sub-plot. Cover was defined as foliage projective cover following Walker and Tunstall (1981). The visually assessed cover-abundance value which was then converted to a presence score. The presence score formed a local shoot frequency score of between 0 and 20 based on the number of times it was recorded within the twenty sub-plots

Analytical procedures

The visually assessed cover-abundance values from the 20 sub-plots were converted to percentage cover using to the mid-points of the particular cover classes in the conversion Table in Chapter 2. To calculate the mean percentage cover scores (and their standard errors), the individual mid-point values were then averaged across the 20 sub-plot observations. The local shoot frequency scores from the 20 sub-plots were summed and expressed as a percentage. A paired two sample Students t-test was performed to examine differences in percent cover in *D. antarctica* across the 20 2x2 m sub-plots in the pre and post disturbance plot samples. The analysis was performed using S-PLUS 6.1 for Windows Professional Edition (Lucent Technologies, Inc. Insightful Corp. 2002).

Experiment 3: Study of tagged *D. antarctica* individuals following cable logging and slash burning

Study Area

An experiment was established to examine the effects of logging and slash burning on the health and regeneration responses of *D. antarctica* and their dependent epiphytic flora (reported in Chapter 6). The study was conducted in the Wayatinah area of central Tasmania at a logging coupe and in reserved forest. The logging coupe used was Pearce 04 (Table 5.4, Figure 5.1), the controls were located 4km east in the same forest type and on the same substrate and aspect in a permanently reserved forest at Heals Spur. The environmental characteristics, pre-logging vegetation, logging and regeneration methods at Pearce 04 are described in Chapter 2. The study area and *D. antarctica* population was selected as being representative of much of the *E. regnans* dominated wet forest in southern Tasmania subject to cable and ground-based logging.

The cable logging treatment was conducted in late 1992 (Table 5.6). A pre-logging vegetation survey conducted in February 1992 indicated the community types were *Eucalyptus regnans* wet forest (Kirkpatrick *et al.* 1988). The study area was on east facing slopes with a clay loam soil underlain by dolerite bedrock.

Table 5.4 *Locational details for study of tagged* D. antarctica *individuals at Wayatinah..*

Sampling Unit	Easting	Northing	Altitude (m)	Forest type (after Stone 1998)	Slope	Aspect
Pearce 04 plot 1	458140	5303810	220	E1bSER	22^0	East (95^0)
Pearce 04 transect section 1	457780	5303550	260	E1dTS	25^0	East (95^0)
Pearce 04 transect section 2	457700	5303530	250	E1dTS	11^0	East (95^0)
Heals Spur plot 2	456400	5305700	470	E1a2dTSM3	19^0	Easterly (140^0)

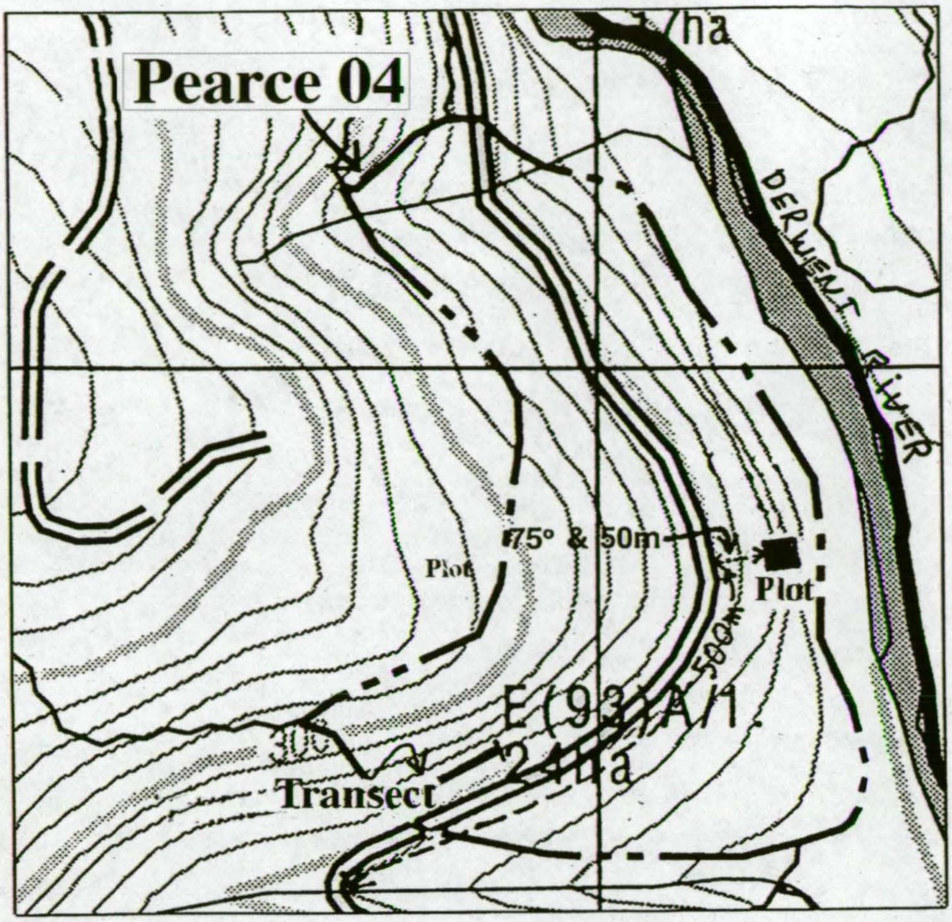


Figure 5.1 *The location of the experimental area at Pearce 04, Wayatinah, central Tasmania. The location of the two sampling areas, the 30x30 m plot and the transect are shown. The coupe boundary (line and two dashes) is indicated by the arrow beneath the Pearce 04 label. The area logged was 24 ha. The plot and transects are approximately 500 m apart, the plot is 50 m below the road at a bearing of 75 degrees. Pearce 04 is also shown from an aerial view in Figure 2.3.*

Experimental design

The design adopted in the study [Before-After-Control-Impact-Pairs (BACIP, Green 1979, 1993)] is common in studies of the effects of disturbance (e.g. Humphrey *et al.* 1995, Faith *et al.* 1995). The experiment was conducted over an area of 3500m², divided across two 30 x 30m plots, and one 170 m x 10 m wide transect (Figure 5.1, Table 5.5). One 30 x 30m plot was located at Pearce 04 in the planned cable logging area, the second at Heals Spur within a larger contiguous stand of permanently reserved forest.

The 30 x 30m plot was used where a consistent treatment, in this case either cable logging (Pearce 04) or control (Heals Spur) could be implemented. The transect was established where two treatments, cable logging or control were being applied in close proximity, and there was less certainty of the precise location of the logging unit boundary (that divides the treatment from the control) along the slope. The transect was divided into two sections, 0-96 m from the start which was subject to cable logging and burning, and 97-170 m which was an undisturbed control. The 30 x 30 m plot within Pearce 04 (Figure 5.1) was positioned 50-80 m. downslope from the Wayatinah Rd at a bearing of 75 degrees towards the Derwent River in an area where *D. antarctica* is relatively evenly distributed across the slope. The 170 m. long transect was also located on Wayatinah Rd, approximately 500 m south of the plot, in an area where *D. antarctica* is frequent and carries a high epiphytic load on its caudex (see Chapter 6 experiment 3). All individual *D. antarctica* within the plots and transect were permanently tagged.

Table 5.5 *Sampling details for study of tagged D. antarctica individuals at Wayatinah..*

Sampling Unit	Dimensions	Tagged individuals	Treatment
Pearce 04 plot 1	30x30m plot	1-65	cable logging, burning
Pearce 04 transect 1	transect (0-96m section)	66-113	cable logging, burning
Pearce 04 transect 2	transect (97-170m section)	114-141	control
Heals Spur plot 2	30x30m plot	142-150	control

Plots were established prior to logging, they were re-measured after logging occurred but prior to burning, then re-measured repeatedly after the slash burn (Table 5.6). Control plots were established at the same time as the treatment plots and were re-assessed twice (Table 5.6). The treatment comprised two steps; cable logging (log falling and yarding) and burning (slash burning of logging residue).

Table 5.6 *The sampling program for study of tagged* D. antarctica *individuals at Wayatinah..*

Event	Date	time[1]
pre-logging assessment of floristics for treatment and control	23-26 February 1992	-9
pre-logging assessment of treatment plots and controls	July 1992	-2
cable logging commences	September 1992	0
cable logging completed	December 1992	3
after logging assessment for treatment plots and controls	24-25 March 1993	6
treatment plots burnt	25 March 1993	6
first after burning assessment on treatments	24 June 1993	9
second after burning assessment on treatments	2 November 1993	12
third after burning assessment on treatments	5 May 1994	17
fourth after burning assessment on treatments	18 January 1995	25
fifth after burning assessment on treatments	8 December 1995	36
second assessment of controls	5 September 1996	45
sixth after burning assessment on treatments	February 2003	~10 years
third assessment of controls	February 2003	~10 years

[1] elapsed since logging (months)

Implementing a logging only treatment, and a burning only treatment would have had obvious benefits in terms of exploring the relative contribution of the two disturbance factors on population dynamics in *D. antarctica*, but it would have logistically very difficult to implement, and of little practical relevance to determining the effects of current forest management practices on *D. antarctica*.

Sampling methods

At establishment, the plots and the transect areas were permanently marked with steel droppers. The two 30 x 30 m plots were sub-divided with a grid of nine 10 x 10 m sub-plots, with each sub-plot intersection point being marked with tagged steel droppers. All *D. antarctica* present were mapped onto these grids and numbered (Table 5.4) with a securely attached steel ring and tag. The steel ring was constructed to fit snugly around the caudex. In the transect, all individuals 5 m either side of the centre line tape had their positions mapped and were scored and numbered in the same way (Figure 5.2).

During the pre-logging assessment of the treatment plot and transect and the control, the height of each *D. antarctica* caudex was measured with a tape measure from the base of the caudex, at the soil surface (coarse litter and fallen fronds were cleared away), to the apical tip following Medeiros *et al.* (1992). For each *D. antarctica* individual, the slope and aspect of the micro-site occupied was measured with a compass and clinometer. Each individual was either mapped into the grid of nine 10 x 10 m sub-plots in the 30 x 30 m plo,or for the transectm its distance along the transect and from the transect centre line was recorded. Each individual was scored for survivorship and orientation (Table 5.1 ie. as either 'dead prostrate', 'dead upright', 'alive prostrate', or 'alive upright') and the percentage of the caudex colonised by epiflora.The variables scored for each *D. antarctica* during after logging and burning assessments are given in Table 5.6.

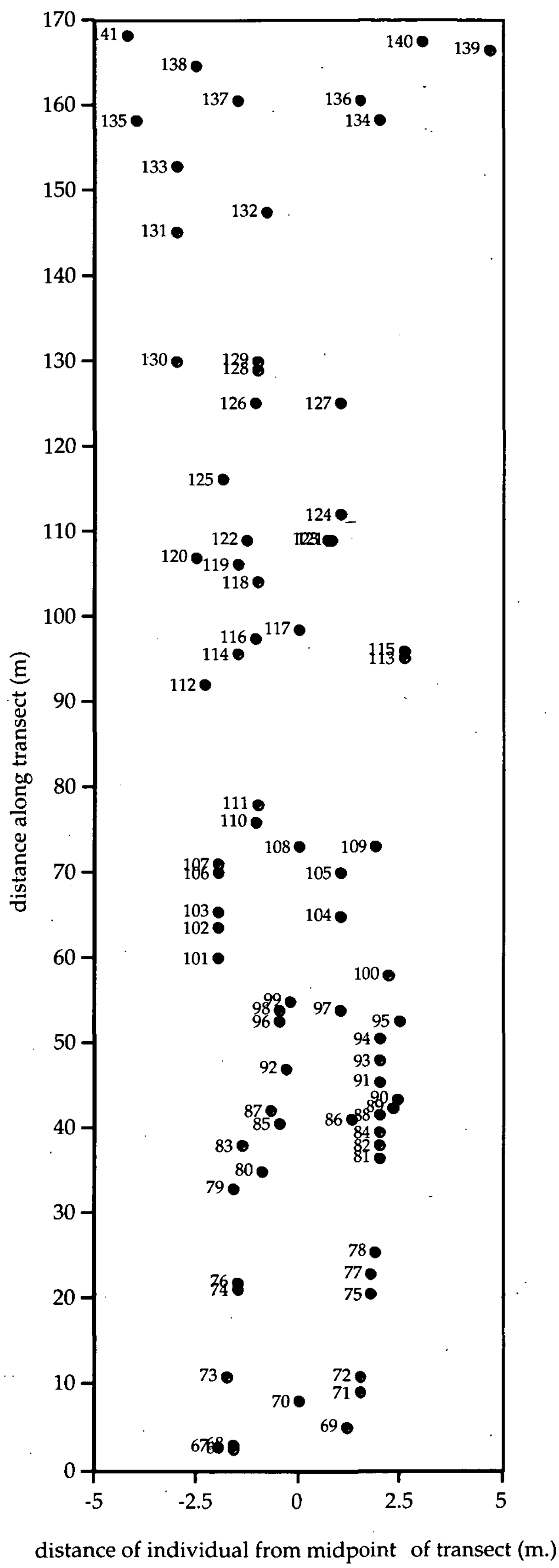


Figure 5.2 *The location of individual* D. antarctica *stems (•) along a transect at Pearce 04, Wayatinah prior to logging. Each stem was identified, mapped, marked with a steel peg and the number attached to a steel ring so stems could be relocated after logging and burning.*

At the first after burning assessment, the circumference of each *D. antarctica* was measured at the caudex mid-point. Pre-burning circumference was obtained from the wire which had been placed snugly around the caudex to hold a steel numbered tag. Two millimetre diameter steel wire is resistant to the fire. Variables scored for *D. antarctica* individuals at subsequent assessments are described in Table 5.7.

Table 5.7 *Variables scored for* D. antarctica *individuals following logging and burning.*

variable	attribute
mortality	dead, alive (0,1)
development stage	new recruit, adult with caudex (0,1)
health	poor, moderate, good (0,1,2)
caudex attachment	rooted to soil or dislodged, ie root mantle broken or pulled free from soil and moved from original position (0,1)
caudex position	upright, intermediate, prostrate (0,1,2)
caudex fire damage	moderate, severe (0,1)
caudex logging damage	none, moderate, severe ie, apical bud destroyed with no means of regeneration and death is inevitable (0,1,2)
*croziers +/-	present or absent (0,1)
crozier development	dormant or unfurling (0,1)
expanded fronds	number
height increment	height increase in cm above burnt caudex apex at 10 years

*a crozier is the developing 'fiddle head' of a frond ready to start expansion

Analytical procedures

Population demographics were explored using frequency histograms of *D. antarctica* caudex heights in 25 cm intervals. The survivorship and orientation categories (Table 5.1) of individual *D. antarctica* was examined by pooling the results from the cable logged and burnt treatments (both plot and transect) and expressing the data as a percentage of the number of observed *D. antarctica* in each category. Changes in mean circumference were explored using t-tests after pooling the data into 25 cm height intervals and averaging within those intervals. The extent of any linear relationships between the variables listed in Table 5.7 for *D. antarctica* individuals were examined using Pearson product-moment correlation analysis (Sokal and Rohlf 1981). Analysis of variance (type III sums of squares) was used to examine the relationship between caudex circumference change following burning and *D. antarctica* survivorship and orientation. Height growth differences between surviving *D. antarctica* individuals and controls at 10 years were compared using a two sample un-paired t-test in S-PLUS 6.1 for Windows Professional Edition (Lucent Technologies, Inc. Insightful Corp. 2002). Variation in caudex heights for *D. antarctica* individuals surviving or dying ten years following cable logging were explored using Box-plots of the quartiles method in S-PLUS 6.1. Differences in caudex heights were tested using a Wilcoxon rank-sum test also in S-PLUS 6.1

Experiment 4: Impacts of cable logging on *D. antarctica* within streamside reserves

Study Areas

Six study areas in northern Tasmania were utilised to examine the response of *D. antarctica* in streamside habitats to different cable logging treatments. The six study areas (all logged 3-5 years prior to sampling) were Roses Tier 126a, Roses Tier 137f, Roses Tier 137g, Loatta 217a, Branchs Creek 60a and Branchs Creek 80a (Table 5.8). These study areas were selected on the basis of similar pre-logging vegetation (from forest type mapping, Stone 1998), stream size, geology, and at Roses Tier, the same cable logging system. Detailed site attributes for each of the locations examined are given in Chapter 4, Branchs Creek 60a is also described by Davies and Nelson (1994).

Table 5.8 *The plot locations, environmental characteristics and logging treatments of the streamside reserve study areas.*

Plot	Year logged	Easting/ Northing	Altitude (m)	Forest type (after Stone 1998)	Stream slope	Side slope	Geology
Roses Tier 126a	1989	457400/5 314400	550-650	E. regnans and Callidendrous Rainforest T(W)M3	5^0	30^0	granodiorite
Roses Tier 137f	1989	458600/5 316600	450-530	E. regnans/E. obliqua and Callidendrous Rainforest M2.T.	5^0	45^0	granodiorite
Roses Tier 137g	1990	458200/5 315500	450-470	E. regnans and Callidendrous Rainforest T(W)M3	5^0	40^0	granodiorite
Loatta 217a	1986	435100/5 392200	350-380	Callidendrous Rainforest T(W).S.	9^0	30^0	shale mudstone
Branchs Creek 60a	1989	475400/ 5335900	160-170	Callidendrous Rainforest T(W) S. E2f.	5^0	$30\text{-}37^0$	quartzwackes
Branchs Creek 80a	1988	473600/5 333500	180-200	Callidendrous Rainforest S.T (W) E2f c/o	8^0	$24\text{-}32^0$	quartzwackes

Experimental design

Stratified random sampling of vegetation was conducted to examine how characteristics of *D. antarctica* populations varied with different streamside cable logging practices, an approach also adopted in northern Tasmania by Davies and Nelson (1994). Cable logging was considered a factor with three levels if intensity; (1) severely disturbed ie a cable logging gap in the reserve (2) a maintained portion of streamside reserve, and (3a) upstream control and (3b) downstream control. The survey was replicated in six locations across northern Tasmania (Table 5.8). Streamside reserve or buffer areas were not considered to be controls as they were generally narrow strips of retained vegetation subject to edge effects. Control plots were sampled in the vicinity of the cable logged area both upstream and downstream. At each location, four sites were identified for stratified random sampling along the streamside, with the total distance between the four plots being

100-200 m. One of each of the four sites was allocated to each of the disturbance types at each location, ie (1) severely disturbed, (2) a maintained portion of streamside reserve, and (3a) upstream control and (3b) downstream control.

Sampling methods

Data were collected on the population characteristics of *D. antarctica* within a single 10 x 2m plot in each of the four stratified types; severely disturbed, reserved portion, upstream control and downstream control. The 10 x 2m plot was located within an area which was homogeneous with respect to the allocated treatment. Within each plot, all *D. antarctica* individuals rooted inside the plot were counted, individuals present as over-hanging fronds were ignored. Each *D. antarctica* was scored as either an alive upright, alive prostrate, dead upright, dead prostrate or new recruits following the previously described categories.

Analytical procedures

The number of individual *D. antarctica* stems present in each the five orientation and survivorship categories was aggregated into three treatment types (gap, reserved portion and control), with the data from the upstream and downstream controls averaged. *D. antarctica* stem count data from the orientation and survivorship categories was grouped by treatment type and then averaged across the six replicate study areas. Mean counts and their standard deviations for each of the five survivorship and orientation categories in each treatment type were presented in with histograms. The density of *D. antarctica* stems in the orientation categories were compared across treatments (the factor) with One-way Analysis of Variance (ANOVA). Density was also compared between treatments where the orientation categories were considered the factor. ANOVA was completed using S-PLUS 6.1 for Windows Professional Edition (Lucent Technologies, Inc. Insightful Corp. 2002).The null hypothesis is that there is no difference between the mean values for the orientation and survivorship categories in the three treatment groups (gaps, retained areas and controls).

5.3 Results

Experiment 1: Chronosequence analysis comparing the response of *D. antarctica* to cable and ground-based logging

Recovery and recruitment of *D. antarctica* in cable logged wet forest

In the unlogged wet forest chronosequence sites, the cover of *D. antarctica* was typically 50-70%. Most of the *D. antarctica* individuals were live upright stems, and approximately one-sixth were living prostrate stems. Dead prostrate stems and dead upright stems were also present (Figure 5.3a) in these unlogged forests.

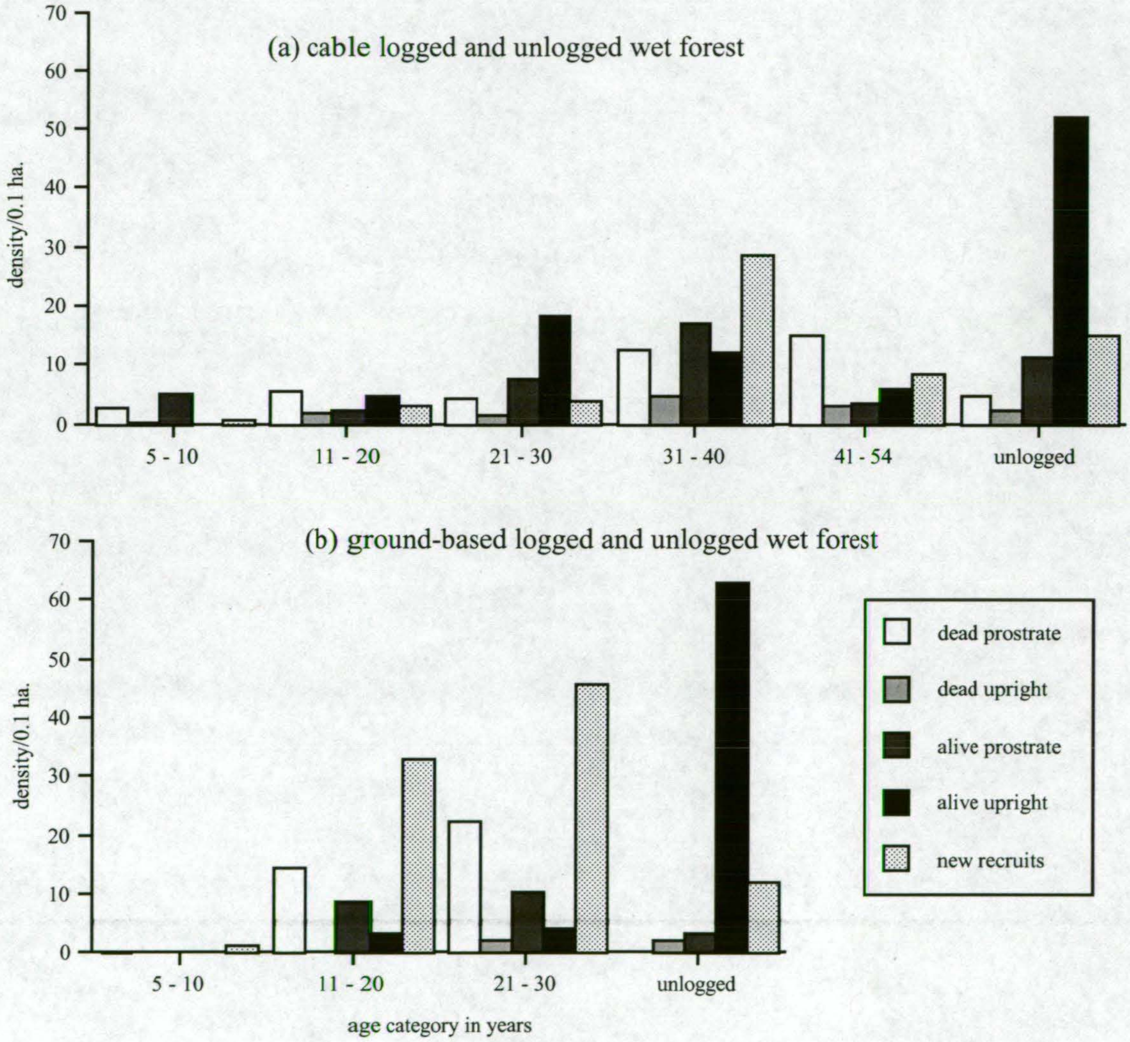


Figure 5.3 a-b *Recovery and recruitment of* D. antarctica *in wet forest in central and southern Tasmania. The values are averaged for each age category. Ground-based logged wet forest were sampled in the 5-10, 11-20, 21-30 and mature age categories only. Standard deviations are large and are not shown.*

The cover of *D. antarctica* was low and variable in regenerating cable logged wet forest until 29 years when cover was in excess of 10% (Figure 5.4a). Based on the average height of individuals (Figure 5.4b) and their orientation as live upright and prostrate stems (Figure 5.3a) it is apparent that one-third to one-fifth of the individual *D. antarctica* present have persisted during logging and burning and have regenerated vegetatively. In the oldest regrowth sampled, the cover of *D. antarctica* was also low, with individuals exhibiting a patchy, clumped distribution.

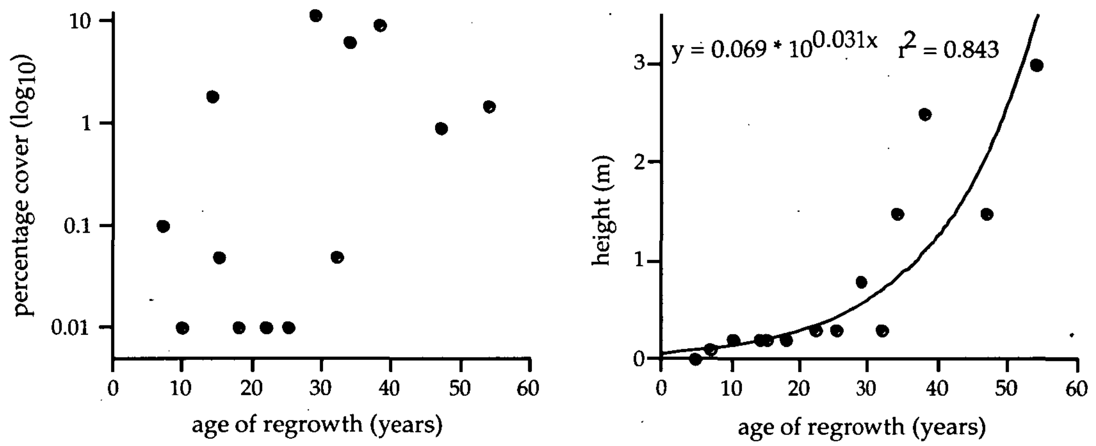


Figure 5.4 a-b *Mean percentage cover and caudex height (exponential curve fitted) of* D. antarctica *in the cable logged wet forest chronosequence.*

New recruits were infrequent in regrowth less than ten years old, and were confined to micro-terraces of heaped soil. With increasing stand age, the abundance of young *D. antarctica* sporophytes increased (Figure 5.3a), as did the diversity of substrates they occupied. (Table 5.9). The most frequently utilised substrate for *D. antarctica* new recruits was the surface of the residual fallen logs, usually from the previous logging event.

Table 5.9 *Host preferences and abundance of* D. antarctica *new recruits recorded in the cable logged wet forest chronosequence. The samples are ordered by stand age since logging and the abundance scores on the particular substrate are rare (1) and moderate (2).*

substrate / age	5	7	9	14	15	18	22	25	29	32	34	38	47	54	100	120	150
D. antarctica				1					1						1	1	
eucalypt buttress				1													
fallen logs										1	1	1		1		2	1
other buttresses												2					
rocks													1				
cut stumps								1			1		1				

Recovery and recruitment of *D. antarctica* in ground-based logged wet forest

In mature wet forest the cover of *D. antarctica* was typically 50-80%. Most of the individuals present in mature forests were live upright stems. Live prostrate stems were present, but relatively uncommon (Figure 5.3b). No caulescent ("becoming stalked, where the stalk is clearly apparent", Jackson 1971) *D. antarctica* were observed in ground-based logged wet forest less than 10 years old (Figure 5.3b). The majority of individuals present in logged forests up to 30 years of age were new recruits growing mainly on fallen logs (Figure 5.3b). Most of the *D. antarctica* individuals which survived logging and burning did so as live prostrate stems (Figure 5.3b). The cover of *D. antarctica* in regrowth stands aged 5-30 years was low and variable (Figure 5.5a). The few alive upright individuals with caudex heights > 2m were restricted to the oldest regrowth sampled (Figure 5.5) and would have predated the logging disturbance.

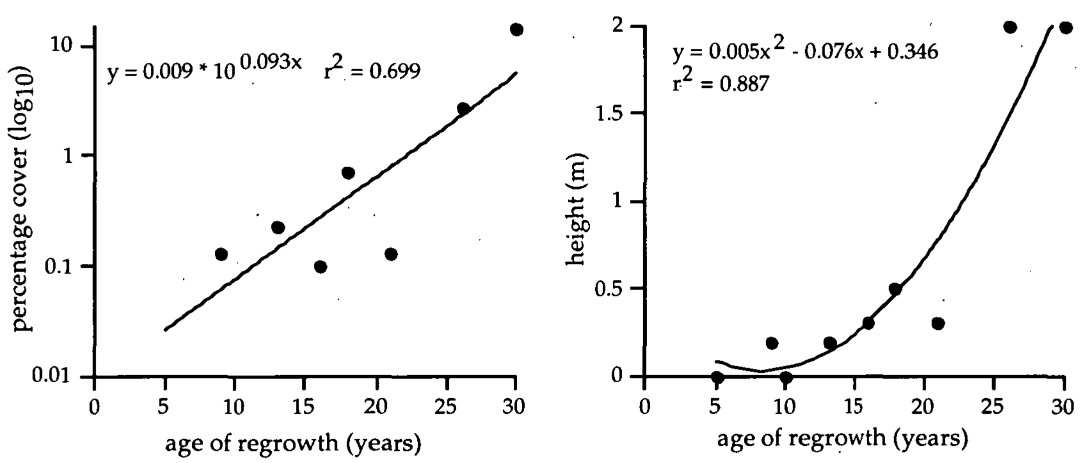


Figure 5.5 a-b D. antarctica *cover (exponential curve) and caudex height (2nd order polynomial fitted) of in the ground-based logged wet forest chronosequence.*

New recruits were infrequent in ground-based logged wet forest <10 years old, observed only on fallen logs and soil heaped along the margins of logging tracks. New recruits were more common in 21-30 year old regrowth on fallen logs and eucalypt buttresses (Table 5.10).

Table 5.10 *Host preferences and abundance of* D. antarctica *new recruits recorded in the ground-based wet forest chronosequence. The samples are ordered by stand age since logging and the abundance scores on the particular substrate are rare (1), moderate (2) and abundant (3).*

age substrate	5	9	10	13	16	18	21	26	30	58	78
D. antarctica										1	1
Atherosperma moschatum										1	
eucalypt buttress									2		
fallen logs		1			1	2	1	1	3	2	2
rocks								1			

Comparison of recovery and recruitment of *D. antarctica* following cable or ground-based logging

Cable and ground based logging have broadly similar effects on populations of *D. antarctica* in wet forest. After either treatment cover was low and variable (Figure 5.6) and approximately one-fifth of the cover in matched mature wet forests. The rate at which cover was re-established following either logging method in wet forest did not differ significantly ($p > 0.05$) when the slopes and intercepts of linear regressions fitted to each chronosequence were compared.

Ground-based logging resulted in proportionally greater mortality to *D. antarctica* based on the density of dead prostrate individuals than cable logging. Ground-based logged sites had a greater proportion of their living *D. antarctica* as live prostrates. By comparison in the cable logged sites most of the living *D. antarctica* were upright following logging.

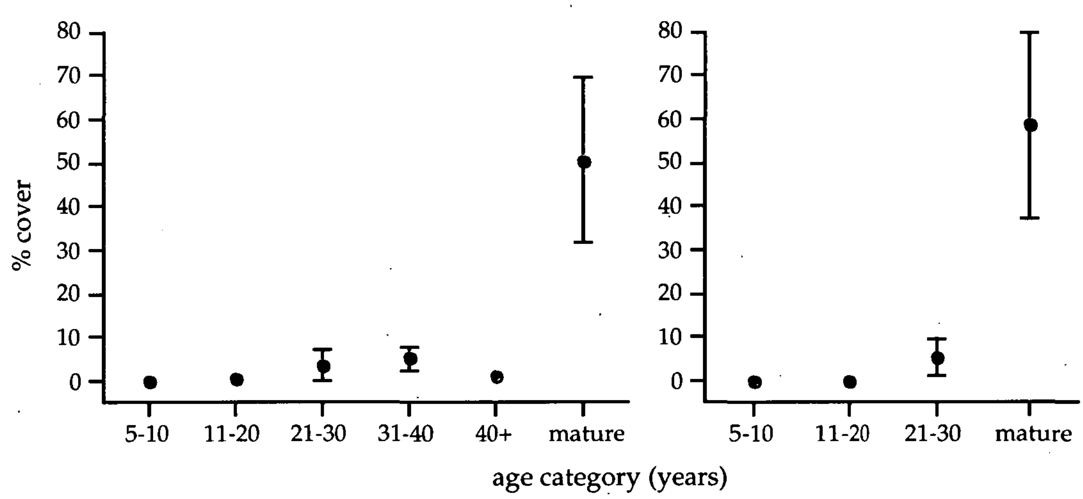


Figure 5.6 a,b *Mean percentage cover (±s.e.) of* D. antarctica *in the cable (a) and ground-based (b) logged wet forest chronosequences against age since logging.*

Recruitment of *D. antarctica* sporophytes into ground-based logged forest occurred at a rate an order of magnitude greater than in cable logged forest in the 11-20 and 21-30 year age categories (Figure 5.3). The substrate preferences of *D. antarctica* new recruits varied between the logging methods. On cable logged sites a wider range of substrates were utlised (Table 5.11), ground-based sites had over-lapping but less well developed substrate use patterns. On ground based sites new recruits were more frequent on disturbed soil heaps compared to the woody substrates favoured in cable logged sites (Table 5.11).

Table 5.11 *Comparative recruitment of* D. antarctica *on different substrates in regrowth wet forests originating from cable and ground-based logging. Data from Tables 5.9 and 5.10.*

	logging treatment	
substrate	cable	ground-based
D. antarctica caudex	rare	-
eucalypt buttress	rare	moderate
other buttresses	rare	-
cut stumps	moderate	-
fallen logs	rare	rare-abundant
rocks	rare	rare
mineral soil	rare	moderate

The gradual increase in *D. antarctica* cover with time since stand initiation following either logging method was inversely proportional to the reduction in overstorey canopy cover over the same period. As the *Eucalyptus* dominated canopy self thinned with time, *D. antarctica* increased in abundance (Figure 5.7).

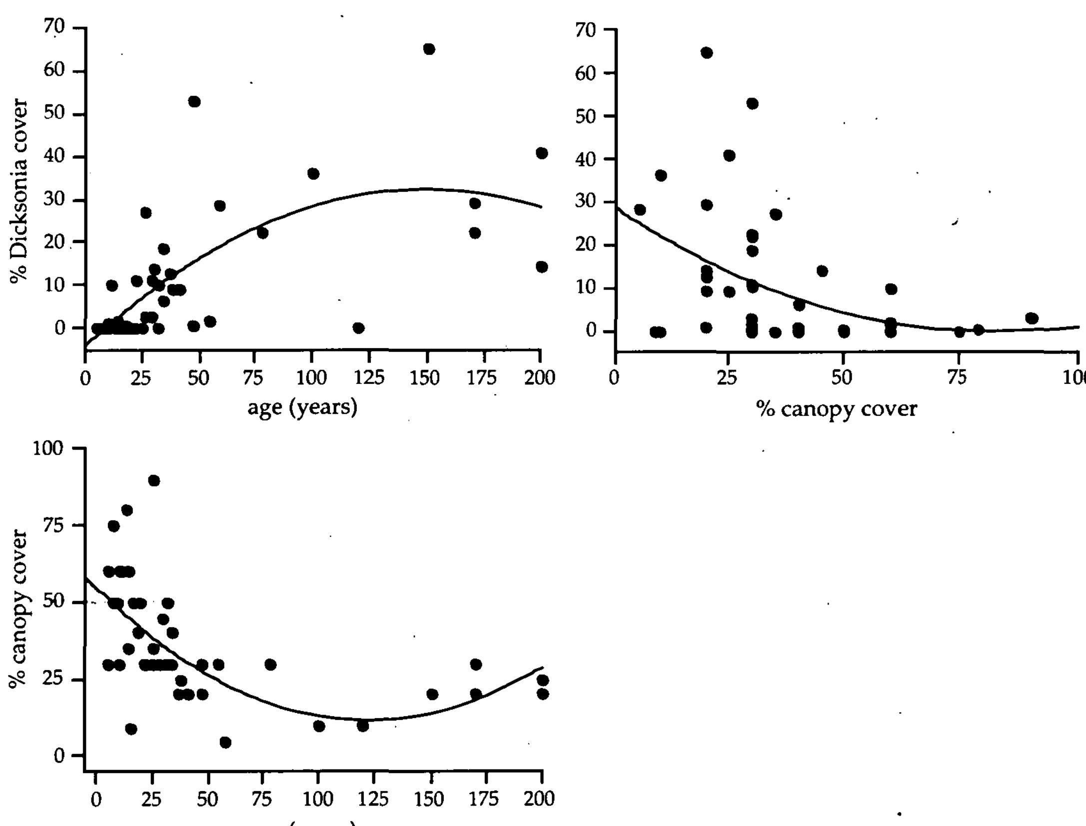


Figure 5.7 *Relationship between* D. antarctica *mean percentage cover, stand canopy cover and stand age in cable and ground-based logged wet forests of southern Tasmania.*

Experiment 2: Impacts of cable logging and burning on the frequency and cover of *D. antarctica* in wet forests

Cable logging and slash burning led to a significant ($p<0.05$) short term reduction in the cover of *D. antarctica* in the four permanent quadrats. The reduction in cover 14 months after the slash burn ranged from 73-97% irrespective of the initial value in undisturbed forest (Table 5.12). The reduction in cover was therefore approximately the same on a proportional basis for sites with low versus high initial covers.

Table 5.12 *Mean percentage cover and percent frequency (based on the sum of 20 2x2 m sub-plot observations) of* D. antarctica *in four permanent plots at Wayatinah before and 14 months after cable logging and slash burning. The significance of the difference in mean percentage cover values are tested with a paired students t-test, at a 95% confidence interval with 19 degrees of freedom*

	pre-logging % frequency	after logging and burning % frequency	pre-logging mean (±s.e.) % cover	after logging mean (±s.e.) % cover	% reduction in cover	significance of difference
Blue 2	50	40	14.3 (4.8)	3.8 (2.0)	73.6	p=0.049
Blue 24	85	25	29.5 (5.4)	0.63 (0.25)	97.8	p<0.001
Pearce 12	35	20	8.6 (3.9)	1.05 (0.75)	87.8	p=0.047
Pearce 5	50	10	1.8 (0.75)	0.17 (0.13)	90.2	p=0.038

Experiment 3: Study of tagged *D. antarctica* individuals following cable logging and slash burning

Pre-logging population characteristics

The caudex heights and circumferences have an approximately normal distribution (Figure 5.8 a,b). Of the 150 individual *D. antarctica* in the study, caudex's ranged from 0.2 to 5.0m in height and 0.3 to 2.1m in circumference (Figure 5.8 a,b). The mean population height was 1.7m (s.e. ±0.07) and circumference 0.99m (s.e. ±0.03).

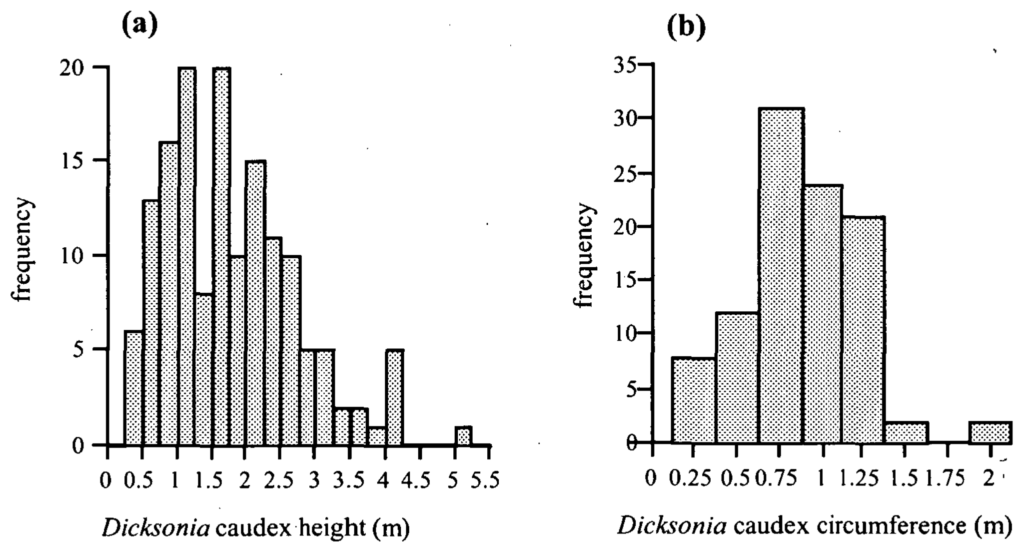


Figure 5.8 a,b *Frequency histogram of* D. antarctica *caudex heights (a) and circumferences (b) prior to logging.*

The relationship between the *D. antarctica* caudex height and circumference is weakly exponential (r^2=0.35, Figure 5.9), the caudex of individual *D. antarctica* does not necessarily become thicker as it lengthens. The density of *D. antarctica* populations averaged 428 stems/ha and 155 new recruit sporophytes/ha across the treatment and control portions prior to cable logging. The basal area of *D. antarctica* within the treatment area was 47.1 m^2/ha. Individuals were randomly distributed along the transect, except in sparse areas where the transect followed the population across the slope and consequently the individuals do not appear to be randomly distributed (Figure 5.2).

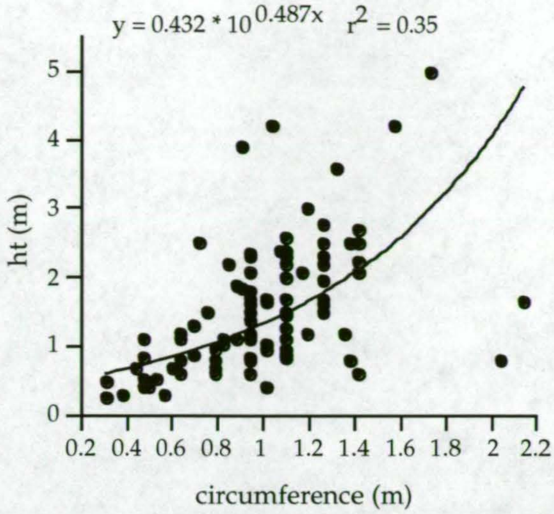


Figure 5.9 *Relationship between* D. antarctica *population caudex height and mid-point circumference prior to logging.*

Impacts of cable logging

The study area (Pearce 04) was clearfelled and the logs extracted by cable. Many of the *D. antarctica* stems were buried 1-1.5 m beneath logging slash debris (Figure 5.10). This led to some re-sampling difficulties with 27.7% of the tagged *D. antarctica* stems unidentified at this stage (Table 5.13), although after the regeneration burn had reduced much of the logging residue, many of these individuals were later identified and the data adjusted accordingly.

Figure 5.10 *A heavy accumulation of logging slash debris just prior to burning at Pearce 04 Wayatinah effectively hid some* D. antarctica *until the debris was burnt.*

Three months after cable logging was completed, survivorship of *D. antarctica* individuals visible in the logging residue was assessed at 64%. The after logging survivorship figure was later revised to 72.9% once the logging residue was burnt and the plot and transect areas could be effectively searched for individuals previously buried under the logging residue. Cable logging increased the proportion of prostrate living individuals (Table 5.13), and decreased the proportion of living upright individuals relative to the initial state of the population, attributed to physical damage during log falling and cable yarding. The reduced proportion of dead specimens three months after cable logging is attributed to the proportion of tagged individuals which were inaccessible in the logging slash for scoring (the nor recovered category).

Table 5.13 *Survivorship and orientation of* D. antarctica *three months after cable logging at Pearce 04. The after logging data is provided only for individuals accessible within the logging residue at this time. New recuits are excluded from the totals but included in Figure 5.20*

	alive prostrate	dead prostrate	alive upright	dead upright	not recovered/ indeterminate
pre-treatment % (n=111)	3.6	5.3	76.8	14.3	-
after logging % (n=77)	30.8	3.1	36.9	1.5	27.7

Of the *D. antarctica* individuals identified under the cable logging residue, most were still attached to the soil by their root systems (Table 5.14), even though they were commonly prostrate. The group of *D. antarctica* individuals which were flattened by logging were significantly (students t-test, $p<0.01$) taller (mean height 1.7 m) than those that remained upright (mean height 1.1 m).

Physical disturbance from cable logging had little effect on survival or resprouting of *D. antarctica*. *D. antarctica* individuals resprouted irrespective of being attached to the soil or having their root mantle overturned and becoming dislodged following cable logging (Table 5.14). Of the living specimens, nearly all had croziers emerging from the terminal bud or new green fronds present within three months of cable logging being completed (Table 5.14).

Table 5.14 *The count of* D. antarctica *individuals three months after logging which were attached to, or dislodged from, the soil and exhibiting signs of vegetative recovery (based on observations of 77 of the 111 individuals in the logged area).*

	living (regeneration present)	dead (regeneration absent)	total
attached	53	9	62
dislodged	15	0	15

Figure 5.11 *Understorey of* D. antarctica *and* Olearia argophylla *immediately after log falling but prior to cable yarding of the logs upslope at Blue 09 Wayatinah, 1992. Most* D. antarctica *are upright and have retained their crown of fronds relatively intact.The removal or fronds and general disturbance to D. antarctica individuals was associated more with log yarding than log falling.*

Figure 5.12 *Understorey of* D. antarctica *and* Olearia argophylla *immediately after cable yarding of the logs upslope at Blue 09 Wayatinah, 1992. Few* D. antarctica *are upright and undisturbed, most are partially buried under logging residue.*

Initial impacts of cable logging and slash burning

Slash burning of the cable logging residue led to a significant reduction in survivorship, from 73% percent following logging only to 43% three months after logging and burning. A greater proportion of those *D. antarctica* individuals failing to exhibit any evidence of after burning recovery were classed as either being severely fire damaged (41.1%) than the living individuals (2%, Table 5.15).

Table 5.15 *Effect of orientation and fire damage on survivorship of* D. antarctica *three months after logging at Pearce 04.*

	orientation		fire damage	
	prostrate	upright	moderate	severe
Alive % (n=48)	41.6	58.4	98	2
Dead % (n=63)	33.9	66.1	58.9	41.1

While there was a clear effect of fire damage severity on survival following the slash burn, it was not related to the orientation of the caudex in terms of individuals being prostrate or upright. (Table 5.15). Severe fire damage was occasionally associated with *D. antarctica* individuals having their caudex burnt almost completely through to the central pith (Figure 5.13), often where a burning log was in contact.

Figure 5.13 *Severe fire damage to individual* D. antarctica's *was often associated with logs which were in contact with the caudex, and subsequently burnt slowly into the caudex through to the central vascular bundle or stele. With the combustion of much of the adventitious root system surrounding the caudex, old stipe bases are exposed.*

Table 5.16 *The relationship between orientation, survival and surface slope for individual* D. antarctica *three months after burning at Wayatinah. The relationship is tested using one-way ANOVA, with the orientation categories (Table 5.1) considered a factor with four levels, and 'not recovered' individuals excluded*

	alive prostrate	dead prostrate	alive upright	dead upright	not recovered/ indeterminate
% (n=111)	29.7	29.7	13.6	15.3	11.7
mean slope in degrees	22.6	23.4	21.2	19.4	-
	df	sums of squares	mean square	F value	probability of F
orientation and survival factor	3	212.3	70.7	3.24	0.0252
residuals	100	2184.5	21.8		

A small proportion (9.4%) of the *D. antarctica* individuals which remained upright during logging subsequently over fell after burning (Table 5.13 and Table 5.16). The propensity for *D. antarctica* specimens to be unbalanced by burning was not related to their caudex height but it was to the slope of the land (Table 5.16). While the relationship between slope and orientation is statistically significant, it is only weakly so, prostrate individuals were found on slightly steeper sites than upright individuals.

D. antarctica specimens exhibiting moderate to severe physical damage from cable logging (which usually involved the loss of the terminal bud) had an increased rate of mortality compared to specimens exhibiting no signs of damage (Table 5.17). Those individuals for which the degree of physical damage was not determined (Table 5.17) may either have been buried beneath logging residue or completely consumed by the fire.

Table 5.17 *Comparative degree of physical damage from cable logging to* D. antarctica *individuals, three months after burning at Pearce 04.*

	physical damage to caudex			
	nil	moderate	severe	not determined (not located)
Alive % (n=48)	96	3	1	-
Dead % (n=63)	50	25	10.7	14.3

Three months after logging and burning, 83.3% of the living *D. antarctica* individuals immediately exhibited vegetative regeneration. Most developed a crop of dormant croziers within their apical bud, and approximately one-third were actively unfurling croziers and expanded fronds (Table 5.18). The remaining individuals developed croziers at a later date.

Figure 5.14 *The regeneration burn at Wayatinah on 25/3/93. Note the volume of logging residue (also Figure 5.10) compared to the after fire volume in Figure 5.15*

Figure 5.15 *Standing and fallen* D. antarctica *individuals at Wayatinah three months after burning.*

Table 5.18 *The form of vegetative recovery exhibited by living* D. antarctica *individuals, three months after burning at Pearce 04.*

	nil[1]	croziers present within terminal bud	croziers unfurling	expanding fronds
Alive % (n=48)	16.7	60.4	20.8	2.1

[1]developed croziers at a later date

Burning alone resulted in a 60% reduction (47.1m²/ha before logging to 18.8 m²/ha) of *D. antarctica* basal area on a population basis, through the loss of individuals and the reduced diameter of retained living individuals. Caudex circumference decreased significantly (two-sample t-test, t=9.32, p<0.001) following burning, from a mean of 0.99 m prior to cable logging to a mean of 0.62 m following burning. The largest proportional reduction in caudex circumference occurred in individuals that had the largest initial circumference (Figure 5.16). Despite the volume of *D. antarctica* caudex's consumed by burning, survival three months after burning was unrelated to the proportion of the caudex burnt (Table 5.19). The categorical assessment of fire damage to *D. antarctica* caudexes, which considered the depth of burning into the central pith of the caudex (Table 5.17) was better predictor of survival than circumference loss.

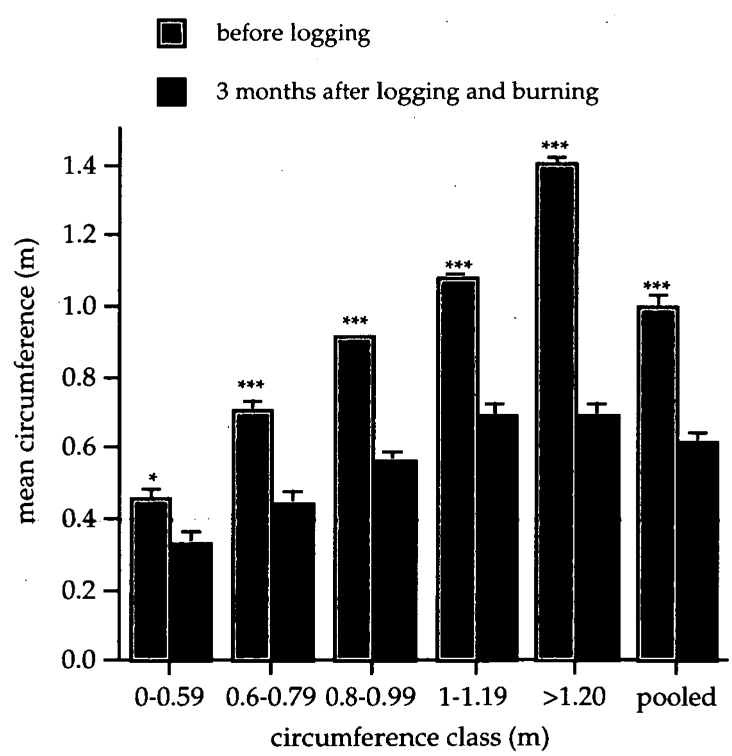


Figure 5.16 D. antarctica *mean caudex circumference and their standard errors at Pearce 04, Wayatinah, prior to logging and three months after logging and burning. Asterisks indicate a significant difference between values before logging and three months after logging and burning.* * $p < 0.05$ *** $p < 0.001$. *The pooled class includes all the previous circumference classes.*

Table 5.19 *Relationship between caudex circumference loss (due to burning) and survival (at three months) of individual* D. antarctica *at Wayatinah. (n=100, individuals with missing values were excluded).*

source of variation	sum of squares	d. f.	mean square	F-ratio	significance level	
co-variates						
alive or dead	277.6695	1	277.6695	0.928	0.3484	NS
	735.0368	1	735.0368	2.456	0.1207	NS
main effects						
circumference class (Fig. 5.16)	1726.8334	4	431.7083	1.442	0.2269	NS
residual	26040.947	87	299.3212			
total	28257.205	93				

Figure 5.17 *The wire band securing the numbered tag to the caudex prior to logging and burning provides a graphical illustration of the degree of caudex burnt.*

Logging and burning also resulted in individual *D. antarctica* stems moved up or downslope from their mapped positions prior to logging. The distances moved were up to 20 m. In most cases the distances moved were 1-2 m.

Short term trends in survivorship – 3-10 years

Three years after cable logging, 29% of the tagged *D. antarctica* had survived (Figure 5.18). An assessment of the control (untreated) plots revealed no changes in survivorship or recruitment of new individuals. The survivorship curve for *D. antarctica* during the study period plateau's three years after logging (2nd order polynomial fitted: $y=0.112x^2 - 5.838x + 97.622$ $r^2=0.940$) (Figure 5.18).

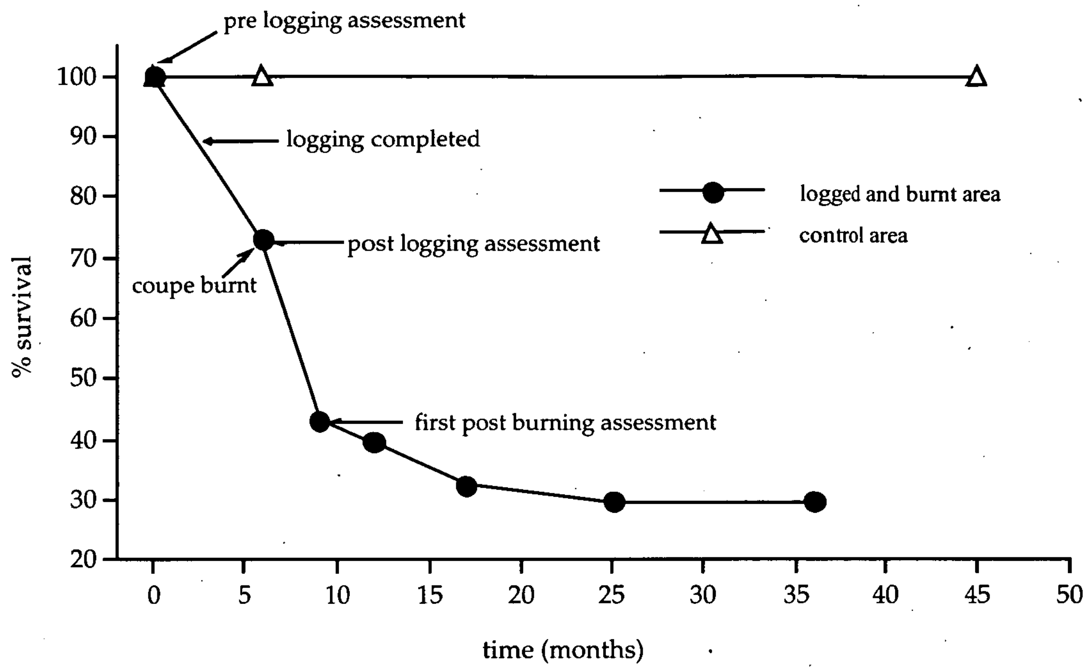


Figure 5.18 *Short term trend in the survival of* D. antarctica *following cable logging and burning at Wayatinah.*

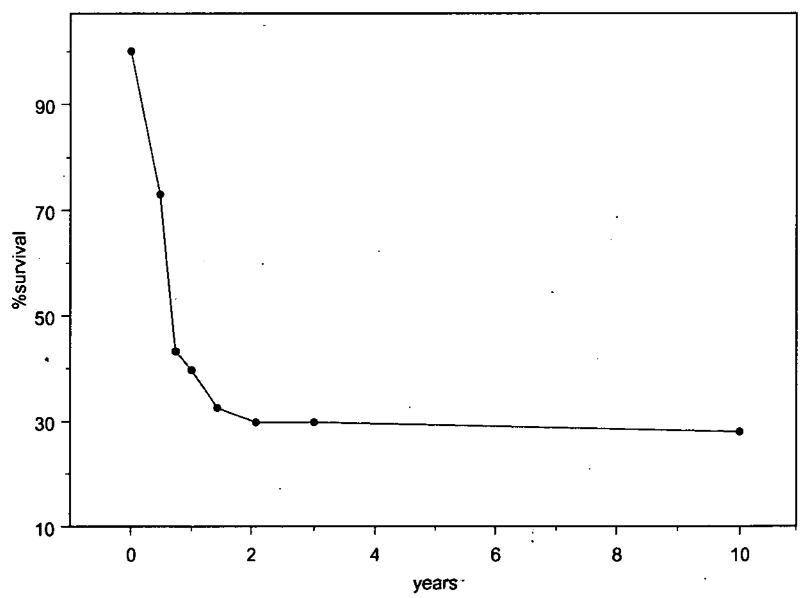


Figure 5.19 *Survivorship of* D. antarctica *ten years after cable logging and burning.*

Between the three and ten year periods, survivorship was nearly stable at 27%, only three further deaths of individually tagged *D. antarctica* occurred (Figure 5.19).

Three years after cable logging and burning most of the living individuals were prostrate, only 21.2% were upright, the reverse of the situation prior to cable logging and burning. Orientation was not a good predictor of survival, there were a greater proportion of dead upright individuals than living upright individuals (Figure 5.20). Over time, dead individuals were more likely to fall over than live individuals. While there was a small decrease in the proportion of living upright stems, the proportion of living prostrate stems remained constant. At ten years no new recruitment had been observed.

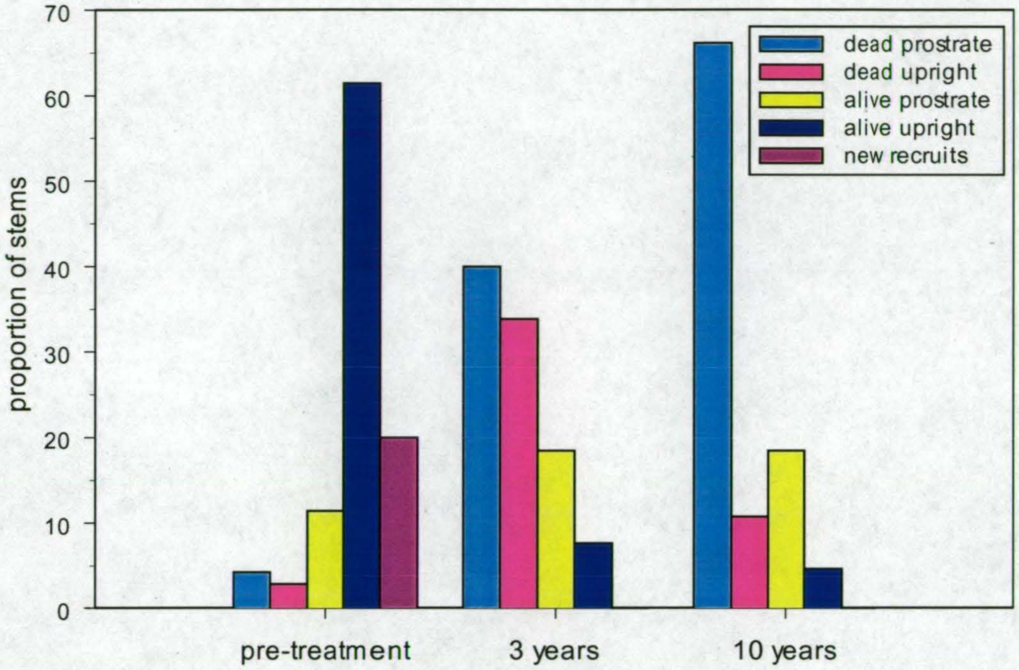


Figure 5.20 *The proportion (%) of the* D. antarctica *population in the orientation (prostrate, upright) and survivorship categories (alive, dead) before, three and ten years after cable logging and burning at Wayatinah.*

Survival of individuals in the *D. antarctica* population after three years was unrelated to their attachment to the soil surface (Table 5.20), a trend also evident in the initial results obtained eight months after logging and burning. Taller individuals were more likely to survive cable logging and burning (two sample Wilcoxon rank sum test Z = 2.1691, p-value = 0.0301), although the difference in mean heights between surviving and dying individuals was not great (Figure 5.21, 5.22).

Table 5.20 *The survival of attached and dislodged* D. antarctica *individuals three years after logging and burning at Wayatinah.*

	attached	dislodged
alive % (n=33)	81.8	18.2
dead % (n=78)	89.7	10.3

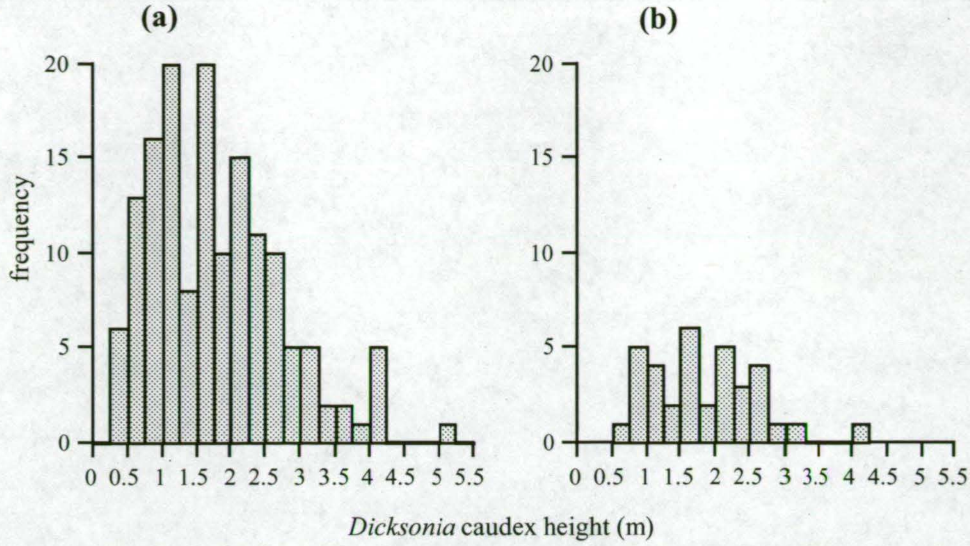


Figure 5.21 a, b *Frequency histogram of* D. antarctica *caudex heights for (a) individuals prior to logging (n=111, mean = 1.51 m, se=±0.111) and (b) those surviving ten years after the logging and burning treatments.*

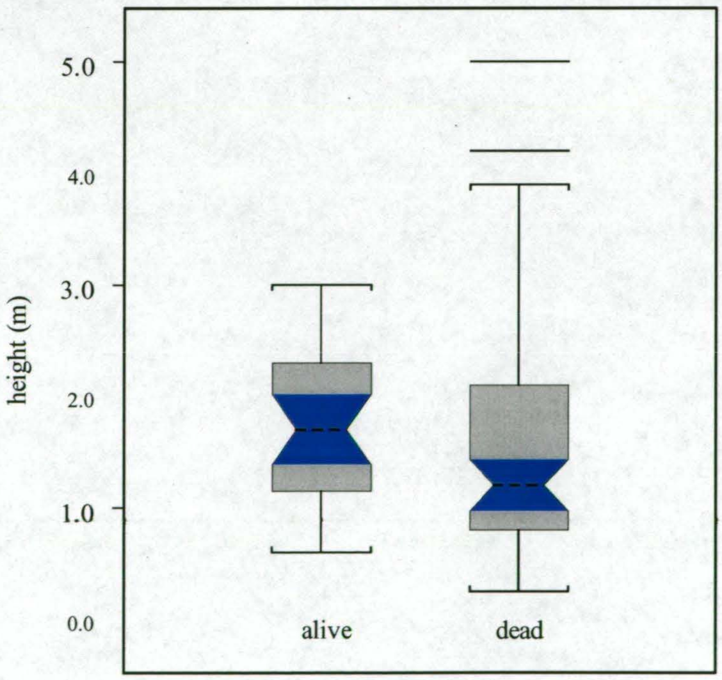


Figure 5.22 *Box-plot of* D. antarctica *caudex heights for individuals surviving (mean height = 1.74 m, s.e. = ±0.116) or dead (mean = 1.5 m, s.e. = ±0.11) ten years after logging and burning at Wayatinah.. The whiskers are the first and third quartiles, the dark shaded portion are the s.e., straight bars are outliers*

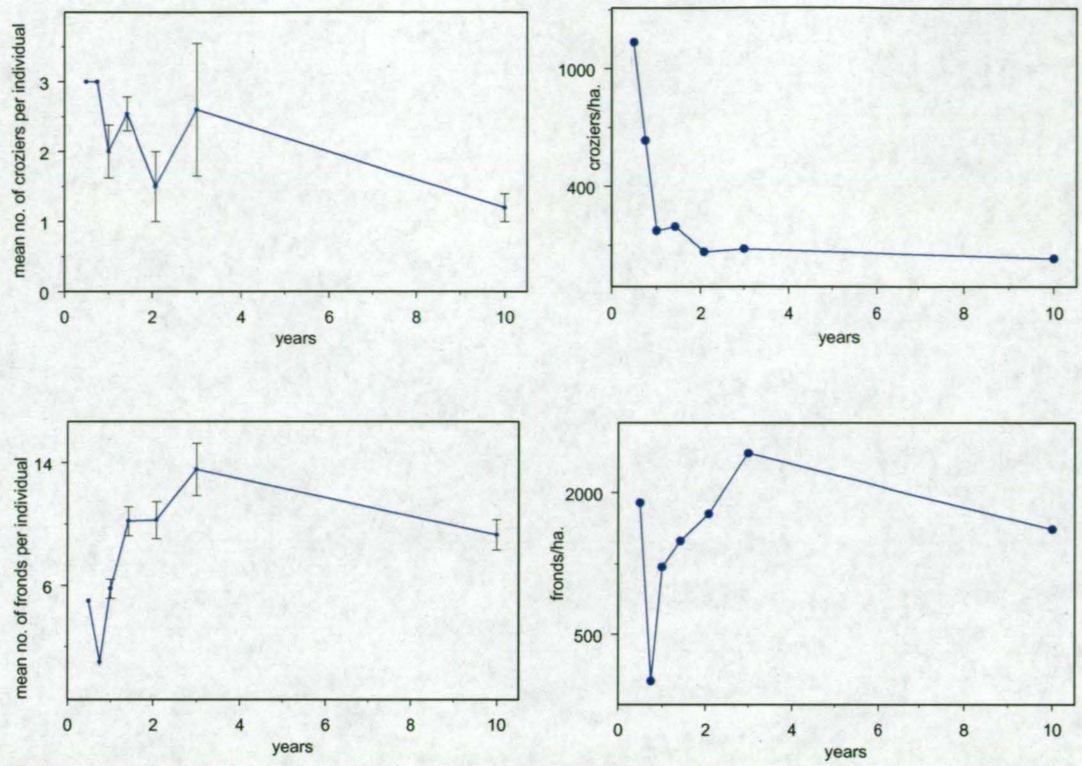


Figure 5.23 *Regenerative vigour of D. antarctica croziers and expanded fronds after logging and slash burning at Wayatinah. Refer to Table 5.6 for an explanation of the sampling times.*

All *D. antarctica* stems surviving three years after logging and burning had expanded fronds. The prostrate specimens (Figure 5.23) had significantly fewer (students t-test p=0.014) expanded fronds (mean=11, se=±1.1) present than the upright specimens (mean=20, se=±4.3). As an indicator of regenerative vigour and health, the mean number of croziers present per specimen and per hectare declined over the ten year period although short term increases in crozier production occurred following logging and burning (Figure 5.23), Much of the decline in the number of croziers per hectare was due to the corresponding reduction in live individuals, especially in the first six months after logging (Figure 5.23). Towards the later part of the monitoring period, most of the living specimens were classified as being of good to moderate health (Table 5.21), although they lacked the growth vigour of croziers and fronds evident immediately following logging and burning.

Table 5.21 *The health of living Dicksonia antarctica 10 years following cable logging and burning.*

	good	moderate	poor
% (n=29)	34.5	48.3	17.2

The number of expanded fronds per individual and per hectare also varies over the observation period, with a gradual decline from three years evident (Figure 5.23). The regenerative response of the monitored individuals was variable, with some specimens failing to exhibit signs of vegetative recovery until 25-36 months after cable logging.

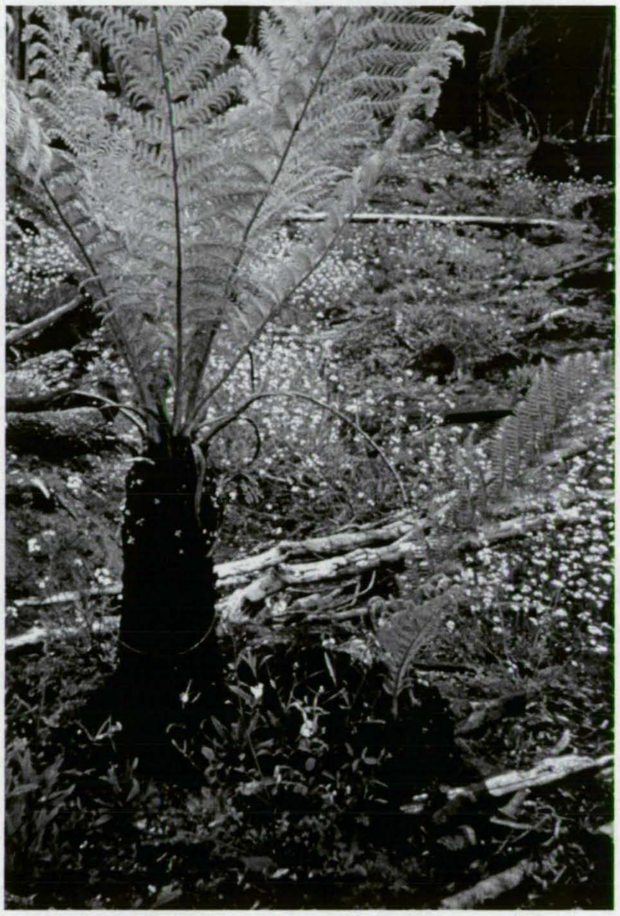


Figure 5.24 *Prolific frond growth of* D. antarctica *at Wayatinah 12 months after burning (November 1993, photographer F. Duncan). Note the reduced vigour of the prostrate individual on the right.*

The vigour of the surviving individuals measured from the increase in caudex height over the ten year monitoring period was approximately equal to the height increase in the control treatment (Table 5.22).

Table 5.22 *The height growth (cm/year) of* D. antarctica *10 years following cable logging and burning and in undisturbed controls. The mean values are not significantly different, (two sample un-paired t test, 95% confidence interval t = -0.1308, df = 45,* p>0.05*).*

	mean height growth/year (cm)	s.e.	minimum	maximum
treatment	2.65	±0.294	0.5	6.0
control	2.75	±0.070	0	14.0

Experiment 4: Impacts of cable logging on *D. antarctica* within streamside reserves

Cable logging resulted in a range of direct disturbance to *D. antarctica* populations, including logging of the streamside habitat where individuals were dislodged, broken, and stripped of fronds. Within the retained and control treatments the proportion of stems in the various orientation categories (alive upright, alive prostrate, dead upright, dead prostrate or new recruits) differed significantly (Table 5.23, Figure 5.25). In the gap treatment however, these density of individual *D. antarctica* in each of the orientation categories did not differ significantly, an indication of the degree of disturbance change in the most severe logging treatment. Across treatments, only the density of alive prostrate and alive upright stems differed (Table 5.24), they were significantly reduced in the logging gap treatment.

Table 5.23 *A one-way ANOVA of the counts of* D. antarctica *in the orientation categories within treatment types (logging gap, streamside reserve, control)*

treatment	df	sums of squares	mean square	F value	probability of F
streamside reserve	4	136620.0	34155.0	21.62	<0.001
residuals	25	39487.5	1579.5		
control	4	207250.9	51812.7	20.62	<0.001
residuals	25	62816.2	2512.6		
gap	4	52413.7	13103.4	0.75	0.5649
residuals	25	434531.2	17381.2		

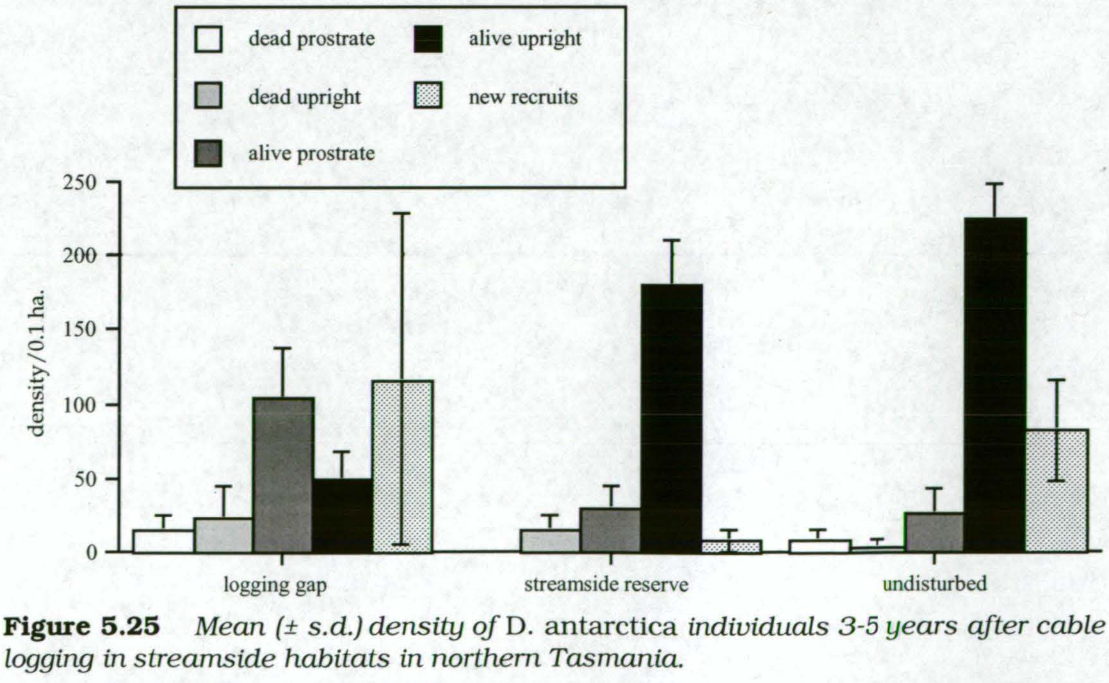


Figure 5.25 *Mean (± s.d.) density of* D. antarctica *individuals 3-5 years after cable logging in streamside habitats in northern Tasmania.*

Logging gaps also had an increased proportion of dead prostrate and dead upright individuals (Figure 5.25).

Table 5.24 *A one-way ANOVA of the counts of* D. antarctica *in the orientation categories across treatment types (logging gap, streamside reserve, control)*

	df	sums of squares	mean square	F value	probability of F
dead prostrate	2	675.0	337.5	1.15	0.3418
residuals	15	4387.5	292.5		
dead upright	2	1068.7	534.3	0.43	0.6534
residuals	15	18309.3	1220.6		
alive prostrate	2	23681.2	11840.6	3.83	0.0451
residuals	15	46321.8	3088.1		
alive upright	2	101074.7	50537.3	13.54	0.0004
residuals	15	55981.8	3732.1		
new recruits	2	37181.2	18590.6	0.67	0.5229
residuals	15	411834.3	27455.6		

Resprouting of prostrate or dislodged individuals was common, including those dislodged onto rock shelves and stream beds or where the streamside vegetation canopy had been logged or burnt (Figure 5.26). The occurrence of new recruits in the three disturbance categories was variable, they were more abundant in the logging gaps however the differences between treatments was not significant (Table 5.24).

Figure 5.26 *Resprouting of* D. antarctica *at Roses Tier 137g within the severely disturbed logging gap section of the creek two years after cable logging was completed.*

5.4 Discussion

The immediate effect of cable logging on *D. antarctica* was defoliation, caudex damage and uprooting. Approximately equal proportions of caudex's were recorded as prostrate as were upright, and the majority were still attached to the soil surface after cable logging. Taller individuals (mean ht of 1.7 m) were more likely to be flattened by logging than shorter individuals (mean ht of 1.1 m). A total of 73% of *D. antarctica* individuals in the Wayatinah experimental area survived cable logging. Most of these were recovering vegetatively within three months of logging, although some took more than six months to exhibit any signs of recovery. Rapid crozier and frond expansion from the protected terminal bud of *D. antarctica* was the immediate response to cable logging damage by surviving individuals. Ash (1987) also observed that tree fern individuals surviving logging exhibit increased growth, however at Wayatinah this initial growth was lost during the slash burn three months later. The proportion recovering from cable logging was approximately equal for upright or prostrate specimens, and those attached or dislodged from the soil surface. Severe physical damaged (usually at the apical bud) to the *D. antarctica* caudex from cable logging event ussually led to the death of the plant. Cable logging also resulted in some movement of individuals within the logging area, however the distances commonly in the order of only 1-2 m.

Slash burning three months following cable logging imposed an additional and more significant disturbance on *D. antarctica* populations. All *D. antarctica* stems were burnt and the initial rapid flush of crozier and frond development after the cable logging was immediate lost in the fire. Mortality corresponded to the severity of the burn not to the volume of caudex burnt. The circumference's of the caudex were reduced by up to 80% with the proportional reduction greater for individuals with large circumference's. The caudex, while being composed of a highly aerated and decomposition resistant material (Prasad and Fietje 1989) is considered the primary mechanism for providing thermal insulation to the apical meristem. The species regenerative strategy relied upon renewal buds hidden between the old stipe bases (Kornas 1985). Specimens with skirts of dry fronds were observed to rapidly combust during the slash burn at Wayatinah, similar to the observed ignition sequence in Chesterfield's (1996) study. Specimens which suffered deep burning damage into their caudex, usually when buried beneath a smouldering log, were most likely to die. Considering the intensity of the logging residue burning, the *Dicksonia* caudex appeared relatively resistant to complete combustion. Logs burnt completely to ash had only partially combusted *D. antarctica* specimens next to them. Mortality after burning was most pronounced for particularly tall and small individuals, either <1m or >2.75m tall. Half the short individuals survived logging and were observed regenerating vegetatively, but none survived the combined effects of logging and burning. Despite Neyland's (1986) suggestion that regeneration burns unbalanced *D. antarctica* caudexes, only a small proportion of individuals were burnt sufficiently to become unbalanced immediately following burning. Neyland's further suggestion that this unbalancing can leading to further mortality could not be tested statistically because of the small number of individuals in this group. It was clear however that a large proportion of prostrate stems at ten years had fallen between three and ten years following disturbance, and that they were already dead before becoming prostrate. Observations such as Neyland's above, where based on an examination of previously disturbed forest stands, may have mis-interpretated the actual disturbance mechanisms. *D. antarctica* individuals unbalanced by burning in the current study were smaller than the mean height in the population, but not significantly so. Post-fire studies of a genus (*Xanthorrhoea*) with a similar gross morphology consisting of a thick caudex and a crown of foliage at their apex also concluded that the youngest and oldest (from size classes, not direct ages) individuals exhibited the greatest post-fire mortality (Curtis 1998, Lamont *et al.* 2000). Mechanisms alluded to in those studies included the caudex of taller individuals breaking after burning, or logs smouldering at high temperatures on the caudex and burning through the persistent leaf bases. Similar patterns of fire damage were observed in this study, with fallen logs burning through *D. antarctica* caudexes and leading to death. In *Xanthorrhoea* mortality following burning is greater where high woody fuel volumes exist in the vicinity of the population (David Keith, NSW NPWS pers. comm. 2000). In *D. antarctica* at Wayatinah, the effect was much more localised to individual stems.

Burning left many individuals prostrate on the ash bed, dislodged, defoliated and without their mantle of roots. While *D. antarctica* will rapidly develop new adventitious root system from the stem base after severing (Hunt 1993), and anecdotal evidence suggests that *D. antarctica* will survive and grow for many years in the absence of a soil substrate or moisture absorption by a root ball (Hunt 1993), none of the fallen or otherwise dislodged individuals at the Wayatinah study area were observed to develop a new adventitious root system three years after logging and burning. Without a functional soil interface, moisture uptake, nutrient absorption and mycorrhizal activity (e.g. Gemma *et al.* 1992) are all potentially restricting the recovery response. Frond production in fern's following defoliation is particularly responsive to water and nutrient inputs (Milton 1987), and these may be critical for the regenerative response of *D. antarctica* following logging and burning. Additionally, the reported slow growth rates (e.g. Neyland 1986) and the physiological characteristics outlined previously may make it inherently susceptible to logging and associated microclimatic changes (e.g. Ashton 1975, Adams, *et al.* 1991).

The limited published data on the response of tree ferns to wildfire only suggests survivorship is either 100% (Chesterfield 1996), as high as 85% (Horn 1988) or varies with fire intensity (Ough and Murphy 1999). If *D. antarctica* individuals can live for centuries (Mueck *et al.* 1996), it is reasonable to assume that *D. antarctica* is tolerant or repeated firing, and of occasional fires of high intensity. Chesterfield (1996) argued that *D. antarctica* in mixed forest with diameters in excess of 50cm would have survived two wildfires, Cremer and Mount (1965), also working in mixed forest, argued that *D. antarctica* will survive regular wildfire. Despite the fact that Cremer and Mount (1965) presented no data, they are widely cited as evidence of this phenomenon.

Three months after burning survivorship was 43%, compared with 73% after cable logging. It continued to decline, reaching a plateau of 29% at 25 months and 27% at ten years. Twenty-nine percent survivorship of *D. antarctica* after three years is similar the survivorship reported by Harris (1986) 1-3 years after logging but greater than the 16.3% survivorship reported by Ough and Murphy (1997) 2.5 years after-logging. At Wayatinah, most of the mortality is attributed to the burning, not the cable logging, although the experiment conducted cannot separate the two effects. The environment of the ash bed (e.g. Chambers and Attiwill 1994) is considered hostile to individual regenerating *D. antarctica*, as low nocturnal temperatures (Warrington and Stanley 1987) and high day shoot and crozier temperatures (Doley 1983) are known to be lethal to expanding tree-fern fronds. Further re-assessments beyond ten years are likely to indicate on-going mortality due to the general lack of vigour in the surviving individuals. The population recovery is likely to be dependent on new recruits. *D. antarctica* increased in cover and frequency with age in the first twenty-five years after ground based logging in another chronosequence study using similar sampling methods (Figure 5.27) although in common with this Tasmanian study the temporal trends were particularly variable. Results from Victoria (Wang 1993, Cook and Drinnan 1984, Harris 1986, Ough and Ross 1992, Mueck and Peacock 1992, Peacock 1995) and Tasmania (Tanjung 1993) during the first decades after ground based logging all described increasing cover with time since logging and burning, although these studies again also reported a large degree of temporal variability.

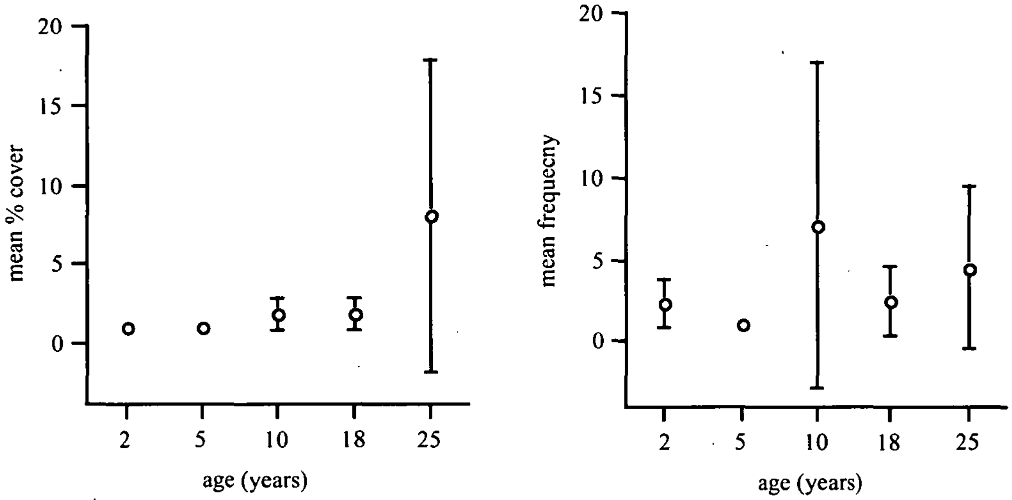


Figure 5.27 *Mean (±s.e.) percentage foliage cover and mean (±s.e.) frequency (score of 0-20) of* D. antarctica *in a chronosequence of ground-based logged wet forest sites in East Gippsland, Victoria. In each age category, there were three spatially replicated sites (data from Peacock unpubl.).*

There appears to be no other data reporting the response of *D. antarctica* to cable or ground based logging only, all of the published studies were concerned with survival following ground-based logging, mechanical site preparation and slash burning. The results from this study at Wayatinah therefore provide a new perspective on the interpretation of all of the previous studies, as the 73% survivorship after logging is twice the amount reported by other studies (16-38%) after ground-based logging and burning (Chesterfield 1996, Ough and Murphy 1997, Harris 1989). While those studies (e.g. Ough and Murphy 1997) attributed mortality in *D. antarctica* to the effects of mechanical disturbance during ground based logging, their conclusions may need to be re-evaluated as an incremental effect of logging and burning, not logging alone. It is also possible that the high reported mortality rates are due in part to sampling error in those studies, as they frequently relied on very limited data sets with no permanently marked individual tree ferns and at times cursory observations.

Differing sampling procedures adopted during studies of *D. antarctica* populations after logging and burning makes comparisons with the current study difficult. For example in the Ough and Murphy (1997) study it appears that the large number (half) of individual ferns in their population samples that could not be located after logging and burning may have masked much of the trends in mortality. Data on trunk orientation after logging was only provided for 18% of their sample, and no new recruitment were recorded either pre or after logging. In a similar forest type in eastern Victoria for example, Chesterfield (1996) recorded new recruitment in the vicinity of 13 000/ha nine years after logging and burning. The low survival (16.3%) of *D. antarctica* 2.5 years after logging in the Ough and Murphy (1997) study compares uncomfortably with Chesterfields 38% survival in a similar forest type and an identical logging method, and the 29% from the cable logged study area at Wayatinah. These apparently conflicting results are difficult to interpret across studies with widely varying sampling procedures.

The rate of recovery *of D. antarctica* cover following either cable or ground based logging did not differ significantly in the chronosequence analysis. They did differ however in the density of surviving individuals in the various caudex orientation categories. In cable logged sites most *D. antarctica* stems survived as upright stems while in ground-based sites they tended to survive at prostrate stems. The greater density of dead prostrate stems in sites subject to ground based logging implied a greater degree of site disturbance and mortality to individual stems, particularly in the 11-30 year old forest comparison. There were also fewer dead uprights stems in ground-based regrowth, potentially a result of the more intensive ground disturbance associated with ground-based logging methods. Barker (1988) interpreted the same trend in mortality in terms of variation in wood utilisation standards (and by inference the intensity of logging traffic) over the past three decades on ground-based logged coupes.

An indirect consequence of cable or ground-based logging is the size of the logging coupe and its potential effect on constraining the recruitment of new *D. antarctica*. While the size of the logging coupe is not a direct consequence of the logging method, in practice in southern Tasmania and elsewhere cable logging areas have a tendency to be larger than ground-based logged areas, due to the scale at which the two logging systems operate at greatest efficiency. Spore dispersal is limited in fern species (Conant 1978), declining markedly with increasing distance from the source population (Penrod and McCormick 1996). Spore viability in the laboratory for tree ferns is usually limited to 2-4 years (Page 1979, Felippe *et al.* 1985, Simabukuro *et al.* 1998), and in *D. antarctica* 2-3 years (Barker 1988), however, field viability of soil stored spores may only be one year (Penrod and McCormick 1996). Cable logged coupes, due to their greater size, and therefore the increased distance to extant populations of reproductively mature *D. antarctica*, may further limit the recruitment of new *D. antarctica* sporophytes into the regrowth vegetation.

New recruits of *D. antarctica* were more abundant within regrowth of the same age originating from ground-based logging than cable logging, suggesting either a greater suitability and availability of substrates for recruitment, more suitable micro-environmental conditions, or potentially a greater number of available propagules. Where logging disturbance involved ground-based methods, *D. antarctica* new recruits appeared well adapted to colonizing heaped soil, often along the margins of snig tracks in sites with soil heaping. Soil disturbance from superb lyrebird activity in *E. regnans* forest at Beenak, Victoria was observed to favour *Cyathea australis* recruitment where litter free soil mounds were available (Ashton and Bassett 1997). Studies elsewhere of tree fern recruitment in Hawaii (Medeiros *et al* 1992), Taiwan (Huang *et al.* 1996) and Nepal (Gurung 1992) also describe scenarios where preferential recruitment occurs on areas of disturbed soil. There are numerous other examples from other environments such as landslides (Myster and Fernandez 1995), soil slumps (Ferwerda *et al.* 1981), barren afforestation areas (Gurung 1992) and exotic pine plantations (Prasad and Fietje 1989) where tree-ferns have demonstrated their colonizing ability following soil disturbance.

Seedling-scale (or in this case prothalli, sensu Hartgerink and Bazzaz 1984) environmental heterogeneity in patterns of soil and debris microtopography induced by logging appears to be an important determinant of the regeneration strategy of *D. antarctica.* Where disturbed soil is not available, for example where dense patches of ground fern exist, tree fern recruitment will instead occur on rotten logs and upturned root balls (Ashton and Bassett 1997).

Several authors have proposed that the ability of *D. antarctica* to recolonise regrowth forest after logging and burning is limited by the availability of suitable substrates (Neyland 1986), including damp woody material (Barker 1991) and other sufficiently moist germination surfaces for the gametophyte-sporophyte life cycle to occur. The observed patterns of recruitment in this study were strongly associated with the substrate preferences of the prothallii. New individuals were recruited mainly on fallen decomposing logs irrespective of logging method in this study, a trend observed elsewhere (20% in the study by Medeiros *et al* 1992), or at least on safe sites (i.e. Fowler 1988) such as windrows (Harris 1986) and the caudexes of fallen *Cyathea* and *Dicksonia* in the East Gippsland study (author pers. obs. from Mueck and Peacock 1992). Fallen logs and *D. antarctica* caudex's were also the preferred substrate for germination in mature forest and the logged forest. Tree fern fibre is a preferred medium for spore germination, prothallii formation and sporophyte growth in tree ferns (Borelli *et al.* 1990), offering high levels of moisture retention and aeration (Dematte and Dematte 1996). While tree fern trunks are relatively nutrient poor (Prasad and Fietje 1989), fern prothalli are extremely responsive to charcoal in the environment (Teng 1997) and there was an abundance of ash and charcoal after burning.

Recruitment of prothalli in cable logged forest was continuous with the recruitment rate increasing with stand age. In the chronosequence analysis, new recruits were uncommon in the first two decades following cable logging. None were observed in the ten year study of tagged individuals, although a small number were observed in the 14 month old permanent plots where new recruits colonised the short burnt *Polystichum* rhizomatous trunks. New recruits however were consistently more abundant in the oldest regrowth sampled compared to mature or unlogged forest. Sporophytes become reproductively mature at ages of 12-23 years (Neyland 1986, Barker 1988) and may therefore also be contributing to spore production and gametophytic production in the later stages of regrowth forest studied.

Broadly similar trends exist for *D. antarctica* in New South Wales subject to ground-based logging. Forests subject to ground-based logging, where soil heaping is common during log snigging, have an increased portion of small (and presumably younger) individuals present, while unlogged mature forests have a greater proportion of larger individuals present (Figure 5.28).

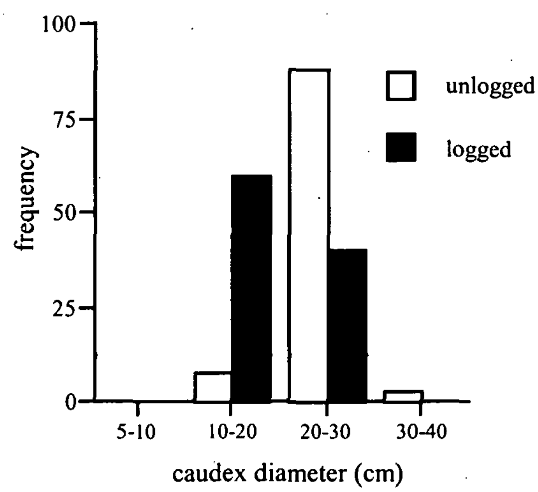


Figure 5.28 *Percent frequency of* D. antarctica *individuals in various size classes within conventionally logged and mature moist forests of north eastern New South Wales (unpublished data from State Forests of NSW).*

Data on the response of tree ferns to physical disturbance, for example hurricanes (Bellingham *et al.* 1995) and tree-falls (Tanner 1983) also suggests they are highly susceptible to damage and exhibit low recoverability. Other natural causes of mortality in tree ferns include physical damage and uprooting from falling branches, disease and senescence (Ash 1986, 1987), self-thinning (Campbell 1990) and smothering by tree epiphytes established on their caudexes (Smale *et al.* 1992). While taller individuals were more likely to be flattened than smaller ones during cable logging, they did not appear to be any more inherently unstable even on the steep slopes studied. They may have simply been more exposed to damage. The consequences of being knocked down during mechanical disturbance or falling over in natural events may include reductions in vigour, frond number, stem diameter, stem elongation and possibly fecundity (Ash 1986). The Wayatinah study confirmed the first of these trends, prostrate individuals had fewer expanded fronds than resprouting upright stems. The interpretation of the causes of physical damage to *D. antarctica* populations from logging is complex, and there appeared to be no single cause of the immediate after logging mortality.

After logging and burning regenerative vigour of *D. antarctica* varied, the number of croziers declined with time and many individuals which had an immediate vegetative response to logging and burning disturbance were observed to be in poor health with new fronds failing to fully expand, or the production of new croziers declining over the three year period. Prostrate individuals exhibited a reduced regenerative response compared to upright individuals, presumably as a result of the disruption to their root mantle, although being much closer to the soil surface may have exposed the croziers to lethal temperatures in summer (Doley 1983) and a greater incidence of frost damage (Warrington and Stanley 1987) in winter. Some *D. antarctica* simply appeared to exhaust their alibility to resprout following burning, after expending their initial effort resprouting following logging and having that regenerative effort lost through burning. Studies on crozier production indicate that frond production can be reduced for a period of up to four years following defoliation (Bergeron and Lapointe 2001). With *D. antarctica* recovery from crown defoliation and then firing beneath a rapidly closing shrubby understorey, the effects of crown defoliation and the species ability to recover will be limited in the shaded environment (Bergeron and Lapointe 2001). The rapid closure of the regenerating understorey at Wayatinah is likely to impose heavy shade over the regenerating *D. antarctica* for up to 15 years. Shading can also limit spore production (Nicholson 1997), growth (Arens and Baracaldo 2000, Durand and Goldstrein 2001, Unwin and Hunt 1996) and recruitment (Arens 2001) in tree ferns. Shade induced etiolation leading to a gradual loss of photosythetically active tissue and reproductive vigour was a widespread phenomenon in regrowth forest up to 30 years old irrespective of forest type or logging method, a phenomenon observed elsewhere in Tasmania (Gilbert 1959, Neyland 1986 and Barker 1988). All of these antagonisms to the species vegetative recovery, and the observation that only approximately one-fifth of the cover characteristic of mature forests was observed in the sampled regrowth up to 50 years old (irrespective of logging method), suggest the species may not reach comparable abundance levels prior to the next harvest event. Mueck and Peacock (1994) similarly found in their East Gippsland chronosequence that the probability of occurrence of *D. antarctica* in regrowth forest aged 25 years was significant lower (Figure 5.29) than in mature forest and they also predicted pre-disturbance abundance levels would not be attained prior to the next harvest event.

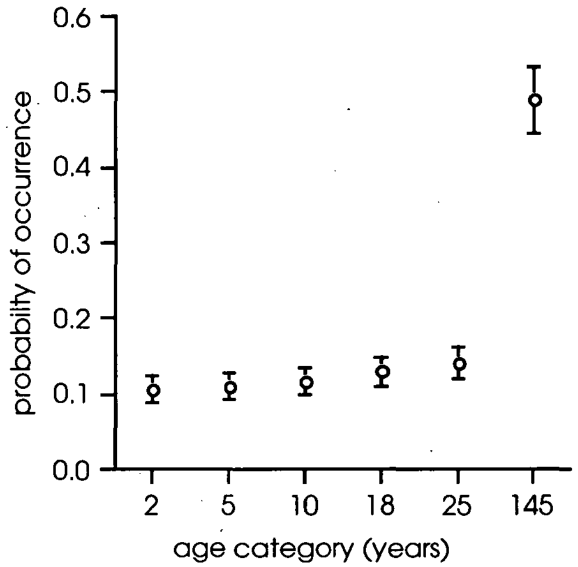


Figure 5.29 *Generalised linear model of the probability of occurrence of* D. antarctica *in wet sclerophyll forest in East Gippsland using stand age as a predictor categorical variable. (data from Mueck and Peacock 1994).*

In addition to the lack of sustained regenerative vigour in *D. antarctica*, it was also a relatively slow species to recruit into the regrowth forest. The chronosequence analyses in areas subject to either cable or ground based logging indicated recruitment was low and minimal in the first decade. Peacock and Doley (1998) found it to be the slowest of the four tree fern species to recruit onto disturbed soil following sub-tropical logging in their study (Figure 5.30), not appearing in permanent plots until 7 years after disturbance.

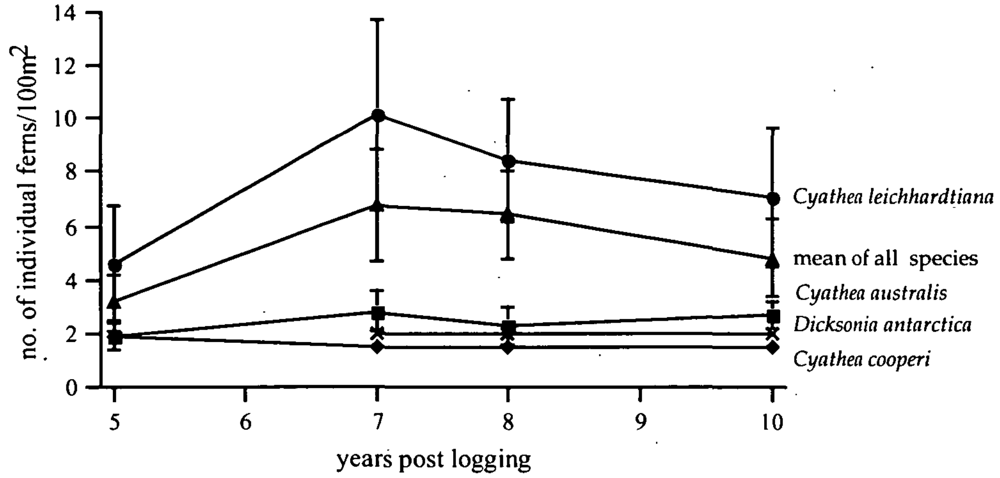


Figure 5.30 *Recruitment patterns of four species of tree-fern on permanently monitoring log dumps in sub-tropical rainforest at North Washpool, north eastern New South Wales (from Peacock and Doley 1998 unpubl.).*

The number of living arborescent *D. antarctica* individuals in the three chronosequences was uniformly low compared to their numbers in unlogged forest. This was also the common conclusion of the previous studies of the response of *D. antarctica* to logging (Ough and Ross 1992, Hickey and Savva 1992, Ough and Murphy 1997). Several authors suggested that disturbance (i.e. logging) was of greater significance in explaining the mortality of the *D. antarctica* following logging than the slash burn, citing as evidence observations that *D. antarctica* was relatively common in adjacent unlogged 50 year old wildfire regrowth forests (Ough and Ross 1992) or was more abundant in small retained areas of understorey habitat within the logging area (Ough and Murphy 1997). While physical disturbance is obviously important, especially where it destroys the caudex, it may only be a partial explanation, the intensity and nature of the disturbance may also be relevant, as several of the Victorian coupes in the studies by Ough had their slash windrowed prior to burning, and none were cable logged. The results from the Wayatinah experiment suggest that the combined effects of burning and logging were of greater significance than logging alone in predicting *D. antarctica* mortality, however the experiment was unable to examine the interaction of logging and burning treatments as all of the logged plants were burnt, the routine silvicultural practice in wet sclerophyll forests.

D. antarctica is widespread and abundant in the tall open forest (Ashton and Attiwill 1994) environments of south eastern Australia, which are subject to irregular but generally catastrophic disturbances such as wildfire (McCarthy *et al.* 1999) at mean intervals of 75-150 years for tree-killing fires and 37-75 years for less intense fires. *Dicksonia antarctica* within production forests have the additional disturbances of logging imposed at intervals of 50-80 years for stands managed for sawlog production, or every 20 years where thinning is also conducted. *Dicksonia antarctica* persists following both wildfire and logging and slash burning. However, logging and slash burning imposes more frequent and often more intense injury to *D. antarctica* than wildfire. This raises the question of the long term survival and ecological contribution of *D. antarctica* in wet forests managed for wood production.

The response of *D. antarctica* populations within streamside reserves to direct and indirect cable logging disturbance differed from their responses described above within clearfelled areas. *D. antarctica* individuals were disturbed directly during log felling and yarding and by subsequent branch and treefalls in streamside reserves. In some cases the residual canopy cover of the streamside reserve was subject to on-going tree health decline, storm damage and fire damage. Provided that the disturbance to the individual *D. antarctica* was not so severe as to destroy the terminal bud, the individual was more likely to recover vegetatively within the streamside reserve, with its abundant moisture, than in the clearfelled areas subject to a less favourable micro-climate.

The range of cable logging treatments which pulled logs across, through or on the margins of retained streamside reserve vegetation effected the survival, orientation and recruitment patterns of individuals, but the treatment effect was only statistically significant for the difference in the number of living upright individuals between the treatments. Gaps through the retained vegetation resulted in the greatest mortality, as measured by the number of fallen or upright dead individuals. Individual tree-ferns were also dislodged from stream banks and rock shelves into creek beds, and soil movement resulting from the logging event may also have contributed to their destabilisation in manner similar to the natural process described by Ash (1986). Most *D. antarctica* responded to the logging or tree falling disturbance by resprouting from their apical meristems, irrespective of whether they were upright, fallen or dislodged. The relatively high survival of *D. antarctica* in streamside areas following logging disturbance is attributed to the ability of the stem to remain moist in the streamside environment even when it has been dislodged. The stem of *D. antarctica* is considered the primary source of moisture for transpiration (Hunt 1993), and adequate moisture is likely to be available in the streamside environment. *D. antarctica* tolerates a wide range of light intensities and is unsuccessful only in deep shade or full sunlight (Hunt 1993). The broken canopy resulting from log yarding would also therefore have provided an environment conducive to vegetative regeneration of the damaged individuals.

Observations from this study suggest the density of mature *D. antarctica* stems in mature wet forests is approximately 600-1500 stems/hectare and this is reduced immediately ten years after logging and burning to approximately 180 stems/hectare. Other published estimates in similar forest types in south eastern Australia (1368/ha in Chesterfield 1996, 1860/ha in Cook and Drinnan 1984, 30-910/ha in Mesibov 1998) and also in New Zealand (2000-2500/ha in Ogden *et al.* 1997, 341/ha in Smale and Kimberley 1993) suggest the Wayatinah population was in the lower range of reported stem densities. The chronosequence analyses of cable logged areas up to ten years old suggest that the density of surviving stems is likely to be in the range of 33-200 stems/hectare. With survival following cable logging and slash burning of 27%, and new recruits becoming reproductively mature at approximately 23 years (Page 1979), the major phase of population re-establishment is likely to be in the early to middle stages of forest development where light (Smale *et al.* 1992, 1997, Ogden *et al.* 1997) and substrate availability is conducive to further recruitment. In forests subject to ongoing thinning or logging events during these early to middle stages of forest development, *D. antarctica's* recovery and recruitment processes are likely to be further disrupted, especially where the population subject to logging is still in the process of development towards optimal site stocking. In populations with greater initial densities of *D. antarctica*, the impacts of logging and burning may be less severe, however in the study area cable logging followed by slash burning resulted in the total loss of the proportion of the population that is less than 30 years old. The 20-30 year stage is the approximately point at which some tree ferns switch their development process from a phase where height growth of the caudex is limited, to a phase where growth is most noticeable in caudex height, not frond diameter (Prugnolle *et al.* 2000) and the space occupied by each tree fern is stable. Gaps in the understorey created by the mortality of individual *D. antarctica* may not be recolonised if the spatial occupation model of Prugnolle for *Cyathea muricata* applies at Wayatinah. A long term population decline in production forests is a probable scenario.

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Chapter 6 Effects of cable and ground-based logging on vascular epiphytes

6.1 Introduction

Vascular epiphytes are non-parasitic plants (with a vascular system) that use another plant as mechanical support but do not derive nutrients or water from their host (Nadkarni *et al.* 2000). Vascular epiphytes can be considered either obligate (a plant that always grows on another plant for structural support), or facultative species (a plant that commonly grows on another plant for support but is not restricted to doing so). Further distinctions can also be made for both accidental epiphytes, a typically terrestrial species which occasionally grows to maturity as an epiphyte, and hemi-epiphytes, a plant that maintains a vascular connection with the soil over part of its life (for further classifications see Benzing 1990, 1995).

Vascular epiphytes have a range of characteristics that make them both inherently susceptible to habitat disturbance and useful indicators of patterns (and processes) of secondary succession following disturbance. Epiphytes tend to have non-random or clumped distribution patterns in forests (Hietz 1997), potentially increasing the susceptibility of individual populations to habitat disturbance. Their preference for shaded and humid environments (Leimbeck and Balslev 2001), and the lower portions of trunks (Hietz and Hietz-Seifert 1995) also disposes them to changes in microclimate resulting from forest disturbance (da Costa 1999). In extreme instances, as in the case of filmy ferns, they can desiccate completely within hours of being exposed to relatively dry air (Hietz and Briones 1998). Vascular epiphytes' degree of micro-habitat specialisation, and their reliance on their micro-habitats for support further ties their post-logging fate to that of their hosts or substrates. The substrates used include plant material (e.g. branches, bark, twigs, cut stumps) and abiotic surfaces. The related term in the epiphytic literature phorophyte ('a support for vascular epiphytes', Benzing 1990) is not used here.

Vascular epiphytic species are highly responsive to perturbations such as wildfire (Adams and Lawson 1984, Adams 1986, McMahon 1987, Ashton and Frankenberg 1976, Chesterfield *et al.* 1990, Barker 1991, Hickey 1993, 1994, Chesterfield 1996), forest fragmentation (Turner 1994), flooding (Leimbeck and Balslev 2001), hurricanes (Oberbauer *et al.* 1996), land clearance (Porembski and Biedinger 2001, Hietz-Seifert *et al.* 1996) and logging (King and Chapman 1983, Hickey and Savva 1992, Mueck and Peacock 1992, Harris 1989 and Ough and Ross 1992). Following logging, species richness, of both generalist and specialist epiphytic fern species are reduced (Binns 1992), with the reduction in richness being proportional to the intensity of logging (Gardette 1996), although compared to trees and shrubs, fewer studies are available on logging impacts (Horne and Hickey 1991). In vascular epiphytes, this reduced diversity is most noticeable in fern and orchid species (Barthlott *et al.* 2001).

In the temperate environments of south-eastern Australia, epiphytic fern species are infrequent or absent in regenerating vegetation in the first three decades following ground-based logging (Cook and Drinnan 1984, Harris 1986, Mueck and Peacock 1992, Ough and Ross 1992, Hickey and Savva 1992). Fallen logs and species such as *Dicksonia antartica* are the preferred substrates for epiphytic re-establishment post-logging and relatively moisture stress tolerant epiphytic fern species such as *Grammitis billardieri* and *Microsorum pustulatum* are the first to re-establish post-logging. Logging disturbance will alter the community organisation patterns of epiphytes, disrupting their vertical patterning in host tree colonisation (Gardette 1996), and in cases of partial or severe habitat disturbance, remnant trees become the primary substrates (Porembski and Biedinger 2001, King and Chapman 1983) for the persisting or recolonising populations.

It is proposed that patterns of re-colonisation of vascular epiphytes following either cable or ground-based logging are dependent on the logging method and how it modifies their specialised microhabitats and substrates. The post-logging environment can limit the ability of specialist species to recolonise where the pre-requisite microhabitats are rare (e.g. Meier *et al.* 1995). The response of vascular epiphytes to ground-based logging differs from their response to wildfire (Hickey 1994), and these types of perturbation vary principally in the nature of the biological and structural legacies they produce (Franklin *et al.* 2002, Lindenmayer *et al.* 1990, 1999, Lindenmayer and McCarthy 2002). Large fallen logs and decayed wood, a primary substrate for epiphytic species, tend to be sparse (Andersson and Hytteborn 1991), and structural diversity (Lesica *et al.* 1991) is reduced following logging. Logging will fragment their micro-environments and temporary habitats, increasing post-logging recolonisation distance (Soderstrom and Jonsson 1992). Specialised substrates, varying in roughness, diameter, inclination, volume and decay stage are also sparse outside mature vegetation, limiting the re-colonisation (Ackerman *et al.* 1996) and growth of epiphytic species (Esseen *et al.* 1996, Andersson and Hyttebom 1991). While much of the literature on epiphytic succession is sourced from tropical environments where vascular epiphytism reaches its peak (Benzing 1990, 1995), temperate environments, the subject of this thesis, are less frequently studied. No general treatment of the composition and distribution of vascular epiphytes is available in Australia, apart from the taxonomic and lifeform ecological overview by Wallace (1981, 1989). Further, the response of the Australian vascular epiphytic flora to disturbance, in particular logging, is poorly documented. No published studies which primarily examine the response of vascular epiphytes to logging in Australia are available. The studies referenced above (King and Chapman 1983, Cook and Drinnan 1984, Harris 1989, Hickey and Savva 1992, Mueck and Peacock 1992, Ough and Ross 1992, Binns 1992, and Peacock 1995) all had a broader aim of describing floristic change following logging, with only a sparing amount of data presented for epiphytic species. Consequently much of the detail inherent in the tropical literature on vascular epiphytism, e.g. substrate preferences, vertical partitioning, community organisation, etc., is missing from the logging response because logging studies were not focussed on epiphytes. Much of the detail of the post-logging response of vascular epiphytes is therefore un-reported.

The lack of data on epiphytic recolonisation following logging, irrespective of method, is attributed to the absence of direct experimentation or observational studies using the appropriate scale of sampling. Of the limited studies available which describe the general vegetation response following cable logging, none report on vascular epiphytic species, as they are generally from environments lacking this life form (e.g. Woodward 1986), or if from the tropics (e.g. Sarre 1992), they do not report on epiphytes. This lack of a general description of epiphytic recolonisation patterns is not unique to the question of their response to logging, even where epiphytism is more pronounced, evidence for epiphytic succession is largely based on analyses of presumed seral stages rather than observed progressions (Benzing 1995). Further, no studies following a single species of substrate are known (Benzing 1990). Both of these limitations of previous studies of logging effects and epiphytic succession in general are addressed in this thesis through the use of complementary chronosequence and permanent quadrat methods, and through the use of repeated observations of the same substrate surfaces.

6.2 Methods

Four experiments examining the response of vascular epiphytes to cable and ground-based logging are presented in this chapter;

1. a chronosequence analysis of the response of vascular epiphytes to cable and ground-based logging in wet forest in southern and central Tasmania
2. a chronosequence analysis of the response of vascular epiphytes within wet forest streamside reserves to different methods and intensities cable logging disturbance in northern Tasmania.
3. an experimental examination of the impacts of cable logging on vascular epiphytic fern communities of wet and mixed forests of the upper Florentine Valley, Tasmania.
4. an experimental examination of the response of vascular epiphytes growing on *Dicksonia* caudexes to cable logging and burning in wet forests of the upper Florentine Valley, Tasmania.

Experiment 1: Chronosequence analysis comparing the response of vascular epiphytes to cable and ground-based logging.

The chronosequence outlined in Chapter 2 was the basis of this study. The overall study aim was to compare the response of vascular epiphytes in wet forests subject to either cable or ground-based logging.

Study areas

Two areas were chosen to carry out the chronosequence surveys, the Florentine/Styx/Tyenna Valleys in central southern Tasmania and the Arve-Blue Hill area in the Arve Valley Southern Forests (Figure 2.1). Both areas have comparable *E. regnans* dominated wet forest communities, the difference being the Florentine Valley forests studied were cable logged and those in the Southern Forests were logged with ground-based equipment.

Seventeen areas were sampled as part of the cable logged wet forest chronosequence (Table 2.3) and fourteen areas as part of the ground-based logged wet forest chronosequence (Table 2.4).

Experimental design

The experimental design (chronosequence) incorporated a regression analysis approach of the vegetation variables of interest (such as vascular epiphytic composition and richness) against stand age. Stand age was categorised as: 5-10, 11-20, 21-30, 31-40, 41+ years and mature forest, with the aim of sampling three independent stands within each category, and those sampled stands encompassing the range of ages within the category (e.g. 5, 7 and 10 years). This was achieved for all categories, except in the 41+ year old regrowth category where only two stands per forest type were sampled because of the relative scarcity of silvicultural regrowth of this age. In the Southern Forests chronosequence only the 5-10, 11-20 and 21-30 years mature forest categories were sampled due to the absence of environmentally similar regrowth aged 30+ years.

Environmental stratification units used in the chronosequence survey design are given in Chapter 2, section 2.2.

Sampling methods

The unit of vegetation sampling adopted was a square quadrat of 30 x 30m or $900m^2$. Each coupe was sampled once in an area considered to be representative of a relatively homogeneous, successfully regenerated part and at least 50m down or upslope from a road or track. Within the quadrat all substrates were surveyed for the presence of vascular epiphytes. Each epiphyte was assigned a visually assessed cover-abundance score from the Braun-Blanquet (Poore 1955) scale described in Chapter 2 experiment 1. Cover was defined as foliage projective cover following Walker and Tunstall (1981). Sampling was completed using the system of ten sub-plots (2 x 2m) spaced at 7.5m intervals along a transect either side of the quadrat's mid-point as described in Chapter 2 experiment 1 (Figure 2.6).

Vascular epiphytes were classified into three groups based on their life forms; ferns, shrubs or trees and herbs. Both obligate, facultative, accident or hemi-epiphytic species were recorded. When an epiphytic species was observed, the type of substrate it occupied was recorded. Biotic substrates were classified into categories such as cut stumps versus living tree buttresses, *Atherosperma* bark versus surface roots, living versus dead *Nothofagus*, and the species of plant upon which its bark was colonised was recorded. Abiotic substrates included rocks, mossy rocks and ledges (lithophytes are included here in the broad definition of epiphytes). Each vascular epiphyte was recorded separately for each unique epiphyte-substrate association (e.g. *Grammitis billardieri* on rocks or *Grammitis billardieri* on *Dicksonia* caudexes) using a relative abundance scale as follows: rare, moderate and abundant. This scale assesses the relative abundance of the epiphytic species by incorporating the degree to which the species occupies the available, suitable substrates (Neyland 1991). Using this recording system, as an example, *Grammitis billardieri* may be recorded as rare growing on rocks and abundant on *Dicksonia* caudexes in the same quadrat.

Analytical procedures

Regression analysis was used to examine the relationship of the dependent variables (Table 6.1) derived from the quadrat sampling of epiphytes and the independent variable stand age. Regression equations for the same dependent variables from the chronosequences sampling quadrats in either cable or ground-based logging treatment were compared to test whether the functional relationships described by the regression equations are the same. These analysis methods (ANCOVA) are described in Chapter 2 section 2.2. Three types of regression predictions were fitted to the data, power, exponential and least squares. The predictions for ground-based quadrats were extended beyond the data points to match the sampled ages for cable logged quadrats.

Table 6.1 *Derived vascular epiphyte variables.*

Variable	Description
cover of obligate epiphytic ferns	cumulative projective foliage cover of obligate epiphytic fern species, derived from the sum of individual mean % cover scores of each species.
obligate epiphytic fern richness	number of obligate epiphytic fern species per 30 x 30 m quadrat.
richness of epiphytic fern-substrate associations	number of different epiphyte-substrate associations, for obligate and facultative epiphytic ferns.
richness of epiphytic fern substrates	number of different substrates colonised by obligate and facultative epiphytic fern species.
richness of fern epiphytes	number of fern species observed growing as epiphytes, either obligate or facultative.

The linear relationships between the variables in Table 6.1 were examined using Pearson product-moment correlations (Snedecor and Cochran 1989) or linear regression. Where linear models are fitted to the data, correlation is described by the r^2 value, being the variance of the data explained by the model. Where Pearson product-moment correlations are used, r = the correlation coefficient. The relationship between epiphytic species occupancy of different substrates is examined using histograms summarising the number of obligate epiphytic ferns present on the various substrates in the age categories above. The relationship between the vascular epiphytic flora of mature forest and regrowth forest originating from either cable or ground-based logging is presented in a cross tabulation of species by substrates. Compositional trends in vascular epiphytic fern abundance on different substrates according to stand age are presented in a two-way table form, with the rows sorted alphabetically by species and the columns sorted by stand age. The fidelity of species to the logging treatments was tested using Indicator Species Analysis (Dufrene and Legendre 1997) with 10,000 randomised starts. Indicator values are defined by combining the values for relative abundance and relative frequency and expressing them as a percentage of perfect indication (see section 2.2).

Experiment 2: Impacts of cable logging and burning on the frequency and cover of vascular epiphytes in wet forests

Data on the frequency and cover of vascular epiphytes from the long term floristic monitoring of two steep country cable logging areas at Wayatinah (Blue 09 and Pearce 04) is presented. The data is restricted to those four permanent quadrats which have been re-measured following after logging and burning. The aim of the experiment is to provide a replicated estimate of the change in vascular epiphytes frequency of occurrence and foliage biomass with time following cable logging and burning that was directly comparable to the estimates derived from the chronosequence analyses (Experiment 1 in this chapter). Comparative data such as this permits the chronosequence analyses to be validated. The description of the study area, experimental design, and sampling methods followed are presented in further detail Chapter 2 as Experiment 2.

Study areas

The experiment was conducted at the Pearce 04 and Blue 09 logging units at Wayatinah in central Tasmania, and are approximately 5 km apart. Both Pearce 04 (22 ha) and Blue 09 (30 ha) consisted of a mosaic of *E. regnans* dominated wet (Kirkpatrick *et al.* 1988) and mixed (Gilbert 1959) forests. Vascular epiphytes are relatively common in the understorey on rocks, logs and shrub and tree substrates. Both Pearce and Blue have moderate to steep slopes and are underlain by Jurassic Dolerite. Pearce and Blue were cable logged using the same machinery in 1992, and burnt and regenerated in 1993 (Table 2.7). The environmental characteristics and treatments applied are detailed in Chapter 2. The location of the four quadrats sampled is given in Table 5.2.

Experimental design

The survey design involving pre and post treatment repeated sampling using fixed monitoring points detailed in Chapter 2. Two permanent monitoring points at Blue 09 (quadrats 2 and 24) and at Pearce 04 (quadrats 5 and 12), representing a sub-set of the pre-treatment quadrats, were re-measured post treatment. The detailed measurement schedule is given in Table 2.9 and 2.11. The pre-logging data was collected in March 1992, post-logging data May 1994. The quadrats were logged in 1992 and burnt in March (Pearce 04) and April 1993 (Blue 09), see Table 2.8.

Sampling methods

The sampling of epiphytic species and their substrates in the 0.09 ha quadrats follows the methods described for the chronosequence study earlier in this chapter.

Analytical procedures

Epiphyte cover was calculated using the methods in Chapter 2 Experiment 1, and expressed as a treatment mean cover score for the mid-point transformed cover-abundance scores. Pre and post cable logging cover scores for epiphytic species are presented in tabular form using the cover scores calculated from the mean of the twenty sub-plot observations (see Chapter 2). Pre and post-logging epiphyte-substrate abundance scores are presented in a two way table of epiphyte-substrate association by quadrat and measurement period.

Experiment 3: Response of the vascular epiphytic flora of *Dicksonia antarctica* caudexes to cable logging and slash burning

The aim of the experiment is to describe the nature of the epiphytic flora of *Dicksonia* caudexes prior to logging, describe the impacts of cable logging and the subsequent regeneration burn, and describe the pattern of recovery in this flora on *Dicksonia* caudexes.

Study area

The experiment established to examine the effects of logging and slash burning on the health and regenerative responses of *D. antarctica* at two sites in the Wayatinah area of central Tasmania was used (Table 5.1, Figure 5.1).

Experimental design

The survey design involved the use of pre and repeated post-treatment sampling of the epiphytic flora of *Dicksonia* caudexes subject to cable logging and burning. The control and treatment quadrats (Table 5.5) were all established prior to cable logging commencing. There were 111 individual *D. antarctica* caudexes surveyed (Table 5.5) in the cable logging treatment (numbered stems 1-113, less 2 lost during logging) and 25 in the control treatment, (numbered stems 125-150).

Sampling Methods

Each of the individually numbered and mapped *Dicksonia* were surveyed for the presence of any vascular species adopting an epiphytic form (filmy ferns, other ferns, tree and shrub seedlings, herbaceous species) on the vertical portion of the *Dicksonia* trunk. This definition of epiphyte included obligate, facultative, hemi- or accidental epiphytes such as rainforest tree seedlings (see introduction). Each vascular species was allocated an abundance scale of (1) rare, (2) moderate and (3) abundant on the individually numbered *D. antarctica* caudexes. Climbing species such as *Clematis aristata* were not recorded as epiphytes of *Dicksonia antarctica* unless they were rooted into the *Dicksonia* caudex. Additionally, species growing at the base of the *Dicksonia* caudex (e.g. seedlings of *Coprosma quadrifida*), intermixed with soil and litter, were not considered epiphytic. The percentage of the *Dicksonia* caudex occupied by hepatics was estimated. *D. antarctica* caudex height was measured with a tape measure from the base of the caudex, at the soil surface (coarse litter and fallen fronds were cleared away), to the apical tip following Medeiros *et al.* (1992).

Analytical procedures

The presence of individual epiphytes on *Dicksonia* caudexes in the cable logging and burning treatment portion of the experiment were summarised to presence only (from the ordinal abundance scale) and expressed as a percentage of the total number of *Dicksonia* stems present in either the logging treatment (n=111) or control portions (n=25). The number of obligate epiphytic ferns species per individual *D. antarctica* was summarised across the population by the proportion of colonised *D. antarctica* caudexes. The relationship between the number of epiphytic fern species present and explanatory variables such as caudex height were explored using bi-plots.

Experiment 4: Impacts of cable logging on vascular epiphytes within streamside reserves

Study areas

Three cable logging coupes in the Roses Tier area of northern Tasmania (Figure 4.1) were used to examine the hypothesis that the intensity of disturbance resulting from different cable logging techniques adjacent to and within streamside reserves effects the richness and abundance of vascular epiphytic fern species, and their substrates. The vegetation characteristics and cable logging treatments at the three Roses Tier coupes are detailed in Table 4.2. Briefly, the cable logging treatment applied at Roses Tier 137f involved cutting narrow cable corridors through the streamside vegetation but not passing logs through them, at Roses Tier 126a cable corridors were also cut through the streamside vegetation and logs were yarded through them, and at Roses Tier 137g the streamside reserve was severely disturbed by cable logging and subsequently burnt.

Experimental design

The three study areas in northern Tasmania were considered replicates and epiphyte sampling was stratified by the intensity of the cable logging treatment imposed on the streamside vegetation (Experiment 1, Chapter 4). Within each of the three Roses Tier replicate coupes, four 10 x 2 m quadrats were sampled, stratified within the coupe according to intensity of cable logging treatment. The survey stratification and quadrat characteristics are described in detail in Table 4.1 and 4.3. Cable logging was considered a factor with three levels if intensity; (1) severely disturbed ie a cable logging gap in the reserve, (2) a maintained portion of streamside reserve, and (3a) upstream control and (3b) downstream control. Streamside reserve or buffer areas were not considered to be controls as they were generally narrow strips of retained vegetation subject to edge effects. Control quadrats were sampled in the vicinity of the cable logged area both upstream and downstream. At each study area, the four quadrats were sampled along a total distance on the streamside of 100-200 m. One of each of the four quadrats was allocated to each of the levels of disturbance intensity above.

Sampling methods

The unit of vegetation sampling adopted was a 10 x 2 m quadrat. Each quadrat was divided into five sub-plots of 2 x 2 m which formed a continuous strip oriented parallel to the streamside. In each sub-plot of 2 x 2 m each epiphyte was assigned a visually assessed cover-abundance score from the Braun-Blanquet (Poore 1955) scale described in Chapter 2 experiment 1. Cover was defined as foliage projective cover following Walker and Tunstall (1981). Data were collected on epiphyte - substrate associations within a 10 x 2m plots following the methods described for the chronosequence sampling for experiment 1 in this chapter. A count of all *D. antarctica* individuals rooted within the 10 x 2 m quadrat was made. The abundance (rare, moderate, abundant) of epiphytes on substrates was scored using the methods in Experiment 1 in this chapter.

Analytical procedures

Epiphytic fern richness were summed from the five 2 x 2 m sub-plots by treatment across the three Roses Tier replicate coupes. Epiphyte cover was calculated using the methods in Chapter 2 Experiment 1, and expressed as a treatment mean cover score for the mid-point transformed cover-abundance scores. In this experiment however only 5 sub-plots, not 20 per quadrat were available. Trends in species cover were then averaged within treatment (logging gap, maintained portion and controls). The faithfulness or fidelity (sensu Bedward 1999) of species to treatments was tested using Indicator Species Analysis (Dufrene and Legendre 1997). It was performed using PC-ORD for Windows (McCune and Mefford 1999) with 10,000 randomised starts. The relationship between the richness of epiphytic fern species and the presence of *D. antarctica* caudex substrates was examined using linear regression. Variation in epiphytic-substrate associations and the logging treatments is examined using cross-tabulations of treatment type by the relative abundance of the various epiphyte by substrate associations.

6.3 Results

Experiment 1: Chronosequence analysis comparing the response of vascular epiphytes to cable and ground-based logging.

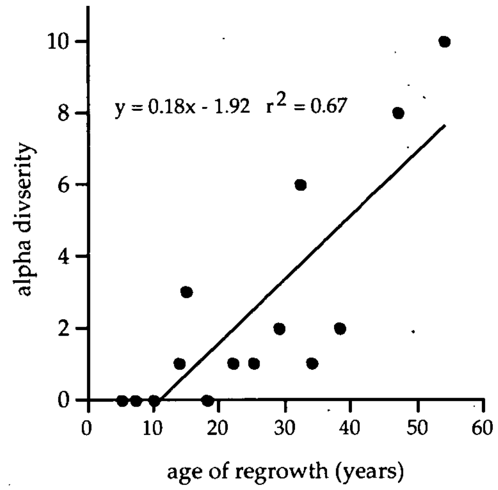
Recovery and recruitment of vascular epiphytes in cable logged wet forest

Recruitment of vascular epiphytic ferns following cable logging of wet forest was a function of time since logging and the availability of suitable substrates. The richness of obligate epiphytic ferns increased steadily with stand age (Figure 6.1a), with an approximate recruitment rate of a new species every five years.

The richness (per 0.09 ha quadrat) of obligate epiphytic fern species in cable logged wet forest reached comparable levels to the mature forest (3-6 species) however their cover was only approximately one-tenth of the levels observed in mature forest. There was a linear relationship between the richness of all fern species growing as epiphytes and stand age (r^2=0.85, $p<0.001$, Figure 6.1e). Facultative fern epiphytes (e.g. *Histiopteris incisa*, *Hypolepis rugosula*) elevated the richness of epiphyte ferns in the first three decades of regrowth, when the obligate fern epiphytes were still uncommon. The richness of fern species growing as epiphytes in the mature forest was similar to the values present in quadrats aged 30 years and over.

The richness of obligate fern epiphyte-substrate associations was closely related to stand age (r^2=0.84, $p<0.001$) with a new association between epiphyte and substrate predicted to occur every three years (Figure 6.1d) with up to 20 associations being present in the oldest regrowth sampled, just exceeding the richness of associations present in the unlogged quadrats. The richness of associations was dependent on the availability of a sufficient diversity of substrate types for colonisation by the obligate epiphytic ferns, the two variables are highly correlated (r=0.91, $p<0.001$). The number of different substrates colonised by obligate and facultative epiphytic ferns increases steadily with stand age. A new type of substrate is colonised at an approximate rate of 1 every 8 years (Figure 6.1c). The chronological arrangement of substrate types (Figure 6.2) indicates that logs and rocks are the first to be colonised by obligate epiphytic ferns, a substrate also favoured by obligate epiphytic ferns in mature forest (Table 6.3). *Dicksonia* caudexes were colonised for the first time at 25 years, and the other biotic substrates, such as *Olearia argophylla* and *Pomaderris apetala* bark, were not colonised until the 41-54 years age group.

The richness of different substrates being colonised by obligate and facultative epiphytic ferns on a quadrat basis equalled the value for the unlogged quadrats by 41-54 years. *Grammitis billardieri* and *Microsorum pustulatum* were the first obligate epiphytic ferns to re-establish following cable logging (Table 6.2). They appeared first at 14 years colonising a boulder (in this case being lithophytic) and fallen log as a substrate. With increasing age since cable logging, *Hymenophyllum* species colonised living substrates.



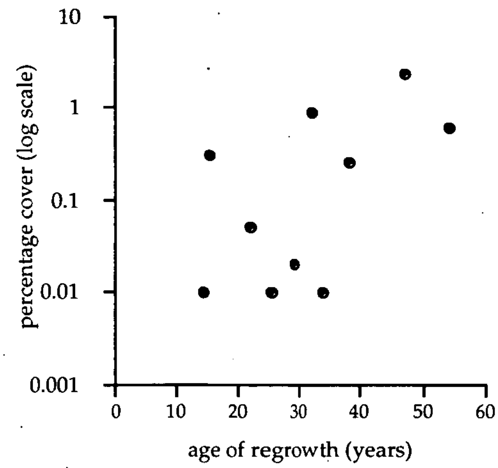
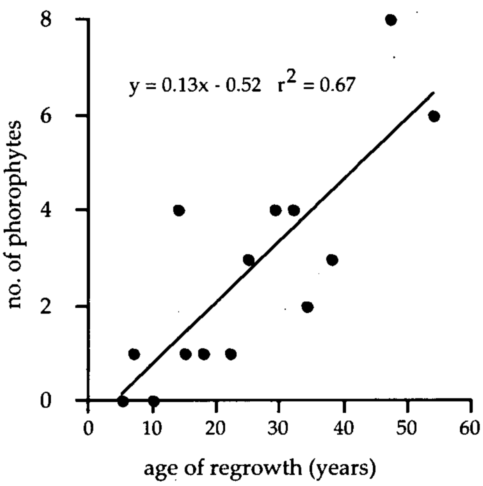


Figure 6.1a Richness of obligate epiphytic fern species

Figure 6.1b Cover ($\log_{10}$) of obligate epiphytic fern species $y=0.03-0.34x$ $r^2=0.36$



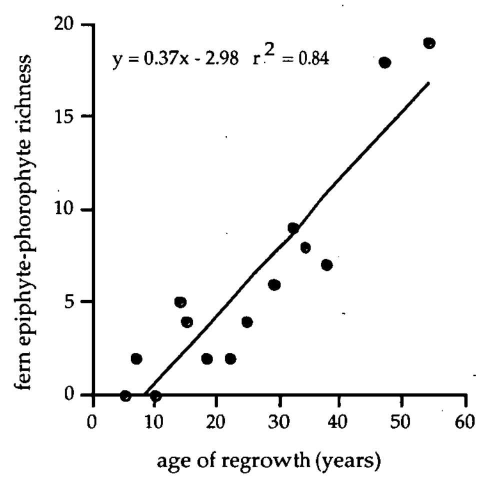


Figure 6.1c Richness of substrates colonised by obligate and facultative epiphytic ferns

Figure 6.1d Fern epiphyte-substrate richness

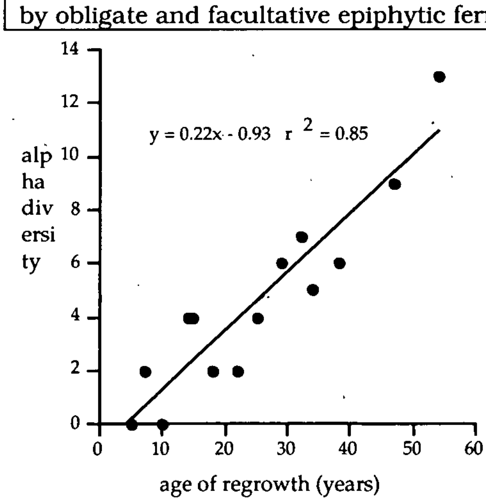


Figure 6.1e Richness of fern species growing as epiphytes, includes obligate and facultative species.

Figure 6.1 *The vascular epiphytic ferns in the cable logged wet forest chronosequence.*

Table 6.2 *Compositional trend in the abundance of obligate epiphytic fern species recorded in the cable logged wet forest chronosequence. The samples are ordered according to the stand age from the youngest to the mature forest sites (from left to right) and species are ordered alphabetically. The species abundance scores on the particular substrate are rare (1), moderate (2) and abundant (3).*

Asplenium bulbiferum on *Dicksonia antarctica*	---------1------3
Asplenium bulbiferum on logs	-------------1---
Asplenium bulbiferum on rock	-------------1---
Asplenium fabellifolium on *Dicksonia antarctica*	-------------1---
Asplenium fabellifolium on rock	----1--------2---
Asplenium flaccidum on twigs and branches	------------1----
Ctenopteris heterophylla on *Olearia argophylla*	--------1--------
Ctenopteris heterophylla on rock	-------------1---
Grammitis billardieri on *Atherosperma moschatum*	--------------2--
Grammitis billardieri on *Bedfordia salicina*	---------------2-
Grammitis billardieri on *Dicksonia antarctica*	--------1---1-2-3
Grammitis billardieri on logs	-----1---2-12-2-2
Grammitis billardieri on *Olearia argophylla*	------------1---2
Grammitis billardieri on *Pomaderris apetala*	------------2-1--
Grammitis billardieri on rock	----1-------2111-
Grammitis billardieri on twigs and branches	---------1--2--2-
Grammitis magellanica on logs	-------------1---
Grammitis magellanica on *Olearia argophylla*	------------11---
Hymenophyllum cupressiforme on *Dicksonia antarctica*	------------1---2
Hymenophyllum cupressiforme on eucalypt buttress	---------------2-
Hymenophyllum cupressiforme on logs	---------1-------
Hymenophyllum cupressiforme on *Olearia argophylla*	-------------1-22
Hymenophyllum cupressiforme on rock	------------2--3-
Hymenophyllum cupressiforme on twigs and branches	---------------2-
Hymenophyllum flabellatum on *Olearia argophylla*	----------------1
Hymenophyllum flabellatum on *Dicksonia antarctica*	-------------12-2
Hymenophyllum flabellatum on logs	------------2-2-2
Hymenophyllum peltatum on logs	---------1----1--
Hymenophyllum rarum on *Dicksonia antarctica*	-------------12--
Hymenophyllum rarum on logs	---------1-12-2--
Hymenophyllum rarum on rock	------------21-2-
Microsorum pustulatum on *Olearia argophylla*	----------------2
Microsorum pustulatum on *Dicksonia antarctica*	-------------1--2
Microsorum pustulatum on logs	------11-11------
Microsorum pustulatum on *Pomaderris apetala*	---------1---1---
Microsorum pustulatum on rock	---11-------1----
Polyphlebium venosum on *Dicksonia antarctica*	--------------1--
Rumohra adiantiformis on *Dicksonia antarctica*	-------------1--2
Tmesipteris obliqua on *Dicksonia antarctica*	------------1-1--
	111
	111122233345025
Quadrat age in years	57045825924874000

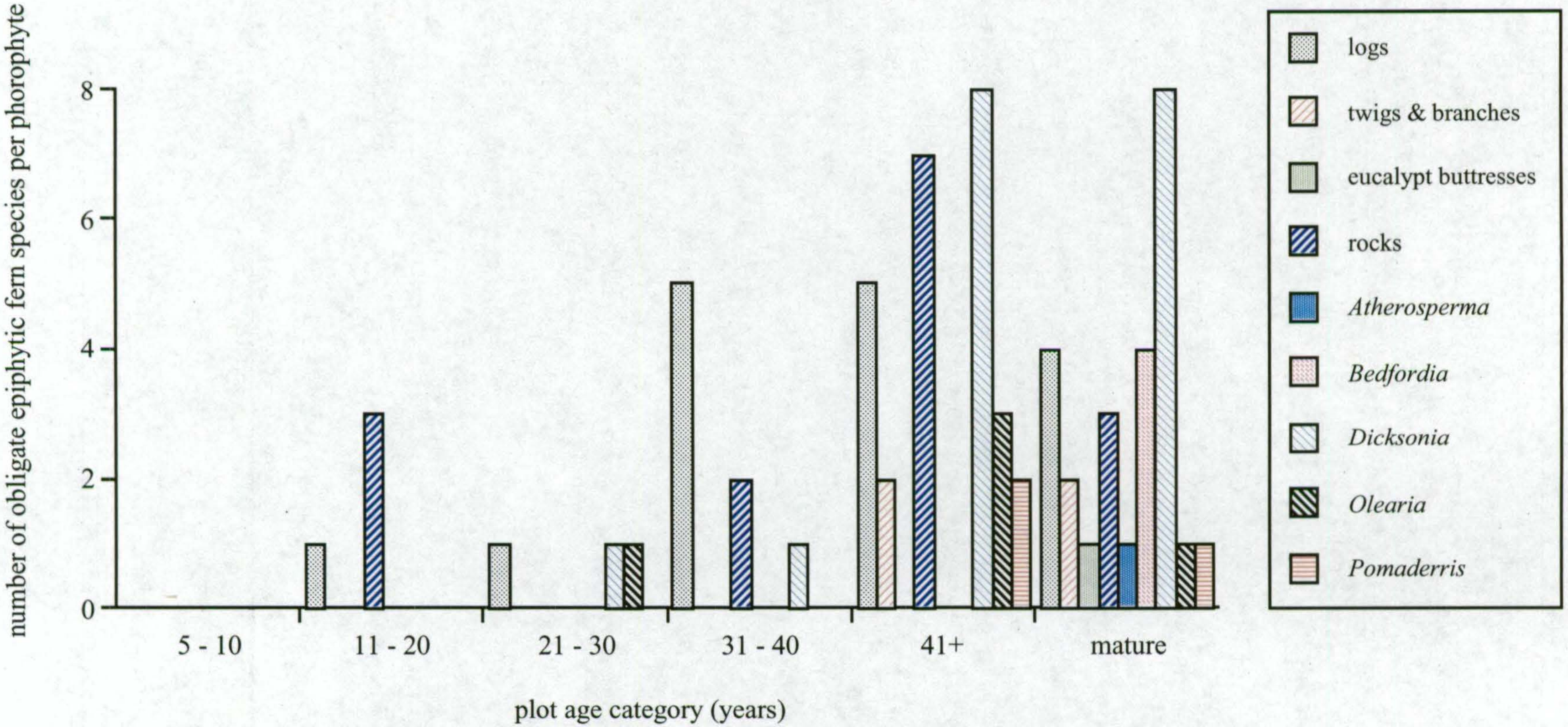


Figure 6.2 *The types of substrates colonised by obligate epiphytic fern species in the cable logged and mature wet forest chronosequence.*

Table 6.3 *The relative frequency of obligate epiphytic fern species in mature forest and forest 5-30 years after cable or ground-based logging. Frequency in mature forest is the number of different obligate epiphytic fern species occupying that substrate. The substrates (rows) are ordered from most frequently colonised to least frequently colonised in mature forest.*

	5-30 year regrowth		species in mature forest																	
substrate	ground-based regrowth	cable regrowth	frequency in mature forest	*Grammitis billardieri*	*Hymenophyllum rarum*	*Hymenophyllum cupressiforme*	*Microsorum pustulatum*	*Rumohra adiantiformis*	*Hymenophyllum australe*	*Asplenium bulbiferum*	*Ctenopteris heterophylla*	*Hymenophyllum flabellatum*	*Hymenophyllum flabellatum*	*Polyphlebium venosum*	*Tmesipteris obliqua*	*Asplenium flabellifolium*	*Asplenium flaccidum*	*Grammitis magellanica*	*Lastreopsis acuminata*
Dicksonia caudex	4	1	13	x	x	x	x	x	x	x	x	x	x	x	x	x			
rock	2	3	13	x	x	x		x	x	x	x	x	x	x	x	x	x		
fallen logs	4	2	13	x	x	x	x	x	x	x	x	x	x	x	x			x	
Nothofagus bark			8	x	x	x	x	x	x				x	x					
Atherosperma bark			8	x	x	x	x	x		x	x				x				
Olearia argophylla bark		1	8	x	x	x	x	x	x		x							x	
twigs and branches			7	x	x	x	x	x					x				x		
Eucryphia bark			6	x	x		x	x	x	x									
Nothofagus stag			6	x	x		x	x		x									x
Pomaderris apetala bark	1		6	x	x	x	x				x	x							
Bedfordia salicina bark			5	x		x	x				x	x							
eucalypt buttress			4	x	x			x				x							
Coprosma quadrifida bark			4	x	x	x	x												
Anodopetalum bark			3	x					x				x						
Beyeria viscosa bark			3		x	x							x						
Phebalium squameum bark			3	x	x		x												
surface roots			3	x				x				x							
Atherosperma stag			2				x			x									
Cyathea caudex			1		x														
other buttresses			1						x										
cut stump			1						x										
Acacia dealbata bark	1		1				x												
Acacia melanoxylon bark			1			x													
Bursaria spinosa bark			1			x													
Cyathodes bark			1	x															
Parsonia brownii bark			1			x													
number of substrates used	5	4	26	17	15	14	14	11	9	7	7	7	7	4	4	2	2	2	1

Recovery and recruitment of vascular epiphytes in ground-based logged wet forest

Recruitment of vascular epiphytic ferns following ground-based logging of wet forest was also function of time since logging and the availability of suitable substrates. The richness of obligate epiphytic fern species increased steadily with age (Figure 6.4a) with an approximate recruitment rate of a new species every eight years.

The richness of obligate epiphytic fern species in cable logged wet forest failed to reach comparable levels to the unlogged forest, with *Tmesipteris obliqua, Polyphlebium venosum* and *Ctenopteris heterophylla* not recolonising in the first 30 years. The cover of obligate epiphytic ferns was low 5-30 years following ground-based logging, reaching 2% in 30 years, compared 3-17% in the unlogged quadrats. There was a linear relationship between the richness of all fern species with an epiphytic or hemi-epiphytic habit (facultative and obligate combined) and stand age in the 5-30 year old regrowth (r^2=0.63, p=0.01, Figure 6.4e). *Dicksonia, Histiopteris, Hypolepis* and *Polystichum*, species which normally grow rooted in the soil, were all recorded as facultative fern epiphytes in regrowth greater than 11 years old. Facultative fern epiphytes such as these reached their peak abundance in regrowth, they were rare or absent in unlogged forest.

The richness of obligate epiphyte fern-substrate associations was closely related to stand age following ground-based logging (r^2=0.87 p=<0.001, Figure 6.4d), with a new association formed approximately every two years, with 14 associations present 30 years after logging. The pearson correlation of the richness of obligate epiphyte ferns and the variety of substrates colonised was high (r^2=0.82, p<0.01, n=9, Table 2.19).

The number of different types of substrates colonised by obligate and facultative ferns increased with stand age, with a new substrate being colonised at an approximate rate of 1 every 6 years (Figure 6.4c). The first substrates colonised by obligate epiphytic ferns were logs and rocks, and with increasing age since logging a greater diversity of substrates were colonised, including the bark of several shrub and tree species (Figure 6.4). The richness of different substrate types colonised on a quadrat basis approached, but did not equal, the level in unlogged forest (Figure 6.5). Substrates not colonised in regrowth included twigs and branches, *Atherosperma* and *Olearia argophylla* bark (Figure 6.5).

Several of the epiphyte-substrate associations in Table 6.4 were unique to unlogged forest in this chronosequence. Epiphytes, which in unlogged forest, occupied a variety of substrates, such as rocks (in this case as lithophytes) and the bark of *Olearia* and *Pomaderris* (Table 6.4), were restricted to logs or *Dicksonia* caudexes following ground-based logging. Following cable logging *Asplenium bulbiferum*, *Microsorum pustulatum* and *Rumohra adiantiformis* were the most abundant epiphytic species (Table 6.4). *Microsorum pustulatum* was the first of the obligate epiphytic fern species to re-establish, observed in 9 and 10 year old regrowth on fallen logs and as a lithophyte on rocks (Table 6.4), then occupying the bark surfaces of tall understorey shrubs in older regrowth (Figure 6.3).

Figure 6.3 *The epiphytic fern* Microsorium pustulatum *on* Phebalium squameum, a *tall shrub of 30 year old ground-based logged regrowth.*

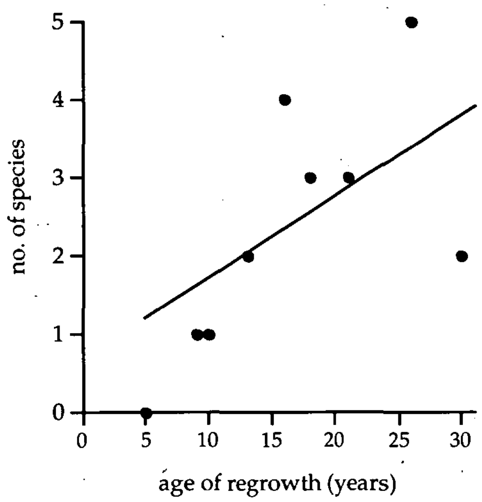


Figure 6.4a Richness of obligate epiphytic fern species y=0.13x-0.17 r^2=0.46

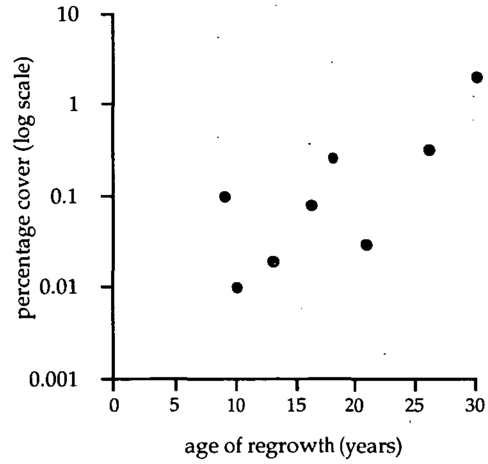


Figure 6.34 Cover of obligate epiphytic fern species y=0.06x-0.62 r^2=0.50

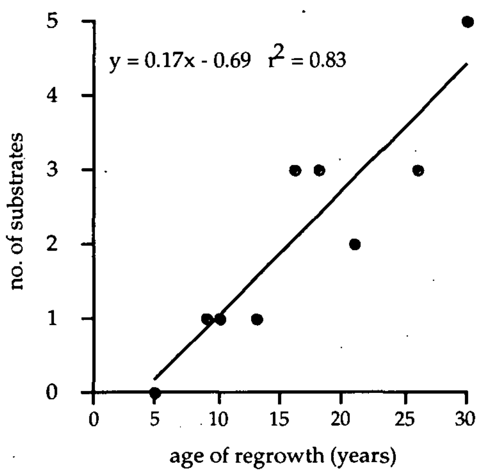


Figure 6.4c Richness of substrates colonised by obligate and facultative epiphytic ferns

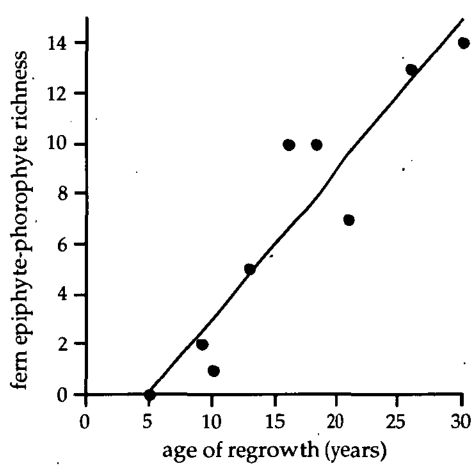


Figure 6.34 Fern epiphyte-substrate richness y=0.59x-2.84 r^2=0.87

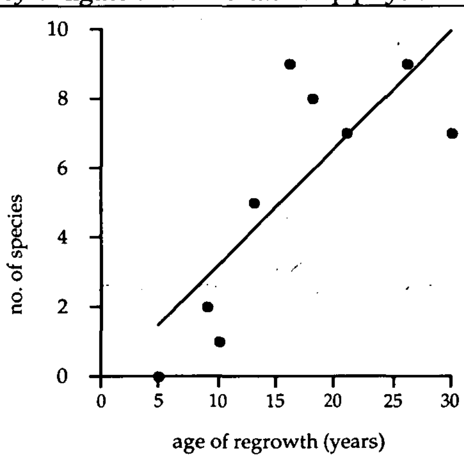


Figure 6 4e Richness of all fern species growing as epiphytes, includes obligate and facultative epiphytes y=0.34-0.25 r^2=0.63

Figure 6.4 *Vascular epiphytic ferns in the ground-based logged wet forest chronsequence.*

Table 6.4 *Compositional trends in the abundance of obligate epiphytic fern species recorded in the ground-based logged wet forest chronosequence. The samples are ordered according to the stand age from the youngest to the mature forest sites (from left to right). The species abundance scores on the particular substrate are rare (1), moderate (2) and abundant (3).*

Asplenium bulbiferum on logs	---1121----
Ctenopteris heterophylla on *Atherosperma moschatum*	---------1-
Ctenopteris heterophylla on *Olearia argophylla*	---------2-
Ctenopteris heterophylla on *Pomaderris apetala*	---------11
Grammitis billardieri on *Dicksonia antarctica*	---------2-
Grammitis billardieri on eucalypt buttress	---------1-
Grammitis billardieri on logs	-------1-22
Grammitis billardieri on *Olearia argophylla*	---------22
Grammitis billardieri on *Pomaderris apetala*	---------11
Grammitis billardieri on rock	---------12
Grammitis billardieri on *twigs and branches*	----------2
Hymenophyllum cupressiforme on *Dicksonia antarctica*	-------1-3-
Hymenophyllum cupressiforme on logs	----1----23
Hymenophyllum cupressiforme on *Olearia argophylla*	---------3-
Hymenophyllum cupressiforme on rock	----------2
Hymenophyllum flabellatum on *Dicksonia antarctica*	-------1-32
Hymenophyllum flabellatum on eucalypt buttress	---------3-
Hymenophyllum flabellatum on logs	---------2-
Hymenophyllum flabellatum on *Pomaderris apetala*	---------1-
Hymenophyllum rarum on *Dicksonia antarctica*	-------1-32
Hymenophyllum rarum on eucalypt buttress	---------2-
Hymenophyllum rarum on logs	---------23
Hymenophyllum rarum on *Olearia argophylla*	---------32
Hymenophyllum rarum on *Pomaderris apetala*	----------2
Hymenophyllum rarum on rock	---------1-
Microsorum pustulatum on *Acacia dealbata*	------1-1--
Microsorum pustulatum on *Dicksonia antarctica*	-------1-2-
Microsorum pustulatum on logs	1--1-2-12--
Microsorum pustulatum on *Olearia argophylla*	---------2-
Microsorum pustulatum on *Phebalium squameum*	--------1--
Microsorum pustulatum on *Pomaderris apetala*	----1------
Microsorum pustulatum on rock	--1-1--1-2-
Polyphlebium venosum on *Dicksonia antarctica*	---------21
Polyphlebium venosum on rock	---------1-
Rumohra adiantiformis on *Dicksonia antarctica*	---------32
Rumohra adiantiformis on eucalypt buttress	--------2--
Rumohra adiantiformis on logs	----121-321
Tmesipteris obliqua on *Dicksonia antarctica*	---------32
	111122357
Quadrat age in years	59036816088

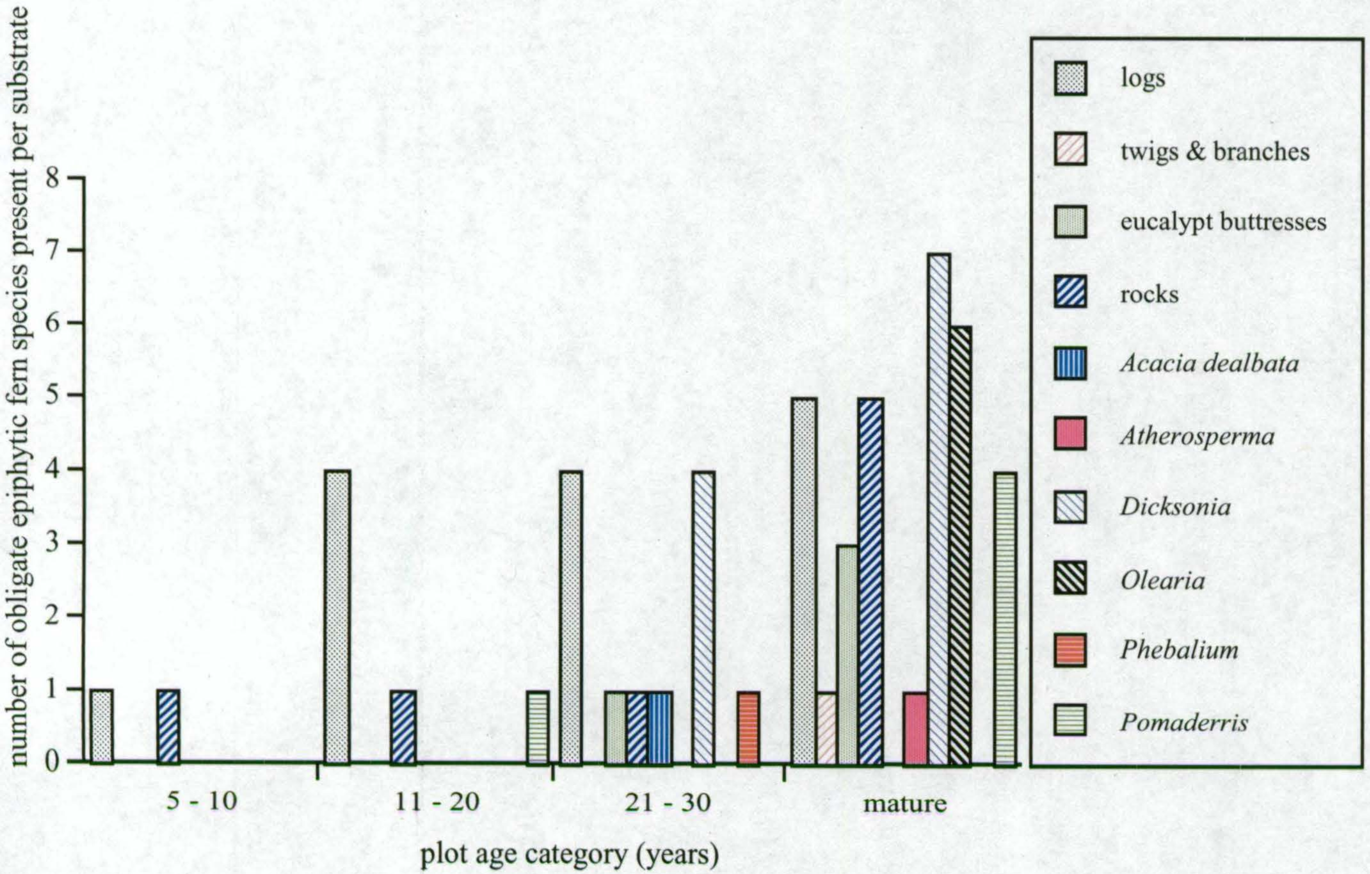


Figure 6.5 *The types of substrates colonised by obligate epiphytic fern species in the ground-based logged and mature wet forest chronosequence.*

Comparison of recovery and recruitment of vascular epiphytes following cable or ground-based logging

The comparison of re-colonisation patterns of vascular epiphytes following either cable or ground-based logging indicated differences in both the rate of species recruitment over time, the type and number of species recruited, and the types of substrates colonised.

Wet forest logged with ground-based machinery exhibited a significantly increased rate of obligate epiphytic fern species recruitment than cable logged wet forest in the first 30 years after disturbance (Figure 6.6, Table 6.5). Similarly, the cover of obligate epiphytic fern species was greater following ground-based logging (Figure 6.6), although not significantly so (Table 6.5). The rate of both obligate and facultative epiphytic fern species recruitment was significantly greater following ground-based logging (Figure 6.10, Table 6.5). Facultative fern epiphytes (*Dicksonia, Histiopteris, Hypolepis rugosula* and *Polystichum proliferum*) were more frequent and abundant, especially when growing on fallen or cut logs following ground-based logging than following cable logging.

Table 6.5 *Analysis of covariance of linear models describing the relationship between epiphytic fern species and their presence following cable or ground-based logging.*

	logging treatment		
	cable	ground-based	significance of difference between treatments
	linear regression equation		
Richness of obligate epiphytic fern species	$y=0.18x-1.92$ $r^2=0.67$	$y=0.13x-0.17$ $r^2=0.46$	$p=0.0011$, $f=12.07$, $df=2,13$
Cover of obligate epiphytic fern species	$y=0.03x-0.34$ $r^2=0.37$	$y=0.06x-0.62$ $r^2=0.50$	ns, $p>0.05$
Richness of substrates colonised by obligate and facultative epiphytic ferns	$y=0.13x-0.52$ $r^2=0.67$	$y=0.17x-0.69$ $r^2=0.83$	ns, $p>0.05$
Obligate fern epiphyte-substrate richness	$y=0.37x-2.98$ $r^2=0.84$	$y=0.59x-2.84$ $r^2=0.87$	$p=0.0011$, $f=10.054$, $df=2,19$
Richness of all fern species growing as epiphytes	$y=0.22x-0.93$ $r^2=0.85$	$y=0.34x-0.25$ $r^2=0.63$	$p=0.0051$, $f=7.05$, $df=2,13$

The richness of obligate epiphytic fern substrate associations was significantly greater in the first 30 years following ground-based logging (Figure 6.9, Table 6.5). Following cable logging, obligate and facultative fern epiphytes were only observed growing on rocks, logs, *Dicksonia* caudexes and cut stumps, however following ground-based logging, they were observed additionally growing on eucalypt buttresses, *Acacia dealbata*, *Phebalium* and *Pomaderris* bark. At the individual species level after ground-based logging, species such as *Microsorum pustulatum* grew on *Acacia dealbata*, *Phebalium* and *Pomaderris* bark, *Dicksonia* caudexes, logs, and rocks, while following cable logging it was only observed growing on logs and rocks.

The number of different types of substrates colonised by obligate and facultative epiphytes did not differ significantly between the cable and ground-based logging treatments (Figure 6.8, Table 6.5). With increasing stand age, both chronosequences exhibited an increased diversity of substrates being colonised by their obligate and facultative fern epiphytes.

The floristic response to cable or ground-based logging differed, a greater richness of obligate epiphytic ferns were present at 30 years following ground-based logging than cable logging, and ground-based sites were more similar floristically to unlogged forest at 30 years than cable logged regrowth of the same age (Table 6.6). Indicator species analysis applied to the two logging treatments (Table 6.7) identified twice as many indicator taxa for ground-based logged sites, many of which were species (e.g. *Hymenophyllum*) characteristically observed in unlogged forest.

Table 6.6 *The percent frequency and percent cover of obligate epiphytic ferns in the cable (n=9) and ground-based (n=9) logged wet forest and unlogged (n=5) chronosequence quadrats. The values are averaged within the respective age categories.*

	cable logged						ground-based logged						unlogged	
age category	5-10		11-20		21-30		5-10		11-20		21-30			
	freq	cov.	freq	cov.	freq	cov.	freq	cov.	freq	cov.	freq	cov.	freq	cov.
Asplenium bulbiferum	-	-	-	-	-	-	-	-	2	<1	<1	<1	4	<1
Asplenium flabellifolium	-	-	2	<1	-	-	-	-	-	-	-	-	-	-
Ctenopteris heterophylla	-	-	-	-	<1	<1	-	-	-	-	-	-	7	<1
Grammitis billardieri	-	-	2	<1	<1	<1	-	-	-	-	2	<1	33	1
Hymenophyllum cupressiforme	-	-	-	-	-	-	-	-	<1	<1	2	<1	20	<1
Hymenophyllum flabellatum	-	-	-	-	-	-	-	-	-	-	<1	<1	24	2
Hymenophyllum peltatum	-	-	-	-	-	-	-	-	-	-	-	-	<1	<1
Hymenophyllum rarum	-	-	-	-	-	-	-	-	-	-	<1	<1	25	1
Microsorum pustulatum	-	-	2	<1	2	<1	3	<1	2	<1	5	<1	7	<1
Polyphlebium venosum	-	-	-	-	-	-	-	-	-	-	-	-	<1	<1
Rumohra adiantiformis	-	-	-	-	-	-	-	-	2	<1	5	<1	7	<1
Tmesipteris obliqua	-	-	-	-	-	-	-	-	-	-	-	-	11	<1

Table 6.7 *Indicator species analysis for sites subject to cable or ground-based logging, based on regrowth forest samples aged 5-30 years. Species are ordered by the treatment in which they reached their maximum indicator value (iv). Higher indicator values approximate greater importance of that species in a particular treatment.*

species	treatment with maximum iv	cable logged iv	ground-based logged iv
Grammitis billardieri	cable	12	5
Asplenium flabellifolium	cable	11	0
Ctenopteris heterophylla	cable	11	0
Microsorum pustulatum	ground-based	14	60
Asplenium bulbiferum	ground-based	0	44
Rumohra adiantiformis	ground-based	0	44
Hymenophyllum cupressiforme	ground-based	0	22
Hymenophyllum flabellifolium	ground-based	0	11
Hymenophyllum rarum	ground-based	0	11

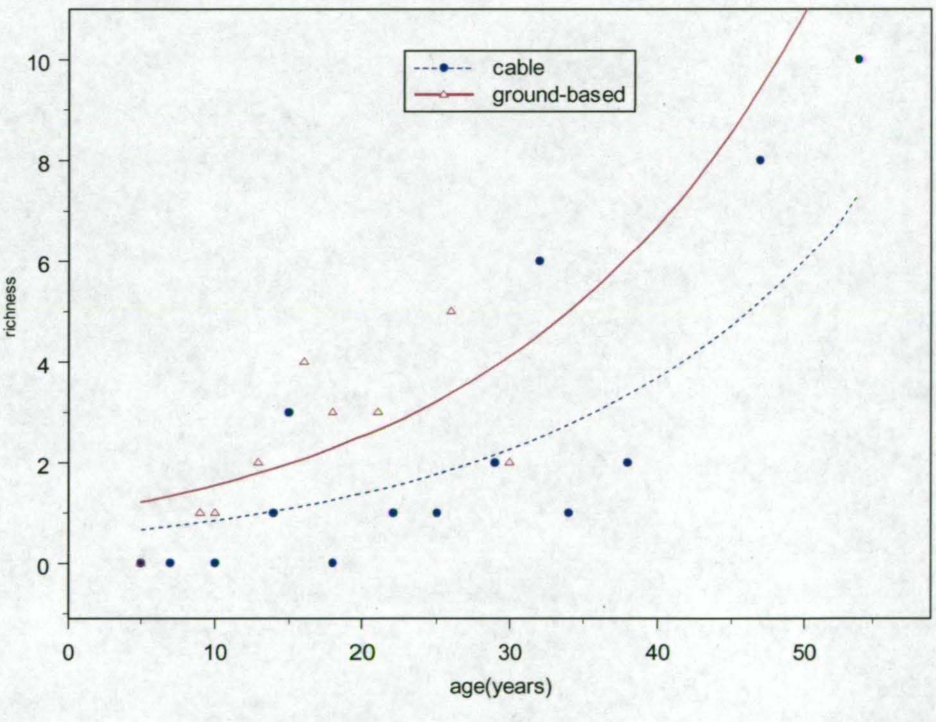


Figure 6.6 *Richness of obligate epiphytic fern species (0.09 ha. quadrats) in the cable and ground-based logged chronosequence. The fitted exponential curve predictions are extended beyond the data points for the ground-based quadrats.*

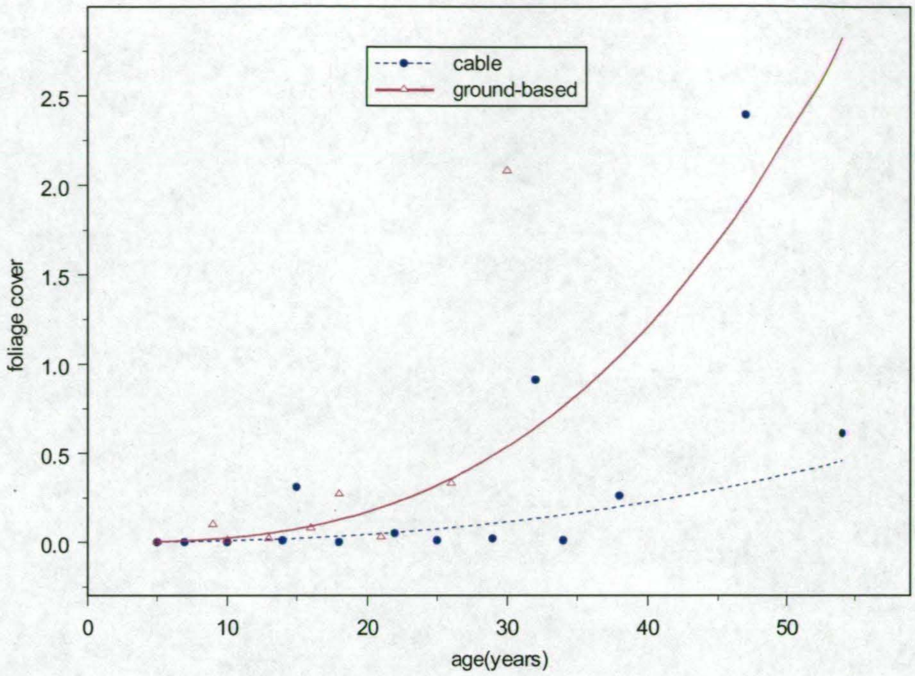


Figure 6.7 *Cover of obligate epiphytic fern species in the cable and ground-based logged chronosequence. The fitted power curve predictions are extended beyond the data points for the ground-based quadrats.*

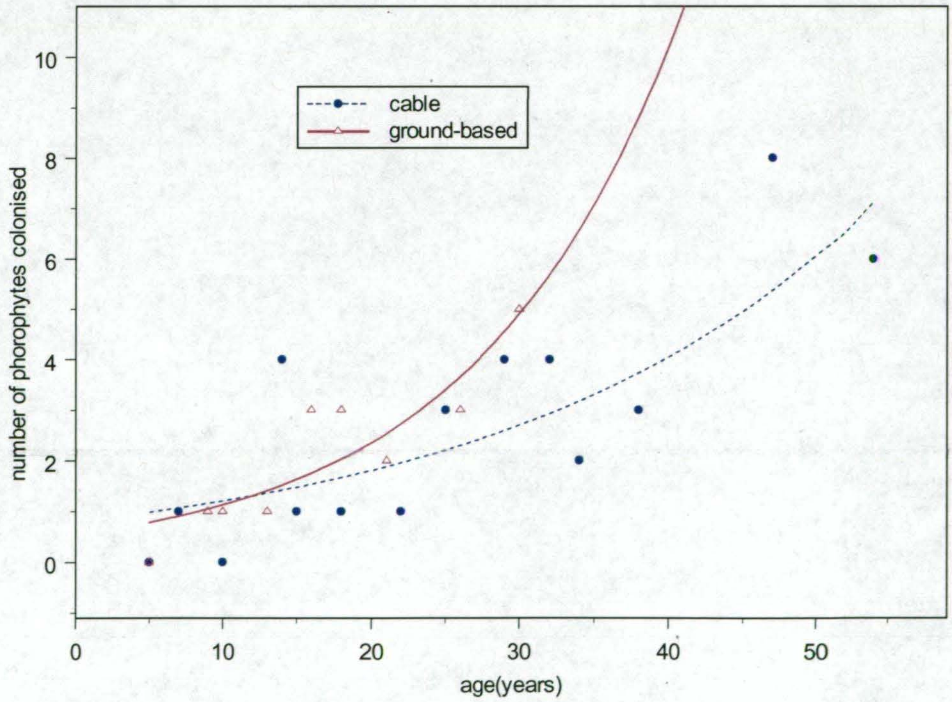


Figure 6.8 *Number of different types of substrates (0.09 ha. quadrats) colonised by obligate and facultative epiphytic fern species in the cable and ground-based logged chronosequence. The fitted exponential curve predictions are extended beyond the data points for the ground-based quadrats.*

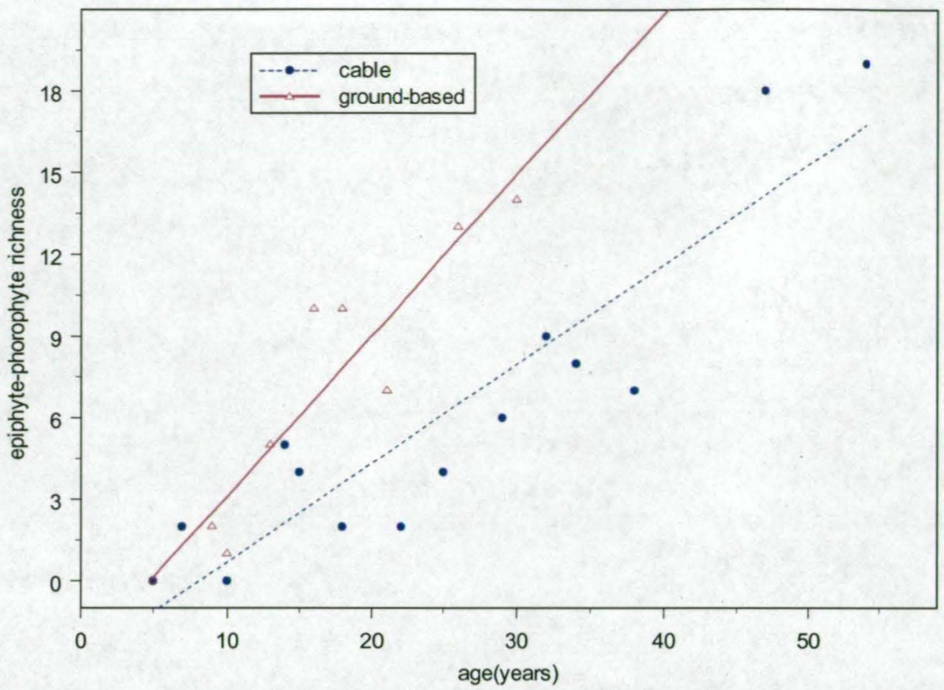


Figure 6.9 *Obligate epiphytic fern epiphyte-substrate richness (0.09 ha. quadrats) in the cable and ground-based logged chronosequence. The linear least squares curve fitting predictions are extended beyond the data points for the ground-based quadrats.*

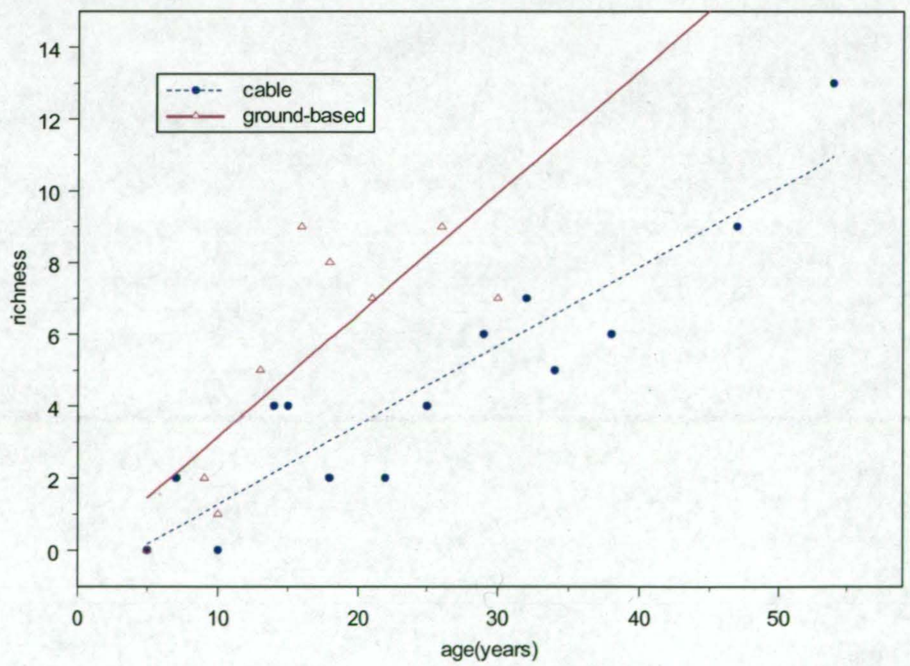


Figure 6.10 *Total richness (0.09 ha. quadrats) of all fern species present as epiphytes in the cable and ground-based logged chronosequence. The linear least squares curve fitting predictions are extended beyond the data points for the ground-based quadrats.*

Experiment 2: Impacts of cable logging and burning on the frequency and cover of vascular epiphytes in wet forests

Two years following cable logging and burning, only two species remained (*Asplenium flabellifolium* and *Microsorim pustulatum*) of the twelve species of obligate vascular epiphytic fern observed prior to cable logging (Table 6.8). Vascular epiphytes persisted in only one of the four quadrats after disturbance, with both species persisting on a rock ledge as lithophytes. No recovery was observed on the remaining quadrats.

For the 15 species of obligate or accidental vascular epiphytic fern species present prior to disturbance, there were between sixteen and seven unique combinations of epiphytic species and substrate observed (Table 6.9). The most frequently colonised substrates were *Dicksonia antarctica* caudexes, rock ledges and *Olearia argophylla* bark.

The trend in persistence of epiphytic shrub and herb species post cable logging and burning was variable. In two quadrats the number increased, in the other two it decreased (Table 6.8). Apart from species occupying niches within rock ledges, all epiphytic shrub and herb species present after cable logging and burning occurred on *Dicksonia* caudexes. *Acacia dealbata* and *Pomaderris apetala* were the most abundant colonising epiphytic shrub species, and had a clear preference for *Dicksonia* caudexes. Cable logging and burning increased the proportion of accidental non-fern epiphytes, and severely decreased the proportion of obligate fern epiphytes.

Table 6.8 *Short term effects (< 2 years) of cable logging on vascular epiphytic fern species at four long term monitoring quadrats at Wayatinah. The abundance measure is the mean percent cover score, based on 20 replicate observations per quadrat.*

	Blue09 site 2		Blue09 site 24		Pearce04 site 5		Pearce04 site 12	
Species	pre	post	pre	post	pre	post	pre	post
Asplenium bulbiferum	0.01	-	-	-	-	-	-	-
Asplenium flabellifolium	0.01	0.01	-	-	0.05	-	-	-
Aplenium terrestre	0.01	-	-	-	-	-	-	-
Grammitis billardieri	0.33	-	0.01	-	0.01	-	0.73	-
Hymenophyllum australe		-	-	-	-	-	0.01	-
Hymenophyllum flabellifolium	0.18	-	0.75	-	0.13	-	-	-
Hymenophyllum peltatum		-	-	-	-	-	0.01	-
Hymenophyllum rarum	0.13	-	0.05	-	0.01	-	0.75	-
Microsorim pustulatum	0.18	0.01	0.05	-	-	-	-	-
Polyphlebium venosum	0.75	-	0.01	-	-	-	0.01	-
Rumohra adiantiformis	0.01	-	0.01	-	-	-	-	-
Tmesipteris obliqua	0.05	-	0.23	-	-	-	-	-
number of species	10	2	7	0	4	0	5	0

Table 6.9 *Short term effects (<2 years) on epiphyte-substrate associations following cable logging and burning. Abundances are (1) rare, (2) moderate and (3) abundant.*

	Blue 2		Blue 24		Pearce 5		Pearce 12	
	pre	post	pre	post	pre	post	pre	post
Asplenium bulbiferum on ledge	1							
Asplenium flabellifolium on ledge	1	1						
Aplenium terrestre on ledge	1							
Dicksonia antarctica on *Dicksonia antarctica*					1			
Grammitis billardieri on *Dicksonia*	2				2		1	
Grammitis billardieri on *Olearia argophylla*	1							
Grammitis billardieri on *Nothofagus cunn.*							2	
Grammitis billardieri on *Atherosperma*							1	
Grammitis billardieri on rocks							2	
Histiopteris incisa on *Dicksonia antarctica*					1			
Hymenophyllum australe on *Dicksonia*							2	
Hymenophyllum flabellifolium on *Dicksonia*	2		2		2			
Hymenophyllum peltatum on rocks							2	
Hymenophyllum rarum on *Dicksonia*	2		2		2			
Hymenophyllum rarum on *Olearia arg.*	1							
Microsorim pustulatum on *Dicksonia*	2		2					
Microsorim pustulatum on fallen *logs*	2							
Microsorim pustulatum on *Olearia arg.*	2		2					
Microsorim pustulatum on ledge	2	1						
Polyphlebium venosum on *Dicksonia*	1		1					
Polyphlebium venosum on *Nothofagus cunn.*							2	
Polystichum proliferum on ledge	2							
Polystichum proliferum on *Dicksonia*					1			
Polystichum proliferum on fallen *logs*					2			
Rumohra adiantiformis on *Dicksonia*	1		1					
Tmesipteris obliqua on *Dicksonia*	1		1		1			
number of epiphytic fern associations	16	2	7	0	8	0	7	0
Acacia dealbata on *Dicksonia*		1		3		1		3
Acacia dealbata on ledge		1						
Acacia sp. on mossy rock					1			
Atherosperma moschatum on *Dicksonia*					1			
Atherosperma moschatum on fallen *logs*					1		1	
Chiloglottis sp. on mossy rocks					1			
Clematis aristata on mossy rocks					1			
Coprosma quadrifida on *Dicksonia*					1		1	
Coprosma quadrifida on ledge	1	1						
Galium australe on mossy rocks					2			
Hydrocotyle sp. on ledge		1						
Noteleae ligustrina on mossy rocks					1			
Pimelea drupaceae on fallen logs			1					
Pomaderris apetala on *Dicksonia*		1		2	2			2
Senecio sp. on *Dicksonia*								
Senecio sp. on ledge		1						
Senecio sp. on mossy rocks					1			
Viola hederaceae on fallen logs							2	
number of epiphytic shrub and herb associations	1	6	1	2	10	1	3	2

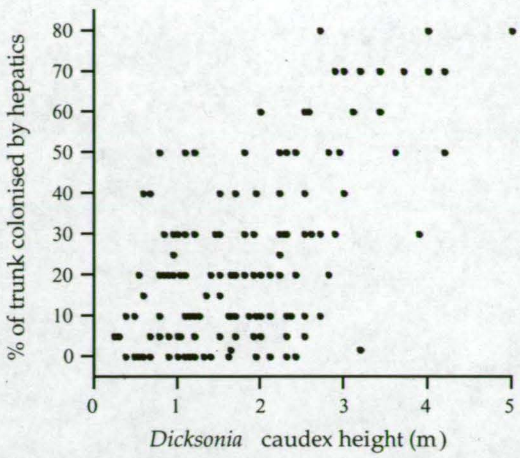
Experiment 3: Study of vascular epiphytic flora of tagged *D. antarctica* individuals following cable logging and slash burning

Pre-logging population characteristics

Seven obligate vascular epiphytic ferns and a further five accidental or hemi-epiphytic shrub and herb species were observed on the 150 (treatment and control) *Dicksonia* caudexes studied (Table 6.10). Prior to logging, vascular epiphytic ferns were the most frequent of the vascular epiphytes on *Dicksonia* caudexes, and of these, *Grammitis billardieri* and *Tmesipteris obliqua* were the two most frequent. Shrub and herbaceous species were infrequent as accidental or hemi-epiphytes prior to logging.

Table 6.10 *The proportional (%) of Dicksonia caudexes (living or dead) colonised by vascular epiphytes prior to logging, after cable logging and after burning at the Pearce 04 experimental area. There were 111 individual D. antarctica caudexes monitored for the 10 year period in the cable logging treatment.*

treatment	pre-logging	post-logging	post-logging and burning				
time (m = months)	0	6 m	12 m	17 m	25 m	36 m	10 years
Species		obligate epiphytes					
Grammitis billardieri	36.9						
Hymenophyllum australe	1						
Hymenophyllum flabellatum	13.5						
Hymenophyllum rarum	8.1						
Microsorum pustulatum	1						0.9
Polyphlebium venosum	2.7						
Rumohra adiantiformis							0.9
Tmesipteris obliqua	25.2	0.9					
accidental and hemi-epiphytes							
Acacia dealbata		0.9	30.6	24.3	24.3	18.0	0.9
Atherosperma	2.7						6.3
Cirsium sp						2.7	
Clematis aristata	0.9						0.9
Coprosma quadrifida	0.9						9.9
Eucalyptus delegatensis				1.8		0.9	
Eucalyptus regnans			16.2	2.7	2.7	6.3	
Gahnia grandis			0.9	0.9	0.9	0.9	
Hydrocotyle sp.				3.6	6.3	16.2	1.8
Hypochaeris radicata						0.9	
Monotoca glauca				0.9	1.8	4.5	1.8
Olearia argophylla				0.9			
Pimelea drupacea			0.9	0.9	0.9	0.9	0.9
Pittosporum bicolor	0.9						
Polystichum proliferum							21.6
Pomaderris apetala	1.8		54.1	45.0	43.2	41.4	0.9
Rorippa stylosa/dictyo.			1.8	0.9			
Senecio bisseratus					2.7	2.7	2.7
Senecio spp.			1.8	0.9	0.9	9.1	
Sonchus sp.		0.9					
Uncinnia tennella							0.9
Viola hederceaea						6.3	1.8
Wahlenbergia spp.						0.9	
Ziera arborescens				2.7	2.7	1.8	
proportion of caudexes occupied by any vascular epiphytic species	50.5	2.7	67.6	62.2	59.5	57.7	39.6



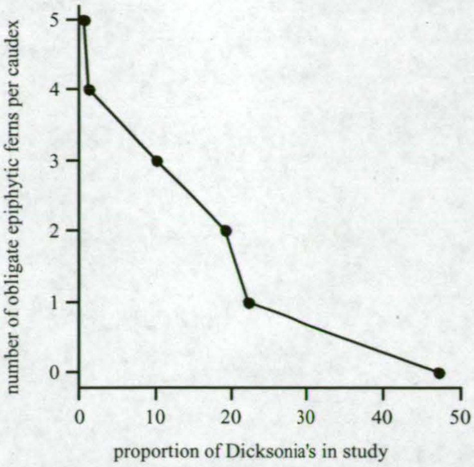


Figure 6.11 *The relationship between* D. antarctica *caudex height and the degree of colonisation by hepatics for the 150 plants in the monitoring study.*	**Figure 6.12** *The relationship between the numberof obligate epiphytic fern species colonising* D. antarctica *caudexes and the total proportion of 150 caudexes monitored.*

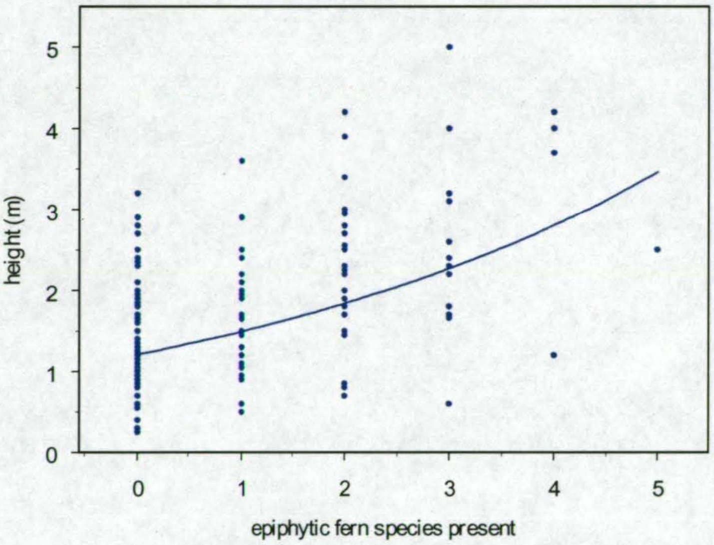


Figure 6.13 *The number of obligate epiphytic fern species occupying individual* D. antarctica *caudexes increased with caudex height prior to logging (n = 150).*

Dicksonia's with a higher cover of both non-vascular and vascular epiphytes on their caudex had an increased richness of obligate epiphytic fern species present (r = 0.73, n=150). Caudex height was only weakly correlated however with the number of obligate epiphytic fern species present (r = 0.49, n=150, Figure 6.13). The number of obligate epiphytic fern species present per caudex prior to logging varied, 47% of the *Dicksonia* specimens in the study had none present (Figure 6.12), those with 2 or more represented only 19% of the caudexes observed. Both non-vascular (e.g. lichens, bryophytes, liverworts) and vascular epiphytes were more abundant on taller specimens of *Dicksonia* (r=0.61, n=150, Figure 6.11 and 6.14).

Figure 6.14 *Obligate epiphytic fern species such as* Hymenophyllum flabellatum *reached their maximum abundance in the unlogged samples on substrates such as* D. antarctica.

Impacts of cable logging

The epiphytic flora of *Dicksonia* caudexes and their micro-habitat relationships expressed through their substrate preferences changed significantly following cable logging and burning (Table 6.10). All vascular epiphytic ferns were eliminated from *Dicksonia* caudexes by cable logging with the exception of one population of *Tmesipteris obliqua* on one caudex. Shrub and herbaceous species as accidental epiphytes which were infrequent prior to disturbance or immediately following cable logging rapidly colonised the burnt *Dicksonia* caudexes. *Acacia dealbata* and *Pomaderris apetala* were the most frequent of these accidental epiphytes. After six months the initial peak of colonising activity of accidental epiphytes onto the burnt *D. antarctica* caudexes declined (Table 6.10), with most of the shrub and herbaceous species being eliminated by 10 years. Minimal recovery (<1% of individual stems) of the vascular epiphytic fern flora of *Dicksonia* caudexes occurred, being limited to two species (*Microsorum pustulatum* and *Rumohra* *adiantiformis*) observed ten years after cable logging and burning. The ground fern *Polystichum proliferum*, which was relatively common on the site prior to logging, at 10 years was the most frequent of the accidental epiphytes of *Dicksonia* caudexes, although it did occupy this niche prior to cable logging.

Initial impacts of cable logging and slash burning

Prior to disturbance approximately half of the caudexes were colonised by one or more species of vascular epiphytic fern. Following cable logging, less than three percent were colonised. After burning however, sixty-seven percent of the caudexes were colonised, not with epiphytic ferns, but with accidental shrub and forb epiphytes.

Only minor changes in abundance in vascular epiphytes of *Dicksonia* caudexes were observed in the undisturbed control during the ten years of observations (Table 6.11). Changes were limited to localised increases in species abundances or recruitment on individual caudexes. No individual populations (defined by their occurrence on a specific caudex) of obligate fern epiphytes were lost during the ten year period in the control area.

Table 6.11 *The proportion (%) of Dicksonia caudexes colonised by vascular epiphytes in the control treatment areas. There were 25 individual D. antarctica caudexes monitored for the 10 year period in the control treatment.*

	pre-treatment	3 years	10 years
obligate epiphytes			
Grammitis billardieri	42.3	46.2	38.5
Hymenophyllum australe	-	7.7	-
Hymenophyllum flabellatum	61.5	61.5	65.4
Hymenophyllum rarum	11.5	15.4	11.5
Microsorium pustulatum	7.7	11.5	3.8
Polyphlebum venosum	-	3.8	-
Rumohra adiantiformis	3.8	7.7	3.8
Tmesipteris oblonga	11.5	15.4	19.2
accidental and hemi-epiphytes			
Aristotelia peduncularis	3.8	3.8	-
Atherosperma moschatum	11.5	19.2	15.4
Blechnum wattsii	3.8	3.8	7.7
Clematis aristata	-	3.8	-
Coprosma repens	-	-	3.8
Dicksonia antarctica	3.8	3.8	0.0
Gaultheria hispida	-	-	7.7
Nothofagus cunninghamii	-	-	7.7
Pittosporum bicolor	3.8	-	3.8
Polystichum proliferum	-	-	3.8

Experiment 4: Impacts of cable logging on vascular epiphytes within streamside reserves

Vascular epiphytic species richness responded in a linear manner to the intensity of cable logging disturbance imposed on their streamside habitats (Figure 6.15). Five to seven species occurred within 20m^2 quadrats sampling undisturbed streamside vegetation, compared to a single species in each of the two high intensity cable logging treatment in the 20 m^2 quadrats. Differences between the moderate cable logging intensity and no disturbance treatments were subtle (Table 6.12, 6.13), compared to the marked differences between high intensity cable logging and moderate intensity cable logging. The abundance of individual obligate epiphytic species was generally greater in the no disturbance treatment, but this was not always the case (Table 6.12).

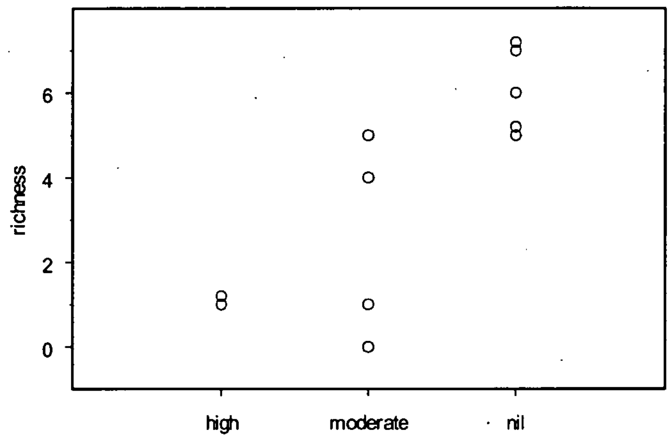


Figure 6.15 *Relationship between the richness of epiphytic vascular epiphytes (20m^2 quadrats) and the intensity of cable logging (high – logging gap, moderate – retained portion, and nil - control) within streamside reserves at Roses Tier, north eastern Tasmania.*

Table 6.12 *Relationship between the mean abundance of obligate epiphytic vascular epiphytes (20m^2 quadrats) and the intensity of cable logging within streamside reserves at Roses Tier, north eastern Tasmania.*

	logging gap (4)	retained portion(2)	control (6)
number of quadrats (n)			
Asplenium bulbiferum			0.085
Asplenium terrestre			0.04
Grammitis billardieri		0.23	0.86
Hymenophyllum australe			1.6
Hymenophyllum flabellatum		9.0	6.6
Hymenophyllum rarum		0.2	4.2
Microsorum pustulatum	0.02	0.2	0.12
Polyphlebium venosum		5.3	4.8
Rumohra adiantiformis			0.21
Tmesipteris obliqua	0.02	0.2	0.15

Table 6.13 *Indicator species analysis (Dufrene and Legendre 1997) for vascular epiphytes (20m² quadrats) and the intensity of cable logging within streamside reserves at Roses Tier, north eastern Tasmania. Higher indicator scores indicate greater association with that treatment. No indicator species were associated with the high intensity cable logging treatment.*

	treatment with maximum indicator value	logging gap (4)	retained portion(2)	control (6)
Asplenium bulbiferum	control	0	0	67
Hymenophyllum australe	control	0	0	67
Grammitis billardieri	control	1	25	59
Hymenophyllum rarum	control	0	2	48
Rumohra adiantiformis	control	0	0	33
Microsorum pustulatum	control	1	28	28
Asplenium terrestre	control	0	0	17
Hymenophyllum flabellatum	moderate disturbance	0	58	42
Polyphlebium venosum	moderate disturbance	0	62	25
Tmesipteris obliqua	moderate disturbance	1	28	21

The presence of epiphytic ferns in each of the three streamside cable logging treatments was correlated with the presence of appropriate substrates (Table 6.14). In general, greater disturbance associated with the more severe cable logging disturbances of streamside reserves removed living substrates and therefore the microhabitat for epiphytic species.

Table 6.14 *Abundance of obligate fern epiphytes (rare, moderate and abundant) on various host substrates in the three disturbance categories two years following cable logging adjacent streamside reserves at Roses Tier, north eastern Tasmania.*

fern species and substrate	logging gap	retained portion	control
Asplenium bulbiferum on *Atherosperma*			rare
Asplenium bulbiferum on *Atherosperma* trunk			rare
Asplenium bulbiferum on *Dicksonia*			moderate
Asplenium terrestre on dolerite bolder			rare
Grammitis billardieri on *Atherosperma*			rare
Grammitis billardieri on *Coprosma quadrifida*			rare
Grammitis billardieri on *Dicksonia*	rare	rare	abundant
Grammitis billardieri on dolerite bolder		rare	moderate
Grammitis billardieri on fallen logs			moderate
Grammitis billardieri on surface roots			moderate
Grammitis billardieri on twigs and branches		moderate	
Hymenophyllum australe on *Dicksonia*			rare
Hymenophyllum australe on dolerite bolder			abundant
Hymenophyllum cupressiforme on dolerite bolder			moderate
Hymenophyllum flabellatum on *Dicksonia*		abundant	abundant
Hymenophyllum flabellatum on surface roots			rare
Hymenophyllum rarum on *Dicksonia*		abundant	abundant
Hymenophyllum rarum on dolerite bolder			abundant
Hymenophyllum rarum on twigs and branches		moderate	
Microsorum pustulatum on *Atherosperma* trunk		moderate	rare
Microsorum pustulatum on *Dicksonia*		rare	rare
Microsorum pustulatum on fallen logs			rare
Microsorum pustulatum on *Nothofagus* trunk			rare
Microsorum pustulatum on twigs and branches	moderate		
Polyphlebium venosum on *Dicksonia*		abundant	abundant
Rumohra adiantiformis on *Dicksonia*		rare	moderate
Rumohra adiantiformis on surface roots			rare
Tmesipteris obliqua on *Dicksonia*	moderate	moderate	abundant

One of the most commonly utilised substrates for epiphytes in streamside reserves are the caudexes of *Dicksonia*. Cable logging disturbed individual *Dicksonia*'s reducing the number of living stems in streamsides at Roses Tier (Chapter 5). The reduction in the number of *Dicksonia* stems was directly correlated with the number of fern epiphytes using *Dicksonia* as a substrate (Figure 6.16). Less disturbed sites, had up to seven epiphytic fern species present on *Dicksonia* within the 10 x 2m quadrats, the most disturbed sites two or less epiphytic fern species present. Remnant *Dicksonia* in poor health often supported small epiphytic populations, also in poor health (Figure 6.17).

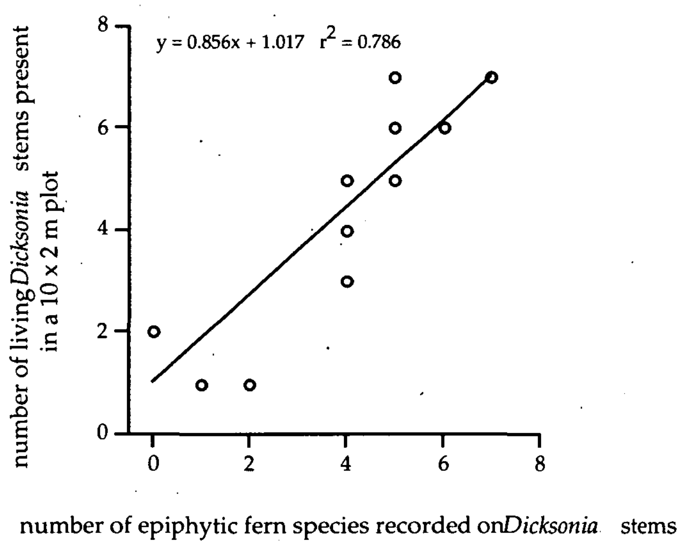


Figure 6.16 *The number of epiphytic fern species recorded on living Dicksonia antarctica trunks within streamside reserves at Roses Tier as a function of the density of trunks available for colonisation.*

Figure 6.17 *Remnant* D. antarctica *in poor health on an exposed stream edge* at *Roses Tier 137g two years after canopy removal during cable logging.*

6.4 Discussion

Vascular epiphytes were observed to be highly responsive to the method and intensity of logging disturbance, in particular the way in which their micro-habitats, or substrates were disturbed and how these substrates re-established in the first three to five decades following disturbance. In most cases, intensive logging followed by slash burning, led to the near complete loss of the vascular epiphytic flora, the exception being localised micro-habitats such as rock ledges which escaped the direct effects of logging and burning.

Chronosequence analysis indicated that while the general patterns of epiphytic species re-colonisation following logging were similar, wet forests logged with ground-based machinery exhibited both a significantly increased rate of species recruitment, relative to cable logging, for obligate and facultative epiphytic ferns and a significantly increased rate of re-establishment of epiphytic species cover in the first 30-50 years after disturbance. Indicator species analysis also suggested that ground-based logged wet forest, at 30 years, had a greater complement of obligate epiphytic fern species (generally considered indicative of moist and protected environments e.g. *Hymenophyllum* species), whereas in the cable logged sites, relatively desiccation tolerant early colonist species such as *Grammitis* and *Asplenium flabellifolium* were the indicator species. Ground-based logged sites also exhibited a significantly greater diversity of colonised substrates than cable logged sites, and individual species, such as *Microsorum pustulatum*, grew on a greater range of living and non-living substrates after ground-based logging. The above conclusions were drawn from wet forests during the first three decades following logging. However, if comparisons were extended to the older (30-54 year old) cable logged wet forest in the same environment, the differences in epiphytic species composition between logging treatments became less significant with time. The longer term amelioration of logging intensity effects, evident in the first decades following treatment, has been noted elsewhere for epiphytic bryophytes (Olsson and Staaf 1995).

The survival of individual populations of vascular epiphytes within areas subject to logging and burning, and their patterns of recolonisation were inextricably linked to the fate of their substrates. The ability of vascular epiphytes to recover post-logging and burning was therefore dependent on the status of their substrates in the regenerating vegetation. This is generally consistent with the broader literature on the ecology of vascular epiphytes where light levels, position in canopies, and moisture and nutrient availability (Benzing 1990) determine species distributions.

Micro-habitats colonised by vascular epiphytes, if retained during logging, can provide a refuge for populations (King and Chapman 1983, Delgadillo and Cardenas 1995), alternately the loss of these substrates can lead to a loss of specialised epiphytes (Ramsay *et al.* 1987 in Norris 1990). The patterns of survival and recolonisation in this study are also consistent with those described in the literature for the effects of fire. Following burning, epiphytic recolonisation is dependent on the re-establishment of their substrates (Equihua 1992, Robertson and Platt 1992), the nature of the fire, the type of substrate utilised (Longan *et al.* 1999), the availability of fire-protected refuges and the time necessary for the microclimate of the stand to redevelop.

Dicksonia antarctica represented the most significant living substrate for vascular epiphytes in mature and logged forest in the southern Tasmanian wet forests. Tree-fern caudexes have a range of characteristics that make them suitable for epiphytic germination (Stone 1965) and colonisation. These include an ability to retain moisture and nutrients (Prasad and Fietje 1989), favourable texture (Beever 1984), longevity, stability and resistance to fauna disturbance (Howard 1973). The importance of *Dicksonia* and tree-ferns as substrates for the germination of a range of tree species in mature forest (Galbraith 1947, Mount 1979, Read and Hill 1983, Cook and Drinnan 1984, Page and Brownsey 1986, Newton and Healy 1989, Ough and Ross 1992, Chesterfield 1996, Ashton 2000), for vascular epiphytes (Petrie *et al.* 1929, Hassall and Kirkpatrick 1985), ferns (Kelly 1985, Heatwole 1993) and bryophytes (Ashton 1986, Beever 1984) has been repeatedly documented, however their role in regenerating vegetation has not. While a reduction in the biomass of tree-ferns can lead to a reduction in the epiphytic fern flora (Medeiros *et al.* 1992, 1993), and alternately, an increase in the abundance of tree-ferns can lead to an increased frequency of epiphytes (Drake and Mueller-Dombois 1993), no quantitative data has previously been presented in the context of logging or using permanently marked individual *Dicksonia* caudex surfaces. Cable and ground-based logging followed by silvicultural burning led to the complete loss of the epiphytic flora on *Dicksonia* caudexes, due to a combination of physical disturbance and desiccation following canopy removal. Larger individual *D. antarctica*, which carried a greater diversity of epiphytic ferns per plant than smaller individuals, were particularly prone to mortality following cable logging and burning. In the immediate (1-3 year) post-logging and burning environment locally referred to as the ash-bed (Pryor 1960), instead of vascular epiphytic ferns occupying the *D. antarctica* caudexes, this substrate was colonised by tree, shrub and herbaceous species which rapidly senesced in the first three to ten years. These accidental epiphytes probably arose as a result of exposure, where the caudexes were subject to increased light, soil mixing and wind dispersed propagules. This process is described as exhibiting a strong influence on epiphytes in disturbed rainforest (Kantvilas 1988, Johansson 1989). Re-colonisation by obligate epiphytic ferns was delayed in these permanently monitored caudexes until ten years following cable logging and burning, and in the chronosequence analysis, until 26-29 years following cable or ground-based logging. The more drought tolerant obligate epiphytic fern species (eg *Grammitis, Microsorium*) were the first to re-colonise the remnant *D. antarctica* caudexes. As the surface area of *Dicksonia* caudexes increased and regrowth forest developed towards canopy closure, the number of obligate epiphytic ferns colonising *Dicksonia* trunks increased. At the individual substrate level, the relationship between host size and epiphytic species richness was not strong, a trend observed elsewhere (Zotz 1997). Some *Dicksonia* caudexes remained uncolonised by epiphytic ferns while neighbouring individuals were densely colonised. Epiphytic patterns of occurrence can be highly localised (Jarman and Kantvilas 2001).

Bedfordia salicina, Olearia argophylla, Pomaderris apetala, Acacia dealbata and *Phebalium squameum* are the tall shrub species with bark that acts as a substrate for obligate epiphytic ferns. *Olearia argophylla* provided the most frequently colonised substrate for epiphytes because it has a spongy bark. Typically it is well colonised by epiphytic ferns and bryophytes (Ashton 1986), providing a successful germination medium for epiphytes such as *Hymenophyllum cupressiforme* (Stone 1965). In the chronosequence analysis, the frequency and abundance of *Olearia argophylla* was low until the third decade after logging and burning, especially on ground-based logged sites. *Olearia argophylla* was first colonised by epiphytic species in 29 year old cable logged regrowth, and was not colonised by epiphytic species in any of the 5-30 year old ground-based logged regrowth examined. *Ctenopteris heterophylla*, an epiphytic fern species with a preference for *Olearia argophylla* bark (Jarman *et al.* 1986) was the only epiphytic fern species absent from the ground-based logged forest. Epiphytes colonising other tall shrub species exhibited a preference for the lower portions of their trunks where the bark surface was more irregular with small buttress growths. Lower trunk micro-habitats (eg. Akinsoji 1991) did not develop on these shrub species for at least 30 years.

The effect of cable logging of different intensities on vascular epiphytes was evident in disturbed streamside reserves for up to five years. Species richness and cover of vascular epiphytes declined with increasing habitat disturbance. Within the most disturbed streamside habitats, the vascular epiphytes of *D. antarctica* caudexes were replaced by short lived ruderal colonists. Few substrates were available for colonisation in the most disturbed streamside habitats. Those epiphytes that persisted did so on remnant *D. antarctica* caudexes, their survival presumed to have been mediated by the mesic stream bank environment. Other substrates colonised in streamside reserves, including root mounds of dominant trees and moss covered fallen logs, were uncommon after significant cable logging disturbance.

In general, those substrates that were preferred by vascular epiphytes in undisturbed forest, we also preferred in regrowth forest where they were available. Where living substrates were absent or infrequent, fallen logs and boulders were colonised. Four times more species colonised fallen logs after logging (irrespective of method) than other substrates, a reflection more of the absence of alternative substrates than the greater suitability of fallen logs. As *E. regnans* forests age, the diameter and moss cover of fallen logs will increase (Lindenmayer *et al.* 1999), which in turn increases the surface area available for epiphyte establishment. Compared to other substrates, logs, with their increased moisture retention, accumulated litter and bryophyte cover (Ashton 1986), provide a favourable regeneration niche (Grubb 1977), especially following clearfelling (Huffman *et al.* 1993). The volume of fallen logs left as residue following clearfell logging of *E. regnans* forest is high on a global scale (up to 1000m^3/ha, Lindenmayer *et al.* 1999). In Southern Tasmanian wet forests logging residue varies from 120-500 t/ha (this study) to 176 t/ha (Grove *et al* 2002).

The inferred patterns of succession of obligate epiphytic ferns following logging and burning are in accordance with the tolerance and facilitation model of Connell and Slatyer (1977), a pattern observed elsewhere in epiphytic succession (Tewari *et al.* 1985). As new species appeared, they did not displace the earlier colonists. The obligate epiphytic fern flora on different substrates tended to converge as succession progressed, as the initial differences between substrates, such as litter accumulation, bark roughness and moisture retention, were reduced. The patterns described in this thesis are generally consistent with those described in relevant wet forest types subject to intensive logging in southern Australia (Cook and Drinnan 1984, Harris 1986, Harris 1989, Mueck and Peacock 1992, Ough and Ross 1992, Hickey and Savva 1992). The greater proportion of epiphytic species recolonising after logging in this study compared to the studies above is attributed to the greater age of regrowth surveyed, the detailed re-scoring on permanent plots and the targeted methods of sampling undertaken.

Benzing (1990) proposed that epiphytic succession could be divided into three phases, the influx of relatively stress tolerant colonists, substratum conditioning prior to arrival of more vulnerable humiphilous (moisture dependent) taxa, and progressive community diversification. In the wet forest environments described in this study, the first phase occurred at 11-20 years after logging and burning, and was characterised by *Grammitis billardieri* and *Microsorum pustulatum* colonising logs and occasional *D. antarctica* caudex. The second phase occurred at 20-40 years after logging and burning and saw the arrival of species with a preference for moist habitats, including the Hymenophyllaceae, and coincided with the emergence of understorey substrates such as *D. antarctica* caudex for colonisation. While many of the obligate epiphytes characteristic if mature wet forests first appeared during this stage of succession, they were significantly less frequent than in mature forest. The third stage, progressive community diversification, was observed on the oldest of the chronosequence samples aged >41 years. Benzing (1990) considers this phase to be dependent upon processes that alter substrates and encourage plant immigration. During the community diversification phase, bark substrates were observed to provide surface conditions more conducive to the development of epiphytic ferns and the interception of diaspores, the number of *D. antarctica* present increased, cut logs continued to decompose and collect litter, and boulders also trapped litter as their bryophyte cover increased. Infrequent obligate epiphytes such as *Tmesipteris obliqua* and *Polyphlebium venosum*, species with a marked preference for *D. antarctica* caudexes (Page 1979, Stevens and Tyson 1989, Kelly 1985, Jarman *et al.* 1986, Ashton 1986) and adapted to low light intensities (Johnson *et al.* 2000) appeared during this phase. The diversity of substrates colonised by vascular epiphytes also peaked.

The time required for the vascular epiphytic flora to return to pre-disturbance levels is difficult to estimate. The age of the mature forest sampled was in the order of 150-300 years, based on the growth stage of the overstorey eucalypts, while the oldest silvicultural regrowth sampled was 54 years. The cover of vascular epiphytic species at 54 years was approximately 8-10% of the mature forest, but the increase in cover over time evident from the chonosequence was not constant, and the cover of epiphytes in mature forests was also variable. Harmon (1989) observed that bryophytes epiphytic upon fallen conifer logs reached a steady state mass after 91 years. Dettki and Esseen (2003) estimated that epiphytic lichens would require in excess of 110 years to recover their pre-disturbance biomass. It is likely that similar time frames would exist in the Tasmanian wet forests studied. It is questionable whether rotation periods in the order of 80-100 years that are being used in wet forests are sufficient to allow the long term persistence of local populations of obligate vascular epiphytes.

Various authors have proposed that micro-habitats occupied by epiphytes should be protected or buffered from disturbance to assist recolonisation (eg. Peck and McCune 1998). This would minimise direct damage to sensitive plant taxa and provide a ready source of propagules for species re-establishment and to 'lifeboat' (Vanha-Majamaa and Jalonen 2001) such taxa over the regeneration phase of stand development where conditions would otherwise be unfavourable to their persistence. These micro-habitats have been referred to variously as understorey islands (Ough and Murphy 1998), key habitats (Hansson 2001), small retention tree groups of late successional areas (Vanha-Majamaa and Jalonen 2001), habitat islands of woody vegetation (Frego 2000), sensitive understorey vegetation (Lindenmayer and McCarthy 2002) or critical habitat components (Lindenmayer and Franklin 1997). Substrate retention during logging, even in small refugia, will assist in the maintenance of epiphyte populations (Raivio 1996), in particular where decomposing logs are retained, or where advance regeneration is protected (Frego 2000). Proposals for substrate retention and habitat islands (Ough and Murphy 1998a, Hickey *et al.* 2001), or lengthened rotation periods to encourage the persistence and recolonisation of vascular epiphytes in production forests (Hickey 1994), will require detailed forest management rarely demonstrated within current clearfell and slash burn silvicultural regimes.

Chapter 7 Discussion

Cable and ground-based logging systems differ in a multitude of ways in their effects on vegetation composition and cover, some of which can superficially be considered to be advantageous or disadvantageous to the vegetation. In general, the vegetation response to the two logging treatments has more attributes in common than differences, and the differences, when examined in detail, are contingent on the age of the regrowth forest surveyed, its pre-logging composition, the type of response variables used (eg compositional dissimilarity, richness or cover), and whether total species composition, functional groups, or even individual species are examined. Local or chance effects then come into play, the role of micro-habitats in the regenerating vegetation can be very significant at the localised scale, the skill of the cable logger and supervising forester can effect the degree to which the forest practice guidelines are implemented and any adverse effects of either cable or ground-based logging result at a particular logging operation.

All of the experimental comparisons of cable and ground-based logging have adopted a common framework in this thesis, that is productive forest types being logged using clearfelling and almost always silvicultural burning. Partial logging systems or various green tree retention systems are yet to be trialed in Australia using cable techniques. With the exception of the dry forest experiment at Wielangta where silvicultural burning was not applied, the comparison of logging systems therefore is made on the basis that the vegetation is all clearfelled and burnt, and regenerated to an overstorey composition resembling the initial composition. In this respect comparisons can be made with cable and ground-based logging systems overseas using clearfell regimes followed by site preparation either by silvicultural burning or by enrichment planting with understorey control. The experiments conducted also permit ready comparison with the majority of the Australian literature examining vegetation composition and cover change following intensive logging and burning.

Differences between the two logging treatments, cable and ground-based logging, were at their greatest immediately after disturbance when the physical environmental conditions are most divergent. As those physical environmental conditions begin to ameliorate, so do the divergent floristic trends resulting from either logging method. This was essentially the conclusion reached by the very few existing comparative studies of cable and ground-based logging (Pothier 1996, Woodward 1986), although they only reported relatively course trends based on tree stocking and understorey cover in regenerating stands. In the short term, allogenic factors (eg., degree of soil and habitat disturbance) related to logging method (cable versus ground-based) were the primary influence on species composition in the wet forest stands examined in southern Tasmania.

In general terms, no successionary model fitted the patterns of persistence, recruitment and community sorting observed in the chronosequences. Competitive sorting models predict that community structure will change during the course of succession, with chance establishment predominating early in succession, and competitive sorting into microhabitats increasing with time. Although the frequently cited models of secondary succession were not specifically tested in this study, generally the findings of this study are consistent with the outputs of such models. This study also agrees with most other logging studies in that post-logging regeneration patterns are consistent with an initial floristic composition model (Bastow-Wilson *et al.*. 1992) with some component of a relay (Egler 1954) floristics and tolerance model (Connell and Slatyer 1977). The wet forests studied exhibited strong community dynamics following either cable or ground-based logging, a trend also described for wet forests following wildfire (Jackson 1968, Ashton 1976, 1981). The Tasmanian wet forests studied also exhibited successionary patterns broadly analogous for European and North American forests subject to intensive logging. Differences in the response between this study and the European and North American studies are only apparent where there is a greater ecological role for ungulate grazing or the young regrowth stands are subject to intensive understorey suppression at canopy closure. The common process constraining the floristic response in both the Australian and overseas literature in intensively logged stands is the role of canopy closure in the first two decades after logging suppressing diversity, and the role of soil disturbance intensity and site impacts generally in predisposing the site to establishing species whose growth traits are advantaged by either intensive or less intensive site disturbance.

Where this study differs from previous model interpretations of secondary succession is the greater emphasis given to the facilitation model of Connell and Slatyer (1977), the extent to which one or more species enable the establishment, growth or development of another species. Facilitation is clearly of greater significance to obligate vascular epiphytes reliant in whole or in part on the provision of appropriate living substrates for physical support. The reasons for there being a greater emphasis placed on facilitation are:-

1) the chronosequences described were longer (by 20 years) than previous studies examining logging impacts in wet forest environments in southern Australia, and
2) ecological census techniques developed for vascular epiphytes have revealed the importance of facilitation processes in the re-establishment of these late successional specialists.

While previous studies described the importance of the general lack of epiphytic species re-colonisation following ground-based logging (Ough and Ross 1992, Hickey 1994, Ough 2001), their chronosequences lacked the longer time frame to be able to describe the processes enabling the eventual re-colonisation of vascular epiphytes, and their vegetation survey techniques did not examine the epiphytic micro-habitats necessary to evaluate the significance of forest management practices on this recolonisation process.

The integration of studies of chronosequences, permanent plots and individual species has led to the prediction of the longer term consequences of different logging techniques and the concurrent testing of those predictions with repeated measurement of permanent plots and permanently tagged individual plants. The integration of these studies has also enabled some of the basic tenets of our existing understanding of patterns of secondary succession following intensive logging and burning to be tested. For example, patterns of species persistence following logging disturbance and burning were frequently deterministic. Individual vascular epiphytes persisting following disturbance were reliant upon localised micro-habitats such as protected boulder faces and unburnt piles of woody debris to avoid logging disturbance. Trends such as these required detailed permanent plot monitoring to recover, and were generally not evident in the existing literature using chronosequence analyses or logged versus unlogged plot comparisons. The use of permanent plot monitoring within the region and forest types studied using chronosequence surveys indicated a proportion of the late successional specialists predicted to recolonise in the 10 to 20 year post-logging period from the chronosequence may in fact colonise at a slightly earlier stage. The study of permanently established control plots to provided a reference point for the magnitude of floristic change following logging then highlighted the fact that the logging treatments need to be assessed not only against their pre-logging state, but also against natural dynamics in un-managed stands. The fact that the control plots in the dry forest experiment at Wielangta over a two year period decreased in overall species richness while the cable logged plots decreased and the ground-based logged plots increased highlights the complications involved in interpreting data such as this.

Detailed studies of the slow growing fern *D. antarctica* via chronosequence surveys, permanent plot studies and 150 individual plants monitored for 10 years provided considerable insight to the detailed process of logging disturbance and the factors influencing survivorship. The medium term consequences of cable and ground-based logging were broadly similar for this species, while ground-based logging led to greater mortality, it also created the microsite conditions for a more rapid recruitment of new individuals. However, the significance of silvicultural burning on population persistence was unexpected. Previous research suggested it was the physical effects of ground-based logging disturbing individual stems that led to most of the mortality (Ough and Murphy 1997). This study indicated the opposite. Most of the population was resilient to the physical effects of cable logging, it was the silvicultural burning that led to the greatest proportion of mortality over the ten year period. A number of theories concerning persistence of *D. antarctica* following logging were also demonstrated to be incorrect, for example

1) the belief that individuals became unbalanced by burning which led to their death (Neyland 1986) was not the case. The ten year observation period identified that most of the specimens which became unbalanced and were scored as dead on the forest floor were in fact already upright and dead.
2) a reduction in regenerative vigour occurred following logging , firstly as a result of burning newly regenerating fronds in the regeneration burn, and secondly as a result of shading at canopy closure.
3) the absence new recruitment over a ten year period at one site (Pearce) casts doubts over the long term viability of some *D. antactica* populations. Broadly similar conclusions have been reached in overseas studies of slow growing species which exhibited low levels of coppice or clonal regeneration and little or no seedling recruitment following logging (Smale 1982, Jules 1998), and more locally, from studies of *Xanthorrhoea*, a genus which also relies on resprouting following fire from a stout caudex (Curtis 1998, Lamont *et al.* 2000). The notion that *D. antarctica* is a keystone species in wet forests (Peacock and Duncan 1994), and acts in a similar manner to other tree ferns which are catalysts for the colonization of other species during succession (Arens and Baracaldo 1998) has merit, even though the whole indicator species concept is problematic (Lindenmayer *et al.* 2000, Lindenmayer and Franklin 2002).

The reduction of *D. antarctica* density following logging to less than one-fifth of the density in mature forests, and the complete removal of the portion of the population estimated to be <30 years old due to the inability of younger plants to survive logging and burning disturbance, has broad implications for the sustainable management of the species in production forests. Statewide population estimates of *D. antarctica* (Forest Practices Board 2001), based on extremely limited empirical data collected in mature forest are being used as the basis of a regulated industry for the species removal and sale for horticultural purposes. The reduction in density of this species recorded from permanent plot studies suggest that the statewide population estimates may be overly optimistic considering the proportion of the Tasmanian forest estate which has been subject to logging and burning. Since these statewide estimates form part of the sustainable yield calculations for the regulated removal and sale of *D. antarctica* for horticultural purposes nationally and internationally (Forest Practices Board 2001), the basis of the predicted sustainable yields of *D. antarctica* populations in the wild may require re-assessment.

The significance of the greatly reduced population of *D. antarctica* following cable or ground-based logging also lies in its role as a preferred substrate for the recruitment of a wide range of epiphytic species. Chronosequences for vascular epiphytes show reduction in recolonisation surfaces in regrowth forests, as has been highlighted previously (eg Mueck and Peacock 1992, Ough and Ross 1992), however this had not been quantified or tested experimentally before. While *D. antarctica* was a preferred substrate for vascular epiphytes in undisturbed forests, fallen logs and boulders were however the first substrates to be recolonised in young regrowth. Immediately following cable logging, *D. antarctica* caudexes were colonised by short lived ruderal and soil seed bank regenerators, and were not colonised by vascular epiphytes until ten years after cable logging and burning, at which time boulders and fallen logs had already been colonised by vascular epiphytes. The temporal sequence described for substrate preferences of vascular epiphytes following cable and ground-based logging is unique in the Australian literature, and while reinforcing the significance of *D. antarctica* as a substrate for recolonisation following intensive logging and burning, it provides a context to its significance as a substrate for epiphytic colonisation in logged stands that is lacking in the literature compared to the previous research which does not consider the role of fallen logs or boulder substrates. These results do however highlight the significance of ecological legacies from the previous forest stand in providing recolonisable substrates for late successional specialist species. This result provides a new understanding of the importance of ecological legacies in southern Australian production forests beyond the published studies on structural components such as stag trees and hollows used by arboreal fauna (eg Lindenmayer and Franklin 1997, Lindenmayer *et al.*. 1990a).

The current study is the most detailed yet undertaken on the vascular epiphytic flora of southern Australian production forests, and is the first to use control plots to:

1) examine changes in undisturbed sites,
2) use pre-logging sampling, and
3) examine the detail of epiphyte-substrate associations and their significance in regrowth forests.

The emphasis given to the examination of the whole vascular epiphytic community - the species, their abundances, the regeneration of their substrates and their recolonisation patterns on those substrates, has been developed from methods employed in studies in tropical forest canopies or in the epiphytic lichen and bryophyte communities in northern Europe. These approaches have provided a more detailed description of the effects of different logging treatments on species composition and of the some of the processes that determine the recolonisation of late successional specialists. Monitoring of epiphytic colonisation on a multitude of single surfaces avoids one of the main limitations of the literature on succession in vascular epiphytes, the retrospective approach (Benzing 1990). Using both monitoring and chronosequence methods, it was evident the re-establishment of species characteristic of late successional environments into the early successional regrowth environment was dependent on the availability of suitable microhabitats such as coarse woody debris and the bark surface of favoured shrub species. While some of these microhabitat features are ecological legacies of the pre-harvest stand structure, others are themselves dependent on the recruitment, survival and growth processes in the regenerating forest.

Implications of this research

Approaches to forest management in Australian forest types where cable logging or other intensive logging practices are applied are highly variable across jurisdictions and are largely dictated by state government legislative and policy imperatives (Forest Practices Board 2000, NRE 1996). Specific recommendations concerning forest practices would need to be made on a state by state basis, which is beyond the scope of this chapter. The results of this thesis however presents a number of general implications for the approaches to forest management and the current paradigm that sustainability can be monitored and reported via Montreal Process sustainability indicators.

Early results from this thesis were made available to the group revising components of the Tasmanian Forest Practices Code (Forest Practices Board 2000). A number of the new provisions in the Code are drawn directly from this research, both in terms of cable logging practices and flora conservation measures in general.

Species associated with late successionary environments can potentially be maintained more effectively in managed landscapes with the use of patches of retained habitat or understorey islands. While this concept has recently been subject to field experimentation (Ough and Murphy. 1998a, Hickey *et al.* 2001), it is not a new concept for forest managers (eg. Spies 1991). Epiphyte rich microhabitats (eg dense areas of *D. antarctica*, rock ledges, course woody debris or mature *Olearia argophylla*) could be excluded from logging damage or potentially silvicultural burning. The increased maintenance of course woody debris in a range of size and decay classes within coupes would provide a ready substrate for epiphyte recolonisation and may ameliorate the micro-environment of the young regenerating forest at a very localised scale. Maintaining the various stages of decay is particularly relevant as it distinguishes the characteristics of coarse woody debris between managed and old growth forest (Franklin 1992). Maintaining a range of decay classes may involve minimising the intensity or extent of slash burning in some parts of the coupe.

A common recommendation of researchers examining the biodiversity impacts of intensive forest management regimes where landscapes are dominated by recent clearfells and stands in the early stages of regeneration or stem exclusion has been for either the extension (Curtis 1997) of forest stand rotation periods (Cunningham and Cremer 1965, Kantvilas *et al.* 1985, Brown *et al.* 1988, Horne and Hickey 1991, Taplin *et al.* 1991, Hickey and Savva 1992, Busing *et al.* 1995, Brown 1996, Dettki and Esseen 2003, Taylor *et al.* 2003) or stand structural manipulation as a means of enhancing the area of late successional forest habitats available to ecological specialist species. More recent proposals include rotating reserves (Brown *et al.* 1988) and habitat islands (Hickey *et al.* 2001). While any approach is likely to involve trade offs between specialist species of different seral stages (Boughton and Malvadkar 2002) all approaches can potentially assist with the persistence of epiphytic species and the mechanisms with facilitate their re-colonisation following clearcutting (Sillett and Goslin 1999, Dettki and Essen 2003).

Forests managed for sustainable volumes of sawlog and veneer are done so on rotations of 80-85 years in Tasmania. This rotation length is also dependent on the intensive management by thinning of the regrowth stand. Thinning by cable or ground-based technology imposed a disruption to the process of secondary succession in these forests, significantly modifying stand structure and in the case of ground-based thinning, encouraging the invasion of herbs and other short lived ruderals. Following cable thinning a dense ground fern stratum of *Histiopteris* and *Hypolepis* would typically form, a pattern observed in numerous overseas studies following thinning. The increased dominance of clonal spreading species following logging can act as a barrier to the establishment of other understorey species (Messier and Kimmins 1991), delaying further forest development (Walker 1994). In some cases dense ground fern understoreys emerging following logging have been considered 'interfering' and subject to chemical control (Horsley 1994). Regrowth thinning appeared to have ecological consequences similar to those described locally by Cunningham and Cremer (1965) following double burning. Where shrub (eg *Pomaderris, Olearia argophylla*) recolonisation is interupted by site disturbance associated with thinning, obligate epiphytic species relying wholly or in part for support on these living bark substrates will be disadvantaged.

The forest practices associated with cable logging were of greatest relevance when logging adjacent to or through streamside reserve vegetation. In a number of the streamsides examined, there was evidence of on-going ill-health if individual, trees and decline of the remnant stands five years after logging was completed. In some cases cable logging had successfully avoided damage to the retained streamside vegetation yet they were subsequently disturbed during fire break construction or incursions from silvicultural burning. In general however, revisions to the forest practices guidelines since the late 1980's have seen a significant improvement in the maintenance of streamside vegetation adjacent to cable logging operations. From the results of this study, the following measures are recommended for consideration during the next revision of the Forest Practices Code as assisting the long term viability of retained streamside vegetation;

1) minimise disturbance to the edges of streamside reserves, either due to tail spar tracks, fire breaks, slash accumulation, damage to individual trees or un-intentional tree falls,
2) consider aspect when planning streamside reserve widths, reserves edges with a north or westerly aspect may benefit from wider reserves,

3) further minimise cable logging and burning disturbance to class 4 watercourses within the net harvestable area, especially in dry forest environments where these watercourses may represent the only habitat for mesic species,
4) if logging disturbance to class 4 watercourses is unavoidable, avoid falling trees into the watercourse and generally attempt to maintain as much of the shrub cover as possible. This will help maintain the constancy of the watercourse micro-environment in the absence of the eucalypt overstorey,
5) silvicultural burning practices occasionally resulted in more damage to retained streamside vegetation than the cable logging operation. Where burning is required, avoid burning within or near watercourses. Watercourse or streamside vegetation will generally recover from logging disturbance more quickly if unburnt,
6) mechanised firebreak construction along streamside reserve perimeters may be more damaging to soil structure and retained vegetation in the long-term than the potential risk of small, localised fire intrusions,
7) minimise direct disturbance to the trunks of trees along the reserve perimeter, (especially *Nothofagus*) as small wounds may act as infection points for myrtle wilt disease. Similarly if it is necessary to pull cables through streamside vegetation, avoid disturbance to *Nothofagus*, and
8) avoid heaping soil, rocks and logging slash against trees on the reserve perimeter. Trees such as sassafras (*Atherosperma*) become more susceptible to windthrow in these circumstances.

Conclusion

It is probable that the conclusions reached by Tappeiner *et al.* (1997) in a review of silvicultural systems and regeneration practices in the Pacific Northwest are just as relevant to the environments studied in this thesis, that is, that in many cases natural succession in regenerated forests will not produce the kinds of stands and habitats as it has in the past. Natural and regenerated stands differ with respect to key attributes such as gross stand characteristics and woody debris (Lindenmayer *et al.* 1990c, Lindenmayer and Franklin 1997, Lindenmayer *et al.* 1999a, Lindenmayer and McCarthy 2002) and vascular plant floristics (Hickey 1994, Ough 2001). While logging disturbance should not attempt to mimic natural disturbance (Lindenmayer and Franklin 2002), greater congruence between natural and imposed disturbance regimes could potentially promote biodiversity conservation. While the oldest of the chronosequence stands sampled (54 years) indicated that most, if not all, of the late successional vascular plant species present in mature forests existed in these regrowth forests, important differences remained, in terms of vegetation structure, the reduced cover of vascular epiphytes and of species such as *D. antarctica*. These results extend the previous data from related forest types where researchers had simply compared logged and unlogged forests and concluded that a range of late successional species were absent from regrowth stands. The study areas chosen in Tasmania are almost unique in southern Australia in having a long history of intensive forest management using logging and silvicultural techniques comparable with current technology. The insights provided in this study were generally not available in previous chronosequences using 20-30 year old regrowth (Mueck and Peacock 1992, Hickey 1994) or more limited logged versus unlogged site comparisons (Ough and Ross 1992). The results presented in this thesis indicate that the differences between logged and mature forest stands in terms of vascular species composition and cover are not as profound as previously considered using these more limited site comparisons.

Cable logging systems, like ground-based systems, are highly variable, some of the complex cable systems under favourable topographic conditions can be highly effective at minimising soil and vegetation disturbance, others can be marginally better than ground snigging. Under some circumstances, gains made in minimising site disturbance during cable logging can be compromised by soil disturbance imposed during fire break construction, or the incursion of silvicultural burning into retained streamside vegetation. The comparison of cable and ground-based logging on vegetation recovery undertaken in this thesis is therefore a complex one, and has been approached broadly using a variety of chronosequence, experimental, observational and population level studies which have extended over years and involved multiple experimental locations and techniques. Simplified notions that cable logging or ground-based logging were either more benign or detrimental to vegetation composition and cover are un-informative. They have broadly similar impacts, and the way in which they differ is dependent on how comparisons are made.

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