Peatland hydrology, fire management and Holocene fire regimes in southwest Tasmanian Blanket Bogs

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Summary

The paradox of fire-adapted vegetation overlying fire-sensitive organic soils requires a solid understanding of the interactions between climate, soils, plants and associated fauna. The purpose of this project was to investigate the impact of past and present fires on the organic soils underlying buttongrass moorlands in southwest Tasmania. This report explores a number of environmental and biological parameters that may be important to existing peatland ecosystems, and explores the impact of fire on their ecological function in peatlands.

Palynological evidence suggests that buttongrass moorlands have survived in a landscape that has been subjected to Aboriginal burning regimes for thousands of years. Evidence of truncated profiles in pollen diagrams also suggests that (wild) fire had a devastating impact on peat in the past. Pollen analyses provide some insight into the past history of moorlands, but their use in describing past burning regimes (fire frequency and intensity) is limited. More research is necessary to provide a landscape/regional scale history of moorlands in the southwest.

Pre- and post-fire samples of soils, vegetation, litter, and water-table levels were taken to examine the impact of fire on soil physical, chemical and biological properties. The topography of the study site confounded pre- and post-fire results. However, the influence of water, from rainfall or water-tables was found to be the most important influence on ecosystem function. Water-tables lay below the soil surface during summer. This finding is important in the context of current global warming scenarios.

Fire removes above-ground biomass from peatlands. The creation of a bare surface is likely to change the physical properties of the surface peat horizon, though these changes were not detected within a year post-fire. The low productivity moorlands of southwest Tasmania accumulate above-ground dead biomass (litter) and below-ground root material very slowly. The process of litter decay is also slow. It is possible that a change in fire management policy, to leave a litter layer over the soil surface, might provide a protective layer for the soil. This aspect of fire management is in urgent need of investigation. In addition, measurements of carbon dioxide emissions from burnt and unburnt peat surfaces also needs to be investigated, in order to develop management mitigation plans with the onset of the global warming threat.
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Chapter 1 – Peatlands in southwest Tasmania

Introduction

The buttongrass (Gymnoschoenus sphaerocephalus) moorlands of Tasmania cover approximately 1,000,000 hectares or roughly one seventh of the State (Jarman et al. 1988). These cyperaceous moorlands are commonly referred to as cultural artefacts. However the extent to which buttongrass moorlands have been created by fire or have extended their range under past fire regimes is under investigation (see Chapter 2).

While debate exists as to the role of fire in maintaining moorland communities (Jackson 1968, Bowman and Jackson 1981, Pemberton 1989, Thomas 1993), management issues are pressing. These fire-prone moorlands are often in close proximity to fire-sensitive vegetation such as is found in rainforest and alpine environments. In addition, the moorlands are mostly underlain by shallow (30 cm) deep, partially-to-well humified organic soils (Pemberton 1989) that will burn when soil moisture is low (Marsden-Smedley 1993, Marsden-Smedley et al. 1999). The high organic content of buttongrass foliage, and the fact that the plant retains its dead foliage as standing mass, leads to increased chances of fire over a relatively short time period. Buttongrass moorlands that have not been burnt for 12 years, will carry an intense fire over a broad range of climatic conditions that is not easily controlled by management agencies (Marsden-Smedley et al. 1999). It is the chance event of an intense ‘wildfire’ that may also lead to the loss of the underlying organic soil profile. While there appears to be a need for some form of fire management of buttongrass moorlands, little published information is available on the impact of fire on the underlying organic soils.

The organic soils underlying buttongrass moorlands are likely to be at least hundreds if not thousands of years old. They are commonly referred to as blanket bogs (Jarman et al 1988, Pemberton 1989) as they are not topographically constrained. Tasmania has the most extensive blanket bogs in the southern hemisphere (Dixon and Duhig 1996, Sharples 2003). These bogs are botanically and chemically distinct from their more extensive northern hemisphere counterparts, making their conservation even more valuable in a global context. In addition, organic soils are carbon stores. In the current climate of anticipated global warming, it is necessary to limit, where possible, loss of carbon from terrestrial stores to the atmosphere (Gore 1991). Appropriate fire regimes are essential in order to manage/limit any carbon loss from terrestrial deposits.
Project aims

The aims of this study are two-fold:

1) to investigate the role of fire in the development of buttongrass moorlands and blanket bogs during the Holocene;
2) to determine the impacts of current day management burns on physical, chemical and hydrological properties of organic soil profiles.

The information gained from these two aims, can only enhance our present knowledge of buttongrass moorlands and the underlying organic soil profiles, creating an improved management strategy to protect diversity (including organic soil profiles) in the World Heritage Area.

Plate 1.1 Fire in a buttongass moorland, southwest Tasmania
**Organic soils and peatland ecosystems**

Peat is an organic soil which consists of ‘… the accumulated remains of dead plants (Clymo 1983:159). Peat forms where the accumulation of organic matter is greater than the degree of decomposition. Cool, wet, acidic environments are conducive to peat formation, due to the influence of these factors on plant decomposition rates. Peatlands are ecosystems that have a soil profile dominated by organic deposits eg bogs and fens. Peatlands cover approximately 4% of the Earth’s total land area (Shotyk 1988) with half of this amount being contained in boreal peatlands which account for approximately 30% of the world’s carbon stores (Post et al. 1982). Peatland ecosystems are present under a variety of vegetation types in Tasmania. However, there is no estimate of areal extent, or volume of peat deposits in the State. While Tasmanian peatlands are shallow, their areal extent is greater than peatlands in many European countries (Bragg 2002, Charman 2002). However, unlike their European counterparts, Tasmanian peatlands are relatively undisturbed and are well reserved.

Mires are a subset of peatlands, and specifically refer to those systems that are currently accumulating organic material (Bragg 2002). As it is not certain that Tasmanian blanket bogs are still accumulating peat, then it is more correct to call them peatlands rather than mires.

Published information on, and classifications of, peatlands is heavily biased in favour of the northern hemisphere (Lindsay 1995, Charman 2002). Peatland ecosystems can be classified according to a number of features, including; morphology vegetation, hydrology, and acidity. The major division between bog and fen is usually determined by the water supply; whether a peatland is fed from ground-water (usually less acidic and more nutrient rich) or from rain-water (usually more acidic and nutrient poor). Sedges tend to dominate in fens, while mosses and shrubs dominate in bogs. These broad classifications were developed in the northern hemisphere are not directly transferable to Tasmanian peatlands. *Sphagnum* bogs are considered to be the most acidic bog type in the northern hemisphere. However, they are displaced in this role by cyperaceous (sedge dominated) moorlands in western Tasmania (Bridle 1994).

Particular morphological types need specific environmental parameters for them to exist/persist. Some peatland ecosystems are more climatically dependent than others. Blanket bogs are peatlands that are not restricted by topography, with the organic soil layer (or peat) extending from valleys to hill slopes and ridges. Blanket bogs occur in maritime regions where high humidity, regular rainfall events and low mean temperatures (for example,
Ireland, Scotland, Wales and western Tasmania, Lindsay et al. 1988). Northern hemisphere blanket bogs may be many metres deep and are associated with *Sphagnum* mosses at some stage of their development (Radley 1965). Their southwest Tasmanian counterparts are shallow and are dominated by sedges and shrubs (Myrtaceae, Epacridaceae, Proteaceae).

**The environment of southwest Tasmania**

Blanket bogs cover extensive areas of western Tasmania, including the World Heritage Area. Their presence in the region is linked to a number of environmental factors. Peat profiles are variable. Peat may overly bedrock and/or gravels, or it can form over a relatively impermeable mineral soil (usually clay). The stratigraphy of peat depends on site history and includes factors such as: dominant vegetation type (past and present); climatic conditions (past and present); site history (any disturbance such as burning, grazing, clearing, draining) (Moore and Bellamy 1974, Charman 2002). The peat profile contains not only the history of the development of the peatland but also the environmental history of the region.

**Climate**

Southwest Tasmania has a cool temperate maritime climate with high levels of precipitation, evenly distributed throughout the year, relatively low temperatures and high humidity levels (Genitilli 1972). Regional climate data show that there may be a precipitation deficit during the summer months, especially during February (Bridle 1994), where the water table may be found at the base of the peat profile for weeks at a time. Temperatures are slightly warmer than those considered optimal for peat growth (Jarman et al 1988, Lindsay et al 1988, Bridle 1994), but rainfall is sufficiently high and widespread over the year to maintain high humidities in support of bogs. The shallow nature of the organic soil deposits of Tasmanian blanket bogs compared to European blanket bogs has lead some authors to conclude that climate of western Tasmania is only marginally suitable for blanket bog formation. Damman (in Jarman et al. 1988) and Balmer (1991) have suggested that rates of peat accumulation and decay reach equilibrium at around 30 cm for much of southwest Tasmania.

Further support for the marginal climate hypothesis comes from the apparently low rates of peat accumulation in Tasmania. These rates have been deduced by examining sediment cores extracted from a variety of peat deposits. They yield accumulation rates of 1-2 cm/100 years. This is low compared to the 5-6 cm/100 years recorded for the northern hemisphere (Bridle 1994).
Vegetation

The vegetation of the region is a mosaic of moorland-heath, scrub, wet sclerophyll forest and rainforest communities. Alpine vegetation is present on the high peaks and ranges (see Reid et al. 1999). The moorlands can be found from sea level to the subalpine zone. The vegetation and underlying soils of these ecosystems have been described in detail by Bowman et al. (1986), Jarman et al. (1988) and Pemberton (1989). Marsden-Smedley (1997a) estimates that there are around 704 200 ha of moorland and associated scrub communities in the southwest region, accounting for 57% of the total area.

Geology

The landforms of southwest Tasmania have been described in detail by Pemberton (1989). Topography varies from low-lying plains to rugged ranges with extremes of aspect and slope. The geology is relatively complex and encompasses considerable variation in bedrock with Precambrian quartzites, dominating along with significant exposures of Cambrian volcanics and mid-Palaeozoic carbonates and clastics (Burrett and Martin 1989). Pleistocene glaciations have impacted on the region and erosional landforms created by small valley and cirque glaciers and ice caps are evident at higher altitudes. At lower altitudes deposition features including outwash tills and moraines are common (Kiernan 1996).

Geomorphology and Soils

The soils of southwest Tasmania are characterised by the predominance (at least in the upper horizons) of organic matter (Pemberton 1989). Organic soils occur on most geological substrates found in the region and across a wide range of topographical situations. They occur in association with most of the major vegetation formations found there including buttongrass moorland, scrub communities, wet sclerophyll forest, rainforest, and subalpine and alpine vegetation. Where intact, the soils of the region are dominated by organic horizons. Red/brown fibrous peats overlying mineral soils of varying composition and depth usually occur under forest vegetation. Very dark brown or black fibric or amorphous (sapric) peats overlying sand, gravels and bedrock form large areas of blanket bog in the region. These latter soils are found on nutrient poor substrates which are very slow to weather. They are usually associated with sedgeland-heath and related scrub communities, termed buttongrass (Gymnoschoenus sphaerocephalus) moorlands (Jarman et al. 1988).

Most of the western Tasmanian blanket bogs are underlain by glacial deposits, gravels or bedrock. There is very little mineral soil beneath the organic layer. Deep peat deposits (greater than 1 m in depth) occur in valleys of the southwest, but many hillslope deposits are
shallow. Deep peat deposits may have more than one hydrologically determined horizon. The acrotelm, the upper horizon closest to the surface of the peatland, is defined by the lowest position of the fluctuating water table. Beneath the acrotelm is the catotelm, a permanently inundated peat deposit, where decomposition is anaerobic. The shallowness of the profile and the climate of the region ensures that the organic layer would be classified as an acrotelm only, and that the catotelm is largely absent from shallow Tasmanian buttongrass moorland organic soil profiles.

**The fire history of southwest Tasmania**

Aboriginal burning practices are likely to have had a dramatic impact on the distribution of blanket bogs in southwest Tasmania, and in the development of the organic soil profile. This topic is dealt with in more detail in Chapter 2. To date there is little information available on the frequency and intensity of aboriginal burning regimes. However, Marsden-Smedley (1997a) demonstrated that there have been dramatic changes in the fire regime in southwest Tasmania in the last 170 years. He proposes that the pre-European fire regime was one of frequent, low-intensity, small-scale burns used by the Aborigines for habitat management. From the 1850s to the 1930s the fire regime was one of less frequent, high intensity, ‘landscape scale’ wildfires. From the 1940s to the 1960s the fire regime was one of medium intensity in spring and autumn. Since the 1970s there has been a policy of fire exclusion from virtually all of the area. Burning has largely been restricted to small fuel reduction fires at a few localities but there have been a few notable exceptions.

**Impacts of fire on the organic soils of southwest Tasmania**

While some researchers suggest that the shallow nature of the Tasmanian blanket bogs is due to climatic constraints, alternative views suggest that fire may have removed the surface peat, creating a truncated profile (Macphail et al. 1999, Fletcher pers. comm.). Pollen and charcoal analyses of cores taken from western Tasmania imply, that the complete removal of the peat profile is likely to have occurred post-fire (Macphail et al. 1999). Present day fires such as the Nook Swamp fire on King Island are testament that vast areas of organic soil may be lost to fire (Plate 1.2, pers. obs.). Large areas of blanket bog (in excess of 100 000 ha) have been degraded by fire and erosion (Pemberton 1988, 1989, Pemberton and Cullen 1995). However, much of the degradation is attributable to accidental and deliberate anthropogenic wildfires in recent times. Most of the degraded bogs evident on aerial photographs and satellite images fall within the boundaries of fires that have occurred in the region since the 1930s (Cullen
unpublished data, Marsden-Smedley 1997a). There have also been dramatic losses of fire sensitive vegetation in the region in historical times (Cullen and Kirkpatrick 1987, Brown 1988, Gibson and Brown 1991).

No data are available on the impact of low intensity fires on peat accumulation in Tasmania. However, low intensity fires are likely to influence the rate of peat accumulation even when peat is not directly burnt. While reducing the risk of wildfire is important, there may be other, less obvious impacts of management burns on organic soil profiles. Fire may decrease the organic content of the soil horizon (Garnett et al. 2000), either by removing accumulated organic material (Racine 1979), or by increasing biological activity in the soil layer (Jeffries 1986) with the release of nutrients (Maltby 1988). Fire may also retard revegetation of the site, by the removal of nutrients post-fire due to wind and water erosion (Maltby 1980). Post-fire impacts on the bare soil surface include desiccation of the surface layer, which leads to the creation of cracks in the soil profile, allowing increased oxidation down the soil profile (Maltby 1980). The exposure of the dark soil surface may also increase soil temperatures, thus increasing humification through increased biological activity (Cole et al. 2002), or it may result in an increase in the impact of frost events on peats in cold climates (Racine 1979). An exposed surface peat is likely to be more humified than protected, vegetated peat (Grover 2002). These impacts have flow-on effects in peatlands.

Elsewhere, studies indicate that peatland ecosystems may switch from being net storages of carbon (organic matter) to net sources of carbon (carbon dioxide) with small changes in soil temperature and soil water table position (Bubier et al. 1999, Johnson et al. 1996, Shurpali et al. 1995). On the other hand such effects may be countered by reductions in evapotranspiration until vegetation regenerates following the fire. Clearly vegetation-soil-fire relationships are not well understood and considerable research is required in this area. If the predictions of CO₂ induced global warming prove correct, then the consequent climate changes may well have a significant impact on peat accumulation. If the climate of southwest Tasmania is only marginal for blanket bog formation at present then small shifts in the prevailing conditions, such as a rise in mean annual temperatures, particularly if coupled with inappropriate fire regimes, may well cause blanket peats to degrade.

The direct physical effects of fire on organic soils have been summarized in increasing order of severity (Pemberton and Cullen 1995). They are:

- Removal of some vegetation but no direct impact on the soil (Plates 1.1, 1.2);
- Removal of vegetation which exposes the soil surface, which may then be vulnerable to wind or sheet erosion;
• Frequent removal of vegetation and litter. Raw material for organic soil development is lost and soil formation is hindered. Ash may be blown or washed away;
• Removal of vegetation, litter, soil seed bank, and soil to varying depths. Ash may be blown or washed away. The more soil removed the longer it is likely to take for the soil to reform because of greater stresses on plant development;
• Soils burnt to bedrock losing thousands of years worth of soil accumulation.

There is no doubt that fire has a pronounced impact upon the development of soils and vegetation in the region but there has been considerable debate as to the exact nature of this impact. To date much of the research and debate has been directed towards understanding the influence of fire on vegetation. Considerably less attention been paid to investigating the effect of fire on the underlying organic soils. Detailed studies in this latter field have been concentrated on the effect of fire on soil nutrition in lowland (Bowman and Jackson 1981, Bowman et al. 1986) and alpine areas (Kirkpatrick and Dickinson 1984) and surveys of soils and soil erosion as a result of fire (Pemberton 1988, 1989).

This report attempts to address some of the issues raised above.

Plate 1.2 Loss of peat profile after a fire in a paperbark swamp, Lavinia Nature Reserve, King Island (This fire was caused by lightning strike)
Chapter 2 – Holocene fire histories in the moorlands of southwest Tasmania

Introduction

There has been much debate over the role of fire in retaining/extending buttongrass moorland vegetation in western Tasmania. Jackson (1968) proposed that buttongrass moorland is a disclimax community that has been maintained far beyond its natural limits by anthropogenic burning. He developed a model of ‘ecological drift’ which suggests that much of the area currently occupied by buttongrass moorland would develop into wet scrub, then wet sclerophyll forest and ultimately rainforest given the absence of fire for a long time period (in excess of 300 years). As vegetation develops along this sequence it becomes more sensitive to the impact of fire but less likely to burn. Repeated burning over long periods by Aborigines and then in recent times by Europeans, seems to have provided a fire regime where the expansion of buttongrass moorland has taken place at the expense of more fire sensitive communities. Thus buttongrass moorlands have become commonly regarded as cultural landscapes, owing their existence to regimes of relatively frequent fires.

This view has been challenged by Pemberton (1989), who suggested that moorland distribution may be primarily determined by geology and soils. Recent developments hypothesise that buttongrass moorlands may be an edaphic climax vegetation type that has extended its range due to Aboriginal burning regimes (Fletcher pers. comm.).

Human occupation

Humans have occupied Tasmania for at least 35,000 years (Flood 1995). Most of the oldest cultural sites on the island are found within southwest Tasmania. Radio-carbon dating of cultural material found in caves in the region indicates that Aborigines occupied inland areas of the region from around 35,000 to around 12,000 years BP. These caves appear to have been vacated as living quarters after this time. There is an unresolved debate as to whether Aborigines generally abandoned inland southwest Tasmania in the face of advancing wet sclerophyll and rainforest vegetation during the late Pleistocene and early Holocene or whether they occupied the region throughout the Holocene up until European colonisation (Cosgrove et al. 1990, 1994, Thomas 1993, 1995).
Pollen sequences obtained from bogs and lakes indicate that the vegetation of lowland southwest Tasmania during the last glacial was open and herbaceous with alpine and subalpine affinities (Jackson 1999). By the early to middle Holocene there is an increasing dominance of pollen associated with closed forest communities. A warmer, wetter climate during the period 9 K to 6 K appears to have favoured the spread of this vegetation. During the late Holocene, decreasing temperatures and precipitation produced an opening of forest vegetation (Macphail 1980) accompanied by increases in charcoal particles in the deposits. This may indicate greater deliberate burning activity by the Aborigines and/or increases in the occurrence of wildfires. There are a number of historical records that indicate that Aborigines were living in inland southwest Tasmania in the early 1800s (Binks 1980, Marsden-Smedley 1997a). These records indicate that Aboriginal people were using fire extensively to modify habitat. European exploration and exploitation of the region resulted in high incidences of wildfire up until relatively recent times (Marsden-Smedley 1997a).

Colhoun et al. (1991) found fairly consistent levels of charcoal associated with pollen representing a range of vegetation assemblages throughout the Holocene at Governor Bog in the King River valley and concluded that fire was ‘an ever-present ecological factor of the changing alpine, subalpine, temperate vegetation/environmental changes. It seems likely that at least some of this firing would have result from human ignition sources.

Little is known of the history of buttongrass moorland or the role of fire in its ecology. Pollen cores taken from the region show the continued presence of charcoal fractions throughout profiles dominated by moorland species (van der Geer et al. 1989, Colhoun et al. 1991). Fossil evidence from western Tasmania indicates that vegetation with strong affinities to modern buttongrass moorland existed in Tasmania long before the arrival of humans (Jordan 1997). This suggests that moorland vegetation has had a long history in Tasmania, pre-dating periods of high fire frequencies, indicating that moorlands can or did persist without the firing regimes they have been subjected to in recent times.

Two pollen sequences from Melaleuca in the far southwest provide vegetation records that span the Holocene (Thomas 1995, Macphail et al. 1999). These show the persistence of open moorland vegetation and blanket bog from the late Pleistocene up until the present in a location that is at low altitude and relatively close to the present coast. Both sequences show that fire was present in the environment throughout this time. Macphail et al. (1999) interpret
a hiatus in the sedimentary record as evidence of a peat burning fire. Neither site provides evidence that the far southwestern coastal plains formed refugia for arboreal taxa, nor do they indicate any successional changes from moorland to forest. A pollen diagram from Melaleuca Inlet (Thomas 1993) indicates that Gymnoschoenus sedgeland co-existed with Poa grassland at the end of the last Glacial under conditions where fire was present but far less significant than at any subsequent time. In other words, moorland was able to prosper under different fire regimes. In contrast a pollen core from an inland location, on the shoulder of Mt Anne shows that over the same period, forests maintained there dominance on the mountain slopes (Harle 1989).

At present the full extent of buttongrass moorland and blanket bog during the Pleistocene and Holocene is unclear. Presumably cold, arid, glacial conditions restricted these ecosystems to wet lowland situations. However this formation may have expanded to occupy more or less its current range during the early Holocene. This would indicate that either there was a sufficiently frequent anthropogenic fire regime to promote moorland vegetation throughout the southwest or that its expansion was favoured by climatic and edaphic factors at this time. Alternatively, buttongrass moorland may have been restricted to low lying sites close to areas frequented by Aborigines until climate change in the early Holocene favoured the spread of fires and the expansion of human activity. Studies of fossil pollen and charcoal from three inland sites now dominated by buttongrass moorland were commissioned as part of the present study to investigate this issue.

**Results and discussion**

**Interpretation of cores – vegetation histories**

Fletcher (2000) took three cores from a site located at Harlequin Hill close to Harle’s (1989) Mt Anne study site. Two of the cores were terrestrial, one from a buttongrass moorland and one from a eucalypt dominated woodland. The third core was taken from a lagoon surrounded by buttongrass moorland (see Fletcher 2000 for details). Tye (2002) took two cores from the Vale of Rassalas, one from a small pond within buttongrass moorland and one from within the moorland vegetation itself (see Tye 2002 for details). A third core was extracted from a small lake on the floor of the Hardwood River Valley, also in buttongrass moorland vegetation. The core from this site is not yet fully analysed (Fletcher pers. comm.).

The results of these studies can be divided into two categories: those from relatively long peat cores obtained lagoons and ponds; and short cores taken from nearby terrestrial locations.
The aquatic cores have been used to reconstruct regional and local vegetation histories and the terrestrial ones have been used to reconstruct local histories for the areas from where they were retrieved.

Fletcher (2000) collected a 118 cm core from a lowland lagoon, a 28 cm terrestrial core from a neighbouring buttongrass moorland, and a 20 cm core from a small copse of eucalypt dominated vegetation. A basal date of 10,350 ± 50 BP was obtained from the lagoon core.

The pollen spectra indicate that the plains around the lagoon have changed little during the Holocene and the site has always supported buttongrass moorland. Charcoal particles occur throughout the core at relatively high levels, suggesting that fire has been common in the area for the whole of the Holocene. It is not possible to determine whether edaphic factors (low nutrient levels, water logging) or relatively frequent fires have maintained this moorland vegetation. The regional vegetation history revealed by the cores taken from Harlequin Hill is in accordance with other studies from the southwest region, i.e. an expansion of rainforest during the early Holocene followed by an expansion of eucalypt and moorland vegetation in the late Holocene. This suggests that although rainforest was able to expand in the early Holocene under warmer, wetter conditions when, presumably, there was less impact from fire, edaphic conditions and/or fire regimes on the moorland areas around the lagoon were such that rainforest could not expand across this site (Fletcher 2000).

Both of the terrestrial cores obtained from the site are much shorter than the lagoon core. Basal dates for these were estimated by applying peat accumulation rates derived from the relatively well-dated aquatic core. These cores appear to be between 1500 and 2000 years old. They indicate that the local vegetation has been dominated by moorland species, with small copses of eucalypt scrub. Conditions have been relatively stable over this period. Both cores show persistent and relatively high levels of carbonised particles throughout.

Both the aquatic and terrestrial cores retrieved from the Vale of Rasselas present a similar vegetation history to those from Harlequin Hill. Tye (2002) presents a date of 4240 ± 40 BP for a 21 cm aquatic core taken from a montane pond. However, no date was taken for the adjacent 32 cm terrestrial (buttongrass moorland) core. Tye (2002) extrapolated that the terrestrial core was likely to be over 900 years old (920 ± 40 BP), and that the surface 7 cm of the aquatic core was the equivalent of the whole of the terrestrial core. The cores indicate the area around the site has been dominated by moorland vegetation for the entire length of the record. Carbon particles are common throughout. Fires appear to have been most frequent between 3660 BP and 200 BP, if Tye’s interpretation of dating and sediment accumulation are
correct. It appears that this site was also dominated by buttongrass moorland. There is limited evidence to suggest that floristic change may be coupled to fire incidence with increases in *Melaleuca* spp., *Eucalyptus* spp. and *Leptospermum* spp. pollen loosely correlated with decreases in charcoal abundance. However the resolution of the study is too coarse for this to be taken as more than reasonable speculation. The date of 200 +/- 40 BP for the last significant peak in charcoal in both cores more or less corresponds with the depopulation of Aborigines from the region and provides support for Marsden-Smedley’s (1997b, 1998) hypothesis that the southwest fire regime changed from one of frequent low intensity fires to infrequent high intensity fires at this time (Tye 2002).

The third site, the Hardwood Valley has not yet been fully analysed. However early results indicate that the pollen and charcoal records show a history is similar to other buttongrass moorland pollen study sites in Tasmania. The local vegetation has been dominated by moorland right throughout the Holocene, fire remains important throughout the sequence, and the regional vegetation history reflects the same trends as else where in the southwest. Here as at Harlequin Hill and at Melaleuca there is an early peak in Poaceae pollen in association with buttongrass moorland taxa, with the Poaceae fraction diminishing over time. This may be explained by loss of nutrients from the system due primarily to fire (Thomas pers. com.) (see Bowman et al. 1986). Nutrient input during the glacial is likely to have been higher because of higher rates of weathering with glacial and peri-glacial activity. The tree line would have been much lower as would have been the limit of peri-glacial activity and areas comprised of more weatherable rocks (phyllites, schists etc which are at lower elevations i.e. not the highest, hardest quartzite peaks) would have been contributing more nutrients.

Table 2.1 Comparison of sedimentation rates (years/cm) in southwest Tasmania: a lowland buttongrass moorland lagoon core (Fletcher 2000), a terrestrial core (Thomas 1993), a high altitude rainforest swamp (Harle 1989) and a high altitude pond surrounded by buttongrass (Tye 2002).

<table>
<thead>
<tr>
<th>Lagoon Core (Fletcher 2000)</th>
<th>Years/cm</th>
<th>Melaleuca Inlet (Thomas 1993)</th>
<th>Years/cm</th>
<th>Lake Timk (Harle 1989)</th>
<th>Years/cm</th>
<th>Pond core (Tye 2002)</th>
<th>Years/cm</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 40</td>
<td>167.75</td>
<td>20</td>
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<td>160 – 180</td>
<td>24.27</td>
<td>0-21</td>
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<td>60</td>
<td>96.5</td>
<td>310 – 330</td>
<td>29.6</td>
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<td>78 – 114</td>
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<td>160</td>
<td>84.64</td>
<td></td>
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</table>
Interpretation of cores – fire histories

Fletcher (2000) and Tye (2002) provide evidence for the continued presence of fire throughout a peat profile. While these data are real, there are many questions to be asked in the interpretation of pollen cores, and also with respect to Jackson’s model of ecological drift (1968).

Tye (2002) extrapolated that the top 7 cm of the pond core correlated with the whole of the 31 cm deep terrestrial core which was estimated to be 920 +/- 40 BP. Similarly, Fletcher (2000) extrapolated that the top 15 cm of his lagoon core correlated with the whole of the 28 cm terrestrial core which he estimated to be between 1800 and 2000 years BP. While these dates are speculative, there appears to be a discrepancy in their results. Tye’s terrestrial core is deeper than Fletcher’s, despite the latter collecting the core from a low altitude site, which is likely to be more productive than Tye’s (2000) montane site. In addition, Tye’s predicted peat accumulation rate is 1 cm in 14 years for the surface 19 cm of the peat. Accumulation rates below this depth are more in agreement with accumulation rates for Fletcher’s findings (1 cm every 30-60 years) and elsewhere (Table 2.1). The accumulation of organic material in the lagoon and the pond core is taken to be much slower than the terrestrial cores (Table 2.1). While it is difficult to see how a montane site could be more productive than a lowland site, given the same vegetation type, there is a possibility that a higher fire frequency at the lower site, might account for differences in peat depths and accumulation rates. There is also a possibility that higher productivity would lead to greater decay at the warmer lowland site, which would contribute less organic material to the peat profile than in a less biologically active situation, even if there is less organic accumulation at higher sites. This seems unlikely given that the deepest peat deposits in Tasmania are found at very low altitudes. The pollen diagrams throw little light onto this conundrum.

Tye’s (2002) pollen data from the fine resolution research shows a general relationship between charcoal counts and abundance of certain taxa, with pollen counts of the woody species Eucalyptus, Melaleuca and Leptospermum increasing when charcoal counts are low. However, the increase in aquatic pollen in both the terrestrial and the aquatic core suggests that site wetness has increased, which confuses the picture somewhat.

Tye’s study (2002) indicates that fire has an adverse effect on peat accumulation. However, once again this is a broad statement that encompasses the removal of the peat layer as an impact on peat accumulation. Whether fire has an impact on peat accumulation when the soil
is not burnt is not easily ascertained from the data presented, especially in the absence of more radio-carbon dates closer to the surface of the aquatic and terrestrial profiles. These fine resolution profiles suggest that fires occurred at least every 30 years, assuming that the 30 cm terrestrial profile is 900 years old. This is all the information that can be gleaned from the profiles for management purposes.

It is interesting to note that as charcoal counts decrease, organic and moisture contents increase (Tye 2002). Organic content is likely to increase on more productive sites and peats with higher organic contents are likely to have higher moisture contents (Bridle 1994). Peats that form in productive environments are likely to be fibric (less humified), but only if decomposition is retarded by the presence of continually high water tables and/or extremely acidic environments. What would be useful from the cores taken from Tye’s study site is data on the degree of humification throughout the profile. Marsden-Smedley (1993) states that the higher the organic content, the more likely the peat profile will dry out and burn. More humified peats may hold less moisture, but they are also generally less organic and less likely to burn (such as the aquatic peat).

Tye (2002) provides evidence of fire from the presence of charcoal in peat cores. The resolution of his work at a 1 cm sample is likely to cover a minimum of 30 years of accumulation. Therefore, while he might conclude that fires were present at least every 30 years, Jackson’s (1968) model of ecological drift would predict a shift from moorland to shrub-dominated communities within that time frame. There was no evidence for such a shift in the pollen profile. Fletcher’s (2000) cores also suggest an increase in wetness through the pollen profile, he records increases in charcoal particles during this phase also.

Until more exact techniques can be developed, it is difficult to determine past Aboriginal fire regimes using charcoal analysis from pollen cores. The lack of dating in the relatively shallow terrestrial peat deposits combined with the difficulty of dating such deposits means that any dates are speculative and are the result of a best estimate using evidence from deeper and less contaminated lagoon cores. Despite these difficulties, there appears to be some agreement over the accumulation rate of terrestrial peat in southwest Tasmania. Extrapolations from lagoon or pond cores estimate peat to accumulate at a rate of 1 cm over 30 to 60 years (Thomas 1993, Fletcher 2000).

If a 1 cm addition to the peat profile takes a minimum of 30 (to 60) years to develop, then an estimate of fire frequency will be restricted to the level of detail to which the samples are subjected. If cores are sampled at 1 cm intervals, which will equate to a minimum of 30 years,
then fire frequencies can only be reconstructed for a 30-year time frame. Given than fire managers are not able to easily control a moorland fire that has been left unburnt for more than 12 years (Marsden-Smedley et al. 1999), then the 30 year time frame poses as many management questions as it attempts to answer. With this resolution there is no way of knowing whether Aboriginal burning regimes were more frequent than this.

It is not known what impact fire intensity has on the size of charcoal particles, and whether angular or large particles are evidence of in situ fires compared to smaller particles that are taken to be evidence of transported or regional fires. Other recent work (Hope pers. comm.) shows that *Sphagnum* bogs will only record charcoal in the profile if the fire passed over the actual site, i.e. there is no evidence of fire in an unburnt patch of bog that is surrounded by burnt bog vegetation immediately after the fire. Whether evidence is later found from the transportation of charcoal grains to the unburnt site is yet to be seen. What is curious about recent research into peat cores underlying buttongrass moorlands, is that they show conflicting results. One study shows a trend towards a decrease in carbonised particles in the surface layers (7 cm) of a terrestrial core (Fletcher 2000), while another shows no real change (Tye 2002). Both studies also record contrasting results for aquatic cores (Fletcher 2000, Tye 2002). The question of whether carbon particles move through the peat profile and at what rate needs to be addressed. While the use of charcoal analysis with pollen counts is useful in providing a range of possibilities as to site history, clearly they do represent site history and are not necessarily able to be extrapolated to other blanket bogs in the region. A decrease or no change in carbon particles did not lead to a decrease in buttongrass pollen grains over the surface 7 cm of cores (Fletcher 2000, Tye 2002). This surface 7 cm would equate to a time frame of 210 years (or 110 years at 1cm/15 years, or 420 years at 1cm/60 years). No real change in buttongrass pollen over this time period, with continued or reduced fire frequencies would challenge Jackson’s model of ecological drift (1968).
Conclusion

There is now a growing body of evidence (Thomas 1993, Macphail et al. 1999, Fletcher 2000, Tye 2002) to support the dominance of buttongrass moorland over extensive tracts of lowland southwest Tasmania throughout the Holocene. While pollen and charcoal analyses show the continued presence of fire in cores taken from buttongrass moorland environments, it is not clear whether fire is necessary to continue the dominance of buttongrass moorlands in the southwest region. It is likely that moorlands have extended their range due to past fire events (Fletcher 2000), but it is also likely that they may be dominant due to their ability to out-compete other species on a particular substrate. In other words, buttongrass moorlands may well be an edaphic climax community on nutrient poor, topographically constrained, waterlogged environments. Whether blanket bogs are a result of Holocene firing regimes can only be speculated. It is possible that the dominance of moorlands over steep slopes on nutrient-poor rock types is due to the impact of past fire events. It is likely that lowland blanket bogs are edaphically controlled, but are extended in range by fire (Pemberton and Cullen 1995).

Tye (2002) suggests that the charcoal found in every 1 cm segment of core analysed in a terrestrial peat, indicates a fire regime of at least one fire every 30 years (the approximate time taken for 1 cm of peat to accumulate). He states that his findings are in agreement with other research (Marsden-Smedley and Kirkpatrick 2000) for the region and that a fire regime of less than 30 years would cause a decrease in plant diversity at a landscape scale. However, in terms of practical management and the protection of other sensitive vegetation types, a 30-year firing regime would not prevent wild fires including intense fires that could remove the peat profile. More research needs to be carried out on the use and reliability of charcoal particles in peat profiles as an indicator of fire frequency and intensity. Reliable dates for both terrestrial and aquatic cores would remove the speculative nature of the relative histories of these cores, and will provide a more accurate determination of peat accumulation in terrestrial environments.

A great amount of knowledge that is relevant to current management of southwest Tasmania was lost with the passing of Aboriginal occupation of the region. At present we have very limited information on which to base future management practices. The extensive losses of fire sensitive vegetation and organic soils in the region in recent times demonstrate that, if we are to manage southwest Tasmania in a sustainable manner, we must proceed with caution. As concluded by Bowman and Brown (1986) and demonstrated by Thomas (1993), Macphail et al. (1999) and Fletcher (2000), experimental investigations and manipulation of fire regimes
in association with appropriate palynological studies are likely to provide the best solutions to these problems. These approaches combined with models of fire frequency in the region suggest that Aboriginal burning practices have shaped vegetation patterns in the southwest over a long period of time. However, more detailed interdisciplinary research needs to be undertaken to determine what the fire frequency was.
Chapter 3 - Regional and local climate constraints on peat formation in southwest Tasmania

Introduction

Previous research and best estimates suggest that the blanket bogs of southwest Tasmania are at the climatic limit for blanket bog formation (Damman in Jarman et al. 1988, Balmer 1991, Bridle 1994). The shallow nature of the deposits is given as evidence for the balance between productivity and decay (Damman in Jarman et al. 1988, Balmer 1991). Climatic ‘evidence’ in the form of dry soil profiles and precipitation deficits during the summer months (Jarman et al. 1988, Bridle 1994) also add support to the marginal nature of the region to support blanket bogs. If the climate of the region is already marginal, any change to temperature and rainfall - as predicted from global warming scenarios - will create a peatland environment that is in decay rather than in equilibrium if this is not already the case. An analysis of the climatic data from the region will highlight any trend in temperature and rainfall (specifically increases temperature and decreased rainfall) with the onset of global warming.

It is probable that a warmer, drier environment will impact on organic soils by drying out the profile more frequently than at present. Under such conditions, it is highly likely that the soil profile will burn in the event of a fire. Whole soil profiles have been lost in this way in parts of southwest Tasmania (Pemberton 1988, 1989, Macphail et al. 1999). Profiles that have taken thousands of years to develop can be destroyed in a few weeks (Pemberton and Cullen 1995).

The impact of fire may be subtle, taking months or years to change the physical and chemical properties of organic soils. Fire removes the vegetative layer that protects organic soils from extreme climatic conditions. Bare, black soil is more likely to heat up, dry out and contract under warm, dry, windy conditions than soil that is protected by vegetation. Therefore, the creation of bare soil surfaces after fire events may further limit the process of peat accumulation in climatically marginal environments. Warmer, drier, soils may also lead to cracks appearing in the soil profile, which would increase oxidation of the organic horizon, ultimately creating a loss of carbon to the atmosphere (Maltby 1980).

This chapter examines the role of present day climate in supporting the continued existence of organic soils at both a regional and a local scale. The relationship between soil temperatures
and soil moisture and the possible impact of global warming on organic deposits of southwest Tasmania will be discussed.

**Methods**

Regional climate data were collated from the four existing meteorological stations of the southwest region: Maatsuyker Island; Gordon Dam; Strathgordon Village; Scott’s Peak (Fig 3.1). The Maatsuyker Island station lies off the southwest coast of mainland Tasmania, at an elevation of 146.5 m a.s.l. The other three stations are located inland at elevations of 320-340m. Maatsuyker Island is the oldest station for the region, dating back to 1950. The Strathgordon station was installed in 1968, and the Gordon Dam station was set up in 1978. The Scott’s Peak station is further inland (a similar longitude to the Airstrip Road study site). This station was installed in October 1998. Daily temperature, rainfall and evaporation data were supplied by the Bureau of Meteorology.

Local climate data were collected from the Airstrip Road study site (531069E 5163012N). This site was chosen for its relative representativeness of lowland moorland landscapes in southwest Tasmania. The fire history of the site has been estimated by Marsden-Smedley (pers. comm.) and is thought to have been burnt in 1898, 1934, 1964, 1972) prior to the start of this experiment. Therefore in recent times the site has been burnt at a mean fire frequency of approximately 26 years. The site is regarded as low-productivity (Marsden-Smedley 1998). Six contiguous 30 x 30 m plots were set up (Plate 3.1). Plots were orientated south-east to north-west in alignment with the dominant slope. Plots 4 to 6 were offset from plots 1 to 3 to account for variability in vegetation and topography at the site. The highest point of plot 1 was at an altitude of 351 m (south-eastern corner), while the lowest point was at 347 m asl at plot 6 (north-eastern corner).

Two macro-loggers and one pro-logger were placed in three of the 6 plots, those that were not to be burnt (2,3,5). The data loggers recorded soil temperature at 5 cm and 15 cm depth for each plot. A mean soil temperature for each depth in each plot was taken by averaging values from 3 randomly located T-type thermocouples. In addition, one data logger recorded air temperature for the site using a thermistor, while another recorded rainfall using a tipping bucket raingauge. A heavy duty 12 volt battery that was charged by a solar panel powered the data loggers. Much data were lost due to; mechanical failure of data loggers, animals chewing cables and software incompatibility over the 3 year period. The electronic data collection period started in March 1999 and ended in January 2002.
Additional rainfall data were collected manually on a fortnightly basis, using a plastic 300 mm raingauge mounted on a 1 m tall metal stake.

![Map of Climate Stations and Study Site](image)

Figure 3.1 Location of the Climate Stations and the Study Site (Airstrip Road)

Ten dip wells (water wells) were randomly placed in each of the six plots (Plate 3.2). A metal pipe, 25 mm in diameter, was used to excavate each hole. The pipe was pushed into the soil until it reached a gravel or clay layer and could be pushed no further. A piece of 25 mm diameter plastic conduit pipe was placed in the hole, to gravel depth. Holes (3 mm diam) had been drilled into the conduit pipe at regular intervals (5 cm) offset along the length of the tube to allow for water movement through the soil and into and out of the pipe. The pipe was then
covered with geofabric to prevent the holes becoming blocked with organic material. Automatic water table recorders, using pressure sensors, were randomly allocated to two of the ten wells in each plot. Water table levels were read manually (from March 2000 to June 2002) on at least a fortnightly basis for the remaining eight wells in each plot.

Plate 3.1 Layout of plots at Airstrip Road study site.
The site was burnt (plots 1, 4 and 6) on 23rd of April 2001 (Plate 3.3). The climate variables on this day were within prescriptions set for management burns for buttongrass moorlands (Marsden-Smedley pers. comm.). Wires to the data loggers for all affected plots were removed on the morning of the burn. Plastic water wells were also removed from the plots destined to be burnt. Each well was marked with a metal stake to prevent the holes from filling up with litter and for relocation purposes. The site was visited one week later (30/4/01) to re-install the water wells and to re-connect the sensors to the data loggers. One data logger was no longer working (from plot 3), therefore all information was collected by the two remaining data loggers. The air temperature probe and screen were removed from plot 3 and placed in plot 2 next to the data logger. The reduction in data loggers meant that water tables could be automatically measured from only one dip well per plot.

**Data analysis**

The annual rainfall and temperature (maximum, minimum) data for each station were plotted for the length of the record to look for any trends over time, and to compare data between sites. Relationships between sites, such as rainfall for Strathgordon versus Gordon Dam/Scott’s Peak road, were explored. Linear regression equations were fitted to the data to determine whether relationships existed between any two stations and to test for the strength of any correlations. Raw data were used where possible. However, log transformations were performed on data to achieve heteroscedasticity. These analyses were used to explore the variability in rainfall and temperature in southwest Tasmania.

Evaporation and precipitation data were available for Strathgordon only. Graphs of rainfall and evaporation data over time were used to determine how frequently the region experienced precipitation deficits (where evaporation is greater than rainfall). These data are important in determining soil moisture levels and thus the overall ‘health’ of the peatland ecosystem, and the ability of organic material to accumulate and decay.

An evaluation of soil temperature data was attempted by comparing data between pairs of adjacent plots (unburnt pre-fire, burnt v. unburnt post-fire), or between temperatures at different depths within the same plot/treatment (5 cm v. 15 cm depth). The paired t-test or its non-parametric equivalent (the Wilcoxon Signed Rank test) was used to analyse these data.
Analysis of variance was deemed to be an ineffective statistical analysis to test for pre- and post-fire or burnt and unburnt differences for these data as the data were influenced by many other factors such as climatic conditions.

Plate 3.2 Dip-well in centre of a 1 x 1 m quadrat in a burnt plot

Plate 3.3 Boundary between plots 3 (unburnt) and 4 (burnt), April 2001
Results

Regional climate trends

There is no apparent trend (increase or decrease) in rainfall over time for the Gordon Dam site and the Strathgordon station (Fig 3.2). The Maatsuyker Island data show a slight decrease in rainfall during the 1990s (Fig. 3.2). Rainfall data from this station are very different from the three inland stations. The Scott’s Peak data were too few to show any trend, however, patterns of rainfall events were similar between Scott’s Peak, Strathgordon and the Gordon Dam site. There was no evidence of a decrease in rainfall over time from the latter two sites (Fig 3.3).

![Graph of annual rainfall (mm) between Maatsuyker Island, Gordon Dam, and Strathgordon](image)

Fig. 3.2 Comparison of annual rainfall (mm) between the climate stations at Maatsuyker Island, Gordon Dam, and Strathgordon.

A linear regression on the rainfall data from Strathgordon and Scott’s Peak suggested that Scott’s Peak receives approximately 60% of the rainfall that Strathgordon receives (based on daily rainfall data) \( y = 0.6938x, R^2 = 0.9006 \).

The Strathgordon village temperature data (maximum and minimum) show no distinct change over time (Fig. 3.4). Maatsuyker Island shows an increase in temperature, corresponding with the decrease in rainfall in the late 1990s (Fig. 3.4).
Fig. 3.3 Comparison of monthly rainfall (mm) between the climate stations at Gordon Dam, Strathgordon and Scott’s Peak.

Fig. 3.4 Comparison of annual rainfall (mm) and mean minimum and mean maximum temperature (°C) for Maatsuyker Island and Strathgordon for the length of the climate record.

Evaporation data are available for the Strathgordon site only. A comparison between evaporation and precipitation for the three warmest months (summer, Dec-Feb) from 1969 to the present, shows that precipitation deficits (evaporation>precipitation) regularly occur. Using temperature and evaporation data for the warmest month only, the Strathgordon village data show precipitation deficits for 60% of all February data from 1970 to 1999 (Fig 3.5). These deficits also occur to a lesser degree in January (40%) but rarely in December (10%).
Temperature patterns at Strathgordon and Scott’s Peak are similar, but the mean maximum temperature at the Scott’s Peak is slightly (but significantly) warmer (by about 1ºC) than Strathgordon (Fig. 3.6). There is no significant difference in minimum temperatures.

Fig. 3.5 Graph showing precipitation deficits (evaporation>precipitation) for Strathgordon during February.

Fig. 3.6 Mean daily maximum and minimum temperatures (ºC) for each month for Scott’s Peak and Strathgordon.
The dominant weather systems of southwest Tasmania are frontal systems from the west. The relationship between rainfall and temperature is illustrated using data from Scott’s Peak (Fig. 3.7). This pattern can be extrapolated to most of lowland southwest Tasmania. Rainfall is low during periods of high temperatures, and it is high when temperatures are low (Fig. 3.7). There is very little frost in the lowland regions of southwest Tasmania due to the maritime, humid environment (Figs. 3.6, 3.7).

![Graph showing relationship between temperature and rainfall](image)

Fig. 3.7 The relationship between mean daily maximum and minimum temperature (°C) and monthly rainfall (mm) at Scott’s Peak.

**Climate of Airstrip Road**

The climate at Strathgordon village is not representative of the climate of the study site at Airstrip Road. Rainfall data for the two sites show that Airstrip Road receives on average 65% of the rainfall that Strathgordon village records (Fig. 3.8). Rainfall at Scott’s Peak station was most closely correlated with rainfall data at the Airstrip Road site (Fig. 3.9).
Limited temperature data also show that the temperature at Strathgordon is not representative of that at the Airstrip Road site (Fig. 3.10). Temperature data from Scott’s Peak are much closer in value to those found at Airstrip Road (Fig 3.11). Scott’s Peak is slightly drier and cooler than the Airstrip Road site. Both sites are drier and warmer than the Strathgordon site.
Figure 3.10 Graph of daily maximum temperature (°C) for March 2001 at Strathgordon and Airstrip Road.

Fig. 3.11 Linear regression of mean daily maximum temperature data for Scott’s Peak and Airstrip Road.
Within Site Climate Data

Data loggers were faulty for much of the time, creating large gaps in the climate data. However some information has been recovered from the data loggers, while other information has been extrapolated from the most suitable permanent meteorological station (Scott’s Peak).

Data on soil temperature are incomplete, due to the problems with data loggers at the site. Air temperature was more extreme than soil temperature (maximum and minimum) at all times (Fig 3.12).

Soil temperatures at 15 cm were greater than those at 5 cm during winter (Fig. 3.12). However, soil temperatures were greater at 5 cm than at 15 cm for all plots during the summer months (eg Fig 3.13).

Towards the end of the study, soil temperatures for plots 1-3 were recorded on one data logger, while temperatures for plots 4-6 were recorded on the other. Very few data exist that compare soil temperatures for all 6 plots at the same time. Where this did occur, it appeared that the temperatures for plots 1-3 had similar values to each other but were different for those recorded for plots 4-6 (Fig. 3.14). When the soil temperature probes were tested under laboratory conditions before and after installation, they were calibrated to each other. This result and further field testing indicated that the difference in soil temperature between the two groups of plots is real.
Fig. 3.13 Mean daily soil temperatures (°C) at 5 cm and 15 cm for plots 4-6 at Airstrip Road during January 2000.

Fig. 3.14 Mean daily soil temperatures (°C) at 5 cm depth for each plot and daily rainfall (mm) during April at Airstrip Road.
Whether fire had an impact on soil temperatures could not be determined from the electronic data set due to faulty data loggers. There were no post-fire temperature data for the only unburnt site (5u) between the two burnt plots 4 and 6 which made any paired comparisons impossible (see Chapter 1 for site layout). There was no obvious effect of fire on the soil temperatures of the burnt/unburnt pairs of plots (plots 1-2, 3-4).

Limited data were collected manually using a temperature probe in March 2003. These data indicate that soil temperatures were at least 1°C higher in bare/burnt plots than in neighbouring vegetated plots. However, these temperature differences are moderated by the presence and amount of soil moisture at any given time.

**Depth to water-table**

Data on water-table depths that were recorded from the data loggers were sparse and unreliable. Therefore all of the results presented here are based on the manual, fortnightly measurements of water-table depth.

The site experienced extensive dry periods over the time of the study. Some water wells were empty for long periods while others were never completely dry (Figs 3.15-3.20). Water-tables rose relatively quickly in response to rainfall events. Mean water table depth was related to position in the landscape, with the downslope plots (4-6), being wetter than the upslope plots (1-3) (Figs. 3.21, 3.22, Table 3.1).

Statistical analyses of the data indicated that within plot variability was as great as between plot variability. There was no distinct impact of burning on water-table measurements at this site.

Mean minimum water-table depths were greater in the upslope plots (1-3) than in the downslope plots (4-6). The fact that the mean minimum water-table depths were not at the soil surface (0 cm) illustrates that the soil profiles of the peatland system are not always water-logged (Table 3.1), and that position of the water-table is related to topographic position.
Fig. 3.15 Water-table fluctuations for each of the 8 dip-wells in Plot 1, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.

Fig. 3.16 Water-table fluctuations for each of the 8 dip-wells in Plot 2, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.
Fig. 3.17 Water-table fluctuations for each of the 8 dip-wells in Plot 3, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.

Fig. 3.18 Water-table fluctuations for each of the 8 dip-wells in Plot 4, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.
Fig. 3.19 Water-table fluctuations for each of the 8 dip-wells in Plot 5, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.

Fig. 3.20 Water-table fluctuations for each of the 8 dip-wells in Plot 6, Airstrip Road. Daily rainfall (data) are taken from Scott’s Peak Road.
Fig. 3.21 Mean depth to water-table (cm) and standard error bars for each dip well in each plot at Airstrip Road.

Fig. 3.22 Mean water-table depth for each plot over the study period.
Table 3.1 Mean and Median water-table depths for each of the 6 plots at Airstrip Road.

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**Discussion**

One of the most important findings of the regional analyses of climate data for southwest Tasmania, is that rainfall in the region is highly variable, even between sites that are only 20 km apart. Nunez et al. (1996) determined a regional rainfall gradient for southwest Tasmania, based on data from a number of different stations and from rain-gauges placed along an altitudinal gradient on three mountains. The results from this study broadly agree with the findings from their research. However, their rainfall map over-estimated summer rainfall in the region of Scott’s Peak/Airstrip Road.

The variability in rainfall within the region has important implications for fire management planning, as the rainfall data from Strathgordon (a commonly used station for the region) are not necessarily indicative of rainfall events in the region. The over-estimation of summer rainfall for inland, lowland regions indicates that issues relating to evaporation deficits and therefore, health of the mire ecosystem, may be more important than indicated by current rainfall maps of the region.

State government authorities carry out management burns in Tasmania according to prescription burning guidelines (Marsden-Smedley pers. comm.). The soil dryness index (SDI) is an important part of the equation of whether to burn or not. In organic soil environments such as buttongrass moorlands, the SDI is used to indicate whether fires lit in moorland vegetation will self-extinguish when they reach a boundary with adjacent ‘wet scrub’ vegetation (Marsden-Smedley et al. 1999), and whether organic soils will burn. The amended prescription cautions the user that the SDI is not reliable for moorland soils and that these soils should be assessed on site before burning (Marsden-Smedley et al. 1999). The chance of organic soils igniting is dependent on the soil moisture and organic content of the
soil at the site, with less organic soils not sustaining fire, independent of soil moisture values (Marsden-Smedley 1993). The guidelines also state that rainfall data for the previous 48 hours should be obtained from the site, as it is rainfall that is one of the major determinants of fuel moisture (Marsden-Smedley  et al. 1999). This is not always possible for remote area burns, in which case, the nearest station to the burn site is used to estimate rainfall. This has important ramifications, given that the Strathgordon data will over-estimate rainfall (soil wetness) for other areas in the southwest region (see Marsden-Smedley et al. 1999).

Our data imply that both Scott’s Peak and Airstrip Road are marginally warmer and drier than Strathgordon village. This is an important result given that Strathgordon village data have been used in the past to determine appropriate climatic parameters under which to carry out management burns.

No information on evaporation can be determined from the Scott’s Peak/Airstrip Road climate data, however, it is likely that evaporation is greater than is recorded at Strathgordon given that these sites experience less rainfall and slightly warmer temperatures. It is commonly believed that the organic soils of the southwest are saturated for most of the year. However, the data presented here indicate that the mean depth to the water-table lies at 10-15 cm below the surface (Table 3.1) for the three upslope plots. This correlates with the dense root mat of many of the moorland plant species at the site.

It is when soil profiles are not waterlogged, that accelerated decomposition of organic matter will take place, through oxidation of the soil profile (Billings et al. 1982 in Carroll and Crill 1997, Chimner 2000). The evaporation data from Strathgordon and the water-table data from the study site indicate that the soil profiles are dry for a period of time every year. Whether profiles are drier for longer periods of time than in the past cannot be determined. However, global warming scenarios suggest that periods of drying will become more frequent and longer lasting under warmer, drier climates. Under these conditions, there will be an increased risk of wildfire destroying the peat profile.

It appears that soil temperatures are heavily influenced by the amount of moisture in the soil, the presence of standing water in the plots and the amount of rainfall. The downslope plots 4-6 are wetter than plots 1-3. When rainfall occurs, plots 4-6 record higher temperatures than plots 1-3. This is reversed during dry periods. The moderating effect of water/soil moisture on soil temperatures is conceivable. Whether fire has an impact on water table levels could not be determined in this study.
The expected increase in soil temperature in burnt plots compared to unburnt plots was not detectable from the data collected by the data loggers. However, the adoption of a different methodology and more reliable data loggers may have given a result. The soil temperature data used here were mean values from 3 probes (at each depth) located randomly in the plot. Data loggers with more stations would have been able to record individual temperature data for each probe, which could have been located within a particular peat type. The fact that there is no result from the impact of burning on soil temperatures is more a result of sampling design (location of burn boundaries) and data logger malfunction. However, the soils within each plot are extremely variable and moisture and organic contents of the organic soil horizon vary dramatically across the plot/site. This study determined that the degree of water-logging has an impact on soil temperatures, therefore the interaction between climate, vegetation, soil type and topography may be such that it may be difficult to determine any impact of fire in the short term.

If the predicted climate scenario of global warming is real for southwest Tasmania, then organic horizons will decay, leading to the destruction of the organic soil profile. Given that western Tasmanian blanket bogs are already ‘warm’ compared to their northern hemisphere counterparts (Bridle 1994), any further warming without an increase in rainfall will be detrimental to the organic soil profile. However, analyses of limited climate data from inland stations in southwestern Tasmania do not show any evidence of a decrease in rainfall (Fig. 3.2). Recent research by Nunez (pers. comm.) suggests that any decrease in rainfall in southwest Tasmania may be offset by orographic effects in the region. Mountainous regions of southwest New Zealand also record no decrease in rainfall, despite this phenomenon occurring elsewhere in the country (Salinger and Griffiths 2001).

Continued research into the complex relationship between climate, topography, soil moisture availability and fluctuations and organic soil profiles is necessary to better understand the mechanics and ecology of the lowland peatland ecosystems of southwest Tasmania. This research is necessary and can be carried out along with more detailed fire research studies on the impact of fire on organic soil profiles.
Chapter 4 – The organic soils of Airstrip Road

Introduction


Organic soils in southwest Tasmania underlie numerous vegetation types, the most dominant vegetation type being buttongrass moorlands (Jarman et al. 1988, Pemberton 1989). These moorlands are considered to be pyrogenic and their widespread presence and perhaps extended range in the landscape of the southwest may be culturally induced (see Chapter 2). Given that organic soils will combust under appropriate conditions, it is highly likely that organic soils underlying pyrogenic vegetation will be impacted upon by fire events. Research suggests that the organic content of soils that are regularly burnt is less than in those that are unburnt (Garnett et al. 2000), and that the depth of the organic soil profile can be related to time since last fire (Bowman and Jackson 1981). Soils with a higher organic content will burn more readily than those with a lower organic content, given the same soil moisture conditions (Marsden-Smedley 1993). However, the organic soils in and around buttongrass moorland environments need to have an organic content of greater than 27% before they will burn, regardless of moisture content (Marsden-Smedley 1993).

The low nutrient environment of buttongrass moorlands, and the slow build up of nutrients post-fire is given as a major reason for the slow rate of succession from moorland to forest in the absence of fire (Bowman et al. 1986, Jackson 1999). However, the nature of water movement through the moorland is likely to have a big impact on nutrient transport post-fire. The greatest concentrations of nutrients are found in the surface soil horizon (Bowman et al. 1986, Jackson 1999). Nutrient losses after fire are restricted to those nutrients held in plant material (dead and alive). However, if the soils are also burnt, nutrient losses will be much greater and more damaging to the recovery of the ecosystem.

This chapter describes the physical and chemical characteristics of the organic soil profiles of the Airstrip Road site. Data of pre- and post-fire chemical analyses are presented and discussed.
Methods

Soil samples were collected around the perimeter of each of the 6 plots at the Airstrip Road site (see Chapter 3 for site layout). Samples were taken at 5 m intervals and 1 m in from the edge of the plot (Fig. 4.1). Soil pits were dug to the gravel/clay layer for descriptive purposes.

A total of 12 samples (approx 5 cm square and 15 cm deep) were collected for each plot. All 12 samples were stored in a fridge until laboratory testing took place. The soils were dried in an oven for 24 hours at a temperature of 105°C. Subsamples were weighed (approximately 15 g per sample) and then placed in a muffle furnace for 2 hours at 550°C. The samples were cooled in a desiccator and then reweighed and the organic content (by loss on ignition) was calculated. Further subsamples were sieved to <2 mm and then mixed with distilled water (1:5 weight to volume). This solution was used to measure soil pH and conductivity. Organic content, pH and conductivity were measured from each sample at 3 depths: 0-5 cm, 5-10 cm, 10-15 cm. Four additional surface soil samples (0-5 cm) were taken from the midpoints of the perimeter of each plot (Fig. 4.1). These samples were used for chemical analyses. The elements measured and the methods used are as follows: Mn, Zn, Cu, (1 part soil: 10 parts 1% EDTA @ pH 4, shaken 30 min.); P (1 part soil: 20 parts 1N sodium bicarbonate @ pH 8 (Olsen), shaken 30 min.); K, Ca, Mg, (1 part soil: 5 parts ammonium acetate @ pH 4.8, shaken 30 min.); Total N (followed method 7A3 (Kjeldahl N with salicylic modification), Raymont and Higginson 1992).

Soil samples for all plots were collected two weeks before the fire (April 2001) and one year later (April 2002). An additional four samples were collected from the three burnt plots (1,4,6) two months after the fire (Table 4.1). These samples were also sent away for soil nutrient analyses.

Soil depths were collected for each of the 30 vegetation quadrats (see Chapter 5) within each plot. Mean soil depth for the quadrat was determined by averaging 4 probe depths within the quadrat. Mean soil depth for the plot was calculated by averaging the mean soil depth for each quadrat (120 probe depths per plot, 4 x 30 quadrats).
Soil samples were collected for chemical analyses at two time periods (pre-fire and one year post-fire) for the unburnt plots (2, 3, 5), and three times (pre-fire, two months post-fire, one year post-fire) for the burnt plots (1, 4, 6).

Table 4.1 List of soil samples taken from each of the burnt and unburnt plots at the Airstrip Road site.

<table>
<thead>
<tr>
<th></th>
<th>Basic laboratory analyses</th>
<th>Soil chemical analyses</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pre-fire</td>
<td>pH, Cond, LOI 0-5, 5-10, 10-15 cm</td>
<td>Mn, Zn, Cu, P, K, Ca, Mg, N 0-5 cm</td>
</tr>
<tr>
<td>1 month post-fire</td>
<td>pH, Cond, LOI 0-5 cm</td>
<td>Mn, Zn, Cu, P, K, Ca, Mg, N 0-5 cm</td>
</tr>
<tr>
<td>1 year post-fire</td>
<td>pH, Cond, LOI 0-5, 5-10, 10-15 cm</td>
<td>Mn, Zn, Cu, P, K, Ca, Mg, N 0-5 cm</td>
</tr>
</tbody>
</table>
Data analysis

Soils data were entered into a spreadsheet for statistical analysis using SigmaStat 2.0 (SPSS 2003). This software automatically tests the data for normality and adjusts the tests to be used if the data do not satisfy the assumptions of a particular parametric test. ANOVA, General Linear Models or the Kruskall-Wallis test were used to test for differences in soil physical and chemical characteristics between plots and within plots at different depths.

The Paired t-test or the Wilcoxon Signed Rank test were used to test for significant differences in soil physical and chemical attributes within plots for pre- and post-fire samples.

The default settings in SigmaStat were used to determine pairwise comparisons (Tukey’s test for parametric data and Dunn’s test for non-parametric data).

Graphs of the mean values (with standard error bars) were created for all soil chemical data.

Statistical analyses were not undertaken between plots due to the topographic locations of the plots. The majority of the burnt plots (4 and 6) are located downslope from the majority of the unburnt plots (2 and 3). In the event of there being differences between these plots, it is as likely that the differences are a result of topographic location as burning treatment. It is noted that statistical analyses of the soil nutrients lacked power due to the small sample size (4 samples for each plot for each time) which was restricted by the cost of the analyses.
Results

Description of the soil profiles

The surface soil layer was fairly homogeneous throughout the study site. The organic soils were generally reddish-brown to brownish-black hemic peats under buttongrass (*Gymnoschoenus sphaerocephalus*) or shrubs and were black, humified (sapric) peats under restionaceous vegetation. Horizon colour broadly represented vegetation type and drainage characteristics and depth of profile, with darker (black) organic horizons occurring underneath lighter (reddish-brown) horizons. While the soil physical properties of the organic horizon were not vastly different, what lay underneath the organic layer highlighted the topographic gradient present at the site. The organic horizon generally overlaid a quartzitic gravel layer which was thicker (up to 10 cm thick) at plot one than it was at plot 6 (less than 5 cm). This gravel layer lay on bedrock for most of plots 1-3, but overlaid silty clay soils from plots 4 to 6. This clay layer reached a depth of 50 cm at plot 6, the lowest point in the study site. It is possible that the presence of these clays is a result of an old water course or lake bed, and may strongly influence the water tables in the 3 lower plots (4-6). The organic horizon was approximately 30 cm deep across the whole site, with all soils analyses being carried out on the surface 15 cm of soil. The majority of plant roots were concentrated in the surface 8-12 cm.

Organic soil depth

Plot 4 had the deepest soils while plots 1 and 5 had the shallowest soils (Table 4.2). Plot 4 was significantly deeper than all of the other plots except plot 3, while plot 3 was significantly deeper than plots 1 and 5. There was no obvious relationship between soil depth and topographic position (eg upslope – plots 1-3, downslope – plots 4-6).

Table 4.2 Mean soil depth (cm) for each plot at the Airstrip Road site.
Mean depth is not significantly different for plots with the same letter.

<table>
<thead>
<tr>
<th>Plot Number</th>
<th>N</th>
<th>Mean</th>
<th>Std Dev</th>
<th>SEM</th>
</tr>
</thead>
<tbody>
<tr>
<td>1a</td>
<td>30</td>
<td>26.667</td>
<td>4.827</td>
<td>0.881</td>
</tr>
<tr>
<td>2ab</td>
<td>30</td>
<td>29.917</td>
<td>5.617</td>
<td>1.026</td>
</tr>
<tr>
<td>3bc</td>
<td>30</td>
<td>31.150</td>
<td>7.272</td>
<td>1.328</td>
</tr>
<tr>
<td>4c</td>
<td>30</td>
<td>34.283</td>
<td>4.854</td>
<td>0.886</td>
</tr>
<tr>
<td>5a</td>
<td>30</td>
<td>26.683</td>
<td>4.387</td>
<td>0.801</td>
</tr>
<tr>
<td>6ab</td>
<td>30</td>
<td>29.433</td>
<td>4.743</td>
<td>0.866</td>
</tr>
</tbody>
</table>
**Pre-fire soil characteristics**

Soil physical properties that were analysed in the laboratory showed significant differences in a number of variables between plots. However, while there were statistically significant differences in values between plots, these differences were usually caused by values from one plot only.

**pH**

Using the complete (15 cm) soil samples, the pH varied significantly between the 6 plots (General Linear Model, df = 5, F = 4.713, P<0.001). However all of the variability was provided by plot 4, which had a higher pH than plots 1, 2, 3 and 6 (Fig 4.2). Soils were less acidic lower down the soil profile (df = 2, F = 3.309, P<0.05). There was no plot by depth interaction.

A comparison of the surface samples (0-5 cm) showed no significant difference in pH between plots. This result agreed with those from the laboratory’s chemical analyses.

![Fig. 4.2 Mean pre-fire pH value for each plot (0-15 cm sample) at the Airstrip Road site. Standard error bars are also shown.](image)

**Conductivity**

Conductivity (15 cm profile) was significantly different between the plots (df = 5, F = 2.334, P <0.05). However, once again the significant result was provided by plot 4 which had a lower conductivity reading than plots 2 and 3 (Fig 4.3). There were no significant differences between any of the other plots. Conductivity increased with depth across all plots (df = 2, F = 2.334, P<0.001). There was no plot by depth interaction.
When the surface (0-5 cm) samples were used, there was no significant difference in conductivity between plots. Again this agreed with the chemical analyses.

![Bar chart showing mean conductivity for each plot (0-15 cm sample) at the Airstrip Road site. Standard error bars are also shown.]

Fig. 4.3 Mean pre-fire conductivity for each plot (0-15 cm sample) at the Airstrip Road site. Standard error bars are also shown.

**Loss on Ignition**

Organic content (15 cm profile) was significantly different between plots (df = 5, F = 3.56, P<0.01). However, these differences were also centred around plot 4 where the organic content was significantly less than in plots 2 and 3 (Fig 4.4). There were no significant differences between any of the other plots.

Organic content decreased with depth down the soil profile (df = 2, F= 20.62, P<0.001). There was no plot by depth interaction.

There was no significant difference in organic content between the plots using the surface 0-5 cm sample. Once again this result agreed with those from the chemical analyses.

The mean organic content of the surface 15 cm of the profile was above 30% indicating that these profiles are organosols under the current Australian classification (Isbell 1996). However, the 5-15 cm horizons of plot 4 and the 10-15 cm horizon of plot 6 record organic contents of less than 30%, indicating that they are organic horizons rather than true peat (Fig 4.4).
Soil chemical analyses

There was no significant difference for any of the elements tested for the surface samples (0-5 cm) between the plots (1-6) before the fire (one-way ANOVA or Kruskal-Wallis H test, n=4 for each plot).

Post-fire soil characteristics

Potassium, calcium and manganese all recorded significantly higher values 2 months post fire than prefire (Figs. 4.5-4.7). Magnesium and nitrogen were higher 2 months after fire for 2 of the 3 burnt plots (Figs. 4.8-4.9). There was no difference in copper, phosphorus, zinc, organic content or pH in the burnt plots for this time period (Figs. 4.10-4.14).

Twelve months after the fire, copper, manganese, potassium and calcium were significantly higher than their pre-fire values, regardless of burning treatment (Figs. 4.5-4.7, 4.10). However, the statistical analyses for copper were not significant (P > 0.05). Phosphorus showed a dramatic increase after 12 months in plots 4-6 but this increase was not related to burning treatment (Fig. 4.11). Magnesium was higher in the three burnt treatments 12 months after fire than the pre-fire values (Fig. 4.8). Zinc decreased over time for all plots except plot one where it increased (Fig. 4.12).
Fig. 4.5 Mean potassium content (ppm) for each plot pre- and post-fire.

(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12 months after fire burnt/unburnt).

Fig. 4.6 Mean calcium content (ppm) for each plot pre- and post-fire.

(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12 months after fire burnt/unburnt).
Fig. 4.7 Mean manganese content (ppm) for each plot pre- and post-fire.
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12months after fire burnt/unburnt).

Fig. 4.8 Mean magnesium content (ppm) for each plot pre- and post-fire.
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12months after fire burnt/unburnt).
Fig. 4.9 Mean total nitrogen content (%) for each plot pre- and post-fire. (u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12 months after fire burnt/unburnt).

Fig. 4.10 Mean copper content (ppm) for each plot pre- and post-fire. (u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12 months after fire burnt/unburnt). Where standard error bars are absent, there was not enough material collected to carry out all analyses (usually from samples with a high degree of woody/rhizomatous material in the sample).
Fig. 4.11 Mean phosphorus content (ppm) for each plot pre- and post-fire. 
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12=12 months after fire burnt/unburnt).

Fig. 4.12 Mean zinc content (ppm) for each plot pre- and post-fire. 
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12=12 months after fire burnt/unburnt).

Organic content was substantially lower for all plots 12 months after fire (Fig. 4.13). However, organic contents measured in the laboratory with larger sample sizes do not support this result (Fig 4.14).
Fig. 4.13 Mean organic content (LOI %) for each plot pre- and post-fire. (u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12 months after fire burnt/unburnt).

Fig. 4.14 Mean organic content (% LOI) of soil samples taken from 0-5 cm for each plot, pre- and post-fire. U = unburnt, (pre-fire), u2 = unburnt (12 months post-fire), b = burnt (12 months post-fire). These samples were analysed at the University of Tasmania.

Soil acidity decreased over time across all plots, irrespective of burning treatment (Fig. 4.15). The larger soil data set (from analyses done at the University of Tasmania) showed no significant change in soil acidity over time (Fig. 4.16).
Fig. 4.15 Mean pH for each plot pre- and post-fire.
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12months after fire burnt/unburnt).

Fig. 4.16 Mean pH of soil samples taken from 0-5 cm for each plot, pre- and post-fire. U = unburnt, (pre-fire), u2 = unburnt (12 months post-fire), b = burnt (12 months post-fire). These samples were analysed at the University of Tasmania.

Conductivity is variable over time, showing no trend between burnt and unburnt plots (Fig. 4.17).
Fig. 4.17 Mean conductivity for each plot pre- and post-fire.
(u=prefire/unburnt, b1=2 months after fire/burnt, b/u12= 12months after fire burnt/unburnt).

**Discussion**

Despite differences in soil horizons underlying the peat, the organic soils of Airstrip Road are fairly uniform. They are peats or organosols (Isbell 1996, organic content >30%), approximately 30 cm deep, overlying gravels or silty clay horizons. Most plant roots are found within the top 8-10cm of the profile, with the organics becoming increasingly humified lower down the soil profile (> 10-15 cm depth).

Soil chemical analyses were only done using the surface 0-5 cm of the soil profile. Variability in soil characteristics across the site are minimal using the surface sample, probably reflecting the uniform nature of the vegetation over the site. Most variability in soil physical characteristics becomes apparent when data from the 0-15 cm soil samples are compared.

The change in soil nutrients post-fire is well documented for organic soils in Tasmania (Bowman and Jackson 1986, Jackson 1999). Most of the immediate changes to nutrients (generally increases in values post fire) are sourced from burnt vegetation (Jackson 2000). Nutrients that showed no real change post-fire were those that were held in the soil profile rather than in above ground vegetation. The limitations of using data from only four samples per plot are acknowledged. However, the results gained are in agreement with soil nutrient studies from elsewhere in southwest Tasmania (Bowman and Jackson 1986).
The results from the soil nutrient analyses become confused over time, which may be due to the small sample size of the nutrient analyses. In addition, the prefire history of the site will also impact on the results from this one experimental burn.

There was no difference in organic content pre and post fire. Again, sample size affects the results (see Figs. 4.13-4.14). Organic content is highly variable within plots and across the site. Samples taken from pools generally have less organic matter per unit weight than those that are taken from higher ground, due to higher decomposition rates in pools. Samples were taken at set distances from the perimeter of the plots. Variability may have been reduced had the samples been taken from particular micro-environments (either pools or raised ground) or under specific vegetation types.

It is unlikely that there would be a change in organic content only one year after a single fire event. Previous fire history is likely to have a much greater influence on organic content than one recent fire event. Data from Tasmania (Kirkpatrick and Dickinson 1984) and elsewhere (Garnett et al. 2000) show lower organic contents in peats that have been burnt than those that have not. But in both cases, whether it was a single fire or multiple fires, the fire occurred at least 10 years before samples were analysed. It may be that repeated burning of buttongrass moorlands may reduce the organic content of the horizon, or that one burn may produce the same effect but over a longer time period than one year. Organic content is directly correlated with moisture content (Bridle 1994), and thus the relationship between soil moisture, organic content and propensity to burn is recognised. Further research needs to be undertaken across fire boundaries within buttongrass moorlands to ascertain the impact of fire events on carbon stores.

Two complicating factors confuse the results from this study. While the planned management burn was undertaken during April 2001, a further ‘accidental’ burn took place in October 2001. None of the research plots were burnt but the fire burnt within metres of the plot boundaries. This fact becomes important when the layout of the study site is taken into consideration. As stated previously (Chapter 3), plots 4-6 are located downslope from plots 1-3, Plots 4-6 are likely to receive water from the upslope plots 1-3, and therefore are likely to show a build-up of nutrients post-fire. Evidence for this is shown in Fig. 4.11 where there is an increase in phosphorus 12 months post fire for plots 4-6. All plots show an increase in potassium 12 months after fire regardless of burning treatment (Fig 4.5). Any build-up of nutrients travelling though the soil profile, will disguise the impact of fire on the experimental plots. The second fire surrounding the research site is also likely to impact on nutrient reserves for the whole site.
Runoff from plot 1 to plot 6 is likely to involve water movement (and nutrient transport) through the unburnt plots 2,3 and 5. Therefore the lack of a significant difference in the soil nutrients between the burning treatments may be a result of water/nutrient transport through the peatland. Infiltration studies and dye traces would determine where water flows occurred and at what rate. It is likely that water will flow fairly rapidly through the more porous upper profile, and over the surface when the profile is saturated. How much nutrient transport has occurred in the past and how this has impacted on organic profiles is not known.

It is interesting to note that most variability in soil chemistry lies with plot 4, the wettest plot on the site. It is paradoxical that plot 4 has the deepest soils of the site, but these soils record the lowest organic content for the top 15 cm of the profile and are (possibly) the least acidic soils. Soils that are less acidic are likely to be less organic if acidity has some control over microbial activity at the site. However the soils may be deeper as they are also more constantly wet, thus retarding any microbial activity that depends on oxygen. Soils at plot 4 may be deeper because they are less likely to be impacted upon by burning. Alternatively the transport of soil particles downslope may also contribute to greater soil depths within this plot.

Soil temperatures are more stable in saturated soil profiles than in better-drained soils (Chapter 3), which may also influence the range of microbes present. In addition the dominant vegetation type may also influence decay (Chapter 5). Fisk et al. (2003) found that microbial activity could be determined in part by pH, but that correlations with other site variables were not clear. Clearly, further research needs to be carried out to determine the complex relationship between these factors.

Many of the results from this study are tentative. However, in general the results support other research carried out in the region. Further research into the role of drainage in blanket bogs would increase the understanding of the driving force of the system.
Chapter 5 – The vegetation of Airstrip Road

Introduction

Plant ecologists (and archaeologists) have argued that buttongrass moorland vegetation is dependent on fire for its continued dominance in the landscape of southwest Tasmania (see Chapter 2). While the use of fire as a management tool may be desirable in order to create a particular suite of plant/animal species, or reduce fuel loads, fire management planners need to balance the desire to burn with the need to preserve the organic soil horizon on the other. Current management practices are focused on reducing fuel and thus the fire risk to surrounding vegetation and infrastructure rather than conserving low productivity peatlands.

The impact of fire on organic soils has been outlined in Chapter 1. However, they will be reiterated here with respect to vegetation. The removal of all plant material (litter included) will result in an increase in bare ground. Burning creates a bare soil surface that is more readily heated than one that is protected by vegetation cover (Thompson 1987), which in turn is likely to affect biological activity at the site (Cole et al. 2002, Wickland 2001) and the physical nature of the soil profile. Decomposition of organic material is faster in burnt soils than in unburnt soils (Jeffries 1986, Morrissey et al. 2000). In addition, the release of nutrients into the soil layer from burnt vegetation is likely to stimulate biological activity by soil flora and fauna (Maltby 1980, Morrissey et al. 2000). The release of nutrients is also likely to promote an initial increase in plant productivity, both above and below ground. Sites with increased biological productivity in the soil profile may also record a change in carbon-dioxide emissions (Cole et al. 2002).

Evapotranspiration in wet heaths is taken to be the same as evaporation from a free water surface (Specht 1981). Therefore, as buttongrass moorlands may be seasonally dry, there may be an impact of an immediate decrease in evapotranspiration after a spring/summer fire. However, species that are able to recover quickly after fire are likely to transpire more in their growth phase than when they reach maturity. Accelerated evaporation losses have been recorded for New Zealand peatlands (Thompson 1987). It is likely that a more dramatic impact on peatland ecology is the removal of above ground biomass which will impact on soil infiltration rates (Mallik et al. 1984) and nutrient removal/deposition (Maltby 1980, see Chapter 4).
An underlying assumption in moorland research in Tasmania, has been that the sedge dominated peats are formed from the decay of above ground plant material. The validity of this assumption has not been tested. It is possible that peat production in buttongrass moorlands is predominantly supplied by below ground biomass, the roots and rhizomes of the moorland plant species. Campbell (1975, 1983) suggests that deep New Zealand peat deposits result from the roots of the restionaceous *Empodisma minus*. Chimner (2000) found that root growth contributed up to 60% of the below ground net primary production for southern Rocky Mountain fens, while Moore et al. (2002) concluded that root production and decay provided substantial inputs into the carbon budget of a *Sphagnum* bog.

Buttongrass tussocks consist of live and dead standing material. The plants are able to draw out essential nutrients from senescing plant parts, which remain attached. The fact that much plant material is retained as dead standing biomass on buttongrass tussocks suggests that if most peat-forming material is obtained from above-ground sources, then the process of peat formation would be extremely slow. This appears to be the case when comparing accumulation rates for southwest Tasmania, to those estimated from bogs in the northern hemisphere (Chapter 2, Bridle 1994). Thormann et al. (2001) found that the decomposition of *Carex* sp. roots was greater than of the leaves, with the result that much of the underlying peat deposit was created from leaves. However, they also found that *Salix* sp. roots contributed more to peat development than did the leaves.

Relationships between shrub characteristics and water-table levels have been reported in the literature (Moore et al. 2002). Bridle (1994) found a relationship between the dominance of shrub species in microhabitats that recorded fluctuating (better drained) water-tables, and sedges dominating micro-environments where water-tables were close to the soil surface. If such a relationship is widespread, then the probability of fire destroying the organic horizon may be predicted by the presence/height/density of particular shrub species. Marsden-Smedley (1993) reports that scrubby vegetation is generally underlain by better-drained fibric peats which dry out more quickly than moorland peats, and thus are more susceptible to fire.

This chapter describes the current vegetation of the Airstrip Road site, and discusses any relationship between vegetation and water-table levels. The following issues are discussed in the context of peat production at the site: the relative importance of above and below ground plant productivity; the impact of fire on litter accumulation and decay; and, the impact of fire on root accumulation.
Methods

Vegetation Survey

Thirty quadrats were located in each of the 6 plots at the Airstrip Road study site (see Chapter 3, Plate 3.1). A 1 x 1 m quadrat was placed around each of the ten dip wells (see Chapter 3, Plate 3.2), with the well being situated in the centre of the quadrat (Plate 5.1). The remaining 20 quadrats were located along two transects (10 quadrats per transect). Each 4.5 m long transect was randomly located within each of the 6 plots at the Airstrip Road site. Metal rods (approximately 1 m tall and 5 mm diameter) were placed at 50 cm intervals along the transects. These rods became the centre point for each 1 x 1 m quadrat.

The presence of all vascular plant species was recorded and each species was allocated a cover value using a modified Braun-Blanquet scale (Mueller-Dombois and Ellenberg 1974): 1 = <1% cover, 2 = 1-5%, 3 = 5-25%, 4 = 25-50%, 5 = 50-75%, 6 = 75-100%. In addition, the mean height of the dominant stratum, the amount of bare ground and the cover of non-vascular species were also noted. The soil depth for each quadrat was determined by taking the average of four probe depths (Chapter 4). The plots were initially surveyed in August 2000 (8 months before the burn). The plots were resurveyed in April 2002 (one year after the fire). Additional data collected at this time was the presence of litter (not attached to the plant). The plant species of buttongrass moorlands are perennial, therefore it was not necessary to collect vegetation data in any particular season. Plant nomenclature follows Buchanan (2002).

Identification and heights of two locally common shrub species (*Sprengelia incarnata*, Epacridaceae and *Leptospermum nitidum*, Myrtaceae) were taken from the 3 tallest individuals within a 50 cm radius of each dip well to determine whether there was any relationship between shrub height and water-table depth. Measurements were restricted to the three tallest specimens of each species within the area. Data were collected during April 2001, just before the experimental burn.

Litter accumulation

Steel discs (washers) approximately 2 cm in diameter, was placed over each metal rod described above (Plate 5.1). These disks were used to determine any litter build-up over the study period. Twenty metal disks from each plot were surveyed in April 2001 (before the fire) and again in April 2002 (one year after the fire). The following information was
recorded: presence and origin (plant species) of litter; depth of litter, amount of disk covered, and the presence of water over disk.

_Litter decay_

Samples of dead, senescing and live buttongrass leaves were collected from the Airstrip Road site in June 2001. Samples were taken from tussocks that had not been burnt in the recent fire. Samples were cut into approximately 5 cm lengths and were weighed into 5-7 g (wet weight) subsamples before being placed in a plastic fly-mesh bag (14 x 12 cm). These bags were buried at 5 cm depth every 50 cm along a 4.5 m transect in each of the 6 plots, giving a total of 10 bags per plot, 30 per burn treatment, 60 per site. An additional 5 bags were placed under rocks on the surface of the vegetation in an unburnt plot (plot 2) to provide a comparison between decay in the soil and at the surface. The bags were left in place for 12 months. Transects were randomly located within the plot, along a line of sight from one plot to the next. Slope was kept as uniform as possible.

It was necessary to know the approximate weight of water content of the original samples, before actual weight-loss (decay) could be determined. Therefore 10 samples of buttongrass leaf material of similar weights were weighed, placed in an oven at the same temperature for 5 days and were weighed on removal. The weight loss resulting from drying the samples was taken to be the moisture content. The mean moisture content for all 10 samples was 52.65%. The original weight of the undried, unburied samples of buttongrass leaves was adjusted by multiplying the wet weight by the dry weight 0.4735 to obtain an approximate dry weight for each sample pre-burial.

After one year, it was decided that only half of the buried bags would be dug up, leaving the other half to be retrieved after 2 years. Therefore 30 bags were dug up from the 6 plots after 12 months. Bags at 1 m intervals were dug up and taken back to the laboratory. The bags were placed in an oven at 55°C and were left for 5 days to dry. After this time, the bags were removed and the contents weighed. The 5 surface bags were also collected after 12 months. The remaining bags were collected after 2 years.

The following formula was used to determine total percentage of organic material remaining (ie not lost to decay processes):

\[ \frac{(a-b)}{a} \times 100 \]

where a = original (dry) weight of the sample, b = (dry) weight of the dug up sample.
Below ground plant productivity

Soil samples to a depth of 15 cm (described in Chapter 4) were taken from each plot. The samples were divided into 5 cm sections (0-5 cm, 5-10 cm, 10-15 cm). These sections were air-dried and sieved (to 2 mm) and the root material was removed from the soil. All roots (greater than 2 mm) were weighed to give an approximate value of belowground biomass for each of the plots. Root weights were divided by air-dried soil weight for each sample to enable comparisons to be made between plots/depths.

The following method was used to determine root productivity over time. An amount of peat was removed from the study site, dried and sieved, with all large roots (> 1 mm diam.) being removed. This peat was then placed in four fly mesh plastic bags or ‘socks’, 50 cm long x 5 cm diameter. In addition four bags were filled with a garden loam mix (pH of 5.5) to determine whether it could be used as an alternative growing medium to peat. One loam and one peat sock were buried in the ground in 4 plots at the Airstrip Road study site, (two pairs in the burned plots (1,6) and two close by in unburned plots (2,5). Randomly placed holes within 5 metres of the treatment combinations were augered out, with the socks being placed in the holes, and the peat/soil scooped into the sock. Average hole depth was 30 cm. Bags were filled to the surface. The socks were buried in February 2002 (Plate 5.2). Four bags were removed after 7 months (September 2002), a peat and loam pair from plot 1 and a peat and loam pair from plot 2. The other bags were removed in February 2003, 12 months after burial. On removal, the bags were taken back to laboratory and stored in a refrigerator until analyses took place.

The contents of each bag was emptied into a plastic tray, and the soils were sorted by hand, picking out any roots (greater than 1 mm). Soil samples were placed in a marked container and dried in an oven at 55°C for 5 days. Large roots were removed by hand. Smaller roots were removed by grinding and sieving the soil samples to < 2 mm. The roots were then weighed and the weight of roots per volume of soil was calculated.
Plate 5.1 Metal washer used to measure litter build-up

Plate 5.2 Peat ‘sock’ buried in unburnt plot
**Data analysis**

**Vegetation survey**

Data for the 180 quadrats were entered into DECODA (Minchin 1990), an ecological database. It was difficult to determine species of Restionaceae during the post-fire survey, therefore Restionaceous species were grouped for the purposes of comparing pre and post fire data. Ordination techniques were used to determine whether there was any difference in plant composition between plots. Analysis of variance (ANOVA or the non-parametric equivalent Kruskal-Wallis H test) was used to determine whether there were any significant differences between plots in the following variables: species cover, mean height of the dominant stratum, shrub height x water-table depth. Analysis of variance was also used to look for differences in vegetation cover pre- and post-fire between burning treatments.

**Litter accumulation**

Data were compared at time one to determine whether any differences existed between the 6 plots. Data were compared at time 2 to determine whether differences existed between the burnt and unburnt plots, and were compared to time one to see how much litter had built up in the unburnt plots over one year. Graphs of the mean values of litter covering discs in each plot were created.

**Litter decay**

Data were entered into a spreadsheet and a statistical package. ANOVA was performed to test for any difference in end weight between plots (1-6), between treatments (burnt, unburnt) and between burial depths (plot 2 only, bags buried at 5 cm and bags buried under rocks on the surface). Data were tested for normality. However, the power of the tests was not great due to the small sample sizes.

**Below ground plant productivity**

Two way analysis of variance was used to determine whether there was any difference in root weigh between plots (individual and wet v. dry) and at different depths. Tukey’s test was used to test for pairwise differences, if the ANOVA was significant. Statistical analyses were inappropriate for the root growth bags due to the limited number of samples. Therefore the results are descriptive only.
Results

Vegetation survey

Pre-fire

Twelve species were common across the study site, 11 of these occurred in every plot (Table 5.1). Eleven species showed significant difference in mean cover values between plots. However most differences were recorded due to strong differentiation of a species from 1 or 2 plots only. Plot 4 was most different to the others, recording a total of 33 significant differences in species cover when compared to the other 5 plots. Plot 6 recorded a total of 19 significant differences with plant species from the other 5 plots. Plots 1, 2 and 5 had the fewest number of significant differences (11) and were identical (in terms of species cover) to each other. The ordination diagram (Fig 5.2) illustrates the differences in floristic composition between plots.

Table 5.1 Mean cover values for each of the major plant species, mean water-table depths, soil depth and mean height of the tallest stratum for each plot.

Table 5.1 Mean cover values for each of the major plant species, mean water-table depths, soil depth and mean height of the tallest stratum for each plot.

<table>
<thead>
<tr>
<th>Plot</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Acanthium hookeri</em></td>
<td>24ac</td>
<td>22ac</td>
<td>28.3a</td>
<td>10b</td>
<td>16ac</td>
<td>26.5c</td>
</tr>
<tr>
<td><em>Baeckea leptocaulis</em></td>
<td>2.5ac</td>
<td>6.3ac</td>
<td>1.5bc</td>
<td>0bc</td>
<td>1.5bc</td>
<td>0a</td>
</tr>
<tr>
<td><em>Bauera rubioides</em></td>
<td>4.5a</td>
<td>8.5a</td>
<td>9.8a</td>
<td>5a</td>
<td>9.5a</td>
<td>7.3a</td>
</tr>
<tr>
<td><em>Boronia pilosa</em></td>
<td>5.8a</td>
<td>8.8ab</td>
<td>6.5a</td>
<td>0.3c</td>
<td>3.5a</td>
<td>1.75bc</td>
</tr>
<tr>
<td><em>Empodisma minus</em></td>
<td>1.5b</td>
<td>1.8b</td>
<td>1.6b</td>
<td>8ab</td>
<td>6.5ab</td>
<td>10.3a</td>
</tr>
<tr>
<td><em>Epacris corymbiflora</em></td>
<td>1.3b</td>
<td>4.3b</td>
<td>1ab</td>
<td>1.8a</td>
<td>3.8b</td>
<td>1.5ab</td>
</tr>
<tr>
<td><em>Eurychorda complanata</em></td>
<td>5.8bc</td>
<td>1.3bc</td>
<td>0.6b</td>
<td>10bc</td>
<td>11a</td>
<td>13.5ae</td>
</tr>
<tr>
<td><em>Gymnoschoenus sphaerocephalus</em></td>
<td>20.8b</td>
<td>20.3b</td>
<td>20.3b</td>
<td>53a</td>
<td>24b</td>
<td>22.8b</td>
</tr>
<tr>
<td><em>Lepidosperma filiforme</em></td>
<td>6.3ab</td>
<td>3ab</td>
<td>10.8a</td>
<td>1.5b</td>
<td>3.3ab</td>
<td>1.5ab</td>
</tr>
<tr>
<td><em>Leptocarpus tenax</em></td>
<td>0.3a</td>
<td>2.3a</td>
<td>1.8a</td>
<td>3a</td>
<td>0.3a</td>
<td>0.8a</td>
</tr>
<tr>
<td><em>Leptospermum nitidum</em></td>
<td>10.1ac</td>
<td>8.3bc</td>
<td>12.5a</td>
<td>5.3bc</td>
<td>10ac</td>
<td>12ac</td>
</tr>
<tr>
<td><em>Sprengelia incanaria</em></td>
<td>11.1a</td>
<td>12.3a</td>
<td>8a</td>
<td>7.8a</td>
<td>11a</td>
<td>11a</td>
</tr>
<tr>
<td>Non-vascular plants</td>
<td>2.8ac</td>
<td>4.35a</td>
<td>6abc</td>
<td>1.8bd</td>
<td>2.3abc</td>
<td>1.8cd</td>
</tr>
<tr>
<td>Bare ground</td>
<td>7.3a</td>
<td>4.5a</td>
<td>8a</td>
<td>3.5a</td>
<td>3.3a</td>
<td>11a</td>
</tr>
<tr>
<td>Mean Height tallest stratum</td>
<td>24.6b</td>
<td>28ab</td>
<td>29.5a</td>
<td>32a</td>
<td>25.2b</td>
<td>25.1b</td>
</tr>
<tr>
<td>Mean soil depth (cm)</td>
<td>26.7</td>
<td>29.9</td>
<td>31.2</td>
<td>34.3</td>
<td>26.7</td>
<td>29.4</td>
</tr>
<tr>
<td>Mean Water-table depth* (cm)</td>
<td>Feb to Apr pre-fire</td>
<td>20.6</td>
<td>22.3</td>
<td>20.0</td>
<td>14.5</td>
<td>18.5</td>
</tr>
<tr>
<td></td>
<td>Feb to Apr post-fire</td>
<td>20.0</td>
<td>21.9</td>
<td>18.6</td>
<td>-11.4</td>
<td>15.2</td>
</tr>
</tbody>
</table>

* a negative value indicates that the water table was below the soil surface.
There was no statistically significant relationship between vegetation cover or height and mean water-table depth in the pre-fire data set. However, some species recorded higher cover values in the wetter plots (4-6) than in the drier plots (1-3): *Empodisma minus* (H = 19.249, d.f. 1, P = <0.001); *Gymnoschoenus sphaerocephalus* (H = 13.894, d.f. 1, P = <0.001); and *Eurychorda complanata* (H = 14.017, d.f. 1, P = <0.001). Species that recorded higher covers in the drier plots (1-3) were; *Boronia pilosa* (H = 23.858, d.f. 1, P = <0.001); *Lepidosperma filiforme* (H = 12.887, d.f. 1, P = <0.001); *Acion hookeri* (H = 23.797, d.f. 1, P = <0.001); and non vascular plants (H = 25.157, d.f. 1, P = <0.001) (Table 5.1).

![Ordination diagram of floristic associations between plots 1-6.](image)

Fig. 5.2 Ordination diagram of floristic associations between plots 1-6.

Height of the tallest stratum was significantly greater in plot 4 than in all other plots except plot 3 (F = 8.021, P < 0.001). However, there was no relationship between height of the tallest stratum in each of the dip-well quadrats and mean water-table depth (Fig. 5.3). Nor was there any relationship between height of three individuals of the two dominant shrubs *Leptospermum nitidum* or *Sprengelia incarnata* within a 50 cm radius of the dip-well and mean water-table depth.
Post-fire

Not surprisingly there was a significant difference in cover of vascular plant species pre- and post-fire (within the burnt plots), and also between the burnt and unburnt treatments over the whole site. However, there was no significant difference in water-table level for the summer/autumn months pre and post fire (Table 5.1).

Non-vascular plant cover was significantly higher post-fire than pre-fire for all plots (Fig 5.4). Non-vascular cover was higher in the burnt plots than in the unburnt plots, and in the wetter plots (4-6) than in the drier plots (1-3) post-fire (Table 5.1).

Percentage cover of bare ground did not differ between plots before the fire (Table 5.1). However, bare ground was significantly greater in plots 4 and 6 than in all other plots post-fire (Fig 5.5).
Fig. 5.4 Non-vascular plant cover (% using midpoints of the cover classes) for each plot pre- and post-fire. Dark coloured columns represent plots that were burnt, light coloured columns represent plots that were not burnt.

Fig. 5.5 Percentage cover of bare ground for each plot pre- and post-fire.
Litter cover was not recorded in the pre-fire vegetation survey, but post-fire there was significantly more loose/detached litter in the burnt plots than the unburnt plots (Fig 5.6). Plots 4 and 1 had significantly more loose litter than all other plots.

![Graph showing percentage cover of loose litter](image)

**Fig. 5.6** Percentage cover of loose litter recorded from the vegetation survey for each plot post-fire. Dark coloured columns represent plots that were burnt. Light coloured columns represent plots that were not burnt.

**Litter accumulation**

The litter that was recorded from the litter tags/discs was generally sparse and was either attached to the plant (such as senescing leaves on buttongrass tussocks) or was dead, unattached vegetation (such as a woody stem or the stem of a rush). The mean depth of unattached material was 0.5 cm, while taller attached vegetation obscured the view of the disc. The greatest decrease in total litter cover (loose and standing) was in the burnt plots (Fig 5.7). There was no significant difference in total litter cover between the pre- and post-burn surveys for the unburnt plots (2,3,5) (Fig 5.7). While standing litter is common across the site, there was a shift from standing litter to loose litter in the burnt plots. Post-fire litter cover was greater in the burnt and wet plots 4 and 6 than in the burnt but drier plot 1.
Litter decay

The decay of buttongrass leaf material was not significantly different between the plots after 12 months, nor was it significantly different between the burnt and the unburnt plots. The five bags that were buried at the surface under rocks in plot 2, decayed less than those buried at 5 cm depth. However, this also was not statistically significant (Fig 5.8). The small sample size (5 bags per plot per time) is likely to play a major part in the lack of significant results.

Fig 5.7 Percentage cover of total litter (loose and standing) recorded from the litter tags for each plot pre- and post-fire.

Fig 5.8 Mean weight (g) of bags buried in each plot pre- and post-fire. 12 = 12 months after fire, 24 = 24 months after fire. 12, 0 = bags buried under rocks on the surface, 12, 5 = bags buried at a depth of 5 cm.
Plots 1 and 6 were heavier after 12 months burial than their initial bag weights. This increase in weight was due to the fine sediment attached to the litter in the bags. All plots showed a loss of litter after 24 months burial, but this loss was not significantly different between plots or between burnt and unburnt treatments (Fig 5.8).

**Belowground productivity**

There was no significant difference in mean root weight between plots, but root weight decreased significantly with depth down the profile (Fig. 5.9). Roots were heavier at 0-5 cm depth than at 5-10 cm or 10-15 cm (Tukey test, $P = 0.009$, $P < 0.001$ respectively), but there was no significant difference in root weight between the 5-10 cm and the 10-15 cm samples. There was no plot by depth interaction for the individual plots, or any difference in root weight grouped according to dry (plots 1-3) and wet (plots 4-6) plots.

![Fig. 5.9 Mean root weight (as a percentage of total dry soil weight) and standard error bars for each plot at each depth, taken from 0-15 cm soil samples (see Chapter 3 for details). 1 = 0-5 cm sample, 2 = 5-10 cm, 3 = 10–15 cm.](image)

Root growth from the ‘socks’ was variable between the two substrates (loam and peat), and between the burnt and unburnt plots (Table 5.2). Socks buried for 12 months had more roots (determined by root weight) than those buried for 7 months. No patterns can be discerned from the limited data presented in Table 5.2.
Table 5.2 Root productivity for each treatment by soil medium after 7 (plots 1 and 2) and 12 (plots 5 and 6) months burial

<table>
<thead>
<tr>
<th>Sample</th>
<th>Treatment</th>
<th>Length of ‘sock’ (cm)</th>
<th>Volume of soil (cm³)</th>
<th>Dry weight of roots (g)</th>
<th>Weight of roots (g) per 1 cm³</th>
</tr>
</thead>
<tbody>
<tr>
<td>Plot 1 loam</td>
<td>Burnt</td>
<td>31</td>
<td>608.68</td>
<td>0.011</td>
<td>0.00002</td>
</tr>
<tr>
<td>Plot 1 peat</td>
<td>Burnt</td>
<td>31</td>
<td>608.68</td>
<td>0.534</td>
<td>0.00088</td>
</tr>
<tr>
<td>Plot 2 loam</td>
<td>Unburnt</td>
<td>24</td>
<td>471.24</td>
<td>0.200</td>
<td>0.00042</td>
</tr>
<tr>
<td>Plot 2 peat</td>
<td>Unburnt</td>
<td>36</td>
<td>706.86</td>
<td>0.342</td>
<td>0.00048</td>
</tr>
<tr>
<td>Plot 5 loam</td>
<td>Unburnt</td>
<td>36</td>
<td>708.86</td>
<td>8.175</td>
<td>0.01153</td>
</tr>
<tr>
<td>Plot 5 peat</td>
<td>Unburnt</td>
<td>31</td>
<td>608.68</td>
<td>6.711</td>
<td>0.01103</td>
</tr>
<tr>
<td>Plot 6 loam</td>
<td>Burnt</td>
<td>35</td>
<td>687.22</td>
<td>9.900</td>
<td>0.01441</td>
</tr>
<tr>
<td>Plot 6 peat</td>
<td>Burnt</td>
<td>30</td>
<td>589.05</td>
<td>4.151</td>
<td>0.00705</td>
</tr>
</tbody>
</table>

Discussion

Fire has an enormous impact on bare ground cover, the amount of litter accumulation at a site and type of litter retained. Buttongrass moorlands retain dead material on the plant, which limits the amount of loose litter available for peat production on the ground. The Airstrip Road study site had not been burnt for 30 years, yet little loose litter was recorded from the unburnt plots (Fig 5.6), with most of this litter being less than 1 cm thick. The removal of litter as a possible source of peat and the creation of bare ground, is likely to have a large impact on peat production at the site. The experimental burn at Airstrip Road was carried out within fire management prescription guidelines (Marsden-Smedley pers. comm.), and resulted in standing litter being converted into loose litter on the burnt plots (Plate 5.3). However, current policy for fuel reduction burns is to remove as much plant material as possible, in order to reduce the likelihood of future fire events (Plate 5.4). The retention of litter on the soil surface is likely to protect the peat surface from drying out, to collect eroding soil particles as they are washed away, and to stem the reduction in infiltration rates.

While litter decay rates are extremely slow (Fig 5.8), litter accumulation and root productivity are also extremely slow (Table 5.2). While the carbon content of roots and buttongrass leaves is over 90% (authors, unpublished data), it is possible that carbon stored in the profile accumulated some time ago. There was no significant impact of fire on the decay of buried litter or root productivity. However, sample sizes were probably too small to detect any difference in decomposition between burnt and unburnt plots. Jeffries (1986) found that
decomposition increased under rotational burning and under burnt peat than in unburnt control moorland sites in the UK. More replicates in the micro-climatically variable plots (Chapters 3 and 4) might indicate whether there is an impact of fire on site productivity. As to how much this productivity is offset by increased decay remains to be determined.

Plate 5.3 Bare peat exposed after fire. Detached litter is visible on the left

Plate 5.4 Post-fire above ground biomass removal
Results of litter and root production at the site suggest that site productivity is low, as estimated by Marsden-Smedley et al. (1993). Post-fire production of loose litter may provide a carbon source for future peat development, but litter decay is extremely slow and the litter is likely to be lost to fire before it can be fully incorporated into the peat profile. It is possible that root productivity and decay contributes at least as much material as above ground productivity to the organic deposits of buttongrass moorlands. The dominance of sapric peats may be due to the nature of the peat forming material, i.e. roots rather than the leaves of silica rich buttongrass or sclerophyllous shrubs. An analysis of the resistance to decay of particular species may help in understanding how peat profiles develop, particularly with respect to physical characteristics such as degree of humification. Macrofossil analyses in conjunction with pollen analyses would be useful in determining the origin of peat profiles.

Balmer et al. (pers. comm.) are in the process of assessing vegetation responses to fire. Above ground biomass may be related to position of the water-table (Moore et al. 2002). However, few relationships were found between vegetation and water-table levels at the Airstrip Road study site. These data are supported by the initial vegetation survey of Barnes and Balmer (2000). Moore et al. (2002) found that shrub root biomass did not extend below the average growing season water table depth (Moore et al. 2002). The data presented here, while not restricted to shrubs, also support this (Table 3.1, Fig. 5.9). Root weight declined with soil depth, but was not related to the relative wetness of the site.

It is possible that the transport of nutrients down-slope to the wetter plots (Chapter 4) is the reason for the greater cover of non-vascular species post-fire in plots 4-6 (Table 5.1, Fig 5.4). The boundary of a recent fire on King Island Tasmania, was easily traced by the presence of moss growing on the burnt site (Plate 5.5). The increase in moss cover over the site is likely to act as a protective layer over the peat. It is unlikely that the shallow rooted mosses will be disturbed by frost events, leading to increased erosion as has been documented elsewhere (Maltby 1980). However, a predicted change to a warmer and drier climate is likely to impact on the frequency and severity of frost events in the region.

The relationships between vegetation, soil, climate and water movement are extremely complex. More research needs to be carried out on how these systems function, to be able to understand more fully the impact of fire. One would expect that the relative productivity of buttongrass moorlands to be high given relatively high air temperatures compared to blanket bog environments elsewhere in the world. However, northern hemisphere peatlands record greater accumulation rates than those of southwest Tasmania. Higher productivity rates should lead to higher decomposition rates (Yu et al. 2001), providing a plausible explanation
for the relatively amorphous nature of the organic horizon underlying moorlands. However, buttongrass plants are slow to produce loose litter, which in turn decays extremely slowly. The question is, what is it about this ecosystem that supports the continued presence/growth of peat? And what is the impact of fire on this support?

Plate 5.5 View of ash and colonising moss in a burnt paperbark swamp, Lavinia Nature Reserve, King Island

While vegetation was relatively homogeneous across the Airstrip Road site, plot 4 was the wettest plot which also recorded the tallest vegetation in the dominant stratum (Table 5.1). Plot 4 also retains much loose litter post-fire (Fig. 5.6) and records the deepest soils (Chapter 4). The value of litter as a trap for silt/organic material particles that have been mobilised post-fire needs to be considered.

It is not known how frequently Aboriginal people burnt buttongrass moorlands. Most recent data suggest that these environments were burnt at least once every 30 years (Tye 2002, Chapter 2). However, evidence of light burns is not always detectable. It is possible that the burns that have been detected are those that were severe enough to remove the vegetation layer and perhaps also impact on the soil profile. More research needs to be carried out to determine the role of loose litter in alleviating impacts from fire, especially on water retention, soil temperatures and sediment trapping.
Chapter 6 - Discussion and management implications

Introduction

This study aimed to present data on two factors dealing with the impact of fire on peatland environments in south-west Tasmania:

1. to clarify the role of Holocene fire regimes on the development and maintenance of buttongrass moorlands, and to determine whether fire frequencies can be identified;
2. to determine the impacts of current day management burns on physical, chemical and hydrological properties of organic soil profiles.

While neither of these aims were met in full, the information that was collected will provide a comprehensive background to guiding more detailed studies in future research.

Holocene fire regimes

While pollen and charcoal analyses show the continued presence of fire in cores taken from buttongrass moorland environments, it is not clear whether fire is necessary to continue the dominance of buttongrass moorlands in the south-west region. It is likely that moorlands have extended their range due to past fire events (Fletcher 2000, King pers. comm.), but it is also likely that they may be dominant due to their ability to out-compete other species on a particular substrate. It is likely that buttongrass moorlands are an edaphic climax community on nutrient poor, topographically constrained, water-logged environments.

The presence of moorland dominated blanket bogs in southwest Tasmania may have been culturally induced by Aboriginal firing regimes and wild fires. The persistence of blanket bogs in a climatically marginal region may now depend on changes in global climatic conditions. However, it is likely that organic soils would have been present even under woody vegetation given the past climate of the region. Many lowland rainforests are underlain by a thick organic horizon of reddish-brown peat to a depth of one metre (Pemberton 1989). If blanket bogs did extend their range as a result of past firing regimes, then the bogs formed through a process of paludification rather than terrestrialisation.

The use of pollen cores to determine the role of Holocene fire regimes on buttongrass moorland development is useful but speculative. More precise techniques need to be
developed in order to determine Aboriginal fire regimes using charcoal analysis from pollen cores. The lack of dating in the relatively shallow terrestrial peat deposits combined with the difficulty of dating such deposits means that any dates are speculative and are the result of a best estimate using evidence from deeper and less contaminated lagoon cores. Despite these difficulties, there appears to be some agreement over the accumulation rate of terrestrial peat in southwest Tasmania. Extrapolations from lagoon or pond cores estimate peat to accumulate at a rate of 1 cm over 30 to 60 years (Thomas 1993, Fletcher 2000). Tye (2002) estimated that 1 cm of peat may accumulate in 14 years in the upper surface of a terrestrial profile. However, this is unlikely, given the productivity of the site (Chapter 5).

If a 1 cm addition to the peat profile takes a minimum of 30 years to develop then fire frequencies can only be reconstructed for a 30-year time frame. Given than fire managers are not able to easily control a moorland fire that has been left unburnt for more than 12 years (Marsden-Smedley et al. 1999), then the 30 year time frame poses as many management questions as it attempts to answer.

Further research is needed to provide definite and reliable dates of terrestrial cores that can then be double-checked against dates and stratigraphies from aquatic cores to test the validity of the assumptions made about peat development with and without the influence of fire.

**Impact of recent fire events on lowland peatlands**

The results from this study indicate that the relationship between climate, vegetation, peat, hydrology and fire are extremely complex. The relatively simple, and small-scale approach adopted here highlights the complexity and provides meaningful information leading to future research.

An important result from this research was the finding that regional climates are variable even within 20 km of a meteorological station. It appears that distance from the sea and proximity to mountain ranges are important determinants of regional climate, with a gradient of decreasing wetness from the southwest to the northeast. Data from the Strathgordon meteorological station suggest that the climate is marginal for blanket bog formation. Given that the climate further inland both at the study site and at the Scott’s Peak meteorological station are warmer and drier than Strathgordon, it is likely that the region is even more marginal than was first suspected. Under global warming, it is possible that the whole region will experience a warmer and drier climate, hence higher evaporation rates than are currently
experienced. It is also possible that there will be no change to rainfall in the region, though there may well be an increase in temperature (Nunez pers. comm.). Under this scenario, plant productivity is likely to increase, which may lead to an increase in decay, leaving less organic material to form peat. Again the influence of hydrology in retarding productivity will be an important determinant.

With the climatic envelope determining which peatland types may form at a regional and global scale, the most important variable in determining peat accumulation and/or decay at a local scale is hydrology. Water-table levels are highly variable even within metres and will fluctuate daily (Fig. 6.1). The degree of fluctuation and response to rainfall event can be related to topographic position. While some dip-wells were often empty (Chapter 3), other seldom were. Topography has a major influence on water-table level, but soil physical characteristics were also highly variable within plots. How much of this variability can be attributed to past fire events is not known. The international literature states that fire can change the physical properties of soils and can cause them to be hydrophobic. Burning is known to reduce soil porosity by blocking pore space (Mallik et al. 1984) and this may lead to an increase in peat formation due to an increase in water-logging and subsequent decrease in decay rates (Charman 1992)

There were visual differences between the organic horizons found immediately below small ponds compared to those found below vegetated areas, but whether the ponds and the differences in soil physical conditions are caused by fire or by the structure of the vegetation is not known. The structure of buttongrass tussocks as a dense, circular mass of rhizomes is likely to influence the micro-geomorphology of the site. Likewise, the presence of burrowing crayfish creates drainage channels for the faster movement of water through the soil. Hydraulic conductivity is usually related to peat structure, with fibric peats having much faster water movement than more decomposed, sapric peats (Hughes and Heathwaite 1995, Charman 2002). Hydraulic conductivity also generally decreases with depth. The shallow and partially decomposed peats at the study site appear to move water relatively quickly as observed with changes in water-table levels with rainfall events (Fig 6.1).
Figure 6.1 Hourly responses of automatic water-table recorders to rainfall
3a, 3b were located in plot 3, 4a, 4b were located in plot 4. Water table levels for each well are relative only to that well and cannot be compared between wells. However, the response of the wells to rainfall or dry periods can be compared.

While failure of the data loggers restricted any comparison of differences in temperature between the burnt and unburnt soils, additional manual data suggests that there is a difference. The next step then is to determine how this difference impacts on soil physical and chemical properties through its impact on above and below ground biological activity.

Plot 4 lay at an extreme for the following variables: deepest soils; least organic soils; least acidic soils Chapter 4); highest mean water-table (Chapter 3, Fig. 6.1); tallest vegetation; most different vegetation composition compared to other sites (Chapter 5). There was no difference in chemical properties for this plot, though the sample number of soils was small. While there appear to be no direct or obvious relationships between these variables as recorded in Chapters 4 and 5, there is likely to be some ecological reasoning for the environmental extremes recorded in this plot. It may be that the location of the plot at a significant break in slope between plots 3 and 4 can account for all the variability. What was noted, was that the site wetness for plot 4 probably contributed to the provision of a litter layer post-fire and that the wetness of the plot may have contributed to less impact of other past fire events. This site was previously burnt in 1898, 1934, 1964, and 1972 (Marsden-
Smedley pers. comm.). If plot 4 is generally wet, it is likely to be less impacted upon during fire events than other drier plots. This might account for the relatively taller vegetation and deeper peats. The less organic soils could be a result of the soils being less acidic with more water flowing through the profile. During fire events, water movements redistribute soil nutrients (Chapter 4), which may enhance biological activity. The wetter plots lost proportionally more litter to decay in 24 months than the 3 drier plots (Fig. 5.8). The wetter plots also recorded less variable soil temperatures than the drier plots (Chapter 3). It is not known what impact previous fires have had on the site. It could be that the impact of one fire is negligible, but that the impact of a number of fires over time could have already altered the peatland system prior to the start of this experiment.

**Management recommendations**

Until more data are available, it would be wise to accept the draft management guidelines for burning vegetation underlain by organic soils (see Appendix 1). However, while these guidelines endeavour to protect the extent of peat over the landscape, they can do nothing without further research to protect the physical properties of the peat profile. It is highly likely that the organic content of frequently fired soils/vegetation types is lower than sites where fire is rare. The structure and water-holding capacity of burnt soils is also likely to differ greatly than for unburnt soils.

**Further research**

Limited data collected during this study suggests that carbon-dioxide emissions are greater in bare soil than from a vegetated surface. If it is necessary to carry out management burns on buttongrass moorlands then more data need to be gathered on the ecological functioning of moorland peats. Perhaps the most pressing research to come out of this project is the analysis of the importance of litter as a protective layer on burnt peat surfaces. An investigation into the ecological importance of a litter layer combined with soil respiration or carbon-dioxide emissions from the peat profile would add greatly to the understanding of this locally extensive and internationally important ecosystem. Baseline data on carbon-dioxide emissions from a number of peatlands across the State would enable managers to determine:

1. whether Tasmanian peatlands are under threat from global warming;
2. whether fire has an impact on carbon-dioxide emissions
3. whether the presence of litter ameliorates carbon-dioxide emissions in burnt peatlands.


Marsden-Smedley, J.B. (1997b) *Fire regimes in southwest Tasmania: implications for management*. Department of Geography and Environmental Studies, University of Tasmania.


Appendix 1

_DRAFT - Interim Guidelines for Hazard Reduction Burning On Buttongrass Moorlands (Blanket Bog terrain)_

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**Background**

Organosols or peats formed under extensive tracts of moorland in western Tasmania are vulnerable to fire and particularly to wildfire. They have been known to burn during and following wildfires often with catastrophic consequences where complete loss of the peat profile can occur leaving only bedrock (Pemberton 1988, Pemberton and Cullen 1995). This can have major ecological consequences on soils which can take up to 6000 years to accumulate. The impacts of management burns are less likely to result in catastrophic loss if care is taken during the burns to ensure adequate moisture in the soil and calm, fairly humid weather conditions prevail (Marsden-Smedley et al. 1999).

These guidelines are provided to assist fire managers plan and conduct management burns in buttongrass moorlands with the aim of minimising impacts on peat soils and their development.

**Setting the Scene**

Organosols form by the accumulation of organic matter. Decomposition of organic matter is retarded in humid environments with low evaporation, low to moderate temperatures and high rainfall. The raw material for peat formation, that is the native vegetation and any fallen dead organic matter is the fuel, which typically concern fire managers. Fire can impact on peat development by:

- removing the vegetative material (live and dead)
- removing the living and dead vegetative material and part of the organic soil horizon
- removing all of the above and all of the organic soil horizon(s) leaving a bedrock pavement or gravel surface with a very shallow interstitial peat layer.

Wildfire can be catastrophic for peatlands, so in the more vulnerable areas around roads, towns, and shack sites, a certain amount of hazard reduction burning is appropriate. Extension
beyond these areas may also be appropriate but will probably require further and continuing research into fire behaviour, passive (natural) extinguishment and the ecological implications of fire for the peatlands. Until this is established, creating low fuel zones adjacent to high risk areas could assist in preventing burns in more distant moorland areas. However there is always the risk, given the right conditions, of wildfires completely destroying peatland environments. From the perspective of maintaining peat forming processes and conserving peat soils, the main aim would be to limit the impacts of wildfire and peat burns. This would depend on successfully containing management burns surrounding high risk areas and a commitment to extinguishing peat fires where possible.

Below is a model of the possible impacts of fire on organic soils, as reported in the literature. These impacts may be applicable to Tasmanian peatlands.

The main objective of these guidelines is to provide managers with guidelines for management burns which could contribute to an ecologically sustainable approach to peatland burning and assist in the control of wildfires. To assist in achieving this the next section is divided into two parts, notably, planning and pre burn assessments and, post burn assessments and review.
Interim Guidelines For Management Burns in Blanket Bog Terrains (Buttongrass Moorlands)

Note these recommendations are based on the best knowledge available and will be refined following further work and appropriate research.

Planning and pre burn assessments

• **Burning Interval.** The preferred burning intervals would be 20 to 30 years as this appears to be the indicative natural burning regime from palynological analysis of Holocene fire regimes in these environments.

• **Autumn Burns.** The preferred time for conducting management burns is autumn under weather conditions and SDI’s established by the PWS (Marsden-Smedley et al. 1999). It is further recommended that soil moisture is checked in the field over a range of slope types at a representative number of sites to ensure soils are adequately saturated. Autumn burns are preferred to spring burns as the bare peat surface heats up significantly in summer, which may have implications for peatland hydrology and decomposition of the organic horizon.

• **Evidence for peat loss.** An assessment should be made of the on ground condition of the peat. Extensive gravel or bedrock outcrop is indicative of a degraded peatland. The fuel loads in these locations are usually lower than in areas with good peat cover so management burns may not be such an issue. It is recommended that any proposed burn area should have less than 15 % gravel exposure for further management burns to be conducted. Soil pedestals and truncated peat profiles (peat edges) are also indicative of soil loss. Future mapping of such environments would greatly assist in planning management burns.

• **Slope and Topography.** Management burns on flats or undulating plains are typically not a problem depending on previous fire history. Burns on slopes over 15° adjacent to mountainous or rugged terrain can lead to serious peat erosion, as these organosols are likely to be more fibrous and will dry out more quickly and completely than those on the flats.

• **Substrate or what underlies the peat.** Sandy substrates result in a freer draining peat, which is more likely to dry out and to dry out faster than peats overlying less porous and permeable substrates. Peats on sandy substrates occur on many islands and in the
coastal zone. Similarly predominantly quartzite bedrock weathers to form coarse sand or gravel which can also dry out quickly. Extra care should be taken in these environments and post fire checks are more important in these areas (see below).

- **Dead Vegetation and dry peat edges.** Dead vegetation and dead roots in particular provide access for fire into the soil. This is chiefly the case if the vegetation occurs on well drained ridge top positions. Extra care is needed when conducting ground inspections to ensure soil saturation in these locations to avoid fire spreading into the soil via the roots or exposed, dry peat edges. Other dry edges can be found next to rock outcrops (which absorb heat and provide better drainage) where contraction may have occurred in drying peats providing an aerated ignition point

**Post burn assessments and review**

- **Aerial assessments.** It is recommended that, if resources permit and aerial ignition has been used for the burns, once the fire front has past through and perhaps the following day an aerial inspection is conducted to investigate smouldering stumps, peat edges or pedestals. These can then be followed up on the ground to investigate whether the peat is alight. Some reports have suggested that even under ideal conditions, peat on the down fire sides of buttongrass stumps can smoulder after a burn has taken place. It is strongly recommended that particularly in sandy areas or areas underlain by highly porous substrates that these investigations are carried out.

- **Surface litter.** Although not ideal from a fire management perspective, saturated or wet litter left on the ground surface is though to protect the soil surface from drying out and it helps to prevent erosion. A ground inspection to estimate how much litter is left could, together with other data collected, assist in planning how to conduct future burns to try to ensure some material is left on the surface.

- **Follow up.** If smouldering peat is identified and steps are taken to extinguish the fire then a follow up after a specified period, (weather depending) is recommended to assess whether this was successful.