Effects of burning of slash piles of *Acacia* spp. and *Eucalyptus camaldulensis* biomass on soil physicochemical properties within Western Cape riparian and terrestrial areas

By

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DECLARATION

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SUMMARY

Removal of woody invasive alien plants from riparian and terrestrial fynbos ecosystems usually leads to the accumulation of large volumes of plant biomass. The three study species, *Acacia mearnsii*, *Acacia saligna* and *Eucalyptus camaldulensis* are well known for their ability to considerably increase above-ground biomass production. While some biomass may be removed for use as timber, or chipped for export, in cases where there is no alternative use for this biomass, or where site accessibility is an issue, burning of biomass in the form of piles or rows is practiced as a way of destroying biomass in situ. However, this approach has been reported to produce high temperatures, which may lead to altered soil properties and destroyed soil stored seed banks. Burn scars may develop on soil surfaces that were exposed to the burning of slash piles, which may remain unvegetated for extended periods of time. As a consequence, restoration may be patchy, uneven or delayed in post-clearing landscapes.

The aim of this study was to evaluate the seasonal and spatial effects of burning of slash piles of *Acacia* spp. and *Eucalyptus* spp. biomass on soil physicochemical properties. Four riparian study areas (Hermon and Robertson, both dominated by *Eucalyptus camaldulensis* prior to clearing, and Wit River and Rawsonville, *Acacia mearnsii* dominated) and one terrestrial study area (Blaauwberg; *Acacia saligna*) were selected within the fynbos biome. Burning was conducted in spring 2014 (Hermon and Blaauwberg) and winter 2015 for all other study areas. *Acacia mearnsii* and *A. saligna* piles had a volume of between 21.01 and 88.17m^3 and *E. camaldulensis* stacks had a volume of between 93.93 and 116.68 m^3. Soil samples were collected from the topsoil layer, 0-10 cm depth, prior to burning, post-fire and three subsequently seasons, from within the burn scars (in the centre, an intermediate position, i.e. between the edge and centre, and the edge), from the soil matrix (about 2 m from the edge), from a recovering reference site and from an invaded reference site. The collected samples were subjected to laboratory analyses for pH, electrical conductivity (EC), total carbon (C) and nitrogen (N), available N, available phosphorus (P), exchangeable cations and hydrophobicity.

At all study areas, soil pH (water), EC, available P and exchangeable cations increased significantly immediately after burning and had returned to pre-fire levels within one year of sampling, with the exception of soil pH, which persisted longer. This was with the exception of the Wit River riparian study area, where soil pH increased significantly and had returned to pre-fire within 3-4 months of sampling and soil EC was not affected at all. Total C and N

responded differently across study areas, where it remained unchanged at Hermon and decreased significantly at Rawsonville. Available N was not initially affected by fire at any of the study areas, but later showed higher levels within fire scars in *Acacia* invaded areas. No such difference emerged within fire scars of *Eucalyptus camaldulensis* invaded areas, suggesting that nitrogen may be more readily available in fire scars of riparian *Acacia* invaded areas. Hydrophobicity increased only at Rawsonville (*Acacia mearnsii*) as a result of fire and was not affected by fire in other areas. At the terrestrial site, soil pH, EC and available P increased significantly, but returned to pre-fire levels after a few seasons, with the exception of pH, which remained significantly higher.

The results from this study indicate that certain parameters such as soil pH, EC, available P and cations generally increase immediately after fire. In addition, the response of other properties including total C and N, available N and hydrophobicity may be governed by the characteristics of the ecosystem, soil type, burn fuel and seasonal variations. The implications of the study are that using fire as a tool for biomass management in post clearing landscapes may introduce unwanted soil physicochemical alterations, which may impact recovery, especially of native species.

OPSOMMING

Die verwydering van houtagtige indringerplante in rivieroewers en terrestriële gebied in die fynbos-ekosisteem lei gewoonlik tot die opeenhoping van groot volumes plantbiomassa. Hierdie studie fokus op drie hoof indringerplantspesies in die Fynbos ekosisteem, naamlik *Acacia mearnsii*, *Acacia saligna* en *Eucalyptus camaldulensis*. Hierdie spesies is bekend vir hul vermoë om die bogrondse biomassa te verhoog. Verder, waar die verwydering van hierdie indringerplante vir kommersiële doeleindes, soos die verkoop van hout nie moontlik is nie, of in gevalle waar daar geen alternatiewe gebruik vir die oorblywende plant biomassa is nie, en waar daar die toeganglikheid van die ooblywende biomass problematies word, word die verbranding van hope of rye biomassa gebruik as 'n manier om van die biomassa ontslae te raak. Hierdie benadering lei tot 'n toename in temperature, wat kan lei tot die veranderinge in grondeienskappe en die vernietiging van die saadbank. Brandletsels kan ontwikkel op grond oppervlaktes weens die impak van vuur. Die impak van vuur verhinder die regenerasie van plante, en areas met brandletsels kan braak le vir lang tye. Die verlies van saadbank na die vuur impak dien as n moontlike faktor hoekom die plant nie kon groei in die gebrande areas nie. Gevolglik kan regenerasie kolsgewys, ongelyk of vertraagd plaasvind in post-restorasie landskappe.

Die doel van hierdie studie was om die seisoenale en ruimtelike gevolge van die brand van *Acacia* en *Eucalyptus* biomassa op grondeienskappe te evalueer. Vier rivieroewer areas (Hermon en Robertson, gedomineer deur *Eucalyptus camaldulensis* voor die verwydering van biomass, en Witrivier en Rawsonville, gedomineer deur *Acacia mearnsii*) en een terrestriële studiegebied (Blaauwberg, *Acacia saligna*) is gekies in die fynbos bioom. Die brand in Hermon en Blaauwberg is uitgevoer in durende die lente in 2014, terwyl die brand in ander areas in die winter in 2015 uitgevoer is. Die volume van die biomassahope van *Acacia mearnsii* en *A. saligna* was tussen 21,01 en 88.17m^3, waar van die volume in *E. camaldulensis* tussen 93,93 en 116,68 m^3 was. Grondmonsters is geneem voor die brand, na die brand en drie seisoene opeenvolgend na die brand. Die monsters bevat grond vanaf die boonste grondlaag (0-10 cm diepte) en die monsters is geneem vanuit die volgende areas: (i) in die middel, (ii) by die intermediêre areas (die areas tussen die rand en die middle), (iii) die rand van die biomassahope, en (iv) die grond matriks (sowat 2 m van die rand). 'n Herstelde verwysings area en 'n indringerverwysings areas is ook in die studie ingesluit. Die versamelde monsters is onderwerp aan laboratorium ontleedings vir pH, elektriese geleiding (EG), totale koolstof (C) en stikstof (N), beskikbare N, beskikbare fosfor (P), uitruilbare katione en hidrofobisiteit.

Al die areas het getoon dat vlakke van grond pH (water), EG, beskikbare P en uitruilbare katione beduidend toegeneem het na die brand. Al die chemiese eienskappe het teruggekeer na die vlakke voor die vuur, en wel binne ‘n periode van een jaar, met die uitsondering van grond pH, wat langer hoog gebly het. Dit is in teenstelling met die Wit river studiearea, waar grond pH toegeneem het na die vuur en teruggekeer het na vlakke voor die vuur binne 3-4 maande, sonder om die grond EG te affekteer. Elke studie area het verskillende tendense in terme van die totale C en N getoon, byvoorbeeld, die Hermon area het minimale verandering getoon waar Rawsonville ’n statisties beduidende verskil getoon het. Die beskikbare N was aanvanklik nie geraak deur die brand nie, maar later is daar beduidend hoër vlakke binne die gebrand areas van *Acacia* gevind. Hierdie bevindinge is nie getoon in areas wat beinvloed was deur die brand van *Eucalyptus camaldulensis* biomassahope nie. Hierdie resultate beteken dat N na die brand meer beskikbaar was in brandletsels in Acacia ingedringde rivieroewerareas. Slegs die Rawsonville (*Acacia mearnsii*) area het n toename in hidrofobisities na die brand getoon, ‘n tendens wat nie by deur ander areas getoon is nie. By die terrestriële studiearea het die grond pH, EG en beskikbare P aansienlik toegeneem, en al die faktore het teruggekeer na die vlakke voor die vuur na 'n paar seisoene, met die uitsondering van grond pH wat aansienlik hoër gebly het.

Die resultate van hierdie studie dui daarop aan dat sekere parameters soos grond pH, EG, beskikbaar P en katione onmiddelik toeneem na ’n brand. Verder, die reaksie van ander eienskappe, insluitend die totale C en N, beskikbaar N en hidrofobisiteit mag moontlik beheer word deur die invloed van die ekosisteem, grondtipe, brandstof en seisoenale variasies. Die implikasies van hierdie studie is dat die gebruik van vuur om oorblywende plant biomassa te beheer (na die verwydering van indringerplante), mag moontlik ongewenste fisiese en chemiese veranderinginge veroorsaak wat die herstel van inheemse spesies kan beinvloed.

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CHAPTER 1

Introduction and literature review

1.1 Preamble

The National Water Act of South Africa (1998) recognises water belongs to all people, and in addition, water protection is necessary to ensure sustainable supply to all users. In South Africa, water resources are threatened by invasive species, especially woody species from genera such as *Acacia*, *Eucalyptus* and *Pinus*, while extraction, diversion and impoundment add additional pressure on water resources. Removing woody invasive plants is a major restoration activity in South Africa, typically garnering immediate results, while biological control measures are receiving more support and funding, however, dividends are only realised in the medium to long term (Enright, 2000). Though cutting and clearing of woody invasive species is seen as an approach that garners immediate results, the resulting biomass may pose new challenges for agencies tasked with clearing. In particular, large quantities of dry biomass may pose a fire risk, and may inhibit natural regeneration of native fynbos species (Holmes and Richardson, 1999). Management fires have been used to control biomass accumulation in fynbos ecosystems (Holmes and Richardson, 1999), along with removal of biomass for fuel, timber and wood chips (Holmes et al., 2008). However, the physicochemical and biological consequences of burning slash piles subsequent to clearing are not known. The aim of this study is therefor to evaluate how burning of slash piles may seasonally and spatially affect soil physicochemical properties in riparian and one terrestrial area.

1.2. Introduction

Water limited countries such as South Africa are faced with the major issue of effectively managing and maintaining the nation's water supply. The country receives an average rainfall below the world average, which amounts to about 500 mm rainfall per year (Friedrich et al., 2009; Knüppe, 2011). In addition, this average rainfall may fluctuate from year to year to year (Kusangaya et al., 2014) and is very unevenly distributed across South Africa. The central, eastern and northern parts of the country are characterised by summer rainfall patterns, and the south-west, which contains the fynbos biome, is a Mediterranean climate region with winter rainfall and plants experiencing pronounced drought stress during summer (Dallas & Rivers-Moore, 2014). The interior of southern Africa, including South Africa, is projected to become drier due to the impacts of global climate change, with more frequent and pronounced droughts (Kusanganya et al., 2014). It is, therefore, important to manage available water resources efficiently in order to maintain supply to urban and rural establishments, agriculture and industry, and to introduce actions that will ensure sufficient water supply into a warmer and drier future (Enright, 2000; Kusanganya et al., 2014). Rivers and associated freshwater ecosystems are central to supply of clean water and other services to rural and urban areas, and mismanagement may result in impairment of this ecological infrastructure, to the detriment of both ecological and socio-ecological systems (Dallas & Rivers-Moore, 2014).

Riparian zones form interfaces between water bodies and terrestrial areas, and are associated with rivers, streams and lakes (Naiman et al., 2005). These zones are complex, dynamic systems which provide habitat to a diverse range of plant and animal species that are adapted to continuously changing riparian environmental conditions (Naiman & Decamps, 1997). Stream waters, which may occasionally rise to flood level, continuously erode sediments from elevated and narrow mountainous areas to flat and wide floodplains where they become texturally sorted and deposited as the stream loses energy (Naiman et al., 2005; Graf-Rosenfellner et al., 2016), resulting in the formation of geomorphologically different landforms (Naiman & Decamps, 1997). It is not only sediments that are moved with the stream, flowing waters and seasonal floods may also contain dissolved solutes, suspended organics and plants seeds which are moved and deposited along the riparian landscape (Naiman & Decamps, 1997). Movement and deposition of both sediments and dissolved solutes results in the formation of spatially variable soils (Naiman et al., 2005; Graf-Rosenfellner et al., 2016), which are colonised by diverse plant communities adapted to the dynamic nature of the riparia (Naiman & Decamps, 1997). Riparian processes are, however, intimately tied to catchment processes in upland areas (Naiman et al., 2005). Many disturbances that affect ecosystem functioning in upland areas (e.g. fire), may, through manifold hydrological and biological connections, also have a bearing on riparian ecosystem functioning as well as on aquatic environments.

1.3. Invasive alien plants in fynbos

In the Western Cape of South Africa, the naturally diverse upland and riparian fynbos vegetation have been affected by human activities through urbanization, agriculture and inappropriate management practices (Holmes & Richardson, 1999). Human activity such as timber plantations, industrial and dune stabilisation has to some extent resulted in the spread of invasive alien plants (IAPs; Cronk & Fuller, 1995; Chamier et al., 2012; Tererai et al., 2013), which have replaced the previously Ericaceae, Proteaceae and Restionaceae dominated fynbos plant communities with dense populations of IAPs such as *Acacia* and *Eucalyptus* spp. (Holmes & Richardson, 1999; Richardson et al., 2007). Flowing streams within fynbos riparian ecotones play an important role in spreading IAP seeds from commercial plantation forests and other stands of IAPs, which upon germination colonise and change riparian vegetation structure (Naiman & Decamps, 1997; Richardson et al., 2007; Dzikiti et al., 2016). In South Africa, of the 1 257 000 hectares (ha) occupied by commercial plantations, about 491 000 ha and 95 000 ha are covered by *Eucalyptus* spp. and *A. mearnsii* respectively (Dye, 2013; Dzikiti et al., 2016), with most of these plantations concentrated in elevated catchments and headwaters in high rainfall regions such as the Western Cape. From these high rainfall regions, the IAPs spread downstream and establish in riparia where they pose threats not only to the native vegetation structure, but may also alter fynbos hydrological properties. Similar to riparian ecotones, within the Western Cape, terrestrial areas including coastal mountain ranges and coastal lowlands are also affected by IAP invasion (Enright, 2000). Invaders such as *Acacia cyclops* and *Acacia saligna* have been highly successful in invading large tracts of land previously covered by terrestrial fynbos (Holmes & Cowling, 1997; Yelenik et al., 2004).

Currently, most information available on streamflow reduction along invaded rivers is based on IAPs'; most of the available literature reports on streamflow reduction along invaded rivers, which is mainly attributed to elevated evapotranspiration rates (Dzikiti et al., 2016) and their deep-rooted nature (Pejchar & Mooney, 2009). Le Maitre et al. (1996) conducted a modelling study based on the data obtained in Kogelberg State Forest and reported that over a period of 100 years stream flow within the invaded catchment catchments areas would be 10.6% less than that of the uninvaded areas. Scott et al. (1998) found that in South Africa plantations that occur mainly along rivers (analogous to riparian IAP stands) reduce streamflow by about 3.2% per year. Le Maitre et al. (2002) estimated that major plant invaders are lowering the Sonderend River by approximately 1856 m^3 ha^{-1} $year^{-}1$ per year, of which *Eucalyptus* spp. and *A. mearnsii* consumed approximately 222 m^3 and 371 m^3 ha^{-1} $year^{-1}$, respectively. The preponderance of evidence suggests that the presence of woody IAPs threatens the hydrological functioning of riparian zones and streams or rivers (Pejchar & Mooney, 2009). Thus, it is imperative for land managers in South Africa to control and reduce the number of IAPs in ecosystems, especially the riparian ecotones. Empirical and modelled data on reduction in streamflow by IAPs in riparian zones, some of which are reported above, served as impetus to reduce stands of *Acacia*, *Eucalyptus* and *Pinus* spp. growing within riparian and terrestrial ecosystems.

Invaded terrestrial ecosystems are often located in moist regions of the catchment areas and they are exposed similar IAPs impacts as the riparian ecotones. These effects include the loss of indigenous fynbos vegetation, elevated evapotranspiration rates and increased above-ground biomass (Holmes et al., 2005; Pejchar & Mooney, 2009). Invasive alien plants in terrestrial areas also contribute to declines in catchment water balance, and may also affect groundwater, both in terms of recharge and quality (Jovanovic et al., 2009). Meijninger & Jarmain (2014) used a remote sensing model based on the Surface Energy Balance Algorithm for Land (SEBAL) to estimate annual evapotranspiration, and found that IAPs invaded areas had significantly higher average annual evapotranspiration rates (895 mm) than native fynbos vegetation (520 mm). Stands of *A. saligna* that displace native vegetation grow to high density, and are often taller, with higher biomass. They are also typically deeper rooted compared to native fynbos vegetation (Morris et al., 2011). Above-ground biomass as a result of woody IAPs invasion has to some extent altered fynbos vegetation structure, altered fire regimes, increased evapotranspiration rates and modified nutrient dynamics of the invaded area (Dye et al., 2001; Cilliers et al., 2004).

1.4. Clearing of woody invasive alien plants

The Working for Water (WfW) programme has been in operation for approximately 20 years, during which it has been tasked with protecting ecosystem services through control or removal of IAPs (Blanchard & Holmes, 2008; van Wilgen et al., 2012). Management of IAPs by the WfW has been done through interaction with local communities, which is aimed at providing employment to the public (Binns et al., 2001). This community interaction approach addresses a number of social factors, including poverty alleviation, raising awareness of IAP impacts and skills development within the communities involved (Turpie et al., 2008; van Wilgen and Wannenburgh, 2016). Through this approach, the WfW has been successful in establishing a link between ecosystem management and socio-economic issues in the South Africa (Hobbs, 2004). Control of woody IAPs has been based on the assumption that if the main stressor (i.e. IAPs) is removed, then the ecosystem would ‘self-repair’ and operate as under natural conditions (Ruwanza et al., 2013b). The success or progress of the WfW has been mainly measured or evaluated by researchers through monitoring native vegetation and/or stream flow recovery.

There are a number of examples of successful passive restoration of native vegetation after clearing of IAPs either by the WfW or through the use of WfW clearing methods. In a short-term study by Morris et al. (2008) along the Sabie River in and around the Kruger National Park, native plant diversity increased after clearing of invasive herbs and shrubs. Within the Berg River in the Western Cape, Ruwanza et al. (2013a) observed increased richness, cover and abundance of native vegetation richness four years after clearing of *Eucalyptus* spp. Blanchard & Holmes (2008) evaluated the recovery of native fynbos vegetation after clearing of IAPs by the methods of cut-and-leave on-site, cut-and-remove from the site, and/or cut-and-burn on-site and reported that the cut-and-remove from site method promoted indigenous plant re-generation more than any other methods. In other situations, clearing of IAPs on its own might not promote or lead to native vegetation recovery; in such cases, planting of selected fast-growing indigenous plants may be necessary (Galatowistch & Richardson, 2005). In addition, continuously monitoring cleared sites is necessary as the primary IAP may re-generate (Morris et al., 2008) and/or secondary alien invasion may occur with time (Blanchard & Holmes, 2008).

When the main stressor i.e. IAPs which consume large volumes of water, has been removed and native vegetation is recovering, evapotranspiration rates should decrease and streamflow improve. For instance, Prinsloo & Scott (1999) conducted a study on three catchments which were cleared of *Acacia longifolia* and *Acacia mearnsii* and reported an increase of approximately 8.8, 10.4 and 12 m^3 ha^{-1} day^{-1} in streamflow along the cleared areas. These findings were similar to those of a study by Rowntree & Beyers (1999), which reported a short-term increase in streamflow following clearing of *A. mearnsii* in the Western Cape. The potential savings in water with managing invasive species can also be empirically determined or modelled, e.g. along the Berg River in the Western Cape, Dzikiti et al. (2016) used a Penman-Monteith based model to estimate *Eucalypts* spp. evapotranspiration rates and found that 2110 m^3 of water can be saved per ha per year by clearing invasive *Eucalyptus* spp. These findings encouraged large-scale restoration of terrestrial and riparian ecosystems through various resource management programmes such as Working for Water, Working for Wetlands and Working on Fire (Carrick et al., 2015).

1.5. Impacts of burning of slash piles as part of woody invasive alien plant biomass management

Approaches for clearing IAPs as outlined in Holmes et al. (2008), include cut-and-leave on-site, cut-and-remove from the site for alternative use such as woodchips, and ring-barking or another form of killing large standing trees. Removing or destroying biomass proved to be the most direct path towards ensuring recovery of native species on site, though other factors may also influence the trajectory of restoration. Thus cut-and-leave on-site may be followed by stacking and burning which is a widely practiced procedure used to reduce wildfire risks and manage localised excess biomass (Holmes et al., 2008; Rhoades et al., 2015). The process involves stacking different sized pieces of plant biomass which are allowed to dry and then burned on-site. This method has been widely used in the Western Cape to dispose of excess IAP biomass accumulated from clearing (Cilliers et al., 2004).

Depending on the load characteristics and microclimatic conditions, burning of slash piles may be very severe and significantly different from natural wildfires (Certini, 2005; Rhoades et al., 2015). Intense fires that last longer (such as burning of slash piles) may produce temperatures ranging between 500-700^{o}C or more (Korb et al., 2004), which are capable of penetrating deep into the soil profile, altering soil physical and chemical properties, soil microbial communities and soil eco-hydrological properties. Esquilín et al. (2007) recorded temperatures of about 300^{o}C beneath the centre and 175^{o}C beneath the boundary zone of the slash piles. Hubbert et al. (2015) reported soil surface temperatures of approximately 200^{o}C for a prolonged period of more than 30 hr under large fuel stacks. Depending on the length of the period of exposure to high temperatures, damage on soils may either be short-termed, long-termed or permanent (Certini, 2005). However, burning of slash piles on site may be the only option, especially where removal proves difficult and costly (Behenna et al., 2008).

In order to lessen the risk of damage to soil and native vegetation seed bank, it has been recommended that pile burning be conducted on moist soil, which is more likely to occur during high rainfall periods (Burt et al., 2002; Holmes et al., 2008). Busse et al. (2005) conducted an experiment on dry and moist soils and recorded temperatures of about 600^{o}C on dry soils and 400-500^{o}C on moist soils. Waterlogged conditions may insulate the soil or delay heat transfer through the soil and thus limit potential damage to soils (Beadle, 1940; O'Donnell et al., 2011;

Zhao et al., 2012), and also be favourable for survival of some soil microbes (Neary et al., 1999).

The fuel load of slash piles will have a major influence on fire severity (Certini, 2005). Large piles are more likely to produce intense heat for extended periods of time and as a result, cause surface and belowground soil changes. According to Cilliers et al. (2004), slash pile burning may destroy both native and alien soil-stored seed banks to a depth of about 15 cm. Hubbert et al. (2015) conducted an experiment on different diameter wood arranged in like-sized piles, i.e. small, large, and mixed wood pieces and found that piles consisting of large wood pieces produced more heat and burnt for longer periods. Severe fires as a result of fuel load might be a problem in areas that are invaded by large stem diameter trees or IAPs which form dense stands and result in a heavy wood load for slash pile burning (Certini, 2005; Holmes et al., 2008).

1.6. Effects of burning of slash piles on selected soil nutrients

The extent of alterations to soil chemical properties and processes will vary depending primarily on fire severity and soil characteristics (Oswald et al., 1998; Giardina et al., 2000b). Loss of soil nutrients through volatilisation often occurs at high fire temperatures, which impacts on the nutrient stock and cycling (Neary et al., 1999). Immediately after combustion of plant biomass, nutrients concentrate in the ash and ultimately get incorporated and enrich the soil surface (Mohamed et al., 2007; Schafer & Mack, 2010). As a result of ash incorporation, nutrient concentrations in soil surface layers generally increase immediately after the fire, and eventually decrease seasonally as a result of run-off, leaching, and the wind, which may displace the nutrient rich ash (Giardina et al., 2000b; Certini, 2005; Mohamed et al., 2007; Schafer & Mack, 2010).

Soil organic matter (SOM) supplies crucial soil macro-nutrients, such as nitrogen and phosphorus, and micronutrients, such as boron (Sparks, 2003). Soil organic matter becomes consumed as a result of fire (Certini, 2005); the quantity of SOM consumed or lost during burning of slash piles is largely influenced by the fire severity, soil moisture content, depth, texture and presence of stable organic compounds such as humus compounds (Forgeard & Frenot, 1996; Certini, 2005; Neill et al., 2007). Forgeard & Frenot (1996) exposed the soil to

150^{o}C and 300^{o}C temperature regimes and found that considerable SOM consumption occurred at 300^{o}C within the 0-2.5 cm soil layer. Granged et al. (2011) also observed that high fire temperatures would consume considerable amounts of soil organic matter. Significant SOM consumption may affect subsequent decomposition processes and thus impact soil nutrient stocks and their cycling. Organic carbon is utilised by microbes during the decomposition process, which leads to the release of micro- and macro-nutrients into the soil, producing CO_2 in the process (Sparks, 2003); thus SOM destruction by fire will have major implications for soil carbon composition. Research has shown that fire-induced temperatures may have non-significant effects on total carbon if fire is less severe (Hinojosa et al., 2012; Fultz et al., 2016) or may significantly reduce total carbon when fires are prolonged with high temperatures (Johnson et al., 2011; Switzer et al., 2012). In addition to changing soil carbon composition, fire may also alter forms of SOM, and thus affect decomposition processes (Nave at al., 2011). Changing decomposition processes affects both carbon and nitrogen cycling, and as a result indirectly influences the carbon: nitrogen ratio (Nave et al., 2011, Naude, 2012).

Soil nutrients in the mineral soil are also consumed at different temperatures (Certini, 2005). For instance, soil nitrogen, including plant available nitrogen (i.e. ammonium and nitrate), becomes volatilised or transformed to gaseous form by less intense fires which produce temperatures of approximately 200^{o}C (Schafer & Mack, 2010). This suggests that nitrogen is highly susceptible to consumption by elevated temperatures during burning of slash piles. However, given variable conditions, post-fire soil nitrogen concentrations may remain unchanged (Hinojosa et al., 2012; Fultz et al., 2016) or it might significantly decline (Switzer et al., 2012; Badia et al., 2014). Plant available nitrogen also shows varying results after fire, for instance, ammonium (NH_4^+-N) may increase significantly (Hernandez et al., 1997; Kulmala et al., 2014; Fultz et al., 2016), significantly decrease (Kutiel & Naveh, 1987) or remain unchanged (Switzer et al., 2012). Nitrate (NO_3^--N) has been also reported to significantly increase after fire (Hernandez et al., 1997; Kulmala et al., 2014). Declines in soil available N could have major consequences for native fynbos species as plant growth is typically constrained by nitrogen and phosphorus (Cramer et al., 2014).

Phosphorus (P) is a soil macro-nutrient that is required for plant growth (Naude, 2012; Cramer et al., 2014) and it is considered to be stable, immobile and not easily leached through most soils (Sparks, 2003; Hinsinger, 2001). However, at high fire temperatures soil P may change into unstable and water-soluble forms, which make it more likely to be leached into soil profile or into flowing stream (Galang et al., 2010). Further, soil P volatilisation often occurs at a temperature of approximately 774ºC. Post-fire, soil available P has often significantly increased (Romanya et al.,1994; Giardina et al., 2000a; Badia et al., 2014), which may be due to an elevated pyromineralization rate of organic P from the combustion of organic compounds (Galang et al., 2010). On the other hand, Castelli & Lazzari (2002) and Wang et al. (2013) reported non-significant effects on available P as a result of burning. Higher availability of P could benefit native species germinating in post-fire environments as plant growth is limited by P, to a greater extent even than N (Power et al., 2010).

Soil exchangeable cations i.e. calcium (Ca), sodium (Na), potassium (K) and magnesium (Mg) are important nutrients for plants and are less vulnerable to heat. They may persist longer than N, P, or C and volatilise only at considerably higher temperatures (Verma & Jayakumar, 2012). As outlined in Neary (1999), Ca, Mg, Na and K volatilise at temperatures of approximately 1240ºC, 1107ºC, 880ºC and 760ºC respectively which may not be reached by most fires. However, after fire, cations become abundant on the soil surface as they are released from surface accumulated ash (Certini, 2005). The elevated concentrations of these plant important cations on the soil after fire may lead to high pH and electrical conductivity (Certini, 2005), as it has been shown by a number of authors (Kim et al., 1999; Menzies & Gillman, 2003; Switzer et al., 2012).

Apart from the direct impacts of fire on soil microbial populations, microbial activity and soil chemical reactions are also greatly, though indirectly, affected by modified pH levels after fire (Sparks, 2003; Certini, 2005). Soil heating and abundant base cations on the soil surface after fire result in high pH values (Certini, 2005). Higher fire-induced pH values may modify the microbial activity, and the release and availability of certain soil bound nutrients (Iglesias et al., 1997; Kulmala et al., 2014). These surface abundant base cations affect not only soil pH, but also soil electrical conductivity (EC), which often become elevated (Hernandez et al., 1997) or on rare occasions decrease after burning (Iglesias, 2010). In addition, high base cations on the surface after fire will also affect soil parameters such as sodium adsorption ratio and the capacity of soil to exchange for cations (Sparks, 2003).

Coarse textured soils that are high in organic matter may naturally develop hydrophobicity or water repellency, which inhibits water infiltration and promotes erosion (Doerr et al., 2000;

Dekker et al., 2001; Fox et al., 2007). This is as a result of SOM decomposition; which may result in development of temporary hydrophobicity on certain soils especially when the soil is dry as opposed to when it is moist (Doerr & Thomas, 2000). However, hydrophobicity may also develop in soils as a result of burning of slash piles of plant biomass which may release waxes that coat soil particles, making it hydrophobic (Mirbabaei et al., 2013). Post-fire, Fox et al. (2007) and Jeyakumar et al. (2014) have observed a significant increase in hydrophobicity; on the other hand, Inbar et al. (2014) indicated a considerable drop in hydrophobicity as a result of burning. On steep landscapes such as narrow valleys, hydrophobicity may lead to movement of large soil sediments by water; and on flat regions hydrophobicity might contribute to the soil surface and subsurface drying because infiltration or wetting is hindered (Neary et al., 1999). The differential responses of soil hydrophobicity to fire may be the result of an interaction between fires and soil properties such texture; sandy soils are more likely to show hydrophobicity compared to clayey soils (Doerr et al., 2000).

In contrast, under natural fynbos conditions where the above-ground biomass is dominated by fine restiods and ericoids, fire may not be as intense as that of burning of slash piles (Kraaij & van Wilgen, 2014). As a result these low intense natural fynbos fires may not expose the soil to very high temperatures, thus the effects on soil would be less severe and less localised than that of burning of slash piles. However processes similar to those of burning of slash piles are also present in fynbos fires. Processes such as volatilisation of soil nutrients, release of plant bound nutrients through combustion and the settling of enriched ash on soil surface post-fire are known to occur (Kraaij & van Wilgen, 2014). De Ronde 1990 reported a loss in cations (Ca and K) and phosphorus as a result of fire, while Stock & Lewis (1986) reported an increase total N and available N which could be the result of enriched ash on surface.

1.7. Formation and properties of burn scars and potential for restoration

Slash pile scars develop on soil surfaces as a result of heating, which alters soil properties to such a degree that vegetation establishment is hindered and soil remains bare (Korb et al., 2004). Severe fire, with heavy fuel distributed in dense patches, may form large scars which are affected more in the centre and less towards the edge of piles (Korb et al., 2004; Rhoades et al., 2015). These usually semi-circular surface scars often occupy significant area within an ecosystem (Fig. 1.1), where their presence may cause soil sediments to be unstable and highly vulnerable to erosion by water, particularly in steeply sloped areas (Neary et al., 1999).

Soil sediments are naturally stabilised by surface vegetation cover, however, this might not be the case in soils exposed to high fire intensities, where fire has altered soil properties and vegetation establishment is not supported (Neary et al., 1999; Korb et al., 2004). Native vegetation establishment may be slow or fast depending on the properties of the burn scar and environmental (Halpern et al., 2014), or in some cases, might require treatment such as seeding in combination with mulching (Fornwalt & Rhoades, 2011).

The residue remaining subsequent to fire, including ash and additional partially combusted plant debris is eventually incorporated into the soil where they elevate nutrient concentrations (Neary et al., 1999; Certini, 2014). The accumulated ash and combusted plant residue (Fig. 1.2.) may however persist long after the leaching process has mobilised enriched nutrients to different parts of the soil profile and the soil surface has returned to pre-fire conditions (Certini, 2005; Schafer & Mack, 2010; Certini, 2014). This persistent ash and partially combusted residue may be moved by surface run-off to adjacent areas and/or might form part of the soil profile where it influences soil forming processes (Certini, 2014). Incorporation of this residue into the soil will change soil organic matter properties since their composition varies from natural humus and as a result, may decompose differently from SOM (Knicker, 2011).

Figure 1.1: Slash pile burn scars (a) within a riparian ecosystem along the Breede River near Rawsonville after burning of dry *Acacia mearnsii* slash piles (Photo credit: J.T. Maubane, 2015) and (b) within a terrestrial ecosystem in Blaauwberg Nature Reserve, Cape Town after burning of dry *Acacia saligna* slash piles (Photo credit: J.T. Maubane, 2014).

Figure 1.2: Layer of ash and partially combusted plant biomass residue accumulated in the burn scar, after burning of a slash pile of dry *Acacia mearnsii* biomass. The photo was taken along the Breede River near Rawsonville (Photo credit: J.T. Maubane, 2015).

1.8. Motivation

Since its formation in 1995, the WfW has been at the forefront of the management and control of IAPs within South Africa's ecosystems, including the Western Cape (van Wilgen & Wannenburgh, 2016). Clearing of these IAPs from ecosystems is important especially in water-limited countries such as South Africa since these species are known to consume larger volumes of water than native vegetation. Effective management of these IAPs through clearing may provide favourable conditions for ecosystem recovery and protecting water resource. One way of measuring success is based on the recovery of native vegetation post clearing (van Wilgen et al., 2012), while estimating how streamflow and groundwater levels would increase after clearing is another important metric (Dzikiti et al., 2016). As a result, most of the information available on Western Cape fynbos invasion is on the impact of IAPs on stream water levels and how native vegetation would recover after clearing of IAPs. Very little has been reported or recorded on how the existence of IAPs and approaches to their clearing affects the Western Cape fynbos soils, particularly on soil chemistry and soil processes (Chamier et al., 2012; van Wilgen et al., 2012; Jacobs et al., 2013). Further, fynbos soils are not well understood in terms of how they are might respond to fire, especially as Certini (2005) suggested that in some situations, the impact of fire may be irreversible.

Some of the IAP control methods may hinder native vegetation recovery and should be practiced carefully within the fynbos biome (Holmes et al., 2008). One such method is burning of slash piles, which in international literature have been reported to be detrimental to soil properties and may lead to an altered soil state to the extent that vegetation establishment becomes hindered (Rhoades et al., 2015). To destroy excess biomass of woody IAPs in South Africa, wood may be stacked to form large slash piles for burning. Large slash piles, with large wood stumps might burn very hot and for extensive time periods, which may result in significant damage to soil properties (Rhoades et al., 2015). Soils exposed to burning of slash piles often develop surface scars, which are bare soils that do not develop vegetation, becoming unstable and vulnerable to erosion by flowing water, or secondary invasion (Korb et al., 2004).

It is, therefore, important to gain knowledge on how burning of slash piles as an IAP clearing method will affect both terrestrial and riparian soil properties. This study aims to evaluate how burning of slash piles as used for biomass management subsequent to clearing of *Acacia* and *Eucalyptus* spp. in the Western Cape riparian and terrestrial ecosystems would affect soil physicochemical properties. After burning of slash piles, selected physicochemical properties were evaluated both seasonally and spatially in selected riparian and terrestrial study areas.

In addressing the main study aim and objectives, certain physicochemical properties were selected to and tested. These parameters include soil pH, EC, total C and N, available N, available P, cations and hydrophobicity. They were selected because they have been reported in most of the literature as being susceptible to effects burning (Neary, 1999; Certini, 2005); and these parameters influence the ability soil to support vegetation (Spark, 2003), hence understanding their response would be beneficial in rehabilitation of fire scars. For instance soil pH have been reported to increase as a result of fire (Neary, 1999; Korb et al., 2004) which may result in the release and mobilisation of soil bound nutrients (Spark, 2003); cations have also been shown to increase after burning (Cetrtin, 2005) which may lead to saline soils (Spark, 2003).

1.9. Research aims, objectives, key questions and hypothesis

1.9.1. Main Aim

This study aims to evaluate how burning of slash piles may seasonally and spatially affect soil physicochemical properties in riparian and one terrestrial area.

1.9.2. Objectives

a) To evaluate the effects of burning of slash piles of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass on riparian soil physicochemical properties.
b) To evaluate the effects burning of slash piles of *Acacia saligna* biomass on terrestrial soil physicochemical properties.

1.9.3. Key questions

a) How will burning of slash piles of *Acacia mearnsii* biomass seasonally and spatially affect selected soil physicochemical properties of riparian soils?
b) How will burning of slash piles of *Eucalyptus camaldulensis* biomass seasonally and spatially affect selected soil physicochemical properties of riparian soils?
c) How will burning of slash piles of *Acacia saligna* biomass seasonally and spatially affect selected soil nutrients of terrestrial soils?
d) What is the trajectory followed by physicochemical properties subsequent to burning of biomass of *Acacia* and *Eucalyptus* spp.?
e) Are there differences in how terrestrial and riparian sites respond to fire?

1.10. Study species

Woody IAPs such as Australian *Acacia* spp. and *Eucalyptus* spp. (Table 1.1) are considered as problematic plant invaders with the Western Cape, South Africa (van Wilgen, 2009). Their ability to grow fast and adapt to various environmental conditions are favourable traits allowing them to out-compete native vegetation (van Wilgen, 2009). The widespread presence of these woody IAPs within the Western Cape fynbos threatens the structure and functions of the fynbos ecosystem (van Wilgen, 2009; Naude, 2012). Clearing of IAPs may thus play an important role towards the recovering of various components of the fynbos ecosystem.

However, clearing also leads to the accumulation of excess biomass, especially since this woody IAPs (*Acacia* spp. and *Eucalyptus* spp.) are known to increase above-ground plant biomass (Table 1.1; Chamier et al., 2012). As a measure to rehabilitating the fynbos ecosystem, it is necessary to effectively manage excess biomass of IAPs such as *Acacia saligna, Acacia mearnsii* and *Eucalyptus camaldulesis*.

Some salient information on the study species is given in Table 1.1. Notable is that *Eucalyptus camaldulensis* grows to very large sizes, and hence can present a major challenge to clearing teams, and especially biomass management. On the other hand, *Acacia* spp. grow in very dense stands, where environmental resources allow, and are nitrogen fixers, which will have some major consequences for ecosystem functioning by impacting soil biogeochemistry following fire.

1.11. Study Area

Study sites were selected near Hermon along the Berg River (33°28’42.83”S, 18°56’13.83”E), near Bain’s Kloof Pass along the Wit River (33°32’36.02”S, 19°10’20.01”E), near Rawsonville along the Breede River (33°43’43.43”S, 19°28’25.63”E), near Robertson along the Breede River (33°50’17.08”S, 19°52’28.33”E) and in Blaauwberg Nature Reserve, Cape Town (33°45’14.61”S, 18°29’35.30”E). These study areas (Fig. 1.3.) fall within the Western Cape Province, South Africa and experience Mediterranean climate type, with warm dry summers and receives most rainfall during the wet cool winters (Naude, 2012).

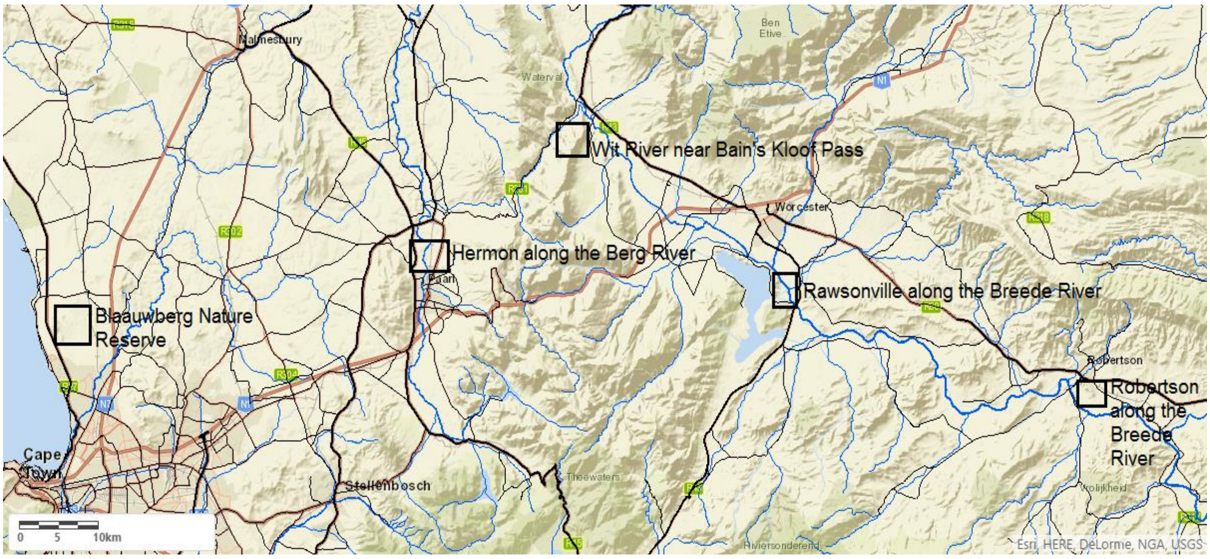

Figure 1.3: Map of study sites located within the Western Cape South Africa. Each study area is represented by a square. The map was created using the South African National Biodiversity Institute (SANBI) online BGIS system (SANBI, 2016).

Table 1.1: Some pertinent information on the three study species (Adapted from Chamier et al., 2012)

Property	*Acacia mearnsii*	*Acacia saligna*	*Eucalyptus camaldulensis*
Family	Fabaceaea (Mimosoideae)[a]	Fabaceaea (Mimosoideae)[a]	Myrtaceae[a]
Common Name	Black Wattle[a]	Port Jackson Willow[a]	Red River Gum[a]
Origin	Australia[a]	Australia[a]	Australia[a]
Introduction in South Africa	1800s[i]	Around 1845[i]	19th century[q]
Occurrence	Mainly along watercourses in high rainfall regions[b], but also in upland areas[o]	Along rivers, and coastal lowlands of the Western Cape[m]	Wide range of habitats, but mainly along perennial rivers[r]
Average height	≈ 20 m[c]	≈ 2 - 6 m[c]	≈ 30 m[s]
Average stem diameter	Range from 10 - 35 cm[p]	Range from 5 - 10 cm[n]	≈ 50 cm[s]
Nitrogen-fixing Habit	Nitrogen-fixing[a]	Nitrogen-fixing[j]	Non-fixing
Water Use	High evapotranspiration rate of about 1500 mm across South Africa[d]	Within South Africa, A. saligna consumes an annual average of 171.13 million m^3 of water[m]	High water consumption through elevate evapotranspiration rates[s]
Biomass	Increases above-ground plant biomass[e,f]	increases above-ground plant biomass[e]	High above-ground biomass production[v]
Management approaches	Clearing through chemical and mechanical methods[h]. Control through biological control agents[g]	Mechanical clearing, which may be followed by herbicide application[k]. Biological control approach through the use of biological agents[l]	Mechanical clearing, followed by burning or herbicide application[w]
Biological control	*Melanterius maculatus* and *Dasineura rubiformis* used as biological agent to reduce seed production[g]	*Uromycladium tepperianum* has been used as a biological control agent[l]	None[t]
Secondary use	Timber, paperboards and tannin[p]. Bio-energy[x]	Bio-energy[x] and fire-wood[n]	Bio-energy[x], timber, and fire-wood[u]

a Coates Palgrave, 2002; b Moyo et al., 2009; c Cronk & Fuller, 1995; d Dye & Jarmain, 2004; e van Wilgen & Richardson, 1985
f Witkowski & Mitchell, 1987; g Impson et al., 2008; h Holmes et al., 2008; i Shaughnessy, 1980; j Yelenik et al., 2004
k Krupek et al., 2016; l Strydom et al., 2012; m Le maitre et al 2000; n Wood, 2012; o Crous et al., 2012
p Praciak & CABI, 2013; q Forsyth et al., 2004; r van Wilgen et al., 2007; s Dzikiti et al., 2016; t Moran & Hoffmann, 2012
u Tererai et al., 2013; u Phiri et al., 2015; w Blanchard & Holmes, 2008; x Mugido et al., 2014

1.11.1. Hermon

The Berg River is a 294 km long perennial river where the catchment has an average annual precipitation of approximately 550 mm, most of which is received during the rainy winter season, with occasional floods (Between May and August) (Ruwanza et al., 2013b; Tererai et al., 2013). Approximately 3800 km^2 of the 7715 km^2 catchment area is urban and agricultural land which extracts water for domestic and irrigation purposes. The upper parts of the river flow into the Berg River Dam and ultimately egresses into the Atlantic Ocean at Velddrift (Ruwanza et al., 2013b; Tererai et al., 2013). The Berg River catchment area comprises of low nutrient acidic soils which originate from sandstones and quartzite of the Cape Supergroup (Rebelo et al., 2006; de Villiers, 2007). In their natural state the riparian zones are nutrient poor and occupied by riparian vegetation which includes *Melianthus major*, *Kiggelaria africana* and *Diospyros glabra, Brabejum stellatifolium Metrosideros angustifolia, Podocarpus elongatus* etc.; however, to a larger extent the area is densely invaded by woody IAPs such as *E. camaldulensis*, *A. mearnsii* and *Populus* spp. (Ruwanza et al., 2013b; Tererai et al., 2013). Within Hermon, these IAPs have displaced the Atlantis Sand Fynbos vegetation type (Rebelo et al., 2006; Tererai et al., 2013).The selected site was invaded and cleared of *E. camaldulensis* and slash piles created ready for burning (Fig. 1.4).

Figure 1.4: A 2013 image of the Hermon study area. The study area has been cleared of *Eucalyptus camaldulensis* and biomass piles allowed to dry. The slash piles are visible as grey dots, with an example indicated with a white arrow (Google Earth, 2016).

1.11.2. Wit River

The Wit River is a small perennial river of approximately 11 km which originates in the Haweques Mountain chain and flows alongside Bain's Kloof Pass to ultimately join the Breede River (Brown et al., 2004; Vosse, 2007; Naude, 2012). The river flows at the bottom of steep mountains in an area that receives an annual average rainfall of about 1200 mm, which mainly occurs between May and August and may be accompanied by occasional floods (Rebelo et al., 2006). The local geology is primarily of Peninsula Formation which is an approximately 1500 m thick quartzite layer contained within the Table Mountain Group (Rebelo et al., 2006). The deep bedrock is overlain by acidic soils with variable sized rock pebbles (Rebelo et al., 2006). In their pristine state, the soils are covered by Hawequas Sandstone Fynbos vegetation type, however, native vegetation has been displaced, primarily by dense stands of *A. mearnsii* species. The selected site is located along the Wit River in Bastiaanskloof (Fig. 1.5), were *A. mearnsii* is being cleared by the private land owner.

Figure 1.5: A 2014 image of the Wit River study area in Bastiaanskloof, invaded by *Acacia mearnsii*. *Acacia mearnsii* stands are indicated with a white arrow (Google Earth, 2016).

1.11.3. Rawsonville

<u>Breede River</u>

The Breede River is located about 250 km from Cape Town. From its origin in the Ceres Valley, it flows south-eastward towards the Indian Ocean (Steynor et al., 2009). The largest river in the

Western Cape, it is made up of seven basins over a catchment area of approximately 12 600 km^2 (Steynor et al., 2009). Tributary rivers which flow into the Breede River include the Holsloot River, Molenaars River and Wit River (Brown et al., 2004). The Breede River drains into Theewaterskloof and Brandvlei municipal Dam storages, with the winter rainy season as the greatest period of water contribution (Steynor et al., 2009).

The two study areas along the Breede River are located in the Worcester region, which falls within the Succulent Karoo biome. The rivers within this area originate in mountainous areas entirely covered with fynbos, and in its pristine state, the riparia is primarily dominated by fynbos vegetation type (Le Maitre et al., 2009). The study area located at Rawsonville (Fig. 1.6) where the average annual rainfall is about 480 mm (Rebelo et al., 2006). The Breede River local geology in this area is mainly made up of Malmesbury Group and Bokkeveld Group shales overlain by fine loamy sands mixed with round alluvial pebbles (Rebelo et al., 2006). These fine loamy sands are covered by the Breede Alluvium Fynbos vegetation type.

1.11.4 Robertson

The second study area along the Breede River is located near Robertson (Fig. 1.7), where most rainfall is received between May and August (Rebelo et al., 2006). Within this area, the Breede River banks are mainly covered by Aeolian sands overlain by Breede Sands Fynbos vegetation type (Rebelo et al., 2006).

Figure 1.6: A 2014 image of the Rawsonville study area along the Breede River at Riverside farm invaded by *Acacia mearnsii*. Stands of *A. mearnsii* are indicated by a white arrow (Google Earth, 2016).

Figure 1.7: A 2014 image of the Robertson study area invaded by *Eucalyptus camaldulensis*. The white arrow shows stands of *E. camaldulensis* (Google Earth, 2016).

1.11.5. Blaauwberg Nature Reserve

Blaauwberg Nature Reserve study area (Fig. 1.8.) is located in the northern section of the city of Cape Town. It receives most of its rainfall (an average of 575 mm rainfall per annum) during cool wet winter months (Krupek et al., 2016). The primary native vegetation includes Restionaceae, Ericaceae, Proteaceae and Asteraceae overlying deep acidic soils (Rebelo et al., 2006). The Cape Flats Sands Fynbos vegetation type has largely been displaced by dense numerous stands of *Acacia saligna* (Krupek et al., 2016).

Figure 1.8: A 2014 image of the Blaauwberg Nature Reserve study area. The image indicates slash piles of *Acacia saligna* (white quadrilateral) and the site currently invaded by *A. saligna* (white arrow; Google Earth, 2016).

1.12. Thesis structure

This thesis contains three chapters with each containing its own reference list. The thesis presents findings on seasonal and spatially effects of burning of slash pile on selected soil nutrients and physicochemical properties in the Western Cape riparian and terrestrial fynbos ecosystems. Some repetition of methods and study area is unavoidable and occurs in data chapters 2 and 3, which are presented as stand-alone chapters for publication purposes.

- Chapter 1: This chapter covers the introduction and a comprehensive literature review of the use of burning of slash piles to manage excess IAPs biomass and how it impacts soils. It also provides the research motivation, aims, key questions and describes the study area.

- Chapter 2: This data chapter is based on experiments conducted at the four riparian study areas. The chapter provides findings of how slash pile burning of *Acacia mearnsii* and

Eucalyptus camaldulensis biomass has affected physicochemical properties of selected riparian sites within the Western Cape.

- Chapter 3: This data chapter is based on experiments conducted at the terrestrial study area. The chapter provides findings of how slash pile burning of *Acacia saligna* biomass has affected certain soil nutrients and physicochemical properties of selected riparian sites within the Western Cape.
- Chapter 4: This chapter will integrate and summarise main outcomes of each data chapter and suggest some implications for management of invasive alien plants in riparian and terrestrial sites.

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CHAPTER 2

Effects of burning of slash piles of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass on physicochemical properties of riparian soils

2.1. Introduction

The biodiversity of the Cape Floristic Region (CFR) is under continuous and expanding threat from establishment and spread of invasive alien plants (IAPs; Roura-Pascual et al., 2009). Despite major efforts by land management programmes such as Working for Water (WfW), IAPs continue to displace native vegetation and may colonise the remaining natural fynbos habitats. Two such IAPs are the native Australian species *Acacia mearnsii* and *Eucalyptus camaldulensis*. *Acacia mearnsii* is an evergreen tree that can grow up to 20 m high and often establishes in dense stands. The species is currently considered as one of the most aggressive plant invaders in the Western Cape South Africa (Cronk & Fuller, 1995) and is classified as a transformer species (Holmes et al., 2005). Within the Western Cape fynbos *A. mearnsii* is mostly, though not exclusively found along watercourses where it has adapted to fire-prone and nutrient-poor conditions, allowing it to continuously spread and replace the previously herbaceous and/or shrub type native vegetation (Dye et al., 2001; Roura-Pascual et al., 2009; van Wilgen et al., 2012). Along these watercourses, *A. mearnsii* forms numerous dense stands and has considerably increased above-ground biomass (Stock et al., 1995; Le Maitre et al., 2002). On average in South Africa, *A. mearnsii* may produce up to 10 times more above-ground plant biomass than native vegetation (Chamier et al., 2012).

Eucalyptus camaldulensis (Red River Gum) is of concern to land managers in South Africa including Mpumalanga and Western Cape (Forsyth et al., 2004). In the Western Cape, *E. camaldulensis* has largely displaced native riparian vegetation and poses a threat to the aquatic ecosystem (van Wilgen et al., 2007) and in some cases may have transformed habitats they invade (Forsyth et al., 2004). This species is able to grow fast and adapt to varying environmental settings such as nutrient-poor soils and fire-prone conditions of the fynbos biome (van Wilgen, 2009; Bush et al., 2013). Such traits and the tree’s ability to release allelochemicals, which hinder germination and growth of indigenous vegetation, has to some degree aided its survival along Western Cape watercourses (Chamier et al., 2012). These allelochemicals are released from *E. camaldulensis* plant tissues (e.g. leaf, bark) and are species specific in terms of how they inhibit germination and/or growth of native vegetation (Ruwanza et al., 2015). The species has an average stem diameter of approximately 50 cm when mature, with a maximum height exceeding

30 m, and along water courses in the Western Cape can reach high densities, with the result that they outcompete native species for light, space, nutrients and water (Power et al., 2010; Dzikiti et al., 2016).

The water resources of the Western Cape are under pressure from IAPs such as *A. mearnsii* and *E. camaldulensis*. These invasive species use large volumes of water from the moist soils, increasing rates of evapotranspiration (Le Maitre et al., 2002). This results in a decrease in both surface run-off and groundwater levels, and thus lowers the stream flow levels of the nearby rivers and desiccates riparian soils (Le Maitre et al., 2002; Moyo & Dube, 2010). It is not only riparian hydrology that is threatened by IAP invasion. Soil microbial activity may also become affected by increased organic matter inputs which may alter decomposition process and thus nutrient stock and cycling (Stock et al., 1995; Baldwin & Mitchell, 2000; Moyo & Dube, 2010).

It is essential to control and limit the spread of *A. mearnsii* and *Eucalyptus* spp. through the Western Cape riparian ecotones because their presence alters ecosystem processes and threatens ecosystem services (Pejchar & Mooney, 2009). This is especially important for countries that have limited water supply such as South Africa (Friedrich et al., 2009). As a measure of control, biological agents *Melanterius maculates* and *Dasineura rubiformis* are used to reduce seed production of *A. mearnsii* and thus limit seed addition to an already large soil stored seed bank (Impson et al., 2008). This, however, does not address the problem of above-ground production of *A. mearnsii*. This is also the case for *Eucalyptus* spp. which is also known to increase above-ground biomass production of invaded reaches (Chamier et al., 2012). Currently, in the Western Cape, *Eucalyptus* spp. biomass is managed by chemical and mechanical methods; this is because biological control has thus far not been effective in managing this IAP (Moran & Hoffmann, 2012). However, due to the large size of the tree, it might be challenging to manage by mechanical methods, and in cases where management has been successful, another major challenge is removal or management of the resultant biomass (Tererai et al., 2013; Dzikiti et al., 2016).

One commonly used mechanical biomass control method is burning of slash piles (Cilliers et al., 2004). This control method involves cutting of standing IAPs, stacking the excess biomass, allowing the wood to dry and then setting the piles on fire (Holmes et al., 2008). The burning of slash piles may produce very high temperatures which have negative impacts on components of the ecosystem, impacting on the recovery of the ecosystem following clearing. Soil-stored seed banks of native fynbos vegetation may be destroyed to various soil depths, depending on the fuel load, which has an influence on the fire intensity (Cilliers et al., 2004; Certini, 2005). Damage to native seeds may offer an opportunity for secondary invasion which is not favourable, especially since often the main objective of clearing is to re-establish indigenous plants and restore the fynbos ecosystems (Halpern et al., 2014). Fire has the ability to alter certain soil properties, which may result in burn scars on severely affected soils such as those exposed to burning of slash piles (Korb et al., 2004). Soil chemical properties have been shown to be significantly altered by high fire temperatures, which combust soil organic matter on surface soils, and pyromineralize nutrients such as nitrogen and phosphorus (Neary et al., 1999). The high fire-induced temperatures often transform certain soil nutrients, which as a result may be lost from the soil. For instance, when exposed to heat, soil nutrients such as phosphorus might be transformed from stable forms into non-stable forms which are leachable by water (Galang et al., 2010); nitrogen may be transformed and released from soil mineral surface into gaseous forms (Neary et al., 1999). Exchangeable cations volatilises at high temperatures ranging from 760ºC for potassium and 1240ºC for calcium (Neary et al., 1999). On the other hand, during burning of plant biomass, nutrients that were plant bound are released and incorporated in ash, which after fire settles on the soil surface (Certini, 2005). By settling on the soil surface, this ash enriches the top layer of soil with ammonium, nitrate, exchangeable cations and available P (Certini, 2005; Schafer & Mack, 2010).

Burning of slash pile may also increase soil pH due to exposure to heat and addition of base cations from ash (Certini, 2005), elevate soil electrical conductivity (EC; Hernández et al., 1997), and induce hydrophobicity, especially on coarse-textured soils (Doerr at al., 2000). The altered soil physicochemical properties eventually return to their pre-fire conditions at rates that vary with soil type, climate and the characteristic of each parameter (Certini, 2005). Some ash enriched nutrients might be leached from the fire affected surface to different parts of the profile or be washed away by surface run-off (Tomkins et al., 1991; Forgeard & Frenot, 1996). Soil EC usually returns to pre-fire level once cations have been leached; soil pH on the other hand may persist prolonged periods (Bodi et al., 2014).

Most of the research conducted on invasion within Western Cape riparian ecotones and terrestrial areas has been driven by the assumption that if the invader has been removed, then the native riparian vegetation and ecosystem services would recover (Ruwanza et al., 2013a). Little information is available on the impacts of clearing methods on ecosystem properties, more specifically on how burning of slash piles may affect soil chemical and physical properties, and what role that may play in recovery or a delay in recovery of native vegetation (Chamier et al., 2012; Jacobs et al., 2013). This chapter aims to evaluate the medium-term (approximately one year) impacts of burning of slash pile of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass on Western Cape riparian soils properties. Three key questions are addressed, firstly, how will burning of slash piles of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass affect soil physicochemical properties over the course of one year? Secondly, what are the spatial impacts of burning of slash piles of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass on physicochemical properties? Thirdly, what is the trajectory followed by physicochemical properties subsequent to burning of biomass of *Acacia* and *Eucalyptus* spp.?

2.2. Material and methods

2.2.1. Study area

The study was conducted at four areas viz. along the Wit River (termed Wit River; 33°32'36.06"S, 19°10'17.90"E; dominant invasive species *A. mearnsii*), along the Breede River near Rawsonville (Rawsonville; 33°43'43.43"S, 19°28'25.63"E; *A. mearnsii*), along the Breede River near Robertson (Robertson; 33°50'17.08"S, 19°52'28.33"E; *E. camaldulensis*) and along the Berg River near Hermon (Hermon; 33°28'42.83"S, 18°56'13.83"E; *E. camaldulensis*). All these study areas occur within the Western Cape and experience Mediterranean type climate of cool wet winters and warm dry summers (Rebelo et al., 2006).

At the Wit River study area (Fig. 2.1), the lowest average temperature in winter is about 4°C and the highest average temperatures at approximately 25°C in summer; it has an average rainfall of 1200 mm per year (Rebelo et al., 2006). The river flows for about 11 km along the R301 (Bain's Kloof Pass) at the bottom of steep mountains and eventually joins the Breede River (Brown et al., 2004; Rebelo et al., 2006). The river's banks consist of coarse textured acidic soils which in their pristine are covered by Hawequas Sandstone Fynbos vegetation type.

Figure 2.1: The Wit River study area. The map shows the burn site where burning of slash piles was conducted, invaded site which is currently invaded by *A. mearnsii* and the recovering site which was previously cleared and is currently recovering (Google Earth, 2016).

The Breede River is the largest river in the Western Cape, and covers a catchment area of about 12 600 km^2, it empties into the Indian Ocean (Brown et al., 2004; Steynor et al., 2009). Of the 12 600 km^2 catchment area, 84 398 hectares is invaded by IAPs, mainly by dense stands of *Acacia mearnsii*. The Rawsonville study area (Fig. 2.2) receives an average annual rainfall of about 480 mm; temperatures drop to an average of about 5^{o}C in winter and peak in summer at an average of around 30^{o}C (Rebelo et al., 2006). The soils are mainly derived from the Malmesbury Group and Bokkeveld Group shales and are mixed with alluvial pebbles of variable sizes which are overlain by the Breede Alluvium Fynbos vegetation type (Rebelo et al., 2006).

Figure 2.2: The Rawsonville study area. The map shows the burn site where burning of slash piles took place, invaded site which is currently invaded by *A. mearnsii*, and the recovering site which represent local native vegetation (Google Earth, 2016).

The Robertson study area along the Breede River (Fig. 2.3) occurs at an altitude of between 200-350 m, is coldest in winter with an average of 5^{o}C and warmest in summer with an average of 30^{o}C; receives an average of 345 mm of rainfall annually (Rebelo et al., 2006). This study area consists mainly of Aeolian sands mixed with small pebbles and which are primarily overlain by Breede Sands Fynbos vegetation type.

Figure 2.3: The Robertson study area. The map shows locations of the burn site where burning of slash piles was conducted, the invaded site which is currently invaded by *E. camaldulensis*, and the recovering site which represents the local native vegetation (Google Earth, 2016).

The Hermon study area (Fig. 2.4), along the Berg River, has an altitude of about 66 m. The Berg River is a perennial river that flows over a distance of approximately 294 km and joins into the Atlantic Ocean at Velddrif (Ruwanza et al., 2013b; Tererai et al., 2013). The river has a catchment area of approximately 7715 km^2, and the study area receives an average of about 550 mm rainfall and has winter temperatures of 11^{o}C and summer temperatures of 22^{o}C (Tererai et al., 2013). The river's banks mainly consist of low nutrient acidic soils derived from sandstones and quartzite of the Cape Supergroup (Rebelo et al., 2006; de Villiers, 2007).

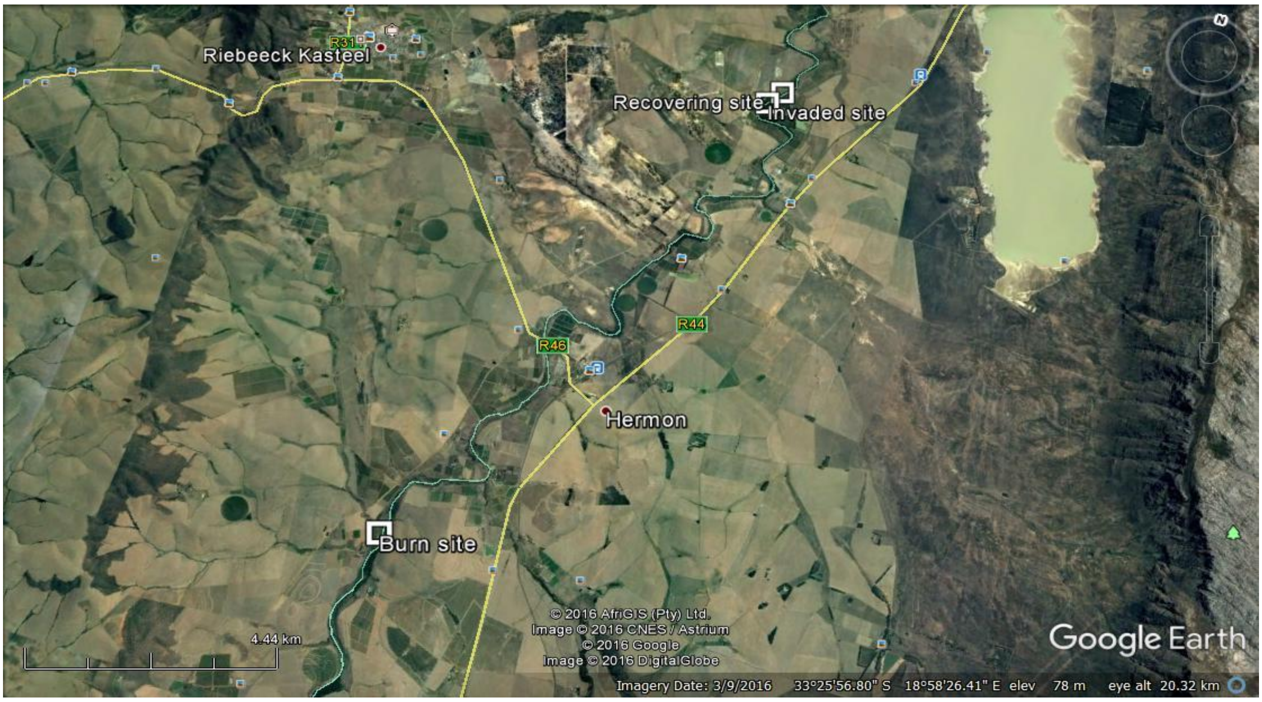

Figure 2.4: The Hermon study area. The map shows the burn site where burning of slash piles was conducted, invaded site which is currently invaded by *E. camaldulensis*, and the recovering site which was previously cleared of *E. camaldulensis* and is currently recovering (Google Earth, 2016).

2.2.2. Experimental design

Each study area is divided into three experimental sites, viz. burn site where burning of slash pile (Table 2.1) of *A. mearnsii and E. camaldulensis* was carried out, the invaded reference site which is currently dominated by either *A. mearnsii* (in the case of the *Acacia* study areas) or *E. camaldulensis*, and the recovering site which represents the local native vegetation or has been cleared of *A. mearnsii* or *E. camaldulensis* and is currently recovering. Within the Wit River, the burn site and the invaded site were located on the property Bastiaanskloof, which is a private land where clearing is conducted by the land owner. The recovering site is located at the Tweede Tol Campsite which has been previously cleared of *A. mearnsii*.

Table 2.1: Description of slash piles in terms of the primary biomass, pile size, texture and burn season volume

Study area	Study species	Average pile size (m3)	Soil texture (tested)	Burn season
Hermon	*Eucalyptus camaldulensis*	116.68	Loamy	Spring 2014
Wit River	*Acacia mearnsii*	21.01	Loamy sand	Winter 2015
Rawsonville	*Acacia mearnsii*	88.17	Sandy loam	Winter 2015
Robertson	*Eucalyptus camaldulensis*	93.93	Sandy clay loam	Winter 2015

Volume = (length x width x height x π) ÷ 6 = volume m^3 (Busse et al., 2013).

The Rawsonville study area is also privately owned (Riverside Farm) and the experimental sites were located within the farm. Clearing and stacking biomass of *A. mearnsii* was conducted by a local contractor. An invaded reference site was selected near the site where clearing operations took place, while a recovering reference site was selected on the section of the river that is more representative of the local native vegetation. The experimental sites along at the Berg River study area were located on two privately owned farms. The burn site was located on one farm, and the two reference sites (invaded and recovering) were located on a farm downstream along the Berg River, where the recovering site was selected on the section of the river that was previously cleared of *E.camaldulensis*. Clearing and stacking of biomass of *E. camaldulensis* was conducted by a local contractor.

The experimental sites at the Robertson study area were located on government property, where clearing and stacking was conducted by a local contractor. All the experimental sites viz. burn site, invaded site and recovering site were located within close proximity along the Breede River. As in the case of the Rawsonville study area, the recovering site was selected within a river section that is representative of local native vegetation – there were no cleared and recovering sites at this study area.

2.2.3. Sampling and laboratory methods

Prior to burning, soil samples within the burn site were collected from either already existing piles in the case of Hermon and Robertson study areas or from newly formed piles in the case of the Rawsonville and Wit River study areas. In the case of existing slash piles, this was done carefully by removing the overlying slash and sampling the soil to 10 cm. In the case of new slash piles, built from scratch, the soils were sampled to 10 cm before the piles were assembled. After burning, soil samples were collected from experimental sites immediately post-fire, and once in the subsequent three seasons. On each sampling occasion, a total of 52 soil samples were collected to a depth of 10 cm using a metal tube. Three samples from each of 8 piles (viz. the centre, position intermediate, i.e. between the edge and centre, and the edge of the burn scar (Fig. 2.5)), one sample from the soil matrix (about 2 m from the edge of the burn scar), ten soil samples from the recovering site, and ten samples from the invaded site.

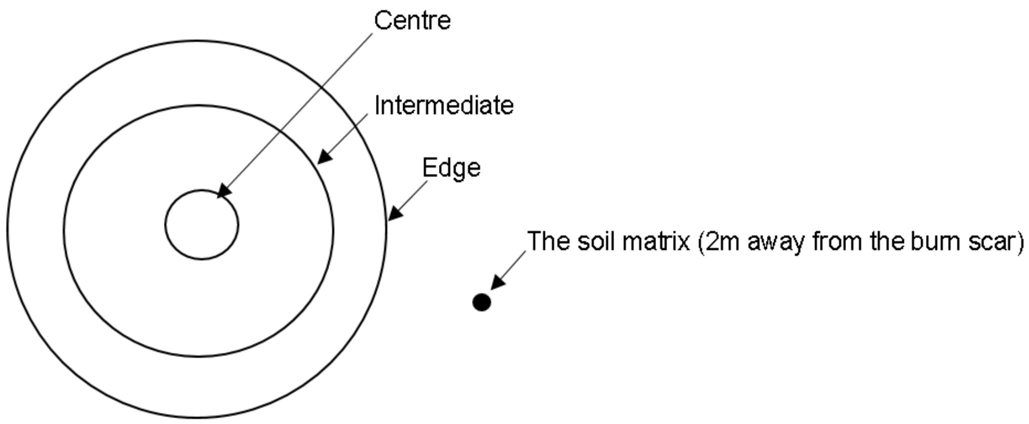

Figure 2.5: Depiction of the sampling positions within the burn site.

Field-collected soil samples were sealed in plastic bags and transported to the laboratory. On arrival, the soil samples were sieved through a 2 mm sieve to remove rocks, pebbles and gravel (Mataix-Solera et al., 2013). Hereafter, the gravimetric soil water content was determined by oven drying samples at 105^{o}C for 48 h (Shafer & Mack, 2010). The soil samples were treated with hydrogen peroxide to remove excess organic matter, then soil texture was determined using the hydrometer method, by dispersing 40-50 g soil with 100 ml sodium hexametaphosphate (HMP) solution in an electric mixer for 5 minutes and measuring buoyancy using an ASTM 152H hydrometer (Gee & Bauder, 1986). Soil hydrophobicity was estimated in the laboratory using the Water Drop Penetration Test (WDPT) on air-dried soil samples using distilled water (Dekker et al., 2001). Soil pH was tested by mixing 10 g of soil and 20 ml distilled water for 30 minutes (1:2) and analysed with a calibrated Hanna pH meter HI 8424 (Robertson et al., 1999). Electrical conductivity of the soil was measured using a 1:2 mixture of soil and distilled water and a calibrated Hanna HI 8733 conductivity metre (Hanna HI 8733) (Miller & Curtin, 2007).

Available nitrogen (NH_4^+-N and Nitrate NO_3^--N) was extracted using a 0.5 M K_2SO_4 solution, then later analysed using Genesys 20 spectrophotometer. The salicylic acid method was used to measure NO_3^--N concentrations and the indophenol blue method for NH_4^+-N concentrations; procedures are as described in Anderson & Ingram (1993). Total soil C and N were measured on a dry combustion CN analyser (Euro EA Analyser) (Sollins et al., 1999). Exchangeable cations were extracted with 1 M NH_4OAc (Simard, 1993) and analysed with a Varian 240 FS atomic absorption spectrometer. Only the Hermon and Rawsonville study areas soils were analysed for total C and N and exchangeable cations. Available P was extracted using the P-Bray 2 solution similar to that of Bray and Kurtz (1945), and the concentration was determined with a spectrophotometer (Genesys 20) based on the molybdenum blue method (Olsen & Sommer, 1982).

2.2.4. Data analyses

The data were analysed using the Statistica version 13 software package (Dell Inc., 2015). All parameters were checked for normality using the Kolmogorov-Smirnov test. Two-way analyses of variance (ANOVA) were used to determine the burn treatment effects seasonally viz. pre-fire, post-fire, summer, autumn and winter. One-way ANOVAs were used determine the burn treatment effects by location viz. the centre, intermediate, edge, matrix, and reference sites. Where the computed ANOVAs showed significance ($p<0.05$), the Bonferroni post hoc tests were performed.

2.3. Results

The results reported in this chapter are for temporal/seasonal and spatial effects of burning of slash piles on soil physicochemical properties. The results for all parameters are based on data from Hermon, Robertson, Rawsonville and Wit River, with the exception of total C and N and exchangeable cations which are based only on data from the Hermon and Rawsonville study areas. Temporal results are reported in comparison to pre-fire properties and the non-burn treatment sites; it must be noted that pre-fire and post-fire fall within the same climatic season. The spatial data is based on the centre, intermediate and edge sampling positions of the burn scar; the recovering reference site; and the invaded reference site. These spatial results are reported as a single mean of all post-fire seasons for each of the sampling positions or location viz. the centre, intermediate and edge position of the burn scar, soil matrix, invaded reference sites and recovering reference sites.

2.3.1. Soil pH and EC

Soil pH increased significantly after burning at all the study areas i.e. Hermon ($p<0.01$; Fig. 2.6a), Robertson ($p<0.01$; Fig. 2.6b), Rawsonville ($p<0.01$; Fig. 2.6c) and Wit River ($p<0.01$; Fig. 2.6d). At the end of the sampling period (approximately 1 year after burning), burn scar soil pH values were still significantly higher than both pre-fire and non-burn sites on all study areas, with the exception of Wit River which had returned to pre-fire levels in summer (about 4 months after burning).

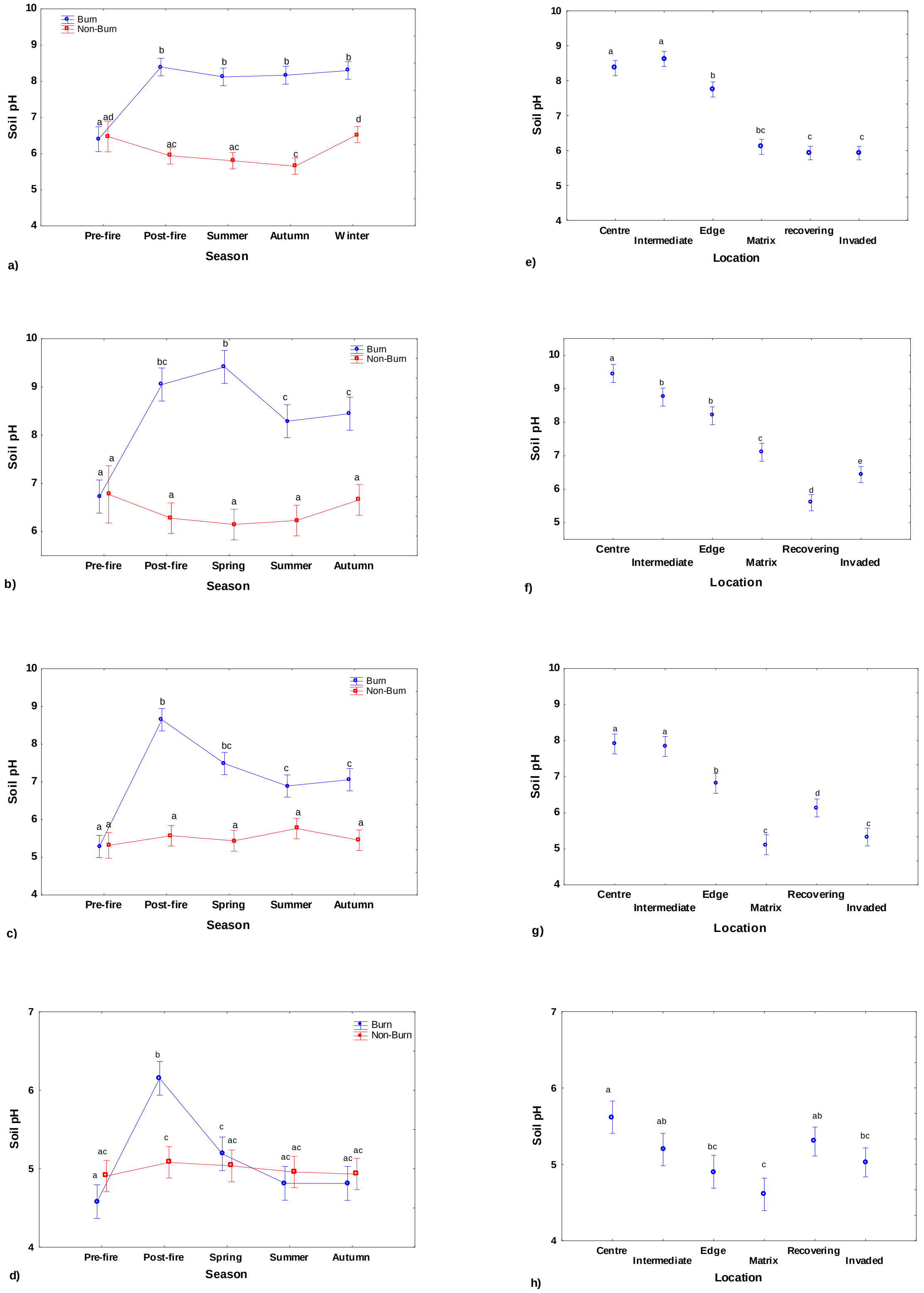

Figure 2.6: Seasonal soil pH values within the burn and the non-burn treatment sites at the (a) Hermon (b) Robertson (c) Rawsonville, and (d) Wit River study areas; spatial soil pH values within the burn scar and non-burn sampling locations at (e) Hermon, (f) Robertson, (g) Rawsonville and (h) Wit River study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil pH means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.6**a** ($F_{[4,\ 217]}=19.41$, $p<0.01$), Fig. 2.6**b** ($F_{[4,\ 230]}=17.75$, $p<0.01$), Fig. 2.6**c** ($F_{[4,\ 240]}=28.21$, $p<0.01$) and Fig. 2.6**d** ($F_{[4,\ 249]}=13.85$, $p<0.01$), and Spatial pH means are based on a one-way ANOVA: location Fig. 2.6**e** ($F_{[5,\ 201]}=143.31$, $p<0.01$), Fig. 2.6**f** ($F_{[5,\ 202]}=129.03$, $p<0.01$), Fig. 2.6**g** ($F_{[5,\ 202]}=77.98$, $p<0.01$), Fig. 2.6**h** ($F_{[5,\ 201]}=10.54$, $p<0.01$).

Spatially, sampling positions within the burn scars (particularly the centre and the intermediate position) had significantly higher soil pH values than the non-burn site i.e. the soil matrix, the recovering and the invaded reference sites. This spatial trend can be seen at three of the study areas viz. Hermon (Fig. 2.6e), Robertson (Fig. 2.6f) and Rawsonville (Fig. 2.6g), with the exception of Wit River, where there were similarities between the burn scar sampling positions and the non-burn sites including the recovering and invaded reference sites (Fig. 2.6h).

At three of the four study areas, soil EC within the burn scars increased significantly after fire compared to the non-burn sites (Fig. 2.7). At Hermon, soil EC increased significantly after fire ($p<0.01$) and returned to pre-fire levels by the end of sampling (about a year after burning) (Fig. 2.7a). At Robertson, burn scar soil EC increased significantly ($p<0.01$) after burning and was still significantly higher ($p=0.02$) than pre-fire in about a year after burning (Fig. 2.7b). Burn scars soil EC at Rawsonville also increased significantly ($p<0.01$) as a result of fire, but had returned to pre-fire levels in spring (approximately 3 months after burning) (Fig. 2.7c). On the contrary, the fourth study area (i.e. Wit River) did not experience changes in soil EC within the burn scar after burning and remained unchanged in spring and summer. However, a spike in EC occurred in both the burn and non-burn sample areas in autumn (almost a year after burning) when soil EC values became significantly higher than pre-fire with both the burn ($p=0.01$) and the non-burn sites ($p<0.01$) (Fig. 2.7d).

Spatially, soil EC values for Hermon, Robertson and Rawsonville followed a similar trend, where the centre and the intermediate sampling positions were significantly higher than the soil matrix, recovering reference site and the invaded reference site (Fig. 2.7). At Hermon the intermediate and centre had similar soil EC values, which were significantly higher than the edge ($p<0.01$), the soil matrix ($p<0.01$), the recovering ($p<0.01$) and invaded reference ($p<0.01$) sites (Fig. 2.7e). At Robertson, the centre soil EC values were significantly higher than the intermediate ($p<0.01$), the edge ($p<0.01$), the soil matrix ($p<0.01$), the recovering reference site ($p<0.01$) and the invaded reference site ($p<0.01$) (Fig. 2.7f). Similar to Robertson, at Rawsonville, soil EC values in the centre were significantly higher than all the sampling positions (Fig. 2.7g). In contrast, the Wit River site did not show spatial significant differences between the burn and non-burn treatment sites (Fig. 2.7h).

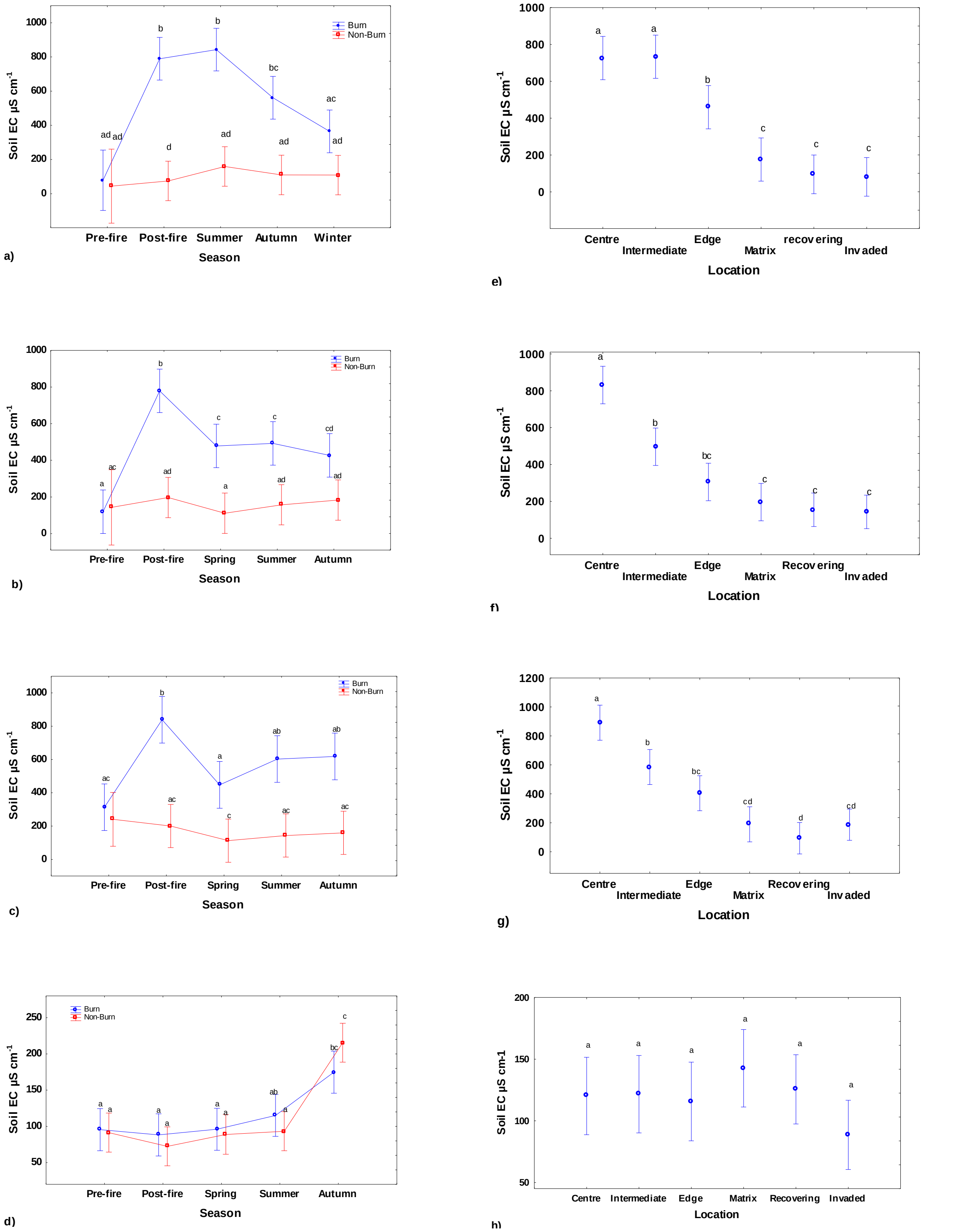

Figure 2.7: Seasonal soil EC values within the burn and the non-burn treatment sites at (a) Hermon, (b) Robertson, (c) Rawsonville and (d) Wit River study areas; and spatial soil EC values within the burn scar and non-burn sampling locations at (e) Hermon, (f) Robertson, (g) Rawsonville and (h) Wit River. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil EC means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.7**a** ($F_{[4, 218]}=7.42$, $p<0.01$), Fig. 2.7**b** ($F_{[4, 230]}=4.83$, $p<0.01$), Fig. 2.7**c** ($F_{[4, 240]}=4.07$, $p<0.01$) and Fig. 2.7**d** ($F_{[4, 249]}=1.50$, $p=0.20$). Spatial soil EC means are based on a one-way ANOVA: location Fig. 2.7**e** ($F_{[5, 202]}=28.45$, $p<0.01$), Fig. 2.7**f** ($F_{[5, 202]}=28.52$, $p<0.01$), Fig. 2.7**g**($F_{[5, 202]}=25.92$, $p<0.01$) and Fig. 2.7**h** ($F_{[5, 201]}=1.40$, $p=0.22$).

2.3.2. Carbon (C), nitrogen (N) and available P (P)

At Hermon, within the burn scar total C did not show any seasonal significant differences as a result of burning. Similarly within the non-burn treatment sites total C also did not change seasonally (Fig. 2.8a). On the contrary, at Rawsonville, total C significantly decreased ($p<0.01$) within the burn scar after burning and was still considerably lower ($p<0.01$) than pre-fire levels in autumns (almost a year after burning); and had remained unchanged in the non-burn site (Fig. 2.8b).

There were no spatial differences in total C at Hermon (Fig. 2.8c). At Rawsonvlille, the total C levels on the recovering site were similar to those of the centre, and were significantly lower than the edge ($p=0.02$), the matrix ($p=0.01$) and invaded reference site ($p=0.02$) (Fig. 2.8d).

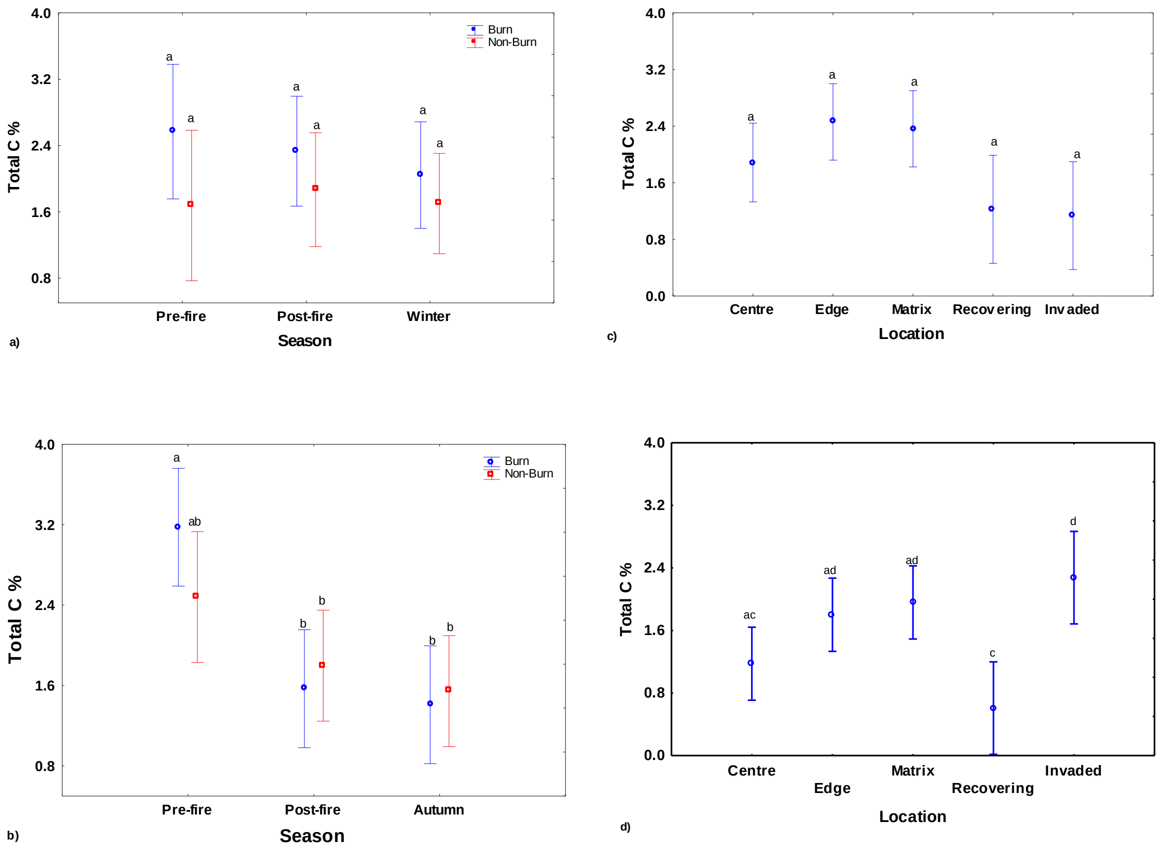

Figure 2.8: Seasonal soil total C levels within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial soil total C within the burn scar and non-burn sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil total C means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.8a ($F_{[2, 75]}=0.27$, $p=0.76$), Fig. 2.8b ($F_{[2, 91]}=1.41$, $p=0.25$). Spatial total C means are based on a one-way ANOVA: location Fig. 2.8c ($F_{[4, 58]}=3.53$, $p=0.01$), Fig. 2.8d ($F_{[4, 63]}=5.70$, $p<0.01$).

Soil total N at Hermon was not affected by the fire within the burn scar and it also did not change seasonally within the non-burn sites (Fig. 2.9a). In contrast, at Rawsonville, total N within the burn scar decreased significantly ($p<0.01$) after fire and was still lower ($p<0.01$) than pre-fire at end of sampling period - approximately a year after fire. No significant changes in total N were observed in the non-burn treatment sites, as they remained statistically similar through the sampling seasons (Fig. 2.9b).

Spatially, at Hermon, total N was similar within the burn scar sampling positions and the non-burn sites (Fig. 2.9c). On the other hand, at Rawsonville, total N levels in the recovering reference site were similar to those of centre, and significantly lower than those of the edge ($p=0.01$), the soil matrix ($p=0.01$) and the invaded reference site ($p<0.01$) (Fig. 2.9d).

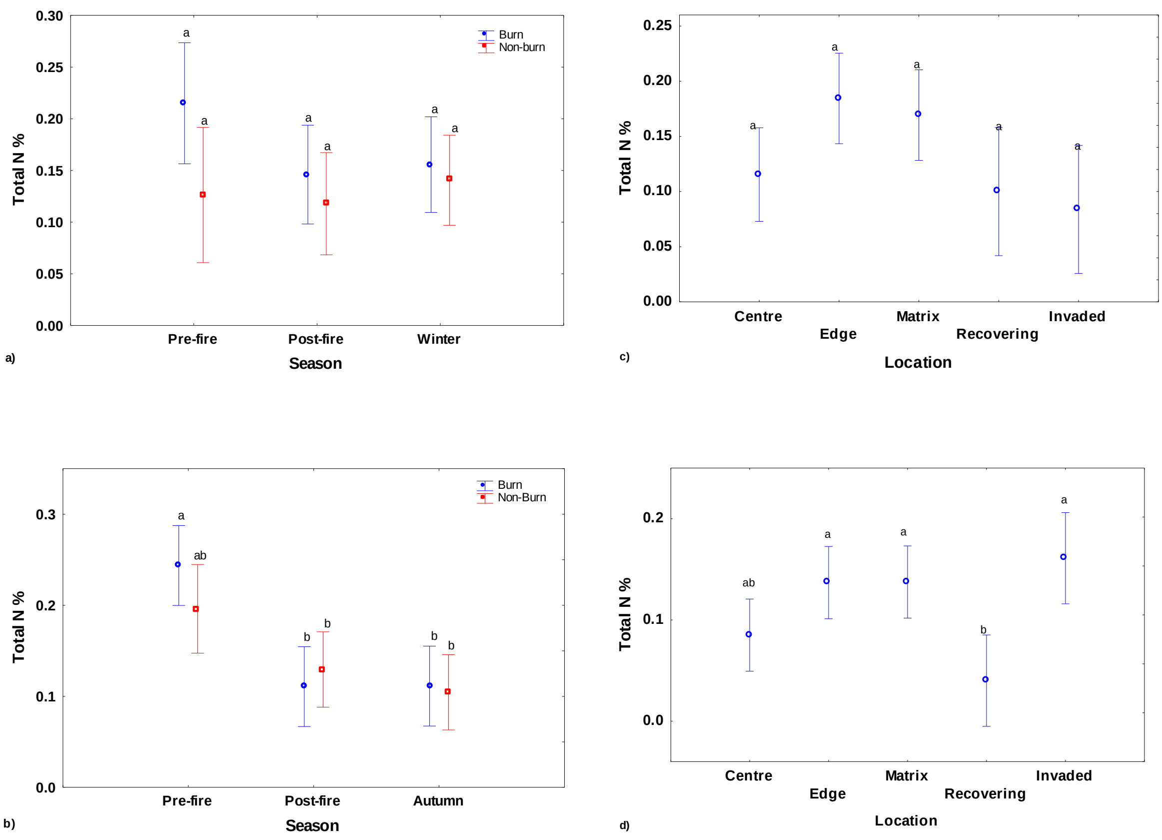

Figure 2.9: Seasonal soil total N levels within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial soil total N levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil total N means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.9**a** ($F_{[2, 75]}=0.96$, $p=0.39$), Fig. 2.9**b** ($F_{[2, 91]}=1.11$, $p=0.34$). Spatial total N means are based on a one-way ANOVA: location Fig. 2.9**c** ($F_{[4, 58]}=3.44$, $p=0.01$), Fig. 2.9**d** ($F_{[4, 63]}=5.27$, $p<0.01$).

Across all the study areas, total available N did not change immediately after fire, within both the burn and non-burn treatment sites. However, in the subsequent seasons, concentrations of available N followed varying trends per study area. At Hermon the available N levels remained relatedly unchanged throughout the sampling period (Fig. 2.10a). At Robertson, in spring (about three months after burning), there were significant drops in available N levels within both the burn scar ($p<0.01$) and the non-treatment sites ($p=0.01$) (Fig. 2.10b). At Rawsonville, six months after burning (summer), available N levels within the burn scar became significantly higher ($p<0.01$) than the non-burn sites, and continued on this pattern toward the end of sampling (Fig. 2.10c). Similarly, at Wit River available N concentrations in the last season of sampling were significantly higher ($p<0.01$) in the burn scar than the than non-burn sites (Fig. 2.10d).

Spatially, the results show that burning had varying effects on available N and there was no specific spatial trend followed by available N levels. At the Hermon, centre, intermediate and edge positions had similar available N levels, which were significantly higher than the recovering ($p<0.01$) and invaded ($p<0.01$) reference sites (Fig. 2.10e). At Robertson, the edge had the highest available N concentrations, which were similar to all the other sampling locations except for the invaded reference site, as it had significantly lower ($p<0.01$) than the edge (Fig. 2.10f). At Rawsonville, available N concentrations in the centre of the burn scar were similar to the intermediate, the edge, and significantly higher than the soil matrix ($p=0.01$), the recovering ($p<0.01$) and invaded ($p=0.01$) reference sites (Fig. 2.10g). At the Wit River, the edge had similar concentrations to the centre, intermediate, and soil matrix, which were significantly higher than the recovering ($p<0.01$) and invaded ($p<0.01$) reference sites (Fig. 2.10h).

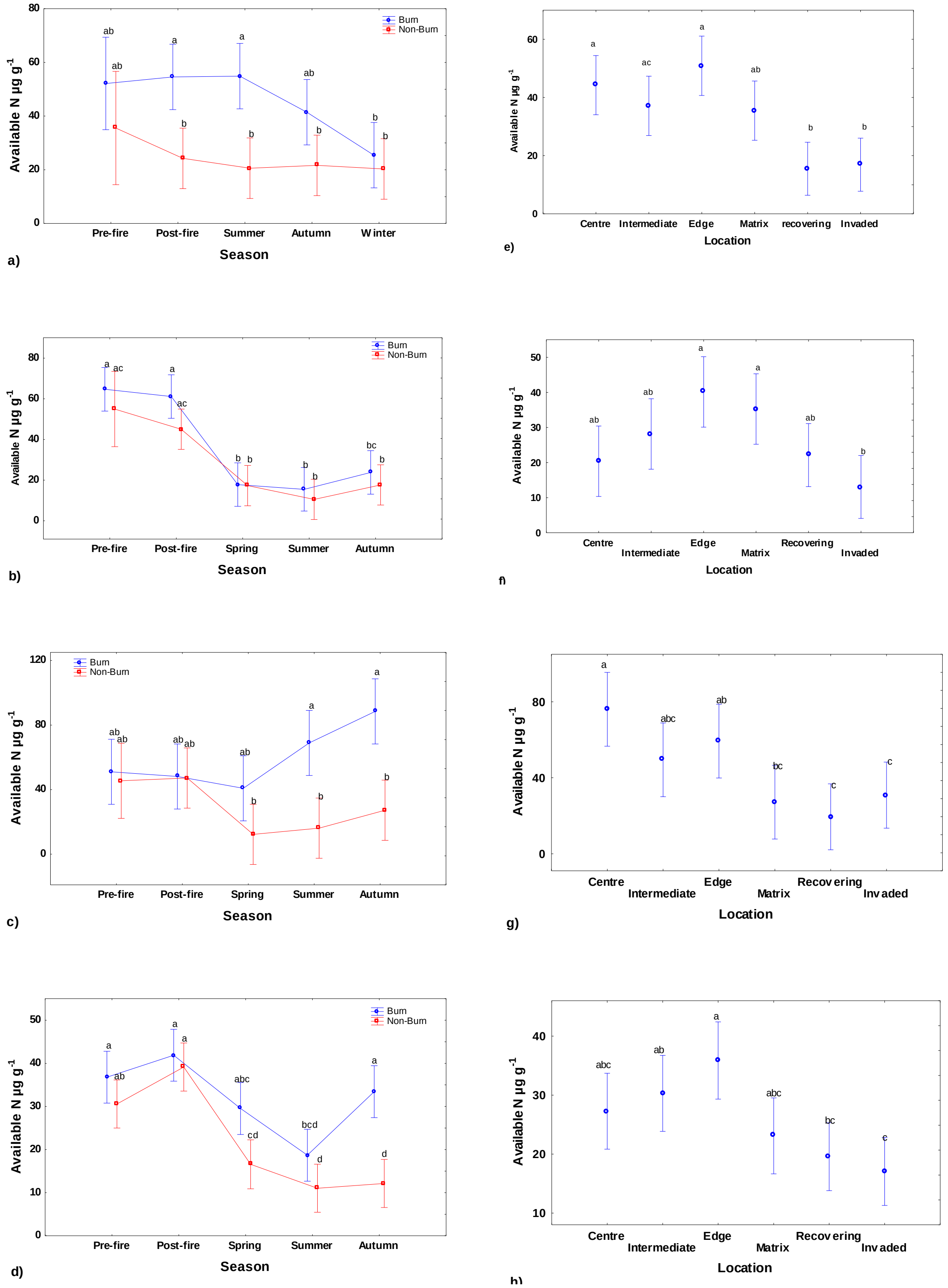

Figure 2.10: Seasonal soil available N levels within the burn and the non-burn treatment sites at (a) Hermon, (b) Robertson, (c) Rawsonville and (d) Wit River study areas; and spatial soil available N concentrations within the burn scar and non-burn sampling locations at (e) Hermon, (f) Robertson, (g) Rawsonville and (h) Wit River study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal available N means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.10**a** ($F_{[4,\ 218]}=1.84$, $p=0.12$), Fig. 2.10**b** ($F_{[4,\ 230]}=0.61$, $p=0.66$), Fig. 2.10**c** ($F_{[4,\ 240]}=3.62$, $p=0.01$) and Fig. 2.10**d** ($F_{[4,\ 249]}=2.99$, $p=0.02$). Spatial available N means are based on a one-way ANOVA: location Fig. 2.10**e** ($F_{[5,\ 202]}=8.77$, $p<0.01$), Fig. 2.10**f** ($F_{[5,\ 202]}=4.25$, $p<0.01$), Fig. 2.10**g** ($F_{[5,\ 202]}=5.21$, $p<0.01$) and Fig. 2.10**h** ($F_{[5,\ 201]}=5.02$, $p<0.01$).

At all study areas, available P concentrations increase as a result of burning. At Hermon, available P concentrations within the burn scar increased significantly ($p<0.01$) after fire, continued to increase ($p<0.01$) to reach its peak in summer, and had returned to pre-fire levels at the end of sampling period - approximately a year after burning (Fig. 2.11a). At Robertson, available P concentrations did not change significantly in post-fire season, a significant increase only came about in spring ($p<0.01$) (three months post-fire), and had returned to pre-fire levels in summer. On the other hand, the non-burn site available P levels did not show any significant changes seasonally (Fig. 2.11b). At Rawsonville, available P concentrations within the burn scar significantly increased ($p<0.01$) after fire and then returned to levels similar to those of pre-fire in spring. On the contrary, the non-burn sites followed a trend opposite to that of the burn scar, where available P decreased significantly ($p<0.01$) after fire, and was still lower at the end of sampling (Fig. 2.11c). At Wit River, available P concentrations within the burn scar increased significantly ($p<0.01$) after fire and had not returned to pre-fire levels at the end of sampling period (about a year after burning). Available P concentrations within the non-burn sites also increased significantly ($p=0.02$) in post-fire season, remained at this level in spring and returned to pre-fire levels in summer (Fig. 2.11d).

At all the study areas, available P concentration showed a spatial trend of being relatively high within the burn scar, and being low on the non-burn treatment site. At Hermon, available P concentrations on the centre were similar to the intermediate and edge, and significantly high than the matrix soil ($p<0.01$), the recovering ($p<0.01$) and invaded ($p<0.01$) reference sites (Fig. 2.11e). At the Robertson study area, available P concentrations on the centre were significantly higher than the intermediate ($p<0.01$), edge ($p<0.01$), matrix ($p<0.01$) recovering reference site ($p<0.01$) and on the invaded reference site ($p<0.01$) (Fig. 2.11f). At Rawsonville, available P levels were relatively high within the burn scar sampling position, with the centre being similar to the intermediate, but significantly higher than the edge ($p=0.03$), soil matrix ($p<0.01$), recovering reference site ($p<0.01$) and the invaded reference site ($p<0.01$) (Fig. 2.11g). The Wit River, the centre available P levels were similar to those of the intermediate and edge, but significantly higher than the matrix soil ($p= 0.01$), the recovering reference site ($p<0.01$) and the invaded reference site ($p<0.01$) (Fig. 2.11h).

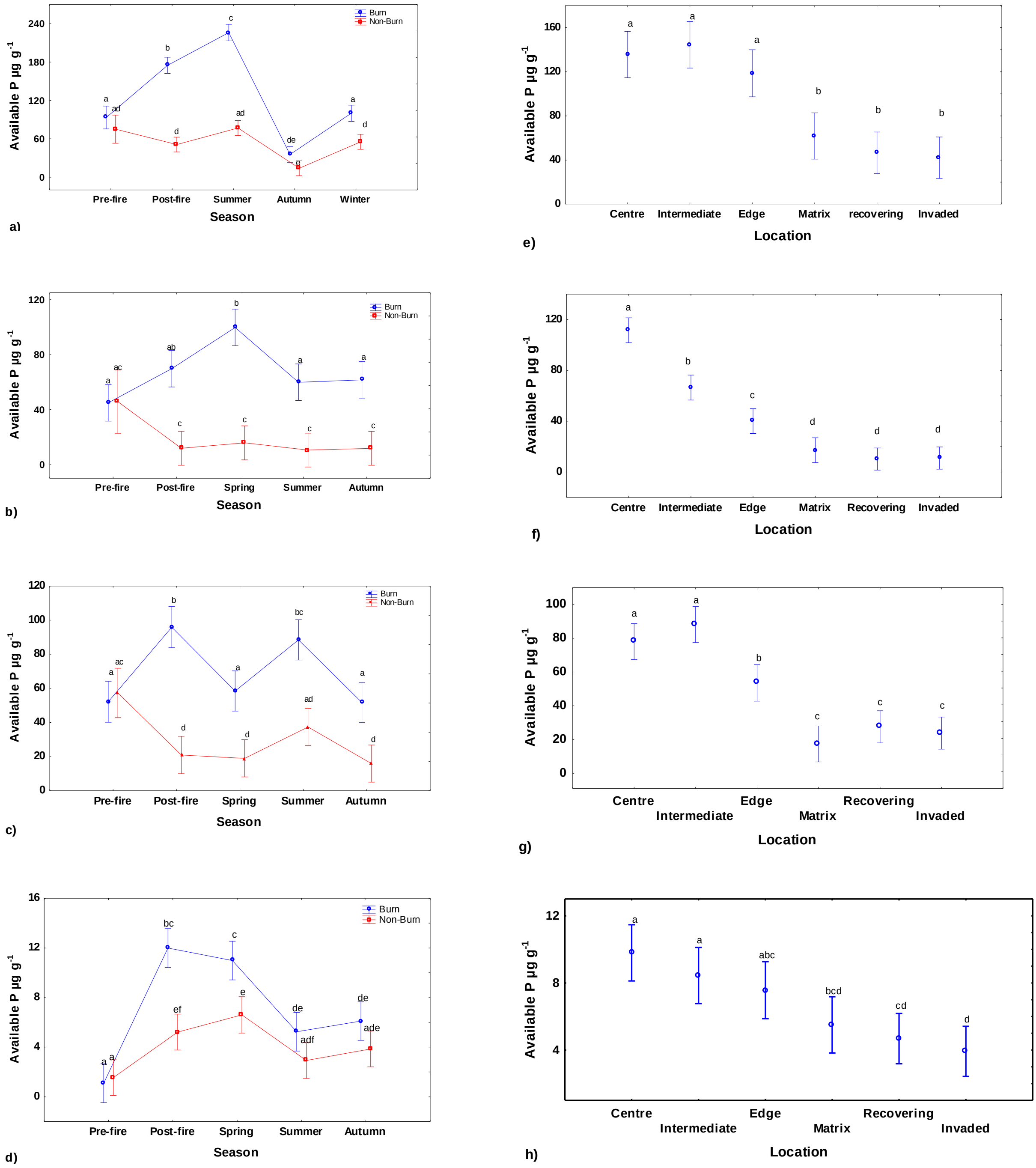

Figure 2.11: Seasonal available P levels within the burn and the non-burn treatment sites at (a) Hermon, (b) Robertson, (c) Rawsonville and (d) Wit River study areas; and spatial available P concentrations within the burn scar and non-burn sampling locations at (e) Hermon, (f) Robertson, (g) Rawsonville and (h) Wit River). Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal available P means are interaction effects based on two-way ANOVA: treatment X season Fig. 2.11a ($F_{[4, 217]}=41.57$, $p<0.01$), Fig. 2.11b ($F_{[4, 230]}=6.93$, $p<0.01$), Fig. 2.11c ($F_{[4, 236]}=10.56$, $p<0.01$) and Fig. 2.11d ($F_{[4, 249]}=6.24$, $p<0.01$). Spatial available P means are based on a one-way ANOVA: location Fig. 2.11e ($F_{[5, 201]}=20.82$, $p<0.01$), Fig. 2.11f ($F_{[5, 202]}=69.28$, $p<0.01$), Fig. 2.11g ($F_{[5, 201]}=32.35$, $p<0.01$) and Fig. 2.11h ($F_{[5, 201]}=8.10$, $p<0.01$).

2.3.3. Exchangeable cations

Temporally, exchangeable cations (i.e. Ca, Mg, Na, and K) within the burn scar increased after fire at both Hermon and Rawsonville, while remaining unchanged on the non-burn site. At Hermon cations within the burn scar significantly increased (p<0.01) after burning and had returned to pre-fire levels approximately a year after burning, while remaining unchanged in the non-burn sites (Fig. 2.12a). Similarly, At Rawsonville burn scar cations concentrations also increased significantly (p<0.01) after fire and eventually returned to pre-fire levels after approximately a year (Fig. 2.12b).

Spatially, at Hermon, cations concentrations were similar on the centre and the edge of the burn scar, and significantly higher than the soil matrix (p=0.04), the recovering reference (p=0.04) site and the invaded reference site (p=0.03) (Fig. 2.12c). On the other hand, at Rawsonville, the centre and edge of the burn scar and the invaded reference sites had similar cations concentrations, which were significantly higher than the soil matrix (p=0.01) and the recovering reference site (p<0.01) (Fig. 2.12d).

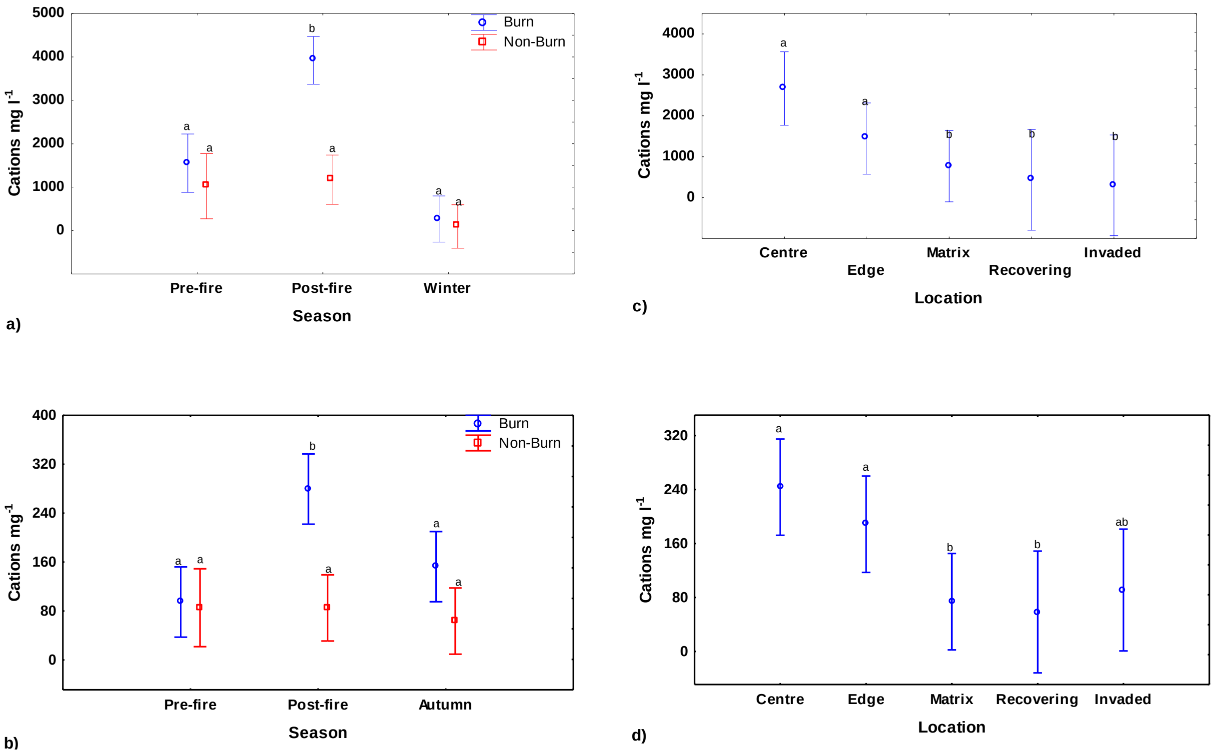

Figure 2.12: Seasonal cations concentrations within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial cations levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, p<0.05). All the seasonal soil calcium concentrations mans are interaction effects based on two-way ANOVA: treatment X season Fig. 2.12**a** ($F_{[2, 75]}$ =12.45, p<0.01), Fig. 2.12**b** ($F_{[2, 91]}$ =5.06, p=0.01). Spatial cations concentrations means are based on a one-way ANOVA: location Fig. 2.12**c** ($F_{[4, 58]}$ =3.87, p=0.01), Fig. 2.12**d** ($F_{[4, 63]}$ =4.53, p<0.01).

2.3.4. Hydrophobicity

The effects of burning of slash piles on seasonal hydrophobicity (time taken for a water drop to infiltrate soil) varied among the study areas. At Hermon, in burn scars, the soil was non-hydrophobic with an average residence time of 5.00 s and it remained non-hydrophobic post-fire (Table A1), which was also the case for the non-burn areas. At Robertson, neither spatial nor temporal data showed any major trends, and soil was on average strongly hydrophobic throughout the study period, including pre-fire (Table A2). At Wit River soils was strongly hydrophobic in both pre-fire and post-fire seasons, and showed a trend of becoming severely hydrophobic towards the drier seasons (spring, summer and autumn) (Table A3). In contrast, at Rawsonville, the soils within the burn scars were non-hydrophobic before burning, they became slightly hydrophobic post-fire and were strongly hydrophobic a year after burning. The non-burn treatment sites ranged between non-hydrophobic and slightly hydrophobic. Spatially, the burn scar positions (viz. the centre, intermediate and edge positions) soils were strongly hydrophobic; and the soil matrix, the recovering and invaded reference sites soils were slightly hydrophobic (Table 2.2).

Table 2.2: (a) Seasonal hydrophobicity values between the burn and non-burn treatment sites at Rawsonville study area, (b) Spatial hydrophobicity values between the burn and non-burn sampling positions at Rawsonville study area. Means represent time in seconds for the Water Drop Penetration Test at each sampling location, with standard errors in brackets.

a) Seasonal

	Burn treatment		Non-Burn treatment	
Season	**Time (s)**	**Category**	**Time (s)**	**Category**
Pre-fire	5.00 (±0.00)	Non-Hydrophobic	5.94 (±0.74)	Slightly Hydrophobic
Post-fire	14.92 (±9.79)	Slightly Hydrophobic	5.00 (±0.00)	Non-Hydrophobic
Spring	115.42 (±75.23)	Strongly Hydrophobic	5.89 (±0.62)	Slightly Hydrophobic
Summer	494.58 (±153.64)	Strongly Hydrophobic	8.07 (±1.91)	Slightly Hydrophobic
Autumn	234.67 (±123.40)	Strongly Hydrophobic	6.68 (±0.94)	Slightly Hydrophobic

b) Spatial

Location	Treatment	Time (s)	Category
Centre	**Burn**	311.53 (±117.83)	Strongly Hydrophobic
Intermediate	**Burn**	190.25 (±90.50)	Strongly Hydrophobic
Edge	**Burn**	142.91 (±73.20)	Strongly Hydrophobic
Matrix	**Non-burn**	8.88 (±1.80)	Slightly Hydrophobic
Recovering	**Non-burn**	5.50 (±0.35)	Slightly Hydrophobic
Invaded	**Non-burn**	5.35 (±0.35)	Slightly Hydrophobic

2.4. Discussion and conclusion

The burning of biomass of *Acacia mearnsii* and *Eucalyptus camaldulensis* in the form of dried stacks affected soil physicochemical properties in multiple ways. Soil pH, EC, cations and available P were affected immediately by burning of biomass, while total C and N, available N and soil hydrophobicity responded variably to burning of slash piles. Some soil properties such as available N were affected only in the medium term, while for physical factors such as hydrophobicity, soils of only one site were affected by stack burning. After one year, most of the tested physicochemical properties had returned to pre-fire levels, with the exception of soil pH, which persisted at elevated values over the study period. In addition, the Wit river study area was exposed to flooding; as a result, some properties recovered earlier than one year.

Soil pH increased immediately after fire at all the study areas, where there was an increase by up to 4.4 pH units at Hermon and Robertson study area and by up to 3.4 pH units in Rawsonville study area (however, only 1.8 units at Wit River). An increase in soil pH is consistent with results obtained by Arocena & Opio (2003) and Granged et al. (2011). Soil pH remained significantly higher than non-burn areas for more than one year after fire at three of the study areas. Granged et al. (2011) observed an increase in soil pH on nutrient-poor acidic soils at the Mediterranean climate type South-Western Spain, which remained high for more than one year. This is also evident when viewing the data spatially (including all sampling dates), since soil pH within the centre and intermediate position of the burn scar were significantly higher than the non-burn treatment sites viz. the matrix and the two reference sites. These results are also consistent with Esquilin et al., (2007), who also found that soil pH in the centre and edge sampling positions differed significantly to that of the non-burn treatment sites. According to Certini (2005), the soil pH increase may be due to high concentrations of base cations i.e. Ca, Mg, Na and K, which concentrate on the soil surface after fire. The ecological consequences of such a major shift in soil pH could be significant as this may lead to a release and increased availability of soil bound nutrients (Arocena & Opio, 2003; Sparks, 2003), while soil microbial communities may also affected by the soil pH (Slabbert et al., 2014). Soil microbial groups such as fungi form symbioses with roots of many fynbos plants, and the medium term modification of soil pH may have negative consequences for mycorrhizal and rhizobial symbioses. The result of stack burning may thus result in a riparian landscape with spatially heterogeneous pH levels in topsoil, which may affect how the landscape recovers following clearing. High soil pH may also affect availability of trace elements such as boron and manganese (Sparks, 2003).

At both Hermon (*Eucalyptus*) and Rawsonville (*Acacia*), all four cations followed the same trend, with major increases immediately following the fire, but returned to pre-fire levels at the end of the sampling period, about a year after the fire. Kennard & Gholz (2001) and Arocena & Opio (2003) reported a significant increase in exchangeable cations within soils that were exposed to burning of slash piles. According to Kim et al. (1999), the increase in exchangeable cations is as a result of leaching from enriched ash which accumulates on the soil surface after fire. The increased cations on the surface gradually return to pre-fire conditions as the processes such as leaching displaces the cations especially from moist soils during rainy periods, or during high flow periods in riparian soils. This is borne out by the Wit River study area, which received flooding following the fire. Here soil pH increased after fire, but declined significantly in the following season, and returned to pre-fire levels four months after burning. These results point to the influence of leaching by floods, in addition to mobilisation of sediment as factors that reset the template and that allow patches affected by fire to recover in terms of chemical properties. In contrast, soil pH on three of the four study areas, those not affected by flooding, remained high. This suggests that factors other than base cations also played a role in elevated pH in the medium term. Denatured organic acids formed during fire may lead to persistence in increased soil pH values (Bodi et al., 2014). It also suggests a dichotomy in how riparian sites respond following fire; where flooding are able to leach soils, soil pH returns to pre-fire conditions in the short term, while drier sites did not return to pre-fire conditions in the medium term.

Soil EC, which often increases after fire as a result of abundant divalent cations on the surface (Inbar et al., 2014), did not increase following fire at the Wit River area. This suggests that flooding might have mobilised the abundant divalent cations in ash and thus mitigated the effects of fire on both soil pH and EC. On the contrary, soil EC did increase significantly at the three study areas that were not affected by flooding. Soil EC increased by up to 710.74 µS cm^{-1} at Hermon (*Eucalyptus*) to become significantly higher than both the pre-fire levels and the non-burn sites; this was also evident spatially where the burn scar sampling locations had higher soil EC values than the non-burn sites. In terrestrial soils, Norouzi & Ramezanpour (2013) and Inbar et al. (2014) also observed an increase in soil EC as a result of burning. However, the trajectory in fynbos riparian areas showed a decrease toward pre-fire levels with time. This mirrors the trends in base cations over time, which are likely leached from the topsoil in the relatively wet soils of fynbos riparian ecotones.

Available P was affected by burning at all study areas, including the Wit River, which was exposed to flooding. At Rawsonville and Wit River (*Acacia*), available P increased to maximum immediately after fire, and then decreased towards pre-fire levels. On the other Hand, at Hermon (*Eucalyptus*), available P increased significantly after fire and continued to increase to reach maximum in summer; and at Robertson (also *Eucalyptus*), available P did not change after fire, instead, the significant increase only came about in summer (there appeared to be a delay the increase in available P at this area). The trajectory showed that available P had returned to pre-fire levels at Hermon, Robertson and Rawsonville; and appeared to approach pre-fire levels at Wit River. This is also complimented by spatial data, where the non-burn treatment sites had significantly lower available P concentrations than the burn site. Romanya et al. (1994), Kim et al. (1999) and Badia et al. (2014) also found an increase in available P concentrations after fire, which declined with time. The increase in available P concentrations may be due to the release of phosphorus from the enriched ash, increased organic P mineralization as a result of heat and release of soil bound P at higher pH values (Hartshorn et al., 2009; Badia et al., 2014). Phosphorus is relatively immobile in soil (Kim et al., 1999) and is naturally low in fynbos soils, and most fynbos plant species are adapted to low phosphorus in soils; except for fynbos legumes which are adapted to environments with limited available P, as they use mechanisms such as root-exuded phosphatase to unlock P (Power et al., 2010). Higher available phosphorus may be beneficial for recovery of native plants in post-clearing riparian areas in the fynbos, however, Vitousek et al., (2002) showed that active N fixers require higher levels of P as it is required by legumes as part of the N-fixing mechanism. Therefore, invasive seedlings, especially legumes, may outcompete native species in the post-clearing environment, and deprive native species from using significant amounts of available P, even though this resource may be more available in patches affected by fire. Soil pH may also play a role in limiting plant uptake, or in enhancing P levels to where they are toxic to native plant species (Power et al., 2010). In addition, the interaction with soil pH may result in the release of soil bound P which may be highly patchily available to plants in riparian areas affected by burning of slash pile.

Burning of slash piles affected soil total carbon variably, where at Hermon the level did not change significantly, while at Rawsonville it decreased significantly. The decrease in total C at Rawsonville are similar to the of a study by Esquilin et al. (2007) which also reported a decrease in total C after burning on loamy soils of Manitou Experimental Forest in Colorado Springs; and Yildiz et al. (2010) also reported a decrease in total C levels following burning in the Mediterranean vegetation of Turkey. On the other hand, Mastrolonardo et al. (2014) observed non-significant differences in soil total C between the burn and non-burn site, which is consistent with the findings at Hermon. This variability in the response of soil carbon to fire may be related to the characteristics of the soils (e.g. texture), or may be related to the characteristics of the fire, which is indirectly related to piles size and size of the main members (Wan et al., 2001).

The slash piles at Hermon study area were larger than those of Rawsonville study area (Table 2.1), which suggests that the fire at Hermon should have induced very high temperatures for prolonged periods (Hubbert et al., 2015). It is therefore suggested that such high temperature fires, for prolonged periods as at Hermon would consume a large amount of soil nitrogen as it is been reported to start volatilising at relatively low temperatures (Neary et al., 1999). However, in this study, it was found that the larger piles at Hermon did not have any significant effects on total N, while smaller slash piles of Rawsonville resulted in a significant decrease in total N. According to Wan et al. (2001), the significant loss of soil nitrogen does not only depend on fire severity, but also the ecosystem properties and the type of plant biomass burnt will have an influence on nitrogen dynamics. Hinojosa et al. (2012) and Fultz et al. (2016) reported non-significant changes in total N following fire, which is consistent with the Hermon study area findings; while Badia et al. (2014) showed a decline in total N level which is similar to the Rawsonville study are results. Effects of fire on both total C and N will have an impact on the C/N ratio, which provides an index of soil microbial activity during decomposition of soil organic matter (Nave et al., 2011; Naude, 2012).

Neither ammonium nor nitrate were significantly affected by the slash pile burning immediately after fire. When combined (total available N), immediately after burning, it did not differ on the burn sites compared to the non-burn sites. Contrary to findings of this study, available N has been reported to increase after fire (Oswald et al., 1998; Fernandez et al., 2009; Schafer & Mack, 2010), which has been attributed to increased N mineralization resulting from fire-induced changes in soil temperature, soil pH and microbial activity (Wan et al., 2001). On occasions, available N (NH_4^+-N) has been shown not to significantly change on burnt soils (e.g. Switzer et al, 2012). However, in the medium term, plant available nitrogen increased in patches affected by fire in riparian areas invaded by *Acacia mearnsii*, compared to the non-burn sites. This is also evident when comparing available N spatially, where the burn scar positions had higher values compared to non-burn treatment sites, but only for areas affected by *A. mearnsii*. This medium-term increase in available N might be due to increased N mineralization resulting from increased pH and elevated soil temperatures during and immediately following biomass burning (Wan et al., 2001). Naude (2012) found higher available N in *Acacia* invaded riparian areas, but no difference in N mineralization rate, compared to natural riparian zones, suggesting modified N cycling in *Acacia* invaded areas. *Eucalyptus* invaded areas (Hermon and Robertson) did not show any clear trends in terms of plant available N, suggesting that burning of biomass may have different consequences for riparian soil biogeochemistry depending on which invader constitutes the bulk of the biomass. Even if N may be more available in patches affected by slash burning (e.g. *Acacia* affected riparian reaches), invasive seedlings may be able to outcompete native species for this resource, as suggested by Morris et al., (2011).

The soils at the Wit River and Roberson study areas are course textured, and were strongly hydrophobic before burning, and they remained strongly hydrophobic after burning. The fine textured soils at Hermon study area were non-hydrophobic and they remained unchanged post-fire. At these three study areas, burning did not induce hydrophobicity, instead, soil texture and seasonal changes may be the main causes of hydrophobicity. On the contrary, the Rawsonville study area was affected by burning; the soils in this area changed from being non-hydrophobic to slightly hydrophobic in post-fire and continued to become strongly hydrophobic in spring. The results at Rawsonville are in line with those of Fox et al. (2007) and Jeyakumar et al. (2014), who also observed an increase in hydrophobicity after fire. This suggests that in fynbos riparian zones, hydrophobicity is less associated with which invader is dominant, is naturally associated with soils of course nature, and that fire may affect levels of hydrophobicity only in certain site-specific circumstances.

This study aimed to address three questions i.e. how will burning of slash piles of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass affect soil nutrients and physicochemical properties in the medium term? Secondly, what are the spatial impacts of burning of slash pile of *Acacia mearnsii* and *Eucalyptus camaldulensis* biomass on soil nutrients and physicochemical properties? Thirdly, what is the trajectory of change subsequent to burning of biomass of *Acacia* and *Eucalyptus* spp.? The results show that changes may either be affected immediately and consistently across all riparian study areas, as in the case of soil pH, EC, available P and exchangeable cations which all increased in response to burning; or vary by study areas as in the case of total C and N, hydrophobicity and available N which varied across the sites. It should also be noted that flooding may have benefits for the speedy recovery of soil after burning of slash piles, because soils that were exposed to water submergence recovered faster than those that were not. In addition, hydrophobicity appeared to be not as a result of burning of slash piles at three of the study areas, rather it was due to soil properties and seasonal changes. Most physicochemical properties that were altered by fire returned to pre-fire levels after one year, with the exception of pH, which remained high. Altered pH may, however, have significant impacts for recovering riparian zones, especially given the interaction between nutrient availability and soil microbial composition as found by Slabbert et al. (2014), working in the same biome.

Based on the findings of this study soil physicochemical properties such as soil pH, EC, available P and exchangeable cations are likely to increase post burning of slash piles. Other physicochemical properties (e.g. available N) may be dependent on the additional factors such as properties of plant biomass to be burnt, the characteristics of the soil and the environment during burning. Soil moisture and/or submergence may play an important role toward the recovery of areas after burning, as it was evident at Wit River area, where flooding lead to the speedy recovery of the soil.

Similar trends in soil physicochemical properties as a result of burning of slash piles may also exists with natural fynbos fires where above-ground biomass has not been affected by alien invasion. These modifications include the loss of nutrients during burning (De Ronde, 1990); increase in soil nutrients after burning (Stock & Lewis, 1986); and the gradual recovery to pre-fire levels with time (Scott & van Wyk, 1992). These natural fires are however less intense (Kraaij & van Wilgen, 2014) and may result in less profound changes in soil physicochemical properties. Scott & van Wyk (1990) measured was hydrophobicity after fynbos windrow stacks burning (similar to slash piles) within the Bosboukloof and reported that surface soils (0-10 cm) decreased from being somewhat-hydrophobic to non-hydrophobic after burning. This findings supports the notion that there are additional factors influencing post-fire hydrophobicity.

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CHAPTER 3

Effects of burning of slash piles of *Acacia saligna* biomass on physicochemical properties of sandy fynbos soils

3.1. Introduction

Control and management of invasive alien plants (IAPs) is among the great challenges currently faced by landowners, conservation agencies and government branches in South Africa (Richardson & van Wilgen, 2004). Successful control and management of woody IAPs will contribute significantly towards restoring and sustaining natural ecosystems which are currently threatened by such species (van Wilgen et al., 2012). One such IAP is *Acacia saligna,* which is a nitrogen-fixing tree that can grow up to 2 - 6 m tall and was introduced into South Africa around 1845 (Cronk & Fuller, 1995; van den Berckt, 2002). Currently, in South Africa, *A. saligna* occurs mainly in and near coastal regions where it was initially introduced to stabilise sand dunes and is now considered one of the top invaders in the fynbos biome. This plant invader is relatively drought tolerant and establishes a large, soil stored seed bank which germinates quickly after wildfires or natural disturbance and grows fast to reach maturity (Cronk & Fuller, 1995; Holmes & Cowling, 1997). Mainly because of these attributes *A. saligna* has been successful in displacing fynbos vegetation, spreading through the Western Cape coastal regions. Currently, there are several biological. Chemical and mechanical methods used to removing from and limit the spread of *A. saligna* the ecosystem.

Biological control methods have to some extent been successful in reducing the spread of *A. saligna* across fynbos ecosystem. One such biological method which has proven to be highly effective in managing this invader involves using the fungus *Uromycladium tepperianum* to infect *A. saligna* trees (Strydom et al., 2012). The effects of *U. tepperianum* on *A. saligna* include reduced tree density and seed production, thus reducing the seed bank (Wood & Morris, 2007). Even though *U. tepperianum* is able to reduce density and limit seed production, *A. saligna* continues to germinate from the existing seed banks, establish, grow and increase above-ground biomass of affected areas (Chamier et al., 2012). To control *A. saligna* biomass production and negate negative influences of the species, various physical and chemical methods have been used, most of which have been shown to be costly and difficult to apply (Wood & Morris, 2007). The main chemical control method is conducted by herbicide application, while the physical methods include hand pulling, ring-barking and slashing which may be followed by burning or chipping of the cut biomass (Blanchard & Holmes, 2008). Although burning of slash piles of *A. saligna* biomass has been conducted on a number of cases within the Western Cape fynbos, it has been reported that it may not promote recovery of native vegetation (Cilliers et al., 2004; Blanchard & Holmes, 2008). However, the method is used to destroy biomass after clearing operations, especially where dry biomass may constitute a fire risk, or where removal of biomass is hampered by accessibility.

Burning of slash piles of may alter soil chemical, physical, biological and hydrological properties (Certini, 2005; Rhoades et al., 2015). Burning of slash in stacked piles might be more severe than natural forest fires, which may lead to the formation of burn scars on the soil surface as a result of altered soil properties (Rhoades et al., 2015). Potential fire damage or changes to soil properties may either be short-term and recoverable, long-term and recoverable or permanent and non-recoverable (Certini, 2005). The magnitude or level of damage to the soil depends on the fire severity, which is influenced by a number of factors including microclimatic conditions and the amount of wet or dry fuel available (Certini, 2005). Fire would have the greatest impact on the soil surface and impacts would decrease as distance into the soil profile increases (Certini, 2005). Soil moisture levels may also reduce the extent of heat penetration into the soil profile and the degree of damage to soil properties. This suggests that dry soil profiles are more likely to experience higher fire temperatures than those that are moist (Busse et al., 2005), because part of the heat produced by the fire will be consumed by evaporation of soil moisture, thus delaying heat transfer through the soil profile (Beadle, 1940).

To some extent the negative consequences of altered soil properties may be mitigated by the addition of ash onto the soil surface during and after burning, which contains nutrient released from plant biomass during burning (Oswald et al., 1998, Certini, 2005). and upon mixing with the soil surface may elevate nutrient concentrations Post-fire nutrients enriched soils often contain more available P, cations and available N (Certini, 2005). The elevated post-fire concentrations of nutrients will differ depending on the characteristics of the combusted biomass and it decreases gradually and return to their pre-fire levels with time (Giardina et al., 2000; Schafer and Mack, 2010). Regardless of nutrient addition onto soil surface, burn scars often still remain bare for extended periods, because burning might have destroyed soil stored fynbos seed banks, and thus hinder vegetation establishment (Cilliers et al., 2004). In addition, the burn scar often has high soil pH values which results from the breakdown of base cation compounds; the latter are highly concentrated in the ash (Korb et al., 2004). Elevated soil pH may lead to modified availability of certain nutrients that might either be beneficial to plants or might lead to toxicity (Arocena and

Opio, 2003). Soil EC also increases in soils that have been exposed to burning of slash piles of plant biomass, indicating modified salinity levels (Certini, 2005). According to Hernandez et al. (1997), the increase in soil electrical conductivity is due to the release of soluble alkaline compounds from burn plant biomass.

Hydrophobicity is associated with coarse-textured soils and may occur naturally as a result of organic matter decomposition, which incorporates certain organic compound into the soil (Doerr et al., 2000; Dekker et al., 2001). However, soil hydrophobicity may also develop on the soil surface as a result of burning plant biomass, which releases certain resins, waxes and oils which settle on the soil surface and coat soil particles, making them hydrophobic (Mirbabaei et al., 2013). Hydrophobic soil particles may hinder or delay water infiltration into the soil and underlying parts of the soil profile and thus enhance surface run-off and result in elevated erosion from sloped landscapes (Mirbabaei et al., 2013; Neris et al., 2013).

It is important to improve our understanding of how Western Cape fynbos soils are affected by the approaches and methods used during clearing of invasive alien plants and also have a better knowledge of how soils will recover after being exposed to such interventions. This study covers aspects of an experiment that was conducted in the Cape Flats Sands Fynbos vegetation type in Blaauwberg Nature Reserve near Cape Town. The experiment was aimed at evaluating how certain physicochemical soil properties are affected in space and over time by burning of slash piles. The objectives of the experiment were (i) to evaluate seasonal the effects of burning of slash piles of *Acacia saligna* biomass on soil physicochemical properties (ii) to evaluate spatial effects of burning of slash piles of *Acacia saligna* on soil physicochemical properties (iii) to monitor the trajectory followed by physicochemical properties subsequent to burning of biomass of *Acacia saligna*.

3.2. Material and methods

3.2.1. Study area

The study was conducted at Blaauwberg Nature Reserve, in the north of the City of Cape Town municipal area in the Western Cape, South Africa (33°45’14.61”S, 18°29’35.30”E). The study area is characterised by deep acidic sandy soils and it experiences a Mediterranean type climate, which is cool and wet in winter, and warm and dry in summer. Most of the rainfall is received between the months of May and August and the mean annual rainfall is 575 mm (Rebelo et al., 2006; Krupek et al., 2016). The native vegetation is dominated by fynbos, with Proteaceae, Ericaceae, Restionaceae and Asteraceae prominent, however, native plant species have almost entirely been displaced by the Australian invasive species *Acacia saligna* which has invaded the area and occurs in dense stands (Krupek et al., 2016).

3.2.2. Experimental design

The study area was divided into three sampling areas, viz. a burn site where burning of slash piles of *Acacia saligna* biomass was carried out in spring of 2014; a recovering reference site which was previously cleared of *A. saligna*; and an invaded reference site which is currently densely invaded by *Acacia saligna* (Fig. 3.1). Five slash piles comprising *Acacia saligna* biomass were selected for the burning of slash piles on sandy soils. Assuming that the slash piles were half ellipsoid shaped, their average volume was calculated to be 28.30 m^3, using the following formula:

Volume = length x width x height x π) ÷ 6 (Busse et al., 2013)

Figure 3.1: Map of the Blaauwberg Nature Reserve study area. The map indicates the burn site where burning of Acacia saligna biomass piles was conducted, the recovering site which was previously cleared and is currently recovering, and the invaded site that is currently invaded by A. saligna (Google Earth, 2016).

3.2.3. Sampling and laboratory methods

Prior to burning, existing slash piles were carefully disassembled by removing the overlying slash and sampling the soil to 10 cm. Hereafter, soil samples were collected from experimental sites immediately post-fire, and once in the subsequent three seasons. On every sampling occasion, a total of 40 soil samples were collected from the experimental sites using a metal tube to a depth of 10 cm. These 40 soil samples were comprised of three samples from five piles viz. the centre, the intermediate position, i.e. between the edge and centre, and the edge (Fig. 3.2), one sample collected from the soil matrix, about 2 m from the edge, ten samples collected from the recovering site, and ten samples from the invaded site.

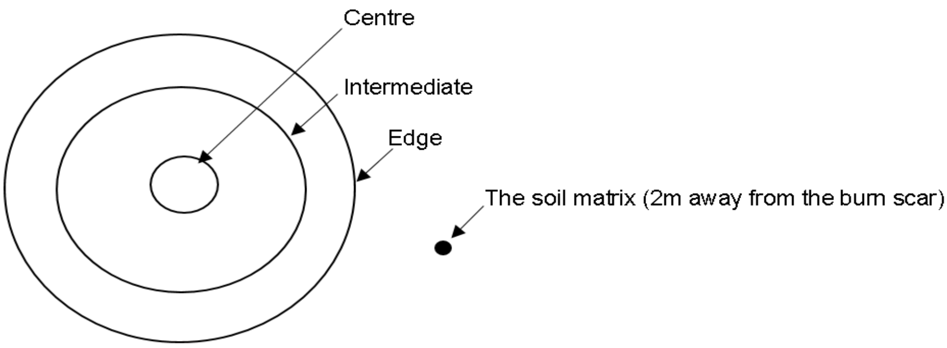

Figure 3.2: Depiction of the sampling positions within the burn site.

Field collected soil samples were sealed in plastic bags and transported to the laboratory. On arrival, the soils were sieved through a 2 mm sieve to remove rocks, pebbles and gravel (Mataix-Solera et al., 2013). The gravimetric soil water content was determined by oven drying samples at 105°C for 48 h (Shafer & Mack, 2010). Soils were treated with hydrogen peroxide to remove excess organic matter, then soil texture was determined using the hydrometer method. This entailed dispersing 40-50 g soil with 100 ml sodium hexametaphosphate (HMP) solution in an electric mixer for 5 minutes and soil texture measured using an ASTM 152H hydrometer (Gee & Bauder, 1986). Soil hydrophobicity was estimated in the laboratory using the Water Drop Penetration Test (WDPT) on air-dried soil samples using distilled water (Dekker et al., 2001). Soil pH was tested by mixing 10 g of soil and 20 ml distilled water for 30 minutes (1:2) and analysed with a calibrated Hanna pH meter HI 8424 (Robertson et al., 1999). Electrical conductivity of the soil was measured using a 1:2 mixture of soil and distilled water and a calibrated Hanna HI 8733 conductivity metre (Hanna HI 8733) (Miller & Curtin, 2007).

Available nitrogen (NH_4^+-N and Nitrate NO_3^--N) was extracted using a 0.5 M K_2SO_4 solution, then later analysed using Genesys 20 spectrophotometer. The salicylic acid method was used to measure NO_3^--N concentrations and the indophenol blue method for NH_4^+-N concentrations; procedures are as described in Anderson & Ingram (1993). Total soil C and N were measured on a dry combustion CN analyser (Euro EA Analyser; Sollins et al., 1999). Exchangeable cations were extracted with 1 M NH_4OAc (Simard, 1993) and analysed with a Varian 240 FS atomic absorption spectrometer. Available P was extracted using the P-Bray 2 solution similar to that of Bray and Kurtz (1945), and the concentrations were determined with a spectrophotometer (Genesys 20) based on the molybdenum blue method (Olsen & Sommer, 1982).

3.2.4. Data analyses

The data were analysed using the Statistica version 13 software package (Dell Inc., 2015). All parameters were checked for normality using the Kolmogorov-Smirnov test. Two-way analyses of variance (ANOVA) were used to determine the burn treatment effects seasonally viz. pre-fire, post-fire, summer, autumn and winter. One-way ANOVAs were used determine the burn treatment effects by location viz. the centre, intermediate, edge, matrix, and reference sites. Where the computed ANOVAs showed significance ($p<0.05$), the Bonferroni post hoc tests were performed.

3.3. Results

3.3.1. Soil pH and EC

Within the burn scars, soil pH (water) increased significantly ($p<0.01$) after burning of slash piles, and continued to increase ($p<0.01$) to reach a maximum in summer and it had not returned to pre-fire levels at the end of the sampling period i.e. winter. There was a difference of 2.90 pH units between the summer values on and off the burn scar. In contrast, the soil pH within the non-burn sites (viz. the matrix and two reference sites) remained unchanged throughout the sampling period (Fig. 3.3a). Spatially, within the burn scar soil pH was highest in the centre, which was similar to the intermediate positions and the edge, but significantly higher than the soil matrix ($p<0.01$), the recovering reference site ($p=0.00$) and the invaded reference site ($p=0.00$; Fig. 3.3b).

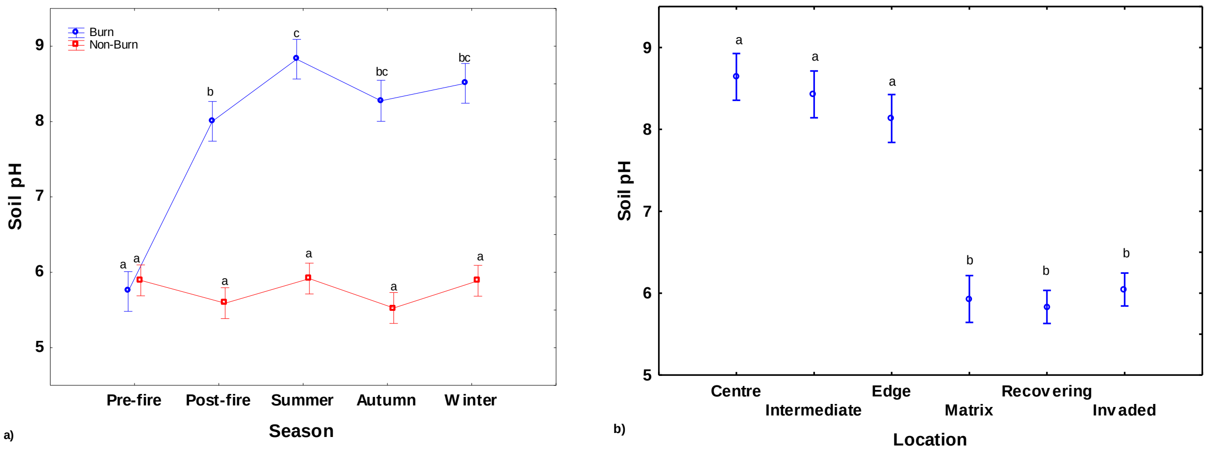

Figure 3.2: (a) Seasonal soil pH (water) values within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil pH values within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil pH means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.3**a** ($F_{[4, 189]}$ =56.71, $p<0.01$). Spatial soil pH means are based on a one-way ANOVA: location Fig. 3.3**b** ($F_{[5, 153]}$ =109.94, $p<0.01$).

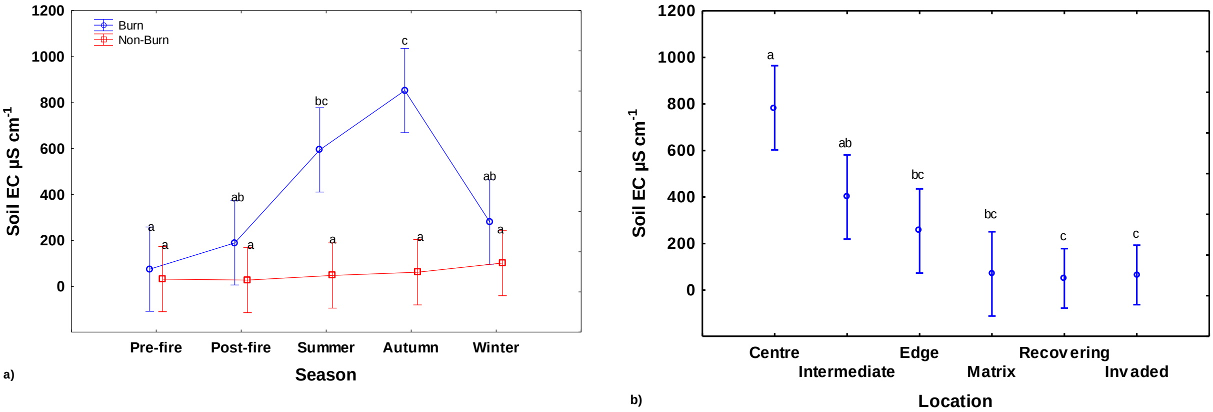

Figure 3.4: (a) Seasonal soil EC values within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil EC values within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil EC means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.4**a** ($F_{[4, 190]}$ =7.08, $p<0.01$). Spatial soil EC means are based on a one-way ANOVA: location Fig. 3.4**b** ($F_{[5, 154]}$ =11.42, $p<0.01$).

Soil EC within the burn scar did not increase immediately after burning; the effects of fire were delayed and were only noticeable in summer (i.e. the season after post-fire), when EC increase significantly ($p=0.01$) and continued to increase ($p<0.01$) to reach its maximum in autumn before returning to pre-fire levels (Fig. 3.4a). There was a difference of 790.72 EC units between the autumn values on and off the burn scar. On the other hand, the non-burn sites viz. the matrix and the two reference sites did not show any major changes through all the seasons. Spatially, EC was highest in the centre, which was similar to the intermediate position, and significantly higher than the edge (p<0.01), the matrix (p<0.01), recovering reference site (p<0.01) and the invaded reference site (p<0.01; Fig. 3.4b).

3.3.2. Carbon (C), nitrogen and available P

Total carbon did not change seasonally as a result of burning of slash piles, and this was observed within both the burn scar non-burn treatment sites (Fig. 3.5a). Spatially, the highest total C levels were recorded in the invaded reference site, which were similar to the edge and recovering reference site and significantly higher than the centre (p=0.01) and the soil matrix (p=0.01; Fig. 3.5b).

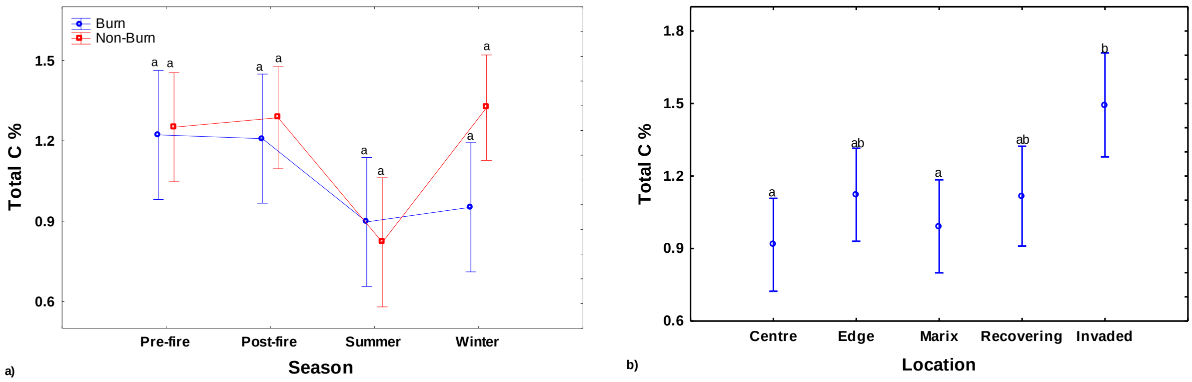

Figure 3.5: (a) Seasonal soil total C levels within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil total C levels within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, p<0.05). All the seasonal soil total C means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.5**a** ($F_{[3, 87]}$ =1.42, p=0.24). Spatial soil total C means are based on a one-way ANOVA: location Fig. 3.5**b** ($F_{[4, 65]}$ =4.58, p<0.01).

Similar to soil total C, total N also did not show any significant differences as a result of fire, and there were also no significant differences between the burn and non-burn sites (Fig. 3.6a). Spatially, total N levels were highest in the invaded reference site; the levels were similar to those of the recovering reference site, and significantly higher than the centre (p<0.01), the edge (p=0.02) and the soil matrix (p=0.03; Fig. 3.6b).

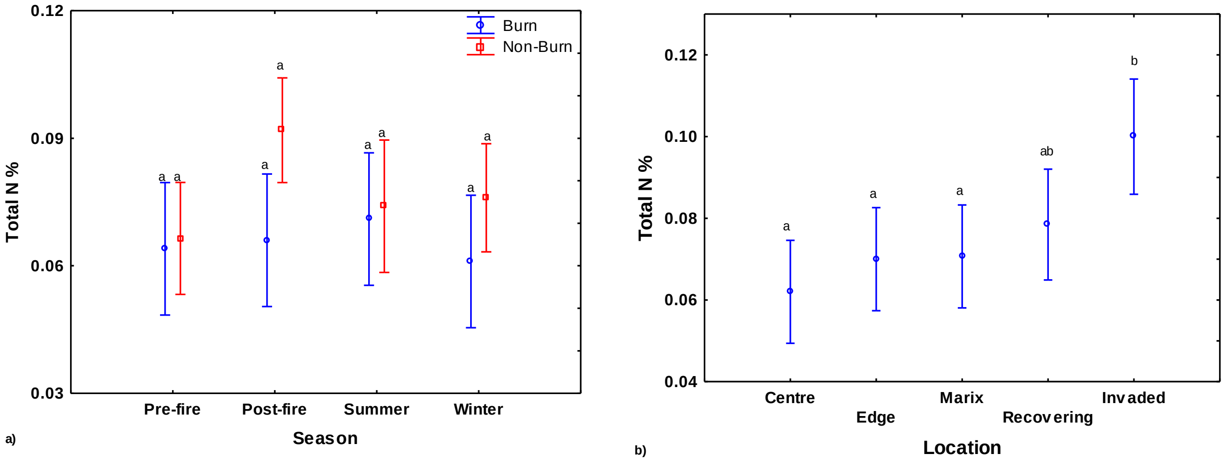

Figure 3.6: (a) Seasonal soil total N levels within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil total N levels within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil total N means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.6**a** ($F_{[3, 87]}=1.18$, $p=0.32$). Spatial soil total N means are based on a one-way ANOVA: location Fig. 3.6**b** ($F_{[4, 65]}=4.53$, $p<0.01$).

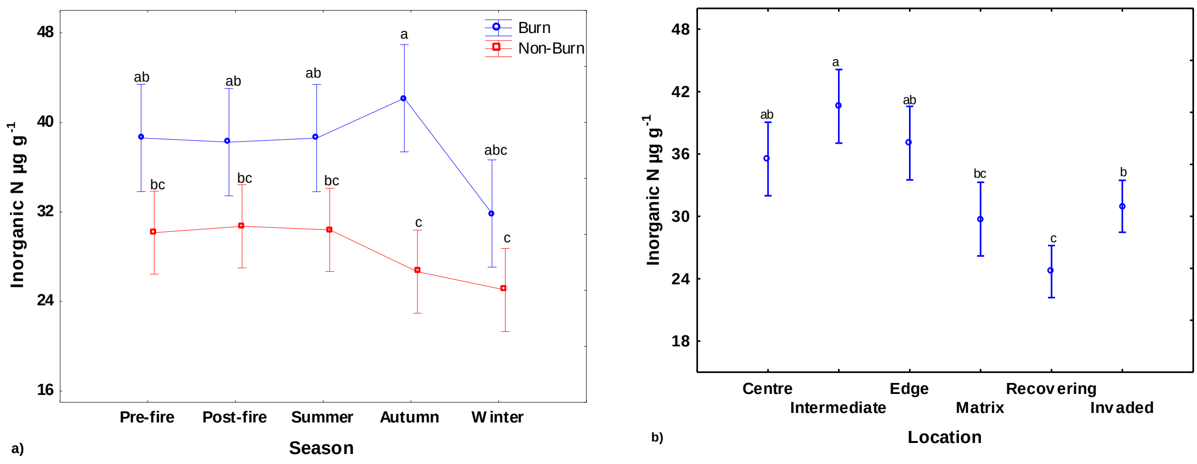

Figure 3.7: (a) Seasonal soil available N levels within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil available N levels within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil available N means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.7**a** ($F_{[4, 190]}=1.31$, $p=0.27$). Spatial soil available P means are based on a one-way ANOVA: location Fig. 3.7**b** ($F_{[5, 154]}=14.04$, $p<0.01$).

Soil available N did not change immediately after fire within the burn scar and remained unaltered throughout the sampling period. The non-burn sites also did not change immediately after fire, but levels were significantly lower than the burn scar ($p<0.01$) in autumn (Fig. 3.7a). Spatially, available N was highest in the intermediate of the scar, which was similar to the centre and the edge, and significantly higher than the soil matrix ($p<0.01$), the recovering reference site ($p<0.01$) and the invaded reference site ($p<0.01$; Fig. 3.7b).

Available P increased significantly (p=0.01) after fire and continued to increase (p<0.01) to reach its highest concentrations in summer; and eventually returned to pre-fire levels in autumn - approximately six months after burning (Fig. 3.8a). The actual values in summer were 7.35 μg g^{-1} in non-burn soils and 120.63 μg g^{-1} in soils of the burn scar soils, a difference of two orders of magnitude. The non-burn treatment sites were not affected by the fire and no differences were found through all the seasons. Spatially, the soils of the centre position had the highest available levels P, which were similar to the intermediate and edge position, and significantly higher than the soil matrix (p<0.01), the recovering reference site (p<0.01) and the invaded reference sites (p<0.01; Fig. 3.8b).

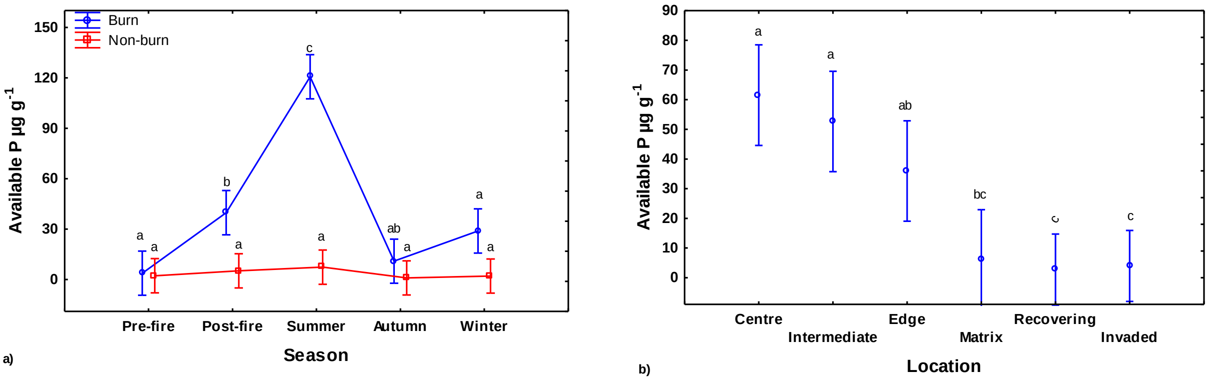

Figure 3.8: (a) Seasonal soil available P levels within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil available P levels within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, p<0.05). All the seasonal soil available P means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.8**a** ($F_{[4, 190]}$ =27.87, p<0.01). Spatial soil available P means are based on a one-way ANOVA: location Fig. 3.8**b** ($F_{[5, 154]}$ =11.82, p<0.01).

3.3.3. Exchangeable cations

Within the burn scar, exchangeable cations (i.e. Ca, Mg, Na and K) did not increase immediately after fire; the increase appeared to be delayed and only became significantly higher than non-burn areas in summer (one season after post-fire); and by the end of sampling period, exchangeable cations had returned to pre-fire levels. There were no significant changes in exchangeable cations concentrations within the non-burn treatment sites (Fig. 3.9a). Spatially, there were no significant differences in exchangeable cations concentrations between all the sampling positions and locations (Fig. 3.9b).

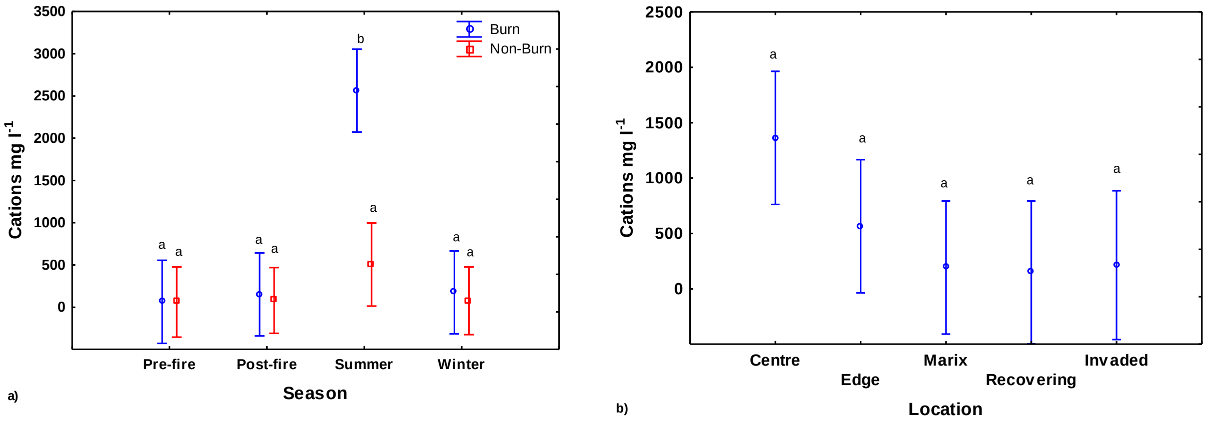

Figure 3.9: (a) Seasonal soil cations concentrations within the burn and the non-burn treatment sites at the Blaauwberg study area; and (b) spatial soil cations concentrations within the burn scar and non-burn sampling locations at Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil cations means are interaction effects based on two-way ANOVA: treatment X season Fig. 3.9**a** ($F_{[3, 87]}=8.56$, $p<0.01$). Spatial soil cations means are based on a one-way ANOVA: location Fig. 3.9**b** ($F_{[4, 65]}=2.80$, $p=0.03$).

3.3.4. Hydrophobicity

Seasonally, during pre-fire, the soils were already strongly hydrophobic within the fire scar and it remained this way post-fire and was still strongly hydrophobic at the end of the sampling period. On the non-burn treatment sites, the soils were strongly hydrophobic pre-fire and at the end of sampling it was severely hydrophobic (Table 3.1a). Spatially, the soils were strongly hydrophobic in the centre and intermediate, and severely hydrophobic on the non-burn treatment sites (Table 3.1b).

Table 3.1: (a) Seasonal hydrophobicity values between the burn and non-burn treatment sites at Blaauwberg study area; and (b) Spatial hydrophobicity values between the burn and non-burn sampling positions at Blaauwberg study area. Means represent time in seconds for the Water Drop Penetration Test at each sampling location, with standard errors in brackets.

a) Seasonal

	Burn treatment		Non-burn treatment	
Season	**Time (s)**	**Category**	**Time (s)**	**Category**
Pre-fire	233.87 (±79.17)	Strongly Hydrophobic	127.12 (±40.73)	Strongly Hydrophobic
Post-fire	504.40 (±134.43)	Strongly Hydrophobic	108.25 (±33.81)	Strongly Hydrophobic
Summer	458.13 (±137.02)	Strongly Hydrophobic	1155.48 (±164.71)	Severely Hydrophobic
Autumn	919.60 (±206.14)	Severely Hydrophobic	1385.32 (±116.53)	Severely Hydrophobic
Winter	363.20 (±119.62)	Strongly Hydrophobic	1158.08 (±133.57)	Severely Hydrophobic

b) Spatial

Location	Treatment	Time (s)	Category
Centre	**Burn**	503.75 (±141.19)	Strongly Hydrophobic
Intermediate	**Burn**	421.45 (±113.66)	Strongly Hydrophobic
Edge	**Burn**	758.80 (±151.02)	Severely Hydrophobic
Matrix	**Non-burn**	814.75 (±148.58)	Severely Hydrophobic
Recovering	**Non-burn**	1030.13 (±141.96)	Severely Hydrophobic
Invaded	**Non-burn**	965.00 (±115.35)	Severely Hydrophobic

3.4. Discussions and conclusion

This study was aimed at evaluating impacts of slash pile burning temporally and spatially on selected terrestrial soil physicochemical properties. Soil pH increased significantly immediately after fire and was still elevated at the end of the sampling period, which was also reflected spatially, where pH was higher within the burn treatment sites compared to the non-burn sites. The increase has been attributed to the abundant presence of base cations contained in the ash deposited on the soil surface after fire (Schafer & Mack, 2010). Xue et al. (2014) also observed an increase in soil pH as a result of fire. Korb et al. (2004) showed soil pH values that were higher in the burn scar and lower on the non-burnt sites within semiarid *Pinus ponderosa* stands. Rhoades et al. (2004) detected soil pH increases which persisted for about 2 years, similar to the results presented here, where after one year, no trajectory toward pre-fire levels was observed. The persistence in high soil pH may either be due to slow dissolution rates, i.e. slow release of base cations from the ash, which result in pH increase or to the denaturing of organic acids during burning (Bodi et al., 2014; Thomaz et al., 2014; Xue et al., 2014).

Both soil cation concentrations and EC increased significantly following burning of slash piles. Kennard & Gholz (2001) compared non-burnt soils with intensely burnt soils (analogues of burning of slash piles), and reported higher concentrations of Ca, Mg and K on burnt soils. Alcañiz et al. (2016) also reported an increase in exchangeable K, Ca, and Mg immediately after fire; within a year Ca and Mg concentrations had returned to pre-fire levels. Ando et al. (2014) observed an increase in K and Ca after burning of slash pile on sandy clay loam soils, mirroring an increase in EC. These cations enriched soils, eventually returning to their pre-fire levels as the surface deposited ash is leached or mobilized, and the topsoil leached (Bodi et al., 2014).

In contrast to soil pH, base cations and soil EC showed a delayed response to fire. In the case of EC, the highest levels were only observed six months after the fire treatment, while cations peaked three months after the fire (no cation results are available for autumn). Moisture during the rainy season has an influence on the mobilizing exchangeable cations (Tomkins et al, 1991). This supports the notion that the rate of leaching of cations from ash into the mineral soil was slow due to a lack of soil moisture, especially since the fire treatment preceded a period of relatively low rainfall – the summer and autumn seasons. Cations concentrations had returned to pre-fire levels at the end of sampling period (winter). Implications of cation enriched soils include increased pH and EC (as observed), which may have an influence on soil microorganisms, leading to altered decomposition processes (Bodi et al., 2014). Soil EC differences could also be seen spatially, where the burn scar sampling positions (especially the centre and intermediate) had high EC levels than the non-burn sites i.e. the soil matrix, the recovering and invaded reference sites, suggesting altered soil biology could manifest in a highly patchy nature in fynbos soils due to slash pile burning.

Available P also increased significantly after burning of slash piles, to reach summer levels of 120.63 µg g^{-1} which was 15 times more than the non-burn sites. These elevated available P levels were also observed when compared spatially, where the burn scar sampling positions had higher available P values than the non-burn sites. Pourreza et al. (2014) also reported an increase in available P after severe burning on sandy clay loam soils. Ando et al. (2014) conducted burning of slash piles experiment in eastern Zambia on sandy loam soils and found that topsoil available P increased. Alcañiz et al. (2016) also reported an increase in available P after prescribed fire which returned to pre-fire level within a year. Within the fynbos, an increase in available P may be beneficial to native vegetation, however, such a large increase may be detrimental to some fynbos species which may experience P toxicity (Power et al., 2010). Yelenik et al. (2004) also suggested that some indigenous fynbos plants may be negatively affected by elevated levels of both N and P. The rapid decline in P concentrations from summer to autumn may be due to uptake from microbes, since fire has the ability to soil P to microbial available P (i.e. orthophosphate; Certini, 2005), thus conversion to unavailable P, though leaching of finer soil particles from the topsoil cannot be excluded.

Total C and N were not affected by burning of slash piles, and they both remained within the same range within both the burn scar and non-burn sites. These soil components showed a similar trends spatially, where both total N and C levels within the burn scar (in the centre and intermediate) were similar to that of the matrix, but significantly lower than soils of the invaded reference sites. The latter could be the result of increased above-ground biomass addition to soils by *A. saligna* (Chamier et al., 2012) which may lead to higher total C and N content. Most research results indicate a decrease in soil total C after prescribed fire (e.g. Johnson & Curtis, 2001; Esquilin et al., 2007; Baird et al., 1999); with the exception of Mastrolonardo et al. (2014), who reported that total C did not change after fire, which is consistent with this study. This could be due to the characteristics of the fuel wood burned (plant species) and the particular properties of the ecosystem, e.g. soil texture (Wan et al., 2001). Total available N (NH_4^+-N plus NO_3^--N) was also unaffected seasonally by burning of slash piles; concentrations remained unchanged after fire and they were similar to those of non-burn sites. In most cases, available N concentrations have been reported to increase after fire, as a result of increase nitrification within burnt soils (e.g. Esquilin et al., 2007; Schafer& Mack, 2010). Switzer et al. (2012) reported an increase in NO_3^--N and no change in NH_4^+-N and attributed it to increased conversion (nitrification) of ammonium to nitrate within the burnt soils. However in this study soil N availability may have been constrained by the lack of rain immediately following burning. These findings support the work of Nardoto & Bustamante (2003) which reported an increase in NO_3^--N during the rainy season.

Hydrophobicity was not affected by burning; there was a decrease in the time it takes for a water drop to infiltrate the soil within the burn scar after fire, compared to the non-burned soils. Whereas soils from the reference sites and the matrix increased in hydrophobicity towards the drier seasons, soils from within burn scars remained lower in water repellency throughout. Research has shown that hydrophobicity often increases after burning (MacDonald & Huffman, 2004; Fox et al., 2007; Malkinson & Wittenbergh, 2011). On the contrary, the results of this study suggest that hydrophobicity was affected more by natural processes and that fire disturbances may lower soil water repellency. Hydrophobicity may occur naturally on coarse textured soils with high organic matter content (Doerr & Thomas, 2000), which may become more profound as the soil becomes drier due to a lack of precipitation. Mirbabaei et al. (2013) evaluated the relationship between hydrophobicity and soil properties; and found that sandy soils that had considerable amounts of organic matter tended to be hydrophobic, sometimes severely hydrophobic. Thus, Blaauwberg soils were strongly hydrophobic not because of the fire; instead, it was likely due to the coarse texture of the soils (sandy soils; Table 2.1), perhaps linked to high organic matter content nature and drier seasons towards the end of the sampling period.

This chapter aimed to answer two key questions, firstly, how will burning of slash piles of *Acacia saligna* biomass seasonally and spatially affect selected soil physicochemical properties? Secondly, what is the trajectory of change subsequent to burning of biomass of *Acacia saligna*? The results show that burning of slash piles of *A. saligna* had variable effects on soil physicochemical properties. Physicochemical properties followed either a trend of immediate increase after fire (pH), a delay of approximately three to six months before the increase was observed (available P, cations and EC) or no changes as a result of fire (soil total C and N and available N). When evaluated spatially, physicochemical properties (i.e. soil pH, EC, available P and cations) showed the highest concentrations within the soil of the burn scar and lowest in the non-burn treatment sites. On the other hand, total C and N had their highest levels on the *A. saligna* invaded reference site, which supports the notion that this plant species produces high quantities of biomass which gets incorporated into the soil over time. The trajectory of changes in physicochemical properties which were affected by burning of slash piles of *A. saligna* biomass indicated that they had returned to pre-fire levels and were similar to non-burn soils within a year of burning. The results indicate that effects of burning of slash piles of *A. saligna* on physicochemical properties of terrestrial soil vary per given parameter properties. In addition, the effects on the physicochemical properties appear to dissipate within a year of burning. This is with the exception of soil pH which was still significantly higher than pre-fire and non-burn levels after a year of sampling, a significant finding.

There is little reported on severity of natural or prescribed fynbos fires (Kraaij & van Wilgen, 2014), which to some extend may be due to the notion that fynbos is dominated by small to large shrubs, hence it may not produce severe fires. There is however reports similar to those of burning of slash piles, where during burning there is a loss of soil nutrients due to volatilisation, in addition, after fire an elevation in nutrients as a result of addition of enriched ash into surface soil, and finally a gradual return of affected parameters to their pre-fire levels (Stock & Lewis, 1986; De Ronde, 1990; Scott & van Wyk, 1992). Soil volatilisation of K, Ca and P after exposure to intense fires (De Ronde, 1990); and enrichment in total N and available N which could be attributed to release during biomass combustion and ash addition to soil (Stock & Lewis, 1986). On the other hand, hydrophobicity has been shown to be controlled by additional factors, in addition to burning, where fynbos fires have resulted in soils that are less hydrophobic (Scott & van Wyk, 1990). This is contrast to studies that have reported an increase in soil hydrophobicity as a result of burning (e.g. MacDonald & Huffman, 2004; Fox et al., 2007).

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CHAPTER 4

Conclusions, recommendations and future research

4.1. Introduction

As invasive alien species, *Acacia* and *Eucalyptus* spp. are known to substantially increase above-ground biomass (Chamier et al., 2012), and large quantities of excess biomass remain after their clearing. *Acacia saligna*, *Acacia mearnsii* and *Eucalyptus camaldulensis* represent three major plant invaders in the fynbos biome, the former in terrestrial areas and the latter two species in riparian environments. The Working for Water programme has been instrumental in clearing of large tracts of land invaded by these and other invasive species. Management of excess biomass accumulated during clearing of IAPs such as *Acacia* and *Eucalyptus* spp. has proven to be challenging to organisations and landowners involved. Burning of IAP biomass slash piles is one of the control methods for destroying excess biomass which accumulates during clearing (Holmes et al., 2008). However, burning of large quantities of biomass may expose the environment (including soils) to high temperatures, for long periods, depending on the characteristics of the slash piles (Cilliers et al., 2012). When exposed to elevated temperatures, soils may experience short, medium or long-term damage, involving alteration of several physicochemical properties; in some cases, changes may be irreversible (Certini, 2005). Damage can be visually observed by the lack of plant germination and establishment on burn scars, thus hampering restoration of the ecosystem in its totality (Blanchard & Holmes, 2008). The findings presented in this thesis address the aspects of soil chemical and physical responses to burning of biomass of the three species in the form of slash piles, a widely practiced method to destroy biomass on site.

4.2. Soil acidity and salinity

Fynbos soils are considered relatively acidic, and also relatively nutrient-poor; native plant species are adapted to such soils (Rebelo et al., 2006). In this study, before burning, in riparian soils, pH (water) ranged between 3.50 and 6.70; after burning topsoil pH changed to being within the alkaline range between 8.30 and 9.05. These changes persisted for more than one year and showed no trajectory of returning to pre-fire levels. This was also the case at the only terrestrial study area where experiments were carried out, Blaauwberg, where there was a shift in soil pH within the burn scar from acidic before burning to alkaline after. One exception is the Wit River study area, where pH of the riparian soil returned to pre-fire levels within four months subsequent to the fire. The reason for this is most likely replacement of some sediment and/or leaching of base cations and other compounds contributing to altered pH following fire. The study area, being relatively low-lying, was exposed to flooding within one month following the burning of slash piles. While there was an initial increase in pH, similar to the other sites, the increase was not as pronounced, and quickly returned to pre-fire levels. However, altered soil pH levels may have significant implications for soil chemistry, rhizosphere processes and plant ecology, since nutrient cycling and microbial activity are both to a large extent influenced by soil pH (Yelenik et al., 2004; Slabbert et al., 2014). Furthermore, burning of large amounts of biomass may result in a mosaic of varying soil pH levels since matrix soils will have soils in the native acidic range, while burned areas will have soils which are basic (at least for the medium term). It is unclear what the long-term trajectory of soil pH is as the experimental period ended after monitoring soils for one year. In other studies (e.g. Bodi et al., 2014), it was found that up to two years is needed for pH to return to pre-fire levels, and if this is the case in riparian and terrestrial sites in the fynbos, this may well affect the trajectory of plant recovery following clearing and burning of slash piles. Soil pH may affect plant growth more directly by affecting availability of nutrients, or indirectly by altering soil microbial composition as some soil microbes are better adapted and more abundant at low pH (Slabbert et al., 2014), while fire affected soil tends to be in the alkaline range.

Soil electrical conductivity also increased significantly at three of the four riparian study areas (except Wit River) and at the terrestrial study area. Soil salinity, of which EC is an index, is known to increase following fire, however, base cations, which are correlated with EC levels, are very mobile in soils (Sparks, 2003). High salinity affects the hydrologic properties of soils, by making them less porous, with limited air and moisture movement (Sparks, 2003). As a result, very wet or inundated soils will experience a rapid drop in EC levels as base cations are leached from the soil profile. Indeed, this is likely the cause of the drop in EC after the *Acacia mearnsii* invaded reach at Wit River experience flooding subsequent to the fire treatment. Throughout the study, there was generally a good correlation between soil EC and cations, with a drop in EC as experienced at Wit River also reflected in a drop in cation concentrations.

4.3. Soil carbon, nitrogen and phosphorus

The fynbos biome is known to have relatively nutrient-poor soils, but with a high diversity of soils, ranging from clay-rich renosterveld to sandy standveld soils (Yelenik et al., 2004; Fey, 2010; Allsopp et al., 2014). The addition of nutrients through burning of slash piles may result in patches within the ecosystem landscape with altered chemical composition, which may generally be beneficial to plant growth. At two study areas, burning did not significantly alter total C and N immediately post-fire nor in the medium term (one year post-fire). However, available nitrogen levels, while not generally affected in the short term, increased in fire-affected soils by the end of the monitoring period. However, this was only the case in riparian soils previously supporting *Acacia mearnsii* stands, and not in terrestrial soils or riparian areas cleared of *Acacia saligna* and *Eucalyptus camaldulensis* stands respectively. Naude (2012) showed that *Acacia mearnsii* invaded riparian soils had higher nitrogen stocks, and the current study shows that fire may pyromineralise stocks of N, and make these available to newly germinated seedlings. Stock & Lewis (1986) also showed that fynbos fires are able to enhance N availability in fynbos soils, and this increased available N stocks are then available to newly germinated seedlings. However, from the results presented here, increased N availability may only manifest in the medium term, and may be site-specific, a response that needs further study as this may have some implications for regeneration post-fire in cleared landscapes. Indeed, the change in nutrient levels within fynbos may provide an opportunity for secondary invasion, maintaining the altered plant species composition characteristic of invaded landscapes (Yelenik et al., 2004).

Soil available P increased significantly at all of the study areas, and in one case (Blaauwberg; terrestrial) it increased by two orders of magnitude. In all cases, immediately following the burning of biomass, elevated levels of available P increased when compared spatially, where the burn scar sampling positions had high levels of available P compared to the non-burn treatment sites. However, soil available P quickly returned to pre-fire levels at Blaauwberg. In contrast, at three of the four riparian reaches, increased levels of available P persisted over the medium term for the riparian sites, though a trajectory towards pre-fire levels was observed. Increased available P may either be beneficial to fynbos vegetation or may lead to competition between invasive N-fixers (since they require P to fix N) and indigenous vegetation (Vitousek et al., 2002; Power et al., 2010). In addition, some fynbos plant species do not respond positively to incensed P and N levels (Yelenik et al., 2004). The interaction between pH and soil P may be significant and prevent newly established seedlings from benefitting in post-fire environments. As in the case of pH and EC, a separate trajectory emerged for Wit River, which rapidly returned to pre-fire soil chemistry. The consequences of fire in the medium term may be different depending on whether the burned area is close enough to the active channel to experience flooding.

4.4. Soil hydrology

There are two ways in which burning of slash piles of *Acacia* and *Eucalyptus* spp. may affect soil hydrological properties. Firstly, through the addition cations in solution, this may result in elevated soil EC level and lead to saline soils, which are less permeable (Sparks, 2003). Such saline soils usually allow for less movement of air and water. Secondly, soil hydrophobicity may develop as a result of burning of slash piles, which hinders water infiltration through the soil (Doerr & Thomas, 2000). Soils at the Rawsonville study area showed fire-induced temporal and spatial hydrophobicity after burning of slash piles, measured using the water drop penetration tests on dried soils *ex situ*. The fine textured Hermon soil did not show significant hydrophobicity, while soils at Robertson, Wit River and Blaauwberg showed naturally high hydrophobicity (Doerr et al., 2000; Doerr & Thomas, 2000), which remained unaffected by fire. Hydrophobicity may develop naturally on coarse textured soils that have high organic matter content, as is the case with these study areas. It can be concluded that soil hydrophobicity may develop post-fire, but this depend on site-specific factors such as soil particle size and organic matter content. In some instances, fire-induced hydrophobicity has been shown to dissipate over time, at rates dependent on fire severity (Huffman et al., 2001; Malkinson & Wittenberg 2011).

4.5. Implications for management

Burn scars are often observed in managed landscapes as semi-circular patches that are not covered by vegetation (Neary et al., 1999). In fynbos riparian environments, soil properties within these scars are different from the surrounding landscape, and this may contribute to a lack of recovery that has been observed on some fire scars, even several years after fire. Indeed, in some cases, the chemistry of the burn patches has not returned to pre-fire conditions after one year, and there is no trajectory towards pre-fire soil chemical conditions. This may result in persistence of bare patches, especially if these are hydrologically disconnected from streams, as in the case of riparian study areas elevated above the active channel. These bare patches may require management efforts to aid in vegetation establishment (Cilliers et al., 2004) and reduce the potential of erosion by wind, or eventual occupation by invasive seedlings, which may be more plastic and more adaptable in terms of soil chemistry for establishment and growth or better able to overcome limitations to compete for elevated nitrogen or phosphorus (Morris et al., 2011). Thus, the process of clearing and burning should also include a monitoring plan which will aid in the recovery of the soil and vegetation establishment on these burn scars (Blanchard & Holmes, 2008).

Precipitation and flooding play an important role towards leaching and washing away of ash containing nutrients and thus lead to a speedy recovery of the soils (Bodi et al., 2014). The role of natural river processes such as flooding can be seen in this study, since the Wit River study area, where burn scars were submerged after burning as a result of flooding, showed a rapid trajectory towards pre-fire soil chemical conditions. At this study area, water submergence likely resulted in the mobilisation of enriched ash, which lead to soil pH recovering within 3 - 4 months and soil EC not changing as a result of fire. Altered soil pH persisted for longer at other study areas, where it may have led to the release of some soil bound nutrients (Sparks, 2003) and affected the soil microbial communities (Slabbert et al., 2014). Thus judicious choice for the location of slash piles (destined for burning) may be an inexpensive approach to ensure that floods are able to flush away or leach some of the ions and ensure pH recovery in the short term.

4.6. Future research

How prevalent is altered soil chemistry and what is the trajectory of soil chemical changes over time? The results presented here suggest some factors, such as soil pH may only return to a trajectory of recovery over the long term. However, also insightful is that this is not the case at all sites, and the factors that may play a role in rapid recovery need to be understood. In riparian soils, the role of flooding in soil biogeochemistry, and in terrestrial soils, the role of precipitation following fire on soil nutrient cycling may shed light on how fire may affect soils. This needs to also be investigated over a reasonable length of time, and in conjunction with vegetation re-generation.

What are the implications of altered soil chemistry for native riparian species as well as native terrestrial species? Some information is already available that illuminates the relationship between riparian soil chemistry, including pH, soil available N and soil available P, and soil microbial composition and diversity (Slabbert et al., 2014). Similar information for various native and invasive fynbos species may assist in understanding and indeed predicting how regeneration of native and invasive species might proceed in cleared landscapes, and assist in deciding on whether assisted restoration is necessary.

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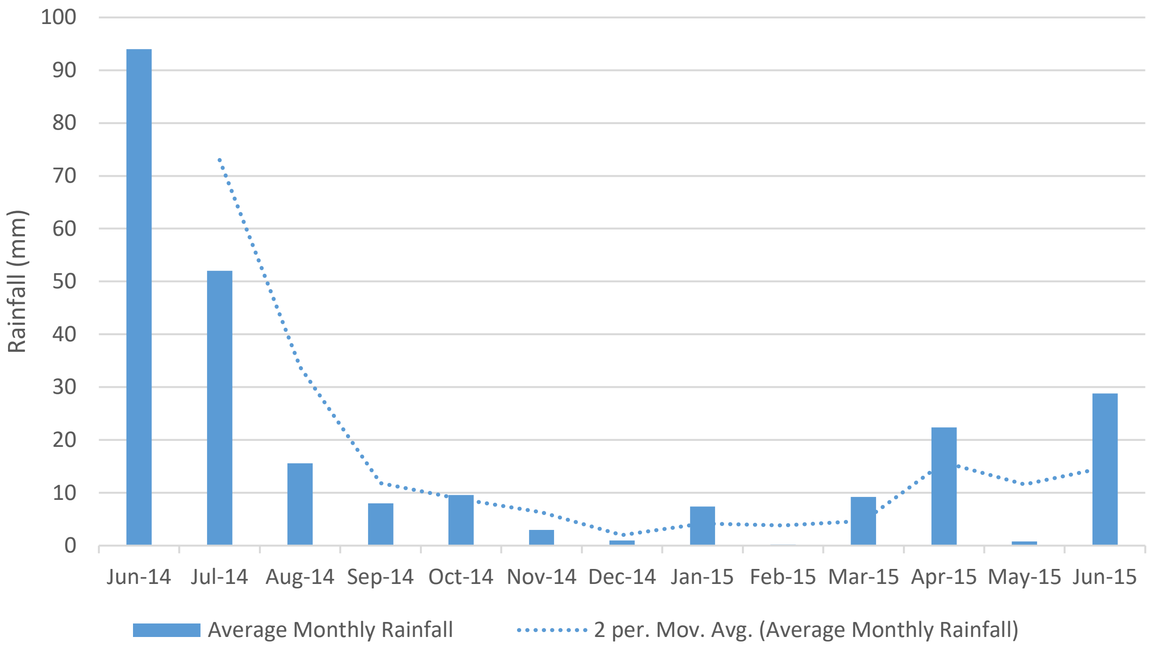
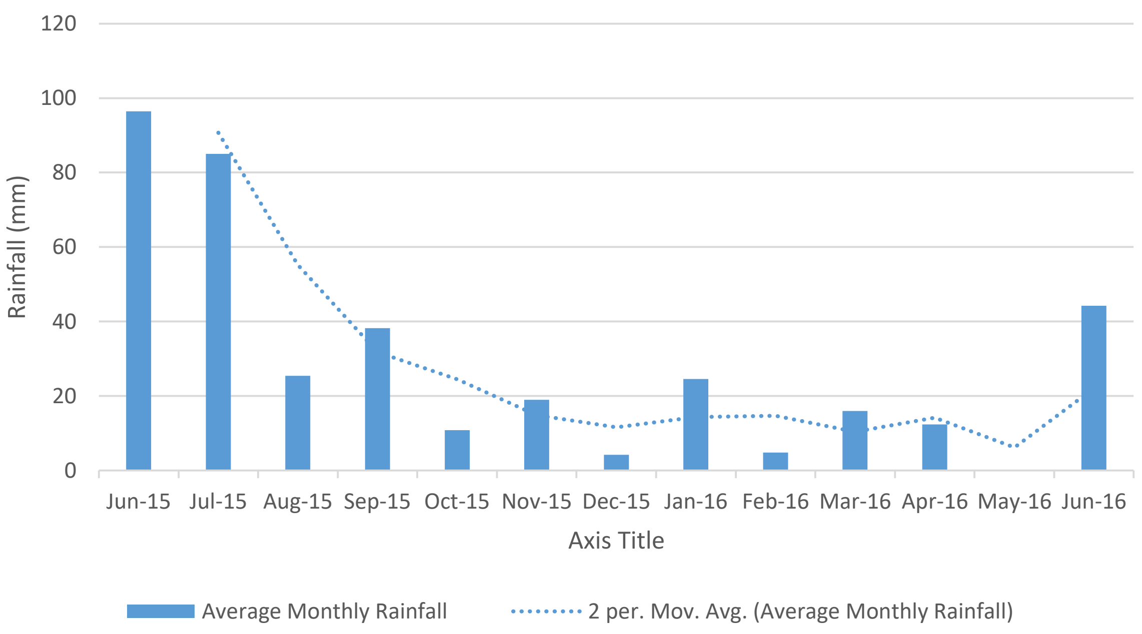
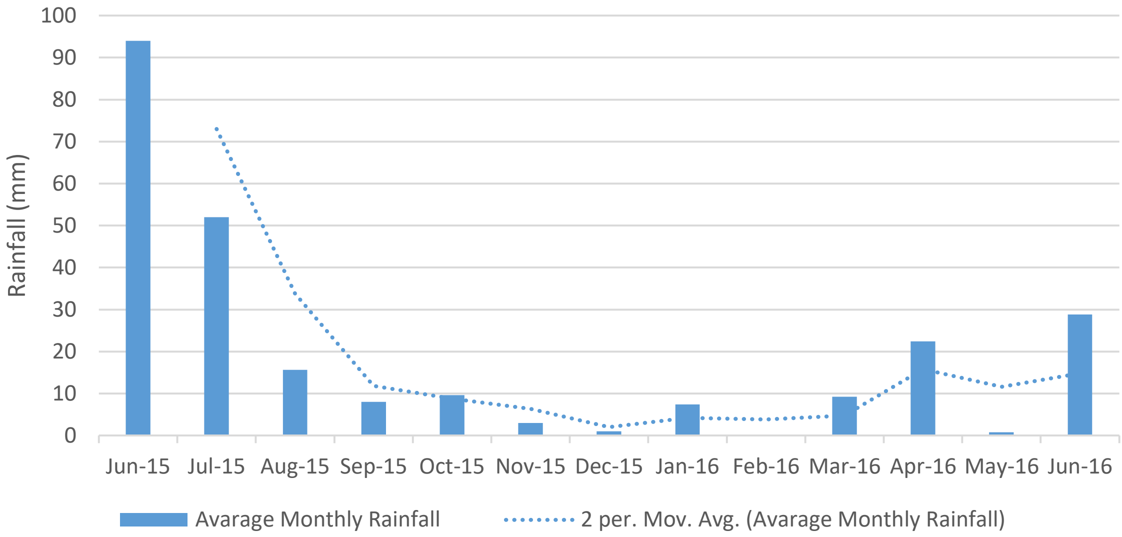
APPENDICES



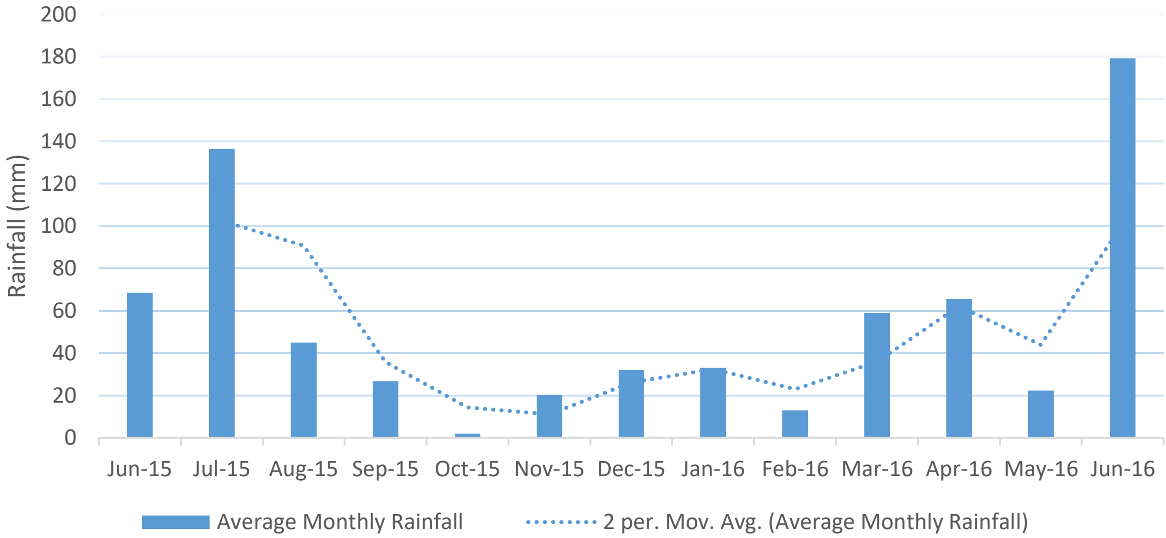
Appendix A1: Rainfall data for the Hermon study area (Jun 14 - Jun 15; credit: WeatherSA)



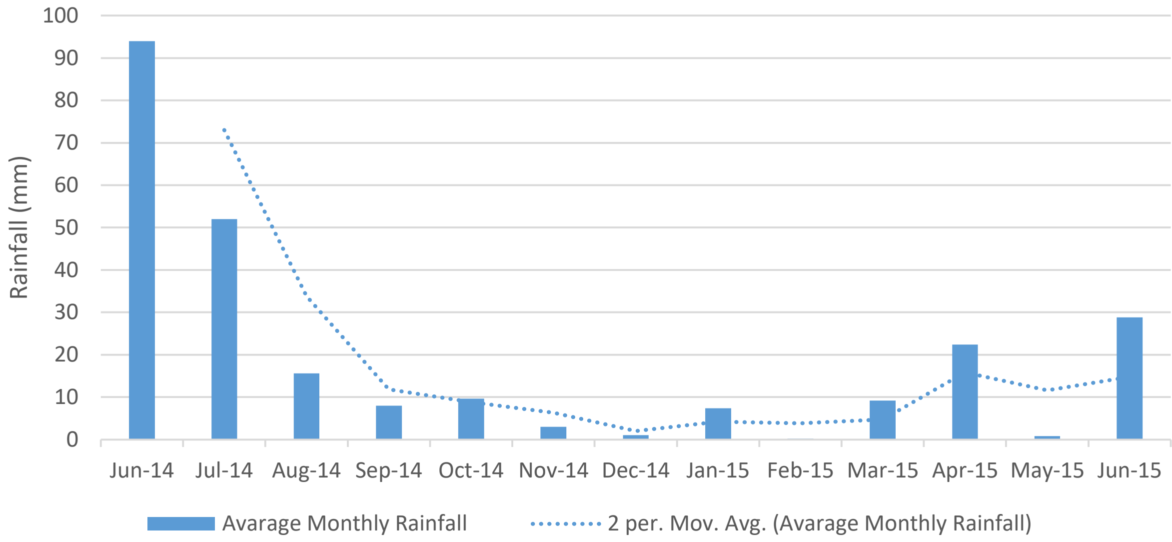
Appendix A2: Rainfall data for Robertson study area (Jun 15 - Jun 16; credit: WeatherSA)



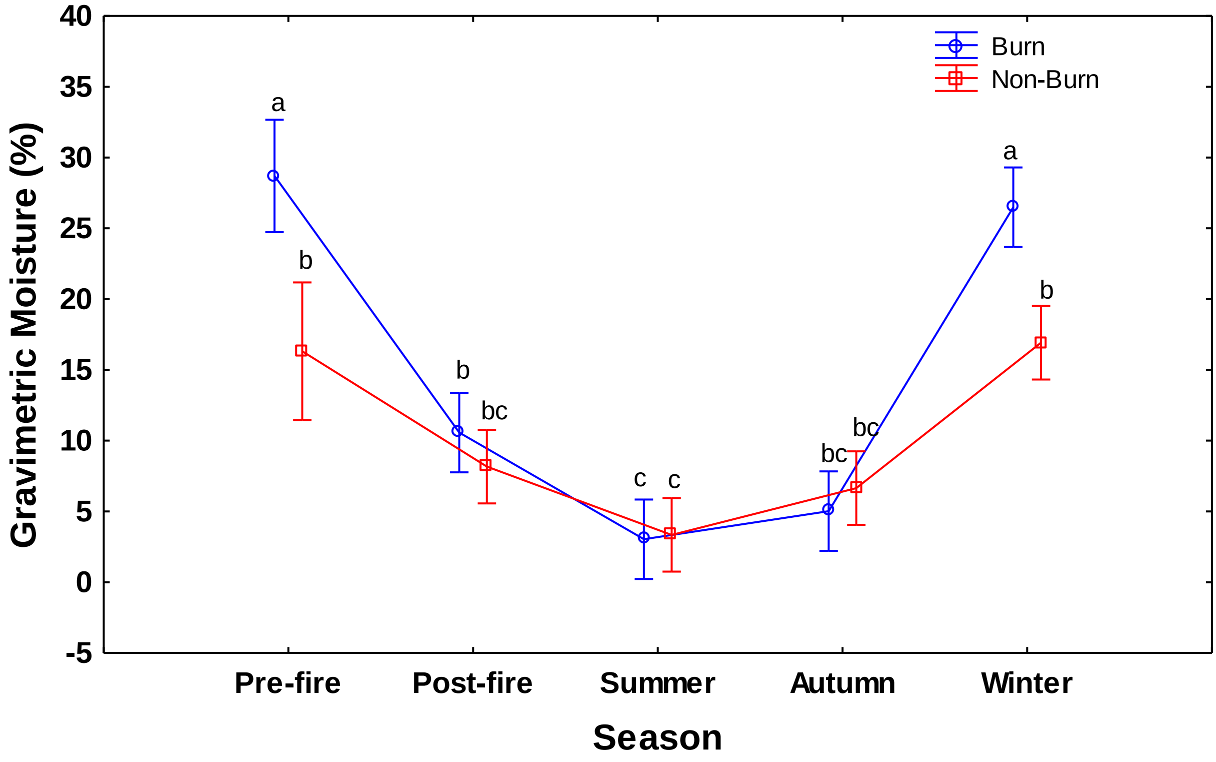
Appendix A3: Rainfall data for the Rawsonville study area (Jun 15 - Jun 16; credit: WeatherSA)



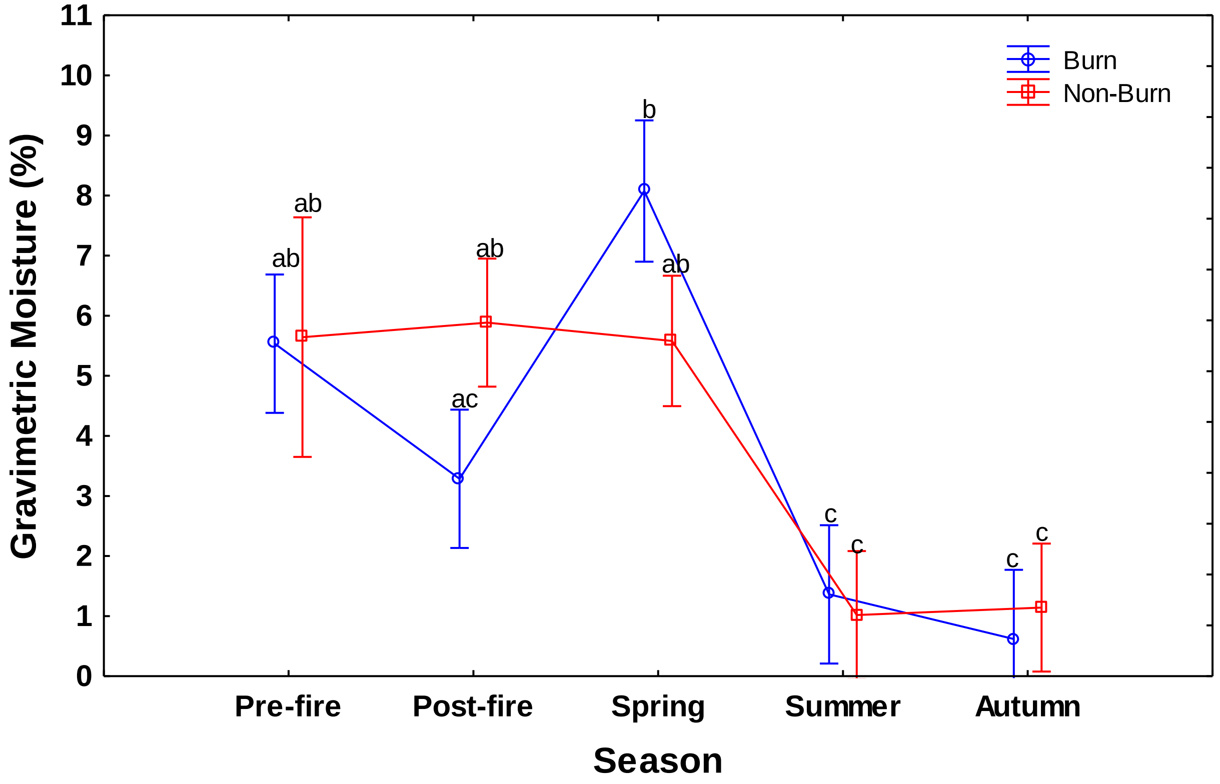
Appendix A4: Rainfall data for the Wit River study area (Jun 15 -Jun 16; credit: WeatherSA)



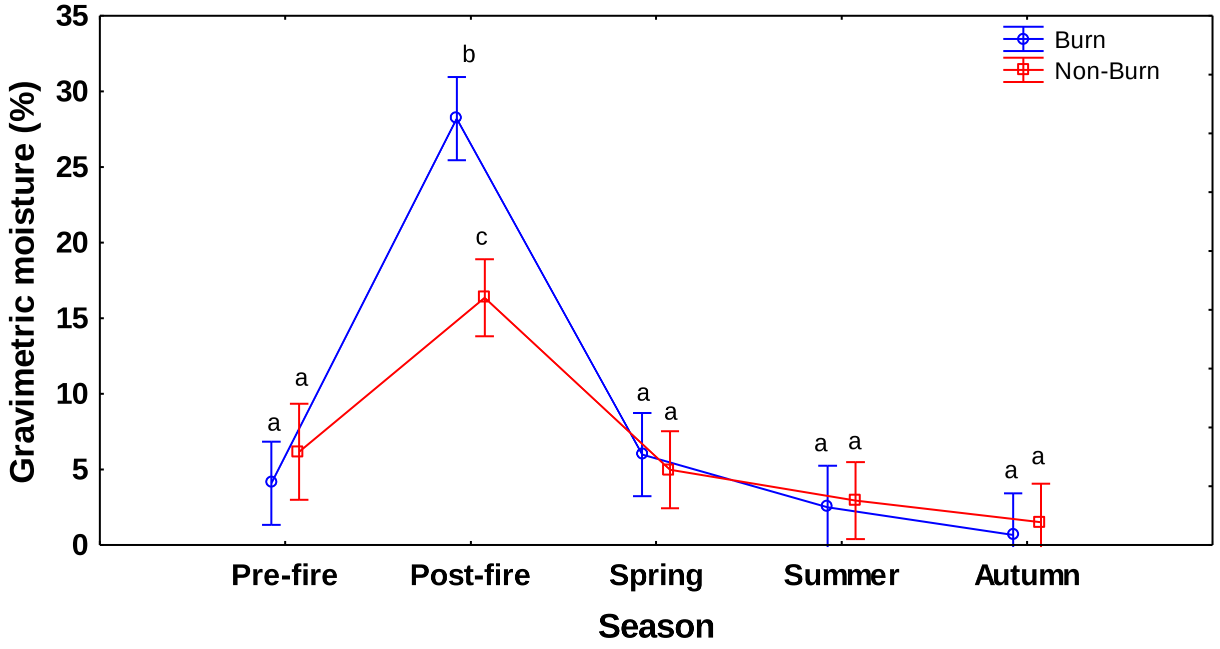
Appendix A5: Rainfall data for the Blaauwberg study area (Jun 14 -Jun 15; credit: WeatherSA)



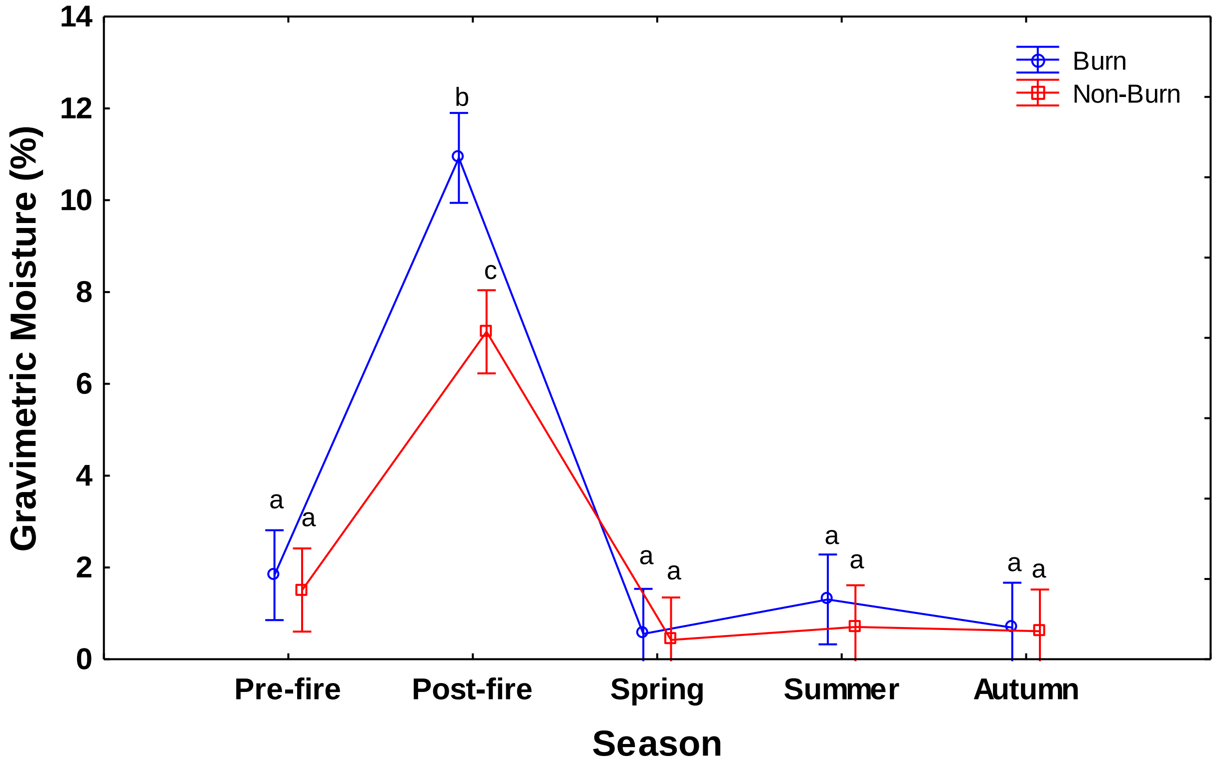
Appendix A6: Seasonal gravimetric soil water content within the burn and the non-burn treatment sites at the Hermon study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). The seasonal means are interaction effects based on two-way ANOVA: treatment X season ($F_{[4, 218]}=7.17$, $p<0.01$).



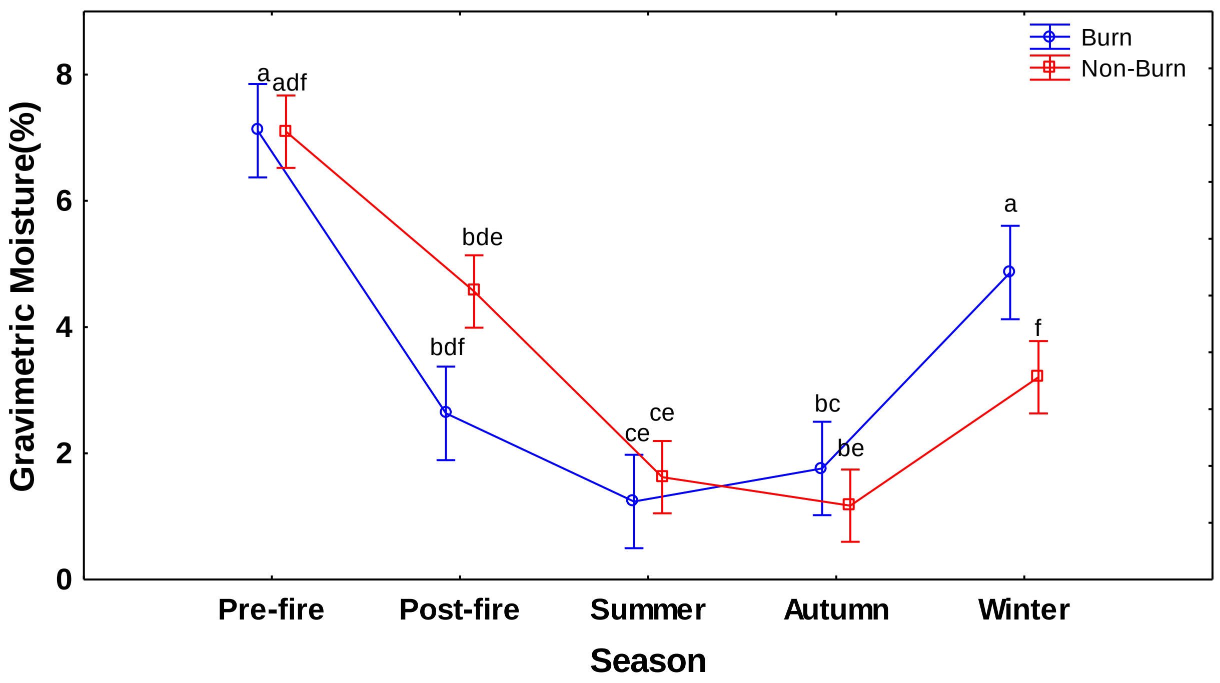
Appendix A7: Seasonal gravimetric soil water content within the burn and the non-burn treatment sites at the Robertson study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). The seasonal means are interaction effects based on two-way ANOVA: treatment X season ($F_{[4, 228]}$ =5.17, $p<0.01$).



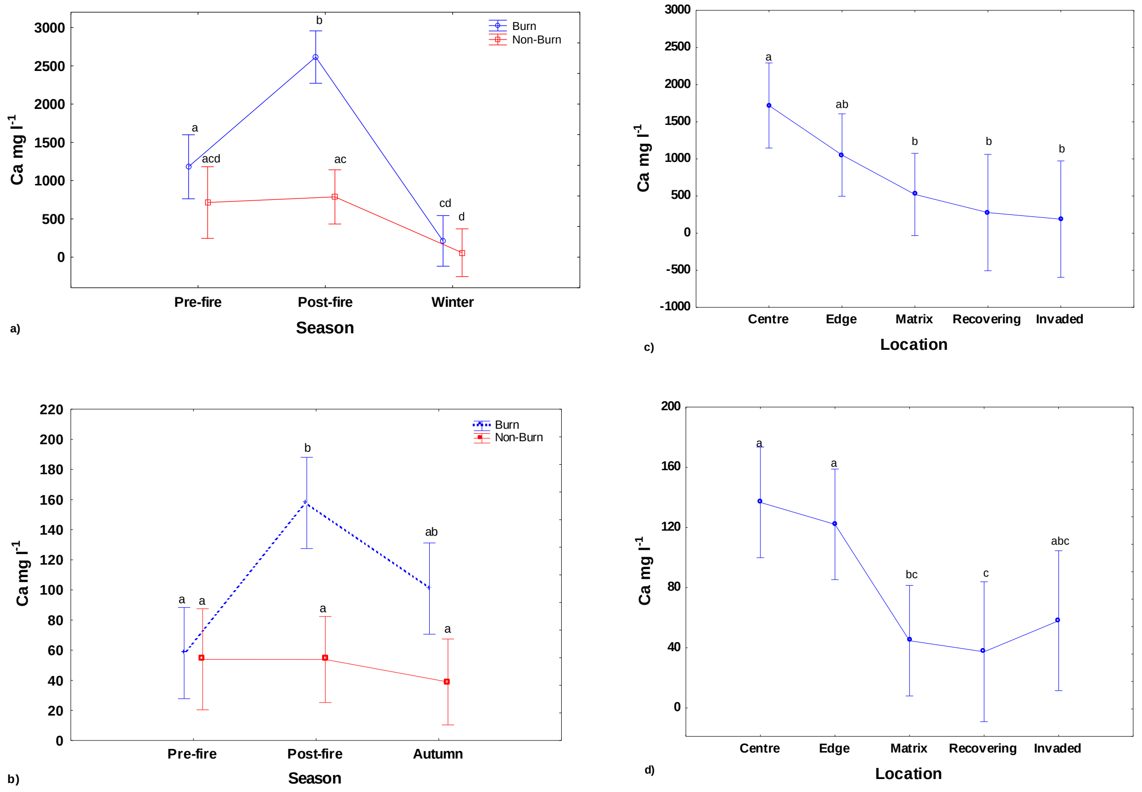
Appendix A8: Seasonal gravimetric soil water content within the burn and the non-burn treatment sites at the Rawsonville study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). The seasonal means are interaction effects based on two-way ANOVA: treatment X season ($F_{[4, 240]}$ =8.65, $p<0.01$).



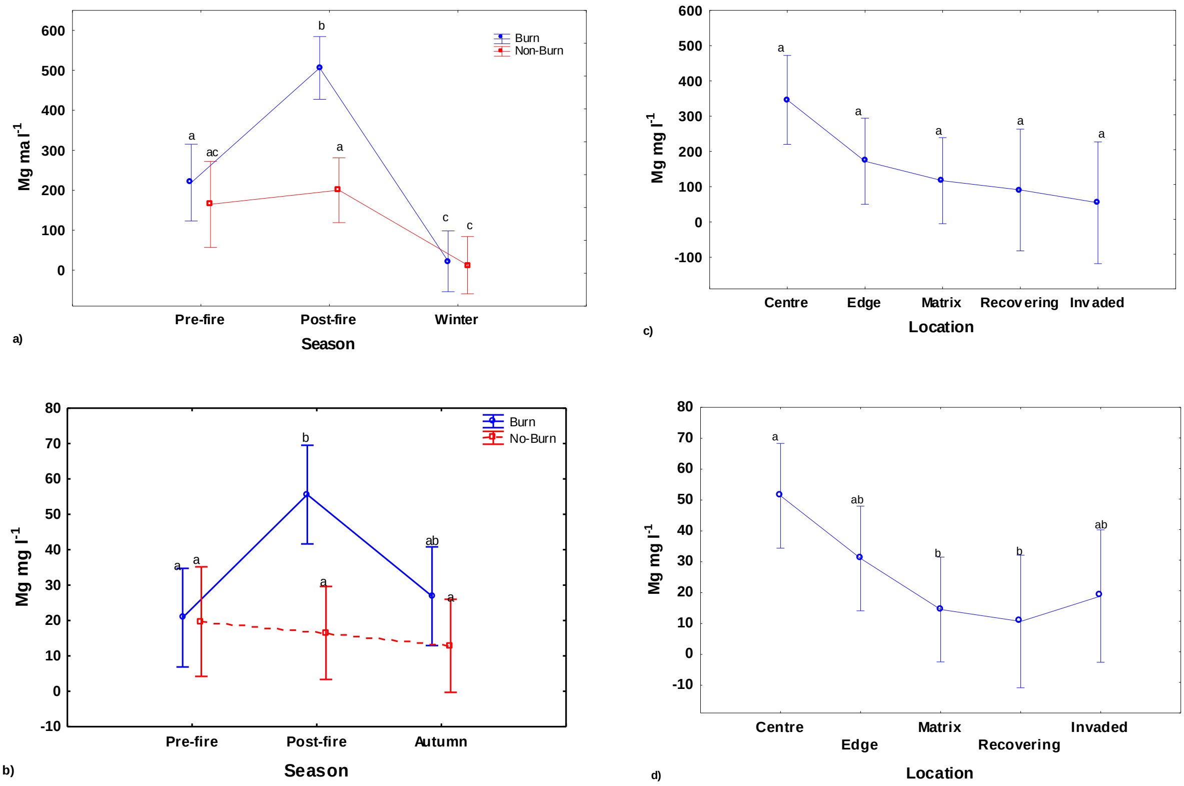
Appendix A9: Seasonal gravimetric soil water content within the burn and the non-burn treatment sites at the Wit River study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). The seasonal means are interaction effects based on two-way ANOVA: treatment X season ($F_{[4, 249]}=5.44$, $p<0.01$).



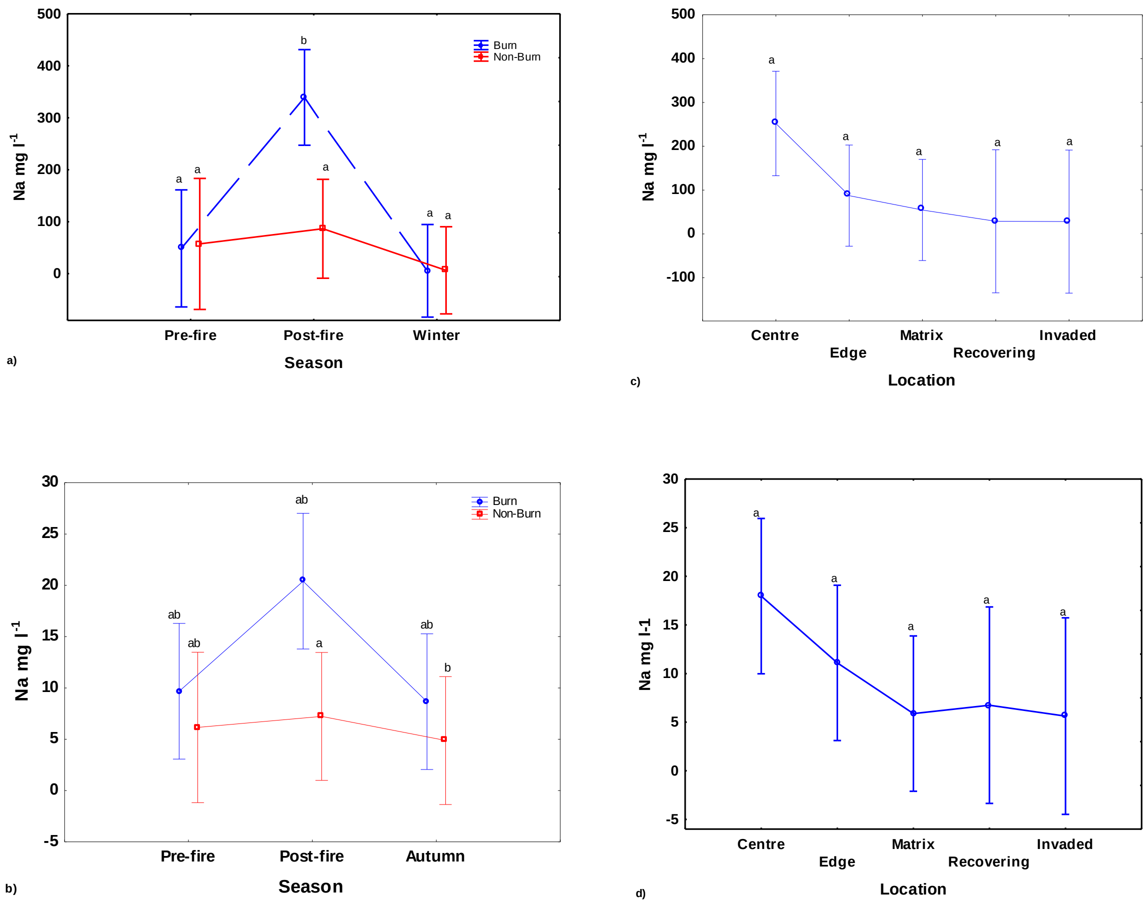
Appendix A10: Seasonal gravimetric soil water content within the burn and the non-burn treatment sites at the Blaauwberg study area. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). The seasonal means are interaction effects based on two-way ANOVA: treatment X season ($F_{[4, 190]}=7.74$, $p<0.01$).



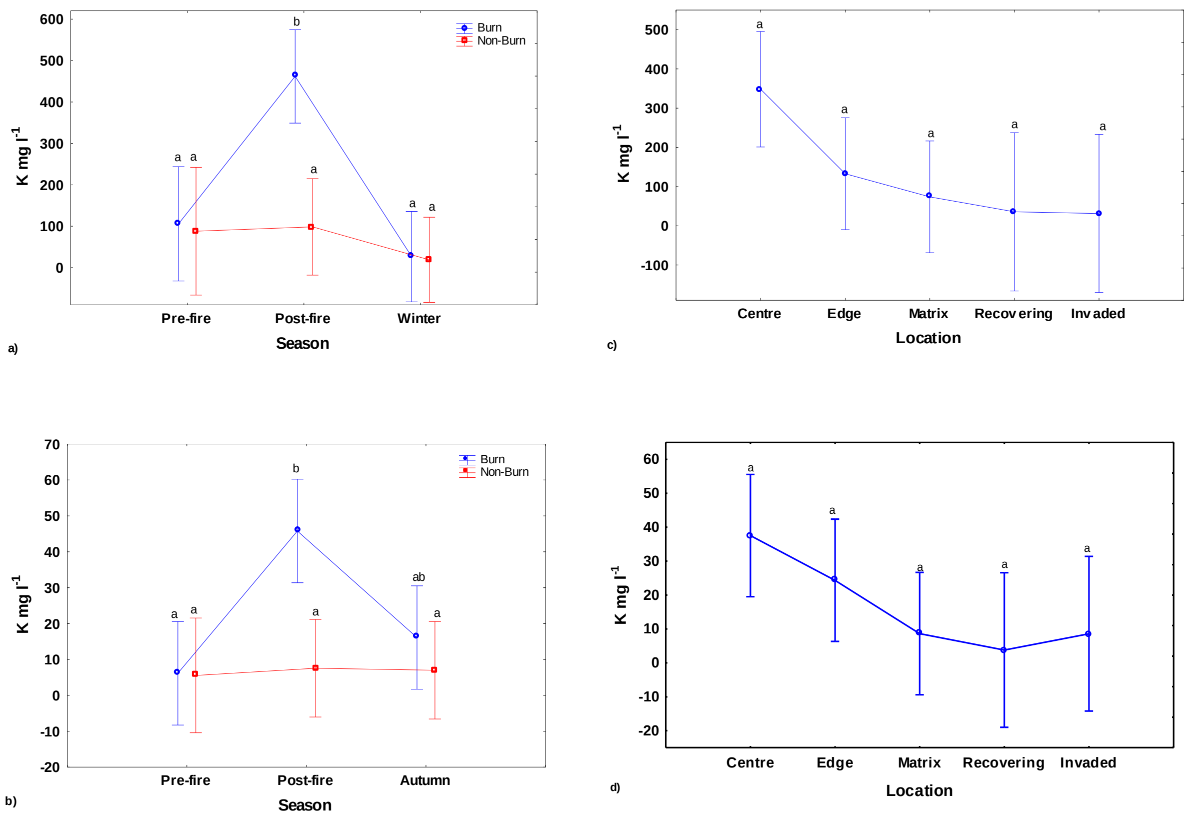
Appendix A11: Seasonal calcium concentrations within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial soil magnesium levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil calcium concentrations mans are interaction effects based on two-way ANOVA: treatment X season **(a)** ($F_{[2, 75]}=13.20$, $p<0.01$), (**b)** ($F_{[1, 91]}=5.21$, $p=0.01$). Spatial calcium concentrations means are based on a one-way ANOVA: location (**c)** ($F_{[4, 58]}=4.04$, $p=0.01$), Fig (**d)** ($F_{[4, 63]}=5.56$, $p<0.01$).



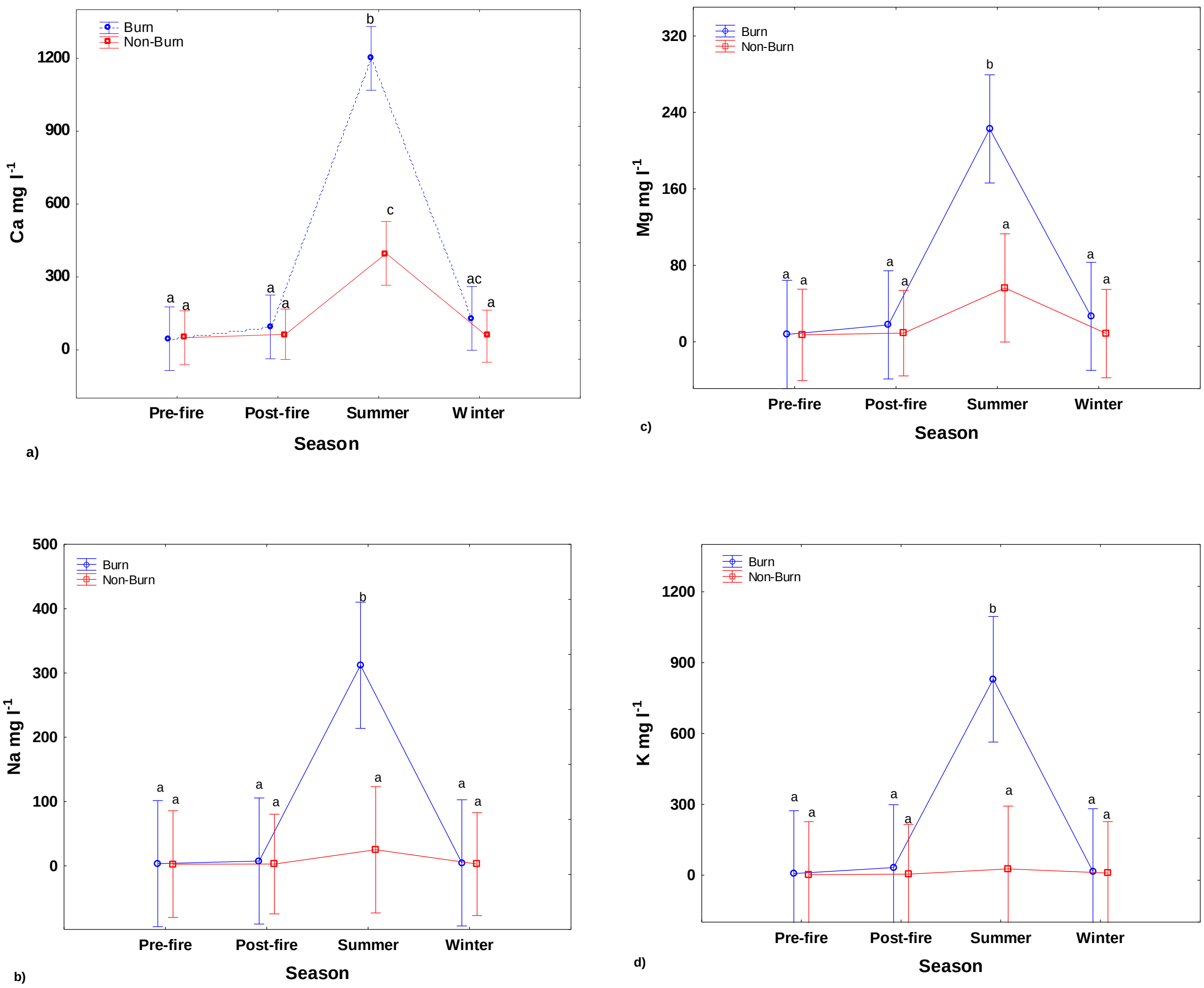
Appendix A12: Seasonal soil magnesium concentrations within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial soil magnesium levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil magnesium concentrations mans are interaction effects based on two-way ANOVA: treatment X season (**a)** ($F_{[2, 75]}$ =7.99, $p<0.01$), (**b)** ($F_{[2, 91]}$ =3.76, p=0.02). Spatial magnesium concentrations means are based on a one-way ANOVA: location (**c)** ($F_{[4, 58]}$ =2.81, p=0.03), (**d)** ($F_{[4, 63]}$ =3.37, p=0.01).



Appendix A13: Seasonal sodium concentrations within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial soil sodium levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal soil sodium concentrations mans are interaction effects based on two-way ANOVA: treatment X season **(a)** ($F_{[2, 75]}=4.77$, $p=0.01$), **(b)** ($F_{[2, 91]}=1.40$, $p=0.25$). Spatial sodium concentrations means are based on a one-way ANOVA: location F**(c)** ($F_{[4, 58]}=2.18$, $p=0.08$), **(d)** ($F_{[4, 63]}=1.57$, $p=0.19$).



Appendix A14: Seasonal potassium concentrations within the burn and the non-burn treatment sites at (a) Hermon, (b) Rawsonville study areas; and spatial potassium levels within the burn scar and non-burn treatment sampling locations at (c) Hermon, (d) Rawsonville study areas. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). All the seasonal potassium concentrations means are interaction effects based on two-way ANOVA: treatment X season **(a)** **(**$F_{[2, 75]}$ =5.60, $p<0.01$), (**b)** ($F_{[2, 91]}$ =3.72, p=0.03). Spatial potassium concentrations means are based on a one-way ANOVA: location (**c)** ($F_{[4, 58]}$ =2.78, p=0.04), (**d)** ($F_{[4, 63]}$ =2.12, p=0.08).



Appendix A15: Seasonal soil exchangeable cations concentrations within the burnt and the non-burnt treatment sites. Mean values are shown by different point symbols and vertical bars indicate ± 95% confidence intervals. Letters indicate significant differences between means (Bonferroni tests, $p<0.05$). Graphs are of interaction effects based on two-way ANOVA: treatment X season, Fig 5.**a** ($F_{[3,\ 87]}$ =17.85, $p<0.01$), Fig 5.**b** ($F_{[3,\ 87]}$ =4.33, $p=0.01$), Fig 5.**c** ($F_{[3,\ 87]}$ = 4.01, $p=0.01$) and Fig 5.**d** ($F_{[3,\ 87]}$ =4.55, $p=0.01$).

Appendix A16: (a) Seasonal hydrophobicity values of the burn and non-burn treatment sites at Hermon, (b) Spatial hydrophobicity values of the burn and non-burn sampling positions at Hermon study area. Means represent time in seconds for the Water Drop Penetration Test at each sampling location, with standard errors in brackets.

a) Seasonal

	Burn treatment		Non-Burn treatment	
Season	**Time (s)**	**Category**	**Time (s)**	**Category**
Pre-fire	5.00 (±0.00)	Non-Hydrophobic	5.00 (±0.00)	Non-Hydrophobic
Post-fire	5.00 (±0.00)	Non-Hydrophobic	5.00 (±0.00)	Non-Hydrophobic
Summer	7.38 (±2.29)	Slightly Hydrophobic	5.25 (±0.25)	Slightly Hydrophobic
Autumn	7.25 (±0.87)	Slightly Hydrophobic	6.41 (±0.85)	Slightly Hydrophobic
Winter	11.42 (±4.20)	Slightly Hydrophobic	8.39 (±3.39)	Slightly Hydrophobic

b) Spatial

Location	Treatment	Time (s)	Category
Centre	**Burn**	5.91 (±0.50)	Slightly Hydrophobic
Intermediate	**Burn**	9.81 (±3.06)	Slightly Hydrophobic
Edge	**Burn**	7.56 (±1.94)	Slightly Hydrophobic
Matrix	**Non-burn**	9.19 (±3.03)	Slightly Hydrophobic
Recovering	**Non-burn**	5.00 (±0.00)	Non-hydrophobic
Invaded	**Non-burn**	5.00 (±0.00)	Non-hydrophobic

Appendix A17: (a) Seasonal hydrophobicity values between the burn and non-burn treatment sites at Robertson study area, (b) Spatial hydrophobicity values between the burn and non-burn sampling positions at Robertson study area. Means represent time in seconds for the Water Drop Penetration Test at each sampling location, with standard errors in brackets.

a) Seasonal

	Burn treatment		Non-Burn treatment	
Season	**Time (s)**	**Category**	**Time (s)**	**Category**
Pre-fire	259.33 (±102.53)	Strongly Hydrophobic	480.00 (±289.06)	Strongly Hydrophobic
Post-fire	269.54 (±96.10)	Strongly Hydrophobic	184.46 (±89.59)	Strongly Hydrophobic
Spring	36.79 (±31.02)	Slightly Hydrophobic	77.68 (±63.94)	Strongly Hydrophobic
Summer	111.63 (±75.80)	Strongly Hydrophobic	172.89 (±91.16)	Strongly Hydrophobic
Autumn	280.83 (±110.03)	Strongly Hydrophobic	177.64 (±80.65)	Strongly Hydrophobic

b) Spatial

Location	Treatment	Time (s)	Category
Centre	**Burn**	104.06 (±53.74)	Strongly Hydrophobic
Intermediate	**Burn**	130.84 (±60.32)	Strongly Hydrophobic
Edge	**Burn**	289.19 (±97.38)	Strongly Hydrophobic
Matrix	**Non-Burn**	2489.44 (±91.28)	Strongly Hydrophobic
Recovering	**Non-Burn**	19.38 (±5.61)	Slightly Hydrophobic
Invaded	**Non-Burn**	210.75 (±84.43)	Strongly Hydrophobic

Appendix A18: (a) Seasonal hydrophobicity values between the burn and non-burn treatment sites at Wit River study area, (b) Spatial hydrophobicity values between the burn and non-burn sampling positions at Wit River study area. Means represent time in seconds for the Water Drop Penetration Test at each sampling location, with standard errors in brackets.

a) Seasonal

	Burn treatment		**Non-Burn treatment**	
Season	Time (s)	Category	Time (s)	Category
Pre-fire	249.42(±85.92)	Strongly Hydrophobic	527.89(±121.47)	Strongly Hydrophobic
Post-fire	206.04(±96.38)	Strongly Hydrophobic	86.29(±56.24)	Strongly Hydrophobic
Spring	668.63(±150.59)	Severely Hydrophobic	480.85(±133.76)	Strongly Hydrophobic
Summer	686.71(±148.11)	Severely Hydrophobic	667.00(±130.91)	Severely Hydrophobic
Autumn	1193.37(±139.84)	Severely Hydrophobic	957.77(±156.63)	Severely Hydrophobic

b) spatial

Location	**Treatment**	**Time (s)**	**Category**
Centre	**Burn**	596.53 (±124.75)	Strongly Hydrophobic
Intermediate	**Burn**	819.50 (±144.34)	Severely Hydrophobic
Edge	**Burn**	664.97 (±127.16)	Strongly Hydrophobic
Matrix	**Non-burn**	575.22 (±129.06)	Strongly Hydrophobic
Recovering	**Non-burn**	932.15 (±123.28)	Severely Hydrophobic
Invaded	**Non-burn**	134.68 (±55.54)	Strongly Hydrophobic