

Quantifying the impacts of invasive N₂-fixing plants on relative sediment loads and nutrient concentrations in mountain streams of the Cape Floristic Region, South Africa

By

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Declaration

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Summary

Riparian zones of headwater rivers in mountain catchments of the Cape Floristic Region (CFR) have been severely transformed and degraded by the invasion of woody trees species. Woody invading alien plants (IAPs), particularly N₂-fixing invasive *Acacia* spp. (*Acacia saligna*, *A. longifolia* and *A. mearnsii*) that are widespread along these rivers, rapidly suppress and replace indigenous fynbos vegetation, reduce stream flow and alter underlying soil carbon (C) and nitrogen (N) cycles. However, the in-stream effects of invasion on sediment and nutrient loading remain unquantified. The impacts of IAPs on water resources and biodiversity have justified extensive IAP clearing initiatives such as the Working for Water (WfW) programme. This program has focussed mainly on the clearing of woody invasive plants with the aim to conserve biodiversity and increase water production. The clearing of IAPs often results in further disturbances and areas are often left to recover without intervention.

The aim of this study was to quantify relative sediment loads and nutrient concentrations in mountain streams associated with natural, *Acacia* invaded and *Acacia* cleared riparian zones. Fieldwork was performed in a combination of mountain stream and mountain stream transitional zones of two perennial river systems within the CFR. Common time-integrating sediment sampling methods were used to monitor temporal and spatial relative sediment loads at three locations on the respective river systems. Each river consisted of one near pristine site (fynbos), one invaded site (predominantly *A. mearnsii*); and one cleared site (cleared from *A. mearnsii* more than 10 years ago). The suspended sediment (SS) samplers were emptied monthly for a period of one year. Laboratory analyses focused on a selection of fine sediment (< 63 µm) properties consisting mainly of particle size composition, total carbon, total nitrogen and total phosphorus content. This was combined with seasonal water samples and instantaneous river stage measurements.

The relative sediment loads were found not to be entirely dependent on catchment contributing area, but also on geomorphic characteristics (i.e. channel morphology, slope and topography) and invasion statuses. Relative sediment loads decrease with an increase in contributing area on the Du Toit's River, resulting in an inverse relationship. It was inferred that the dominant sediment source on the Du Toit's River is from channel bank erosion and sediment deposits on the riverbed. Invasion and clearing appeared to have no effect on sediment loads at the Du Toit's River. This suggests that the channel and riparian zone characteristics might reduce immediate sediment connectivity to the channel, thus limiting the effect of invasion and clearing on relative sediment loads. In contrast, the magnitude of increase in relative sediment loads on the Wit River scales well with catchment contributing

area, which is indicative of a system where sediment supply is predominantly derived from a combination of in-channel and catchment sources. Thus, the elevated relative sediment load in the invaded and cleared sites could reflect enhanced input of sediment from the *A. mearnsii* invaded and cleared riparian ecotones. Although relative sediment loads were reduced to levels similar to natural fynbos sites, relative sediment loads at the cleared site on the Wit River remained around two times higher after more than 10 years of clearing. The analysis of temporal and spatial variation showed that nutrient concentrations differed significantly amongst invasion statuses and across seasons. Mean total nitrogen content in infiltrated sediments of the invaded site on the Wit River were up to 30% and total phosphorus up to 43% higher than in the fynbos site. These results support the hypothesis that *A. mearnsii* invasions and clearing may have a significant influence on sediment-associated nutrients in streams of the CFR. Nutrient concentrations in sediments were significantly lower during the wet winter months compared to warm summer months. The temporal results suggest that peak litterfall rates, relative suspended sediment loads and hydrometeorological factors (i.e. discharge, stream temperature and rainfall) are important drivers of temporal dynamics. No significant spatial trends were noted in the concentrations of nutrient concentrations in surface water with regards to invasion statuses. While clearing may have reduced nutrient concentration to levels that resemble a natural state, P levels remained significantly higher than in the fynbos site on the Wit River and are suggestive of a long-lasting effect after the removal of invasive *A. mearnsii*. The significant relationship between iron and phosphorus concentrations in these systems suggests that P might accumulate in these systems and is biologically unavailable. Furthermore, the impacts of invasion and clearing on sediment and nutrient loads seemed to be strongly dependent on the level of channel-riparian connectivity and influenced by channel morphology.

This study contributes to the knowledge of sediment dynamics and in-stream biogeochemistry of fynbos mountain streams and the extent to which these processes may be altered by *A. mearnsii* infestations and clearing activities. In addition, improved understanding of how different catchments and channels route sediments and associated nutrients can provide valuable insights into the use of site characteristics as a management tool to mitigate and reduce the downstream export of sediment and nutrients.

Samevatting

Oewersones van hoofwater riviere in bergopvangs gebiede van die Kaapse Floristiese Streek (KFS) is hewig getransformeer en degradeer deur houtagtige uitheemse indringer plante (UIPs). Houtagtige UIPs, veral stikstof fikserende Australiese *Acacia* spp. (*Acacia mearnsii*, *A. longifolia* en *A. saligna*) wat wydverspreid langs hierdie riviere voorkom, onderdruk en vervang inheemse fynbos plante, verminder stroomvloei en verander onderliggende grond se koolstof en stikstof siklusse. Die in-stroom effekte van die UIPs op sediment -en voedingstof ladings bly egter onbeslis. Die impak van die UIPs op water hulpbronne en biodiversiteit het die implimentering van grootskaalse opruimings inisiatiewe soos die Werk vir Water (WfW) program regverdig. Hierdie program is hoofsaaklik gefokus op die verwydering van houtagtige UIPs met die doel om biodiversiteit te bewaar en water produksie te verhoog. Die verwydering van UIPs vanuit oewersones lei dikwels tot verdure versteurings en die gebiede word gereeld oorgelaat om op 'n natuurlik wyse te herstel.

Die doel van hierdie studie was om relatiewe sediment ladings en voedingstof konsentrasies in bergstrome wat geassosieer word met inheemse fynbos, uitheemse *Acacia mearnsii* en *Acacia*-verwyderde oewerwalsones te kwantifiseer. Veldwerk was uitgevoer in 'n kombinasie van bergstroom en bergstroom-oorgangsones van twee standhoudende rivier stelsels binne in die KFS. Gewilde tyd-geïntegreerde sediment monsterneming metodes was gebruik om tydelike en ruimtelike relatiewe sediment ladings op drie plekke op die onderskeie rivier stelsels te monitor. Elk van die geselekteerde rivier stelsels het bestaan uit; een inheemse fynbos, een uitheemse (hoofsaaklik *A. mearnsii*) en een skoongemaakte (skoongemaak van *A. mearnsii* meer as 10 jaar gelede) oewerwal-areas. Die sediment monsternemers is maandeliks vir 'n tydperk van een jaar geneem. Laboratorium ontledings was gefokus op 'n seleksie van fyn sediment ($< 63 \mu\text{m}$) eienskappe wat hoofsaaklik bestaan uit deeltjie grootte samestelling, totale koolstof, totale stikstof en totalefosfor inhoud. Dit was gekombineer met seisoenale rivier water monsters -en stadium metings.

Dit is gevind dat relatiewe sediment ladings nie net afhangklik was van die bydraende opvanggebied nie, maar ook van geomorfologiese eienskappe (bv. kanaal morfologie, helling en topografie) en indringer status. Relatiewe sediment ladings verminder met 'n toename in bydraende opvanggebied op die Du Toit's Rivier, wat 'n omgekeerde verhouding tot gevolg het. Daar is afgelei dat die oorheersende sediment bron op die Du Toit's Rivier bestaan hoofsaaklik uit bank erosie en sediment deposisie in die rivier kanaal. Indringer -en skoongemaakte oewersones het geen effek gehad op sediment ladings in die Du Toit's Rivier nie. Dit dui daarop dat die geomorfologiese eienskappe van die kanaal en rivier oewersones

die onmiddellike kanaal-oewerwal konnektiwiteit verminder, en die effek van UIPs en verwydering daarvan op relatiewe sediment ladings beperk. In teenstelling hiermee was die toename in relatiewe sediment vragte in die Wit Rivier goed korreleer met die groote van die bydraende opvanggebied. Dit dui op 'n stelsel waar sediment lewering afgelei word uit 'n kombinasie van rivier oewersones en sediment deposisie in die rivier. Dus, die verhoogde relatiewe sediment lading in die uitheemse en skoongemaakte areas veroorsaak was deur verhoogde inset van sediment uit die uitheemse *A. mearnsii* en skoongemaakte oewersones. Alhoewel relatiewe sediment ladings in die skoongemaakte areas minder was as in die uitheemse *A. mearnsii* area in die Wit Rivier, was dit steeds ongeveer twee keer hoër as in die fynbos area na meer as 10 jaar van opruiming. Die ontleding van temporale en ruimtelike variasie het getoon dat voedingstof konsentrasies beduidend verskillend was tussen indringer status en oor seisoene. Die gemiddelde totale stikstof inhoud in sediment van die uitheemse *A. mearnsii* area op die Wit Rivier was tot 30% en die totale fosfor tot 43% hoër as in die fynbos-area. Hierdie resultate ondersteun die hipotese dat uitheemse *A. mearnsii* plante en verwydering daarvan 'n beduidende invloed op sediment verwante voedingstowwe in fynbos bergstrome van die KFS kan hê. Ten opsigte van die tydelike patrone, was dit duidelik dat voedingstof konsentrasies laag was gedurende die nat winters maande met beduidende hoër konsentrasies in die warm somer maande. Hierdie patroon dui daarop dat blaarval syfers, relatiewe sediment ladings en hidrometrologiese faktore (bv. Rivier vloeï volume, stroom temperatuur en reënval) 'n belangrike rol speel in die temporale dinamika. Geen beduidende tendense is opgemerk in die voedingstof konsentrasies in die rivier water ten opsigte van indringer status nie. Terwyl die konsentrasies van voedingstowwe in die skoongemaakte area merkwaardig laag was, bly fosfor vlakke aansienlik hoër as in die fynbos-area op die Wit Rivier en dui dit op 'n lang-durige effek wat voortduur na die verwydering van uitheemse *A. mearnsii* plante. Die beduidende korrelasie tussen yster en fosfor in hierdie stelsels dui daarop dat fosfor biologies onbeskikbaar is en opbou in hierdie stelsels. Verder is die impak van UIPs en die verwydering daarvan op sediment- en voedingstof ladings sterk afhanklik van die kanaal-oewer konnektiwiteit en word hoofsaaklik beïnvloed deur kanaal morfologie.

Hierdie studie dra by tot die kennis van sediment dinamika en in-stroomse biogeochemiese prosesse van fynbos bergstrome en die mate waartoe hierdie prosesse beïnvloed word deur uitheemse *A. mearnsii* plante en opruimings aktiwiteite. Daarbenewens sal 'n beter begrip van hoe verskillende opvanggebiede en kanale sediment en geassosieerde voedingstowwe vervoer, sal waarde volle insigte lewer oor die gebruik van opvangsgebied – en kanaal eienskappe as 'n bestuurs instrument om die vervoer van sediment en voedingstowwe na riviere te verminder.

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Chapter 1

Introduction and literature review

1.1 General introduction

Rivers in South Africa account for less than 0.4% of the country's total area (Moran et al. 2013) and are heavily degraded, as is the case with most rivers worldwide (Nel et al. 2007). The factors leading to degradation of rivers include water diversion and extraction, high nutrient loads and salinisation (DWA, 2011; Chamier et al. 2012). Already, two thirds of South Africa's streams and rivers are classified as irreversibly modified (Nel et al. 2007). Further deterioration of intact streams and rivers is likely to have detrimental impacts on the country's economy, social welfare, human health and well-being. However, the most significant challenge to South Africa's water resources is the invasion of alien woody plants (Le Maitre et al. 1996; Chamier et al. 2012). Invasive alien plants (IAPs) are amongst the most important factors leading to riparian habitat degradation and transformation (Jansson et al. 2000; Holmes et al. 2005; Moran et al. 2013).

Streams and adjacent riparian zones are closely linked (Mineau et al. 2011). Riparian ecosystems form the interface between rivers and the surrounding terrestrial matrix, supporting multiple ecosystem services that are essential for good water quality and aquatic biodiversity (Richardson et al. 2007; Vigiak et al. 2016). Riparian habitats provide essential ecosystem services, including controlling the flux of materials between terrestrial and streams, such as nutrients, organic matter, runoff, and sediments (Decamps, 1993; Brown et al. 2004), regulating stream temperature and light availability by shading (Hill and Dimick, 2002; Sabater and Bernal, 2011) and stream banks stabilization (McGlynn and Seibert, 2003; Naiman et al. 2005). The River Continuum Concept developed by Vanotte et al. (1980), suggests the longitudinal changes in stream physical, biogeochemical and biological variables along a stream network are a result of changes in physical and biotic components of riparian ecosystems. However, the interaction between these longitudinal patterns and changes in native riparian ecosystems structure and function along the streams remain poorly understood, particularly in the fynbos biome.

Alien invasive woody plants, especially those belonging to the genera *Acacia* (Richardson et al. 1997; Brown et al. 2004) take up more water than the indigenous fynbos species (Le Maitre et al. 2002; van Wilgen et al. 2012). These woody invasive alien species have specialised mechanisms to obtain nutrients in nutrient poor environments, a characteristic of many plant species in the fynbos biome (Stock et al. 1995; Yelenik et al.

2004). IAPs are also known to take advantage of disturbance events (e.g. floods and fires) and establish themselves faster than indigenous fynbos species (Morris et al. 2011). Together these attributes have led to the successful invasion by *Acacia* spp. in the Western Cape province of South Africa. The province has high cover of *Acacia* spp., especially *A. mearnsii*, which has subsequently increased pressure on aquatic ecosystems due to reductions in stream flow and groundwater recharge (Enright, 2000; Görgens and van Wilgen, 2004; Jovanovic et al. 2009).

Several *Acacias* spp. (*Acacia saligna*, *A. longifolia*, and *A. mearnsii*) are widespread and form dense, impenetrable, near monocultures that largely suppress and replace indigenous vegetation along fynbos rivers in the Western Cape (Holmes et al. 2005; Blanchard and Holmes, 2008). These species have higher growth rates and often increase the proportion of aboveground biomass (Chamier et al. 2012), which reduce ground cover and understorey vegetation (Scott et al. 1998; Beater et al. 2008; Gaertner et al. 2009). As a result, local bank and soil erosion rates increases in areas densely invaded by woody alien trees and this may affect catchment sediment yield and river geomorphology where they invade riverbanks and channels (Cowling et al. 1976; Rowntree, 1991; Holmes et al. 2005; Beater et al. 2008). Therefore, extensive invasions in riparian zones and riverbanks could lead to increased sediment loads downstream and impaired in-stream habitats due to reduced surface and bank stability (Beater et al. 2008). In addition to suppressing indigenous fynbos growth, IAPs also alter nitrogen and carbon cycles (Ehrenfeld, 2003). The invasion of woody nitrogen (N_2)-fixing plants such as *Acacia* spp. impact nutrient cycling more than herbaceous non-nitrogen fixing plants (Liao et al. 2008). Thus, invasion by N_2 -fixing acacias in nutrient poor fynbos areas can increase the nitrogen input and thus altering soil fertility. Consequently, these species can proliferate and out-compete indigenous species (Stock et al. 1995). Through this, *Acacia* spp. have been recognised as transformer species and are classified as the most damaging IAPs in the fynbos biome (Richardson et al. 2000; Brown et al. 2004; Reinecke et al. 2007).

Le Maitre et al. (1996) modelled the predicted impacts of *Acacia* spp. on water resources. These models suggested that if no management initiatives were put in place that the catchments of Cape Town could lose up to 87 million cubic meters of water per year to invasive species. The high water demand of these species is especially noticeable in water scarce regions like South Africa, with mean annual precipitation (MAP) of ± 500 mm (compared to a global average of 860mm) (Dallas and River-Moore, 2014). For these reasons, the Working for Water (WfW) Program was established in 1995 in order to control alien invasive woody species and their adverse impacts in mountain catchments and riparian zones (van Wilgen et al. 2011). This program has focussed mainly on the removal of N_2 -fixing legumes and subsequent restoration of riparian areas to restore hydrological functioning and

conserve biodiversity, while creating job opportunities for previously disadvantaged communities (van Wilgen et al. 1998; Turpie et al. 2008). It is unclear how the progressive disturbances to riparian zones and vegetation characteristics (type, cover and density) along fynbos mountain streams by IAPs will affect sediment dynamics and water quality of streams. Therefore, this research seeks to add to the existing knowledge of IAPs in Fynbos catchments by investigating the impacts on sediment availability and the potential link to the quantity of sediment and nutrient loads in streams. Understanding the suspended sediment and nutrient dynamics can improve knowledge of the catchment level disturbances from invasion, the effects of clearing within the catchment and the potential drivers and mechanisms of suspended sediment and nutrient loads in fynbos mountain streams.

1.2. Invasion of fynbos catchments by woody nitrogen fixing trees and changes in geomorphology

1.2.1 Fynbos community composition and structure

Broadleaved woody sclerophyllous species, specifically small trees and perennial shrubs (Cowling and Holmes, 1992) characterize natural Fynbos riparian zones of steep and constrained headwater rivers in mountain catchments (Reinecke et al. 2007). The well-developed understorey and ground cover vegetation consists mainly of graminoids, forbs and perennials, with woody plants occurring within the sub-canopy (Reinecke et al. 2007). The stable ground cover and fine root systems provided by the indigenous fynbos vegetation effectively bind the upper soil surface, inhibiting erosion and making fynbos riparian ecosystems well adapted to the flash floods that occur in most catchments (Macdonald and Richardson, 1986; Rowntree, 1991). Fynbos catchments have typically very low sediment yields, less than one tonne per hectare per year (Scott et al. 1998). The dense vegetation and protective ground cover is an important factor in limiting sediment mobilisation as it protects soil surfaces from erosive rainfall, increases infiltration rates, and reduces runoff through increased surface roughness, and acts as a sediment buffer (Fryirs et al. 2007a; van der Waal and Rowntree, 2014).

Terrestrial landscapes in the Fynbos biome of the Western Cape Province and in particular, riparian ecosystems have been extensively degraded and transformed by alien invasive woody plants (Richardson et al. 2007; Moran et al. 2013). The ability of woody invasive species to grow faster and taller than indigenous fynbos species enable them to reduce light penetration and alter biomass production (Holmes et al. 2000). Invasive alien plants affect the structure and diversity of native ecosystems (Richardson et al. 2007; Gaertner

et al. 2009). Studies have suggested that where invasive *A. mearnsii* forms dense stands with canopies exceeding 50% cover, ground cover and understorey vegetation are dramatically reduced (Macdonald and Richardson 1986; Richardson et al. 1989; Cowling and Gxaba, 1991; Holmes and Cowling, 1997). For example, in their study on non-native tree invasions of mountain fynbos, Richardson and Cowling (1992) report a decrease in native plant richness at a local and community level. The increase of non-native tree canopy led to a decrease of native plant cover (Beater et al. 2008). The above mentioned studies suggest that shading, which is associated with the dense overgrowth of IAPs and larger above-ground biomass is the main reason why woody IAPs, such as *Acacia mearnsii*, displace native species and inhibit protective ground cover and understorey vegetation (Holmes et al. 2000; Jasson, 2005; Gaertner et al. 2009).

The decrease in vegetation cover exposes the soil to erosive rain, decreases slope roughness and reduces sediment binding by roots (Rowntree et al. 2004; Sandercock and Hooke, 2011). This allows water and sediment to move freely between vegetated patches and into adjacent streams (van der Waal and Rowntree, 2015). Changes in vegetation cover have been shown to exert a first order control on erosion, sedimentation and deposition (Clifton, 1989; Molina et al. 2007; Eviner and Chapin III, 2003). Catchment sediment availability has long been recognised as an important variable controlling sediment loads in rivers (Walling, 1977). For example, in the KwaZulu-Natal Province, Grenfell and Ellery (2009) related the decrease in sediment concentration throughout the wet season to decreased sediment availability as a result of the increase in vegetation cover.

1.2.2 Impact of woody IAPs on sediment availability and yields in Fynbos catchments

Sediment availability in catchments is particularly sensitive to changes in land use activities and vegetation cover, and sediment loads in rivers will closely reflect both increases and decreases in soil erosion (Walling, 2008). Therefore, suspended sediment load of rivers is often used as a measure of catchment level disturbance caused by external (e.g. climate change) or internal (e.g. change in vegetation cover or soil erodibility) factors (Foster et al. 2012). The suspended sediment load that is transported by rivers depends primarily on two factors: 1) the availability of sediment eroded from the in-channel and catchment sources 2) the ability of the river to transport this sediment (Jonker and Shand, 2010). Therefore, rivers are characterized as either sediment supply limited or sediment transport limited (Jonker and Shand, 2010). In the case of when a river's sediment transport capacity exceeds the amount of sediment available for transport, erosion will take place, which results in scouring of the

banks or a lowering of the riverbed (degradation) (Jonker and Shand, 2010). In contrast, when the sediment transport capacity is less than required to move sediment, deposition will occur (Jonker and Shand, 2010). Rooseboom (1992) reported that sediment loads for South African rivers are generally determined by the availability of sediment rather than the transport capacity. The two main sources of suspended sediments that are present and available to rivers are 1) in-channel sources (mainly derived from streambeds and banks), and 2) catchment sources (i.e. bare soils prone to erosion) (Grimshaw and Lewin, 1980; Wood and Armitage, 1997).

1.2.2.1 Catchment sources of sediment

The rate of sediment supply to river systems from catchment sources depends mainly on catchment sediment availability and the linkage between the river channel and catchment (Warburton et al. 2003; Croke et al. 2013). The fynbos biome in the CFR comprise mainly of Peninsula sandstone that is very hard, being 95% quartz, and erodes slowly, resulting in low soil erosion rates, less than 1 t ha yr^{-1} (Scott et al. 1998). The well-established and undisturbed indigenous fynbos vegetation and ground cover is effective at binding the soil and protects it from erosive rainfall, reduce runoff and overland flow through increased surface roughness, consequently trapping sediment from uplands (Sieben, 2000; Naiman et al. 2005). The sediment load of the rivers that drains these undisturbed fynbos mountain areas is usually very low, while the carrying capacity is high (van Wyk, 1982; Scott et al. 1998; Sieben, 2000). However, landscapes in semi-arid regions are particularly susceptible to soil erosion that are driven by the variability in seasonal rainfall patterns and the occurrence of drought, which in combination change the type and density of vegetation cover (Foster et al. 2012).

The invasion of fynbos ecosystems by IAPs (e.g. *A. mearnsii*, *Eucalyptus* spp., *Pinus* spp.) leads to a change from a shrubby to a tree dominated vegetation structure. The shift in dominant native vegetation cover (Fynbos shrubs to Australian *Acacia* spp.) can lower the erosion threshold in catchments, increasing erodibility. According to Cowling et al. (1976), the absence of ground cover and the fine root systems of indigenous fynbos species could increase soil erosion rates in areas densely invaded by alien trees. In their study, Kosmas et al. (1997) used runoff plots to establish the effect of land use on runoff and concomitant soil erosion rates in the Mediterranean region of Europe. The exotic *Eucalyptus* trees limit the growth of perennial indigenous vegetation, leaving the understorey almost bare and are subject to soil erosion compared to soils beneath natural Mediterranean vegetation (shrubland). The average sediment loss measured under *Eucalyptus* plantation compared to shrubland was two times higher. The increase in erosion was associated with reduced ground

cover under the trees. Van der Waal (2009) quantified soil loss using erosion plots in three river valleys on *Acacia*-invaded, cleared and natural (control) fynbos slopes on the Kouga Mountains of the Eastern Cape province of South Africa. The average erosion rates on fynbos and cleared slopes were not significantly different; however, erosion rates were almost twice as high on *A. mearnsii* invaded slopes. Van der Waal (2009) further found that soil erosion rates had increased up to 90% on the slopes of the Witteklip River that had been cleared \pm 11 years ago compared to fynbos slopes.

However, the significance of catchment sediment sources for suspended sediment supply largely depends on the level of sediment connectivity with the river channel (Croke et al. 2013). Sediment deposition at the base slope and the adjacent floodplain prevents the movement of sediment from reaching the adjacent streams (Walling, 1983). A good example is presented in a study by Evans and Warburton (2005), which indicated that despite high rates of hillslope erosion and reduced sediment connectivity between the stream channel and sources might negate the importance of catchment sources compared to in-channel sources. The hydrological connectivity is also critical in maintaining the release of sediments from catchment and channel sources to the rivers (Croke et al. 2013). For instance, higher water levels during heavy rains in the wet season enable the stream flow to temporarily access sediment sources on riverbanks and areas proximal to the river channel (Bull, 1997) and increased overland flow connects catchment sediment sources to the river channel (Walling, 1983; Campbell, 1985).

1.2.2.2 In-channel sources of sediment

Riparian vegetation is a key site variable controlling channel shape and bank stability (Rowntree, 1991). In fynbos catchments that are relatively undisturbed, riverbanks are covered with a high diversity of dense fynbos vegetation. Invasion by woody IAPs (especially *Acacia longifolia* and *A. mearnsii*) have altered headwater rivers in recent years, out-competing indigenous vegetation and degrading ecosystem function (Galatowitch and Richardson, 2005). IAPs may alter features of riverine areas by changing channel morphology through altered sediment dynamics. The lower root-to-shoot ratio of the dominant invader, *A. mearnsii*, allows it to be easily uprooted during high flows (MacDonald and Richardson, 1986; MacDonald and Cooper, 1995). The exposed riverbanks are then subject to accelerated erosion and soil loss (MacDonald and Richardson, 1986; Rowntree, 1991; Mack and D'Antonio, 1998; Enright, 2000).

Several woody IAPs (*P. pinaster*, *A. saligna*, *A. longifolia* and *A. mearnsii*) have been shown to reduce bank stability and exacerbate channel bank erosion South African rivers

(Versfeld and van Wilgen, 1986), which ultimately leads to the alteration of the geomorphology of rivers. Rowntree (1991) investigated the potential impacts of invasive woody vegetation such as *A. mearnsii* on channel processes and form. Rowntree (1991) noted that where *A. mearnsii* had invaded the riverbanks of steep mountain reaches, the channels could become progressively wider due to lateral erosion of riverbanks compared to reaches where the banks supported natural vegetation cover. Similarly, Versfeld (1995) found that the bank erosion associated with the establishment of *A. mearnsii* along the Disa River in the Western Cape led to channel widening. Further, several field investigations have since been carried out by Esau (2005) and Pietersen (2009). Pietersen (2009) reported that channels invaded by dense *A. mearnsii* were on average 40% wider than those under indigenous fynbos cover. The observed effects of *A. mearnsii* have been mainly attributed to the loss of protective ground cover and the shallow rooting system that encourage undercutting and scouring. Although several studies have attempted to determine and identify the dominant bank erosion processes operating as a result of IAPs, no links have been made to the quantity of sediment in streams.

Channel bank erosion can be an important source of suspended sediment (Collins and Walling, 2004). Several authors estimated that in-channel sources such as bank erosion could contribute up to 90% of a catchments suspended sediment yield (Caitcheon et al. 2012; Kronvang et al. 2013; Olley et al. 2013). The contributions of channel bank erosion as dominant sediment source in river systems has been documented in Queensland, Australia (Caitcheon et al. 2012; Olley et al. 2013), Loess area of Midwest United States (Wilkin and Hebel, 1982; Simon et al. 1996), in Denmark (Kronvang et al. 1997; Kronvang et al. 2013) and River Ouse, United Kingdom (Walling et al. 1999). However, this aspect has not yet received attention in South Africa and remains largely unquantified.

Together the above-mentioned studies indicate that dense stands of alien invasive woody trees along on riverbanks may increase sediment availability and alter river geomorphology. The influence of alien invasive woody trees on sediment transfer from invaded riparian zones and sediment regimes in river systems remains unconfirmed. Thus, estimating relative sediment loads in rivers draining areas invaded by woody IAPs and land cleared of IAPs would provide a good estimate of the level of disturbance these changes in vegetation structure have on river systems. The transport of these excessive sediments to and in rivers could also affect its water quality (Chamier et al. 2012). For instance, sediments exert an important control on the transport and fate of nutrients such as phosphorus (Hygarth et al. 2005), particulate organic carbon and heavy metals (Neal et al. 1999) from catchment sources to stream channels. The capacity of sediments to bind nutrients and their protracted movement

can also lead to an increase in the residence time and thereby increasing exposure times in rivers (Hamilton, 2011).

1.3 Impacts of N₂-fixing IAPs on riparian soil biogeochemistry and catchment nutrient exports

Riparian vegetation filters and traps sediment and reduces non-point sources of pollution from agricultural pesticides and fertilizers in rangeland and cropland areas, thus improving water quality in streams (Dosskey et al. 2002; Richardson et al. 2007). Riparian vegetation also provides organic material that encourages microbial denitrification in soils and results in the release nitrogen from plant leaf litter into the environment (Schade et al. 2002). In riparian zones of undisturbed mountain streams, leaf litter from riparian vegetation is the major nutrient resource to riparian and adjacent aquatic ecosystems. Large amounts of N can be derived from nutrient rich leaf litter and root exudates, especially from N₂-fixing plants, and can significantly enhance nutrient cycling, i.e. N mineralisation and nitrification, and thus increase N in underlying soils (Helfield and Naiman, 2002). As a result, nutrient poor riparian areas have the potential to become sources and not sinks of nutrients when N₂-fixing invasive plants dominate the canopy sinks (Compton et al. 2003).

1.3.1 Impacts of N₂-fixing IAPs on fynbos riparian soil biogeochemistry

Indigenous fynbos vegetation normally occurs on and are adapted to oligotrophic soils which are nutrient poor, acidic and sandy (Witkowski and Mitchell, 1987; Cowling et al. 2004) and nitrogen cycling appears to be very slow (Stock et al. 1995). Very few fynbos plants have the ability to fix nitrogen, and the N₂-fixing plants occur for only short periods after fires (Cocks and Stock, 2001). These plants are short-lived and only contribute 0.5% (c. 24 kg N ha⁻¹) of the total nitrogen to the ecosystem (Cocks and Stock, 2001). In the fynbos biome of the CFR soils have low total N and P concentration, averaging 973 mg kg⁻¹ and 87 mg kg⁻¹ respectively (Stock and Verboom, 2012). IAPs such as *Acacia* spp. are effective N₂-fixers and can deposit large amounts of nitrogen into the soil (Richardson and Cowling, 1992; Musil, 1993). These nitrogen fixing IAPs have altered nutrient cycling processes in many parts of the fynbos biome (Ehrenfeld, 2003). Invasive alien plants tend to have higher aboveground biomass, primary production and faster growth rates compared to fynbos, and thus supply additional nutrients to nutrient poor ecosystems (van Wilgen and Richardson, 1985; Versfeld and van Wilgen, 1986; Ehrenfeld, 2003). Elevated nitrogen levels in leaf litter increase the nitrification rates,

change soil carbon, and nutrient pools in soils (Ehrenfeld, 2003). The increases in soil nitrogen content and rates of nitrogen cycling are often associated with excessive annual leaf litter inputs of the *Acacia* spp., higher foliar N (Milton 1981; Yelenik et al. 2007), and rapid rates of leaf litter decomposition and nutrient mineralisation (Milton 1981; Mitchell et al. 1986).

Research has been conducted on the changing levels of nitrogen in soils between natural fynbos, cleared and invaded. Yelenik et al. (2007) studied the effect of a N₂-fixing *A. saligna* on Sand Plain Fynbos. The *Acacia* stands increased nitrogen cycling rates within the nutrient poor environment of fynbos. Soils from *Acacia* infestations had a higher nitrogen concentration, with no difference between invaded and *Acacia* cleared sites. Litterfall was four times greater in the *Acacia* stands than in natural fynbos on similar soils. In addition, *Acacia* litter had 2.5 times higher nitrogen concentration, resulting in a 10 times greater nitrogen return to the soil. Similarly, phosphorus, calcium, magnesium and potassium were found at higher concentrations in soil under *Acacia* spp. than soil under fynbos.

In their study, Jacobs et al. (2012) studied the impacts of N₂-fixing *Acacia mearnsii* on carbon, nitrogen and phosphorus stocks and nutrient cycling in riparian areas of the Western Cape. Soil nitrogen stocks were enhanced in invaded riparian areas, and exhibited a trajectory towards a natural state after the removal of *A. mearnsii*. Their findings on available phosphorus suggest that the phosphorous cycle appears to be altered in invaded riparian soils, even though no significant differences were found between treatments. The mechanisms that underlie the effects of N₂-fixing IAPs on nitrogen cycling in soils of nutrient poor ecosystems are reasonably well understood and documented. However, the impact of N₂-fixing IAPs on phosphorus cycling mechanisms and processes is less well understood and the reported effects on soil chemistry are variable. Furthermore, no links have been made to the impacts of intact invaded stands and alien clearing projects in close proximity of water sources in South Africa. There is a significant gap in our understanding of the alterations in riparian soil biogeochemical processes associated with N₂-fixing alien invasive plants and clearing thereof, especially regarding the resulting changes in water quality (Chamier et al. 2012).

Riparian ecosystems are hotspots of nutrient cycling and can strongly influence nutrient concentrations in-stream along the stream network (Harms and Grimm, 2008; Bernal et al. 2015; Pinay et al. 2015). The alterations to biogeochemical cycles in association with invasion of N₂-fixing species may reduce the ability of riparian vegetation to efficiently absorb excess nutrients and thus increase leaching and mobilisation of nutrients into groundwater and surface water systems (Dosskey, 2002). N₂-fixing riparian trees also affect stream nitrogen concentrations and primary production (Compton et al. 2003; Mineau et al. 2011; Wiegner et al. 2013).

1.3.2 N₂-fixing IAPs and related water quality deterioration

Nutrient resources in undeveloped headwater streams are mainly derived from riparian vegetation (Vannote et al. 1980; Webster and Meyer, 1997; Polis et al. 1997). The most important pathways of the movement of terrestrial resources to streams are litterfall, particulate and dissolved organic matter, which is delivered through surface runoff and direct litterfall input (Minshall 1967; Wallace et al. 1997; Polis et al. 1997).

The presence of invasive N₂-fixing trees in riparian areas increases the biomass of an invaded area, and by excessive litterfall, add additional dissolved nutrients to local soils and streams (Ehrenfeld, 2003). The excessive production of leaf litter from invasive plants is typically higher in nitrogen and phosphorous content compared to many native plants and often results in changes in soil chemistry by either the higher rate of litter decomposition or nitrogen fixing (Witkowski, 1991; Stock et al. 1995; Ehrenfeld, 2003;). Additional nutrients in riparian soils have been identified as a key factor controlling changes in stream nitrate (NO₃⁻) concentrations and fluxes in both semi-arid Mediterranean and temperate headwater catchments (Medici et al. 2010, Ross et al. 2012, Duncan et al. 2015). Studies on the impacts of invasive nitrogen fixing trees in riparian areas on water quality have been documented in Pacific Northwest rivers in the United States (Compton et al. 2003), and in the semi-arid western United States, as shown more recently (Mineau et al. 2011) and Hawaii Island streams (Wiegner et al. 2013). However, this aspect has not been well documented in South Africa.

Compton et al. (2003) found increased (NO₃⁻) and dissolved organic nitrogen (DON) in streams invaded by *Alnus rubra* in the Pacific North Forests, while, Mineau et al. (2011) showed that stream reaches found in the intermountain west region of USA, invaded by nitrogen fixing *Elaeagnus angustifolia* had higher organic N concentrations than reference reaches. Likewise, Wiegner et al. (2013) found increased nitrate and nitrite (NO_x) of up to 600 percent in surface water of Hawaiian streams below reaches invaded by N₂-fixing *Falcataria moluccana*, with no differences between dissolved phosphorus concentrations, ammonium (NH₄⁺) and particulate nitrogen. However, the documented effects of invasive N₂-fixing trees on stream phosphorus concentrations remain largely unknown. Only one study reported an increase in total P concentrations in streams of the Pacific Northwest draining invasive *A. rubra* forest (Volk et al. 2003), while several others reported no effects on phosphorus concentrations in streams draining N₂-fixing invaded riparian areas (Mineau et al. 2011; Wiegner et al. 2013). Therefore, more investigation is required to determine the impacts of invasive N₂-fixing trees on P concentrations to understand their impacts on eutrophication

processes (Chamier et al. 2013). The modification of nutrient cycling including carbon cycling and nutrient concentration in streams may result in increased algal densities and changes in trophic status, which adversely affect human health (Dallas and Rivers-Moore, 2014).

Jovanovic et al. (2009) investigated the impact of invasive *Acacia saligna* and clearing thereof on soil N and groundwater N in the Western Cape, South Africa. The invasion of *Acacia saligna* and its removal, increased NO_x concentrations groundwater up to 12 mg l⁻¹, and was attributed to the quick release of the residual nitrogen out of the soil into groundwater. In their study on the effects of land use change on stream water quality and streamflow of the Touws River (South Africa), Petersen et al. (2017) noted that indigenous fynbos vegetation showed inverse relationship with NO_x whereas degraded Outeniqua Plateau fynbos, which was invaded by *A. mearsnii*, was significantly associated with NO_x in surface water.

The movement of sediment through aquatic systems can be protracted due to intermittent deposition and remobilisation (Hamilton, 2011). Given the key role of the transport and fate of suspended material in the transfer of nutrients in river basins, the impacts of N₂-fixing trees on nutrient concentrations in suspended material (< 0.45 µm) have not been well documented. The only known studies that have examined the effects of N₂-fixing on suspended material in streams are by Volk et al. (2003) and Wiegner et al. (2013). Volk et al. (2003) reported that the N percentage in suspended material was not significantly different between streams surrounded by *Alnus rubra* stands and those from coniferous old-growth forests. In contrast, particulate carbon concentrations were significantly lower in rivers with old-growth coniferous than in *A. rubra*. Like Volk et al. (2003), Wiegner et al. (2013) found that particulate nitrogen concentrations in Hawaiian streams were the same below and above *F. moluccana* stands. A similar pattern was found to that of particulate nitrogen with regard to particulate carbon concentrations. This was attributed to the processing of tree litterfall in the riparian zones rather than in the streams (Wiegner et al. 2013).

These results suggest that invasion by IAPs and the subsequent clearing are capable of altering soil chemistry, which affects the water quality (Ehrenfeld, 2003; Chamier et al. 2012; Jacobs et al. 2012). River systems in South Africa are already plagued by high nutrient levels, exceeding drinking water quality guidelines (De Villiers and Thiart, 2007). For example, CSIR (2010) stated that many of South Africa's freshwater resources are classified as moderately to highly eutrophic due to high average phosphorus concentrations. However, little attention has been paid to quantify the amount of nutrients that N₂-fixing alien invasive plants may be adding to streams. There remains a significant gap that needs to be addressed to improve our understanding of the impacts of N₂-fixing invasive plants in riparian zones on in-stream biogeochemical cycling. Thus, continued research is needed to determine the mechanisms

that are primarily responsible for the elevated nitrogen concentrations in streams draining riparian zones invaded by non-native N₂-fixing trees (Wiegner et al. 2013).

1.4 IAP clearing and riparian restoration

According to a South African governmental institution, the Department of Water Affairs (DWA), IAPs are causing 6.5 billion rands damage annually to South Africa's economy (De Lange and van Wilgen, 2010) and are the biggest threat to the country's biological biodiversity, water resources and ecological functioning of natural systems. Despite the huge investment, with initial budget of R25 million (Marais et al. 2004) by the national government to remove invasive and restore cleared habitats, the invasion of IAPs has only been retarded, rather than reversed (van Wilgen et al. 2012). The effectiveness of the Working for Water program is mostly measured by the hectares of IAPs removed, native vegetation recovery, gains made in terms of enhanced stream flow and poverty relief (Holmes et al. 2008; Turpie et al. 2008; van Wilgen et al. 2012). This had the result in clearing taking place with little emphasis on lessening the impacts on abiotic components of fynbos ecosystems such erosion, biogeochemistry and fluvial geomorphological processes. Recently, there has been a call that more needs to be invested in research to alleviate or improve this situation, as there is a poor understanding of the abiotic disturbances of invasion and clearing on riparian zones and rivers (Richardson et al. 2007).

1.4.1 Success and challenges

The influence of alien clearing operations on hydrology and economic returns of water appears to be beneficial (Le Maitre et al. 2002; Görgens and van Wilgen, 2004). In their study, Marais and Wannenberg (2008) estimated that the clearing of IAPs resulted in an increase of stream flow by approximately 46 million cubic meters per annum between 1997 and 2006. Clearing of invasive alien species may initiate restoration in previously invaded ecosystems, but the resulting changes to the native ecosystem functioning with regards to enhanced N availability and erosion rates may result in significant conservation challenges.

Cleared sites are often left with the expectation of recovery to the pre-invasion state without additional intervention (Galatowitsch and Richardson, 2005). In fact, clearing led to a reduced recovery of indigenous riparian species diversity, with widespread colonization of invasive alien grasses and regeneration of woody invasive aliens. Several authors have suggested that the slow recovery of indigenous species might be attributed to altered soil

nutrients cycles which can persist for extended time intervals following clearing (Brown et al. 2004; Marchante et al. 2009). However, there is still limited understanding of fynbos recovery after clearing due to the complex interaction between plants and their physical environment (Naude, 2012). Through this, large areas may be left exposed and vulnerable to soil erosion as a result of soil surfaces exposed to rainfall and increased erosive forces of increased overland flow, which leads to local soil erosion. In addition, the removal of woody IAPs in the riparian zones can also lead to the release of substantial amounts of sediments into stream, because of channel widening and soil erosion (Bruton et al. 1989; Rowntree, 1991).

In their study, Scott et al. (1998) stated that increased sediment yields should be expected following major disturbances such as flooding subsequent to clearing of vegetation. Similarly, Brown et al. (2004) suggested that the sand that has accumulated over time beneath invaded vegetation will be flushed out during the first winter storms (wet season) immediately after clearing before the protective vegetation cover has re-established. In river catchments where invasion led to transformation and degradation exceeding a threshold where ecosystems are not capable to recover naturally, active rehabilitation and restoration is required (Le Maitre et al. 2011). Planting or seeding with indigenous species is one way of altering the trajectory towards recovery (Galatowitsch and Richardson, 2005; Holmes et al. 2005; Le Maitre et al. 2011). Although re-planting of indigenous riparian vegetation could be essential to restoration and rehabilitation efforts, growth rates are often slow. Marsh et al. (2004) monitored erosion and sediment yield in streams with re-vegetated (treatment) riparian zones after clearing relative to a reference (forested) and control (pasture Kikuyu grass) sites in *Eucalyptus* forest in Australia. The mean annual suspended sediment yield for the catchments was 14.5 - 87.8 t km² yr⁻¹ for the control stream, 12.3 - 212.2 t km² yr⁻¹ for the treatment stream and 3.0 - 78.0 t km² yr⁻¹ for the reference stream. They further predicted that sediment load in the treatment streams will continue to increase until riparian vegetation is fully established and resulting channel changes are complete.

The long lasting biogeochemical changes (legacy effects) in riparian areas after the clearing of N₂-fixing alien invasive plants could introduce another major challenge (D' Antonio and Meyerson, 2002; Marchante et al. 2009). Naude (2012) found that a long lasting effect with respect to nitrogen availability in riparian soils after the removal of invasive *Acacia* spp. in the south-western Cape. This may lead to water quality impacts through surface runoff and soil erosion processes (Ehrenfeld, 2003; Chamier et al. 2012). The impact of the legacy effects of exotic species in riparian ecosystems on water quality of rivers is less well understood. Although several studies have focussed on the on-site effects of alien clearing projects, the downstream effects of these initiatives and their impacts on sediment and nutrient dynamics and ultimately on water quality remain largely unknown (Chamier et al. 2012). It is also thought

that although clearing would lead to restored nutrient cycling in soils of previously densely invaded riparian zones, some of the nitrogen will be mobilised causing a flux of nitrogen into streams (Follstad Shah et al. 2010; Jacobs et al. 2012). These gaps in our understanding, identify several important aspects for ecosystem recovery that need to be considered during restoration management. They include both biotic and abiotic aspects of restoration, such as approaches to alter existing biogeochemistry (e.g. N enrichment) and geomorphology (e.g. erosion) to facilitate recovery (Holmes et al. 2005; Richardson et al. 2007; Esler et al. 2008; Reinecke et al. 2008).

1.5 Problem statement

Invasion by alien woody trees, particularly fast growing Australian *Acacia* spp. (*Acacia mearnsii* and *A. longifolia*) have altered river systems in the Cape Floristic Region (CFR), replacing native vegetation and altering ecosystem functioning (Galatowitsch and Richardson, 2005; Blanchard and Holmes, 2008). Several field investigations reported that invasions of riparian zones by woody alien vegetation such as *A. mearnsii* have an adverse effect on bank resistance and stability (Rowntree, 1991; Versveld, 1995; Esau, 2005; Smith-Adao and Scheepers, 2007; Petersen, 2009) and soil erosion rates (Kosmos et al. 1997; van der Waal et al. 2012). The observed effects of *A. mearnsii* were mainly attributed to the lack of protective groundcover and a shallow root system that makes the banks and soil surfaces highly vulnerable erosion. The increase in riverbank and soil erosion rates could ultimately alter the input of sediment to streams.

The efforts directed at clearing them by the WfW program often result in further disturbances as cleared sites are expected to restore without active intervention (Galatowitsch and Richardson, 2005; Esler et al. 2008). The WfW program has been largely successful in eliminating dense stands of invasive alien plants in riparian zones of headwater reaches. The effectiveness of these alien clearing operations will depend on the ability of streams to remove excess sediment and the rate at which native riparian vegetation is fully established. However, native fynbos riparian plants recovery appears to be slow and exceedingly reduced after clearing, resulting in bare soil surface or regeneration of woody alien invasive plants (Galatowitsch and Richardson, 2005). The focus is on biotic aspects of recovery, with little consideration for other possible interventions targeted at the abiotic aspects such as riverbank erosion and soil loss (Richardson et al. 2007; Esler et al. 2008). The benefits of increased water availability through alien clearing could be negated by the increased delivery of sediment

downstream. Indeed, in South Africa, to date no studies have quantified the amount of sediment that is released by invasion and subsequent clearing.

Fynbos riparian soils and streams are generally oligotrophic systems. Few fynbos plants species have the ability to fix nitrogen, and most are pioneers that are short-lived, and replaced by other plants as part of succession (Cocks and Stock, 2001). Invasive *Acacia* spp. (*A. mearnsii*), by contrast are effective nitrogen fixers, altering C and N cycles through biological N₂-fixation (Witkowski, 1991) and excessive production of litter. Yelenik et al. (2004) found that *Acacia* stands increase nitrogen cycling rates within the low-nutrient environment of fynbos. Similarly, Naude (2012) and Jacobs et al. (2012) found increased nutrient stocks in fynbos riparian areas invaded by *Acacia* spp. in the Western Cape. This difference has been mainly attributed to the larger stocks of N-rich leaf litter which can enhance microbial N mineralisation and nitrification, and thus, the storage of N in the riparian surface soils (Helfield and Naiman, 2002). The additions of N to riparian soils may increase N stocks and shift the riparian zones from sinks to sources of added nitrogen to streams, which may lead to elevated N concentrations in surface water (Follstad Shah et al. 2010; Mineau et al. 2011). The presence of *A. mearnsii* in riparian zones may therefore be a source of biological available nitrogen in aquatic ecosystems (Tye and Drake, 2012). In addition, clearing of N₂-fixing invasive woody vegetation may result in a long lasting effect on N availability and may contribute to eutrophication in surface water systems (Jovanovic et al. 2009).

The impacts of invasive N₂-fixing trees in riparian zones on stream nutrient concentrations have been largely documented in streams in the Pacific Northwest of the United States (Compton et al. 2003), and drier parts of western United States (Mineau et al. 2011) and Hawaii Island streams (Wiegner et al. 2013). However, the link between increases in nutrient cycling in riparian areas due to invasive N₂-fixing woody trees and a possible nutrient flux into adjacent aquatic ecosystems has not yet been determined in South Africa. There is also limited understanding in the mechanism by which N₂-fixing riparian trees affect in-stream nutrient concentrations (Wiegner et al. 2013), therefore requiring further investigation. While much attention has been focused on terrestrial soil biogeochemistry and water quality changes in streams as a function of invasive nitrogen fixing vegetation, much less is known about the role of suspended sediment (SS) as carriers and sinks for nutrients. Understanding of these inter-related aspects is imperative for permanent solutions to the restoration of degraded riparian zones to support approaches to reduce nutrient enrichment, soil erosion and changes to geomorphology brought by invasion (Holmes et al. 2005; Richardson et al. 2007).

1.6 Rational and Motivation

Stream restoration in South Africa is focused on the removal of woody alien invasive plants and the restoration of riparian areas. However, little attention is paid to in-stream effects of riparian restoration, or indeed invasion, and it has simply not been quantified. Yet, such data will add significantly to the body of knowledge informing management decisions in order to minimize the impacts of IAPs and to aid in effective riparian zone management.

Management of fynbos riparian ecosystems is hampered by this lack of understanding of the impacts of IAPs, in particular on fluvial geomorphological properties such as sediment chemical and physical properties, soil erosion and sediment dynamics. These are crucial considerations to advance our understanding of water quality and ecosystem response to future global change, especially with increased invasion and climate change. This research seeks to aid in the development of a novel approach to quantify and demonstrate the likely impacts of N₂-fixing invasive alien plants on water quality, and the definition of plausible management approaches based on sound ecological and geomorphological understanding.

Although there is evidence that consider alien invasive plants affect the quality of water, this aspect is still not well understood in the Fynbos Biome. This research proposes to investigate the potential mechanisms responsible for the a) change in water quality in streams draining areas invaded by N₂-fixing woody alien invasive plants, b) determine the effects of invasive *A. mearnsii* and clearing on water and sediment quality, and c) to determine the impacts of invasive *A. mearnsii* and clearing on the sediment loading and transport of rivers.

The present research is also expected to contribute to improve the effectiveness the WfW programme by influencing its restoration practices aimed at restoring and increasing the ecological integrity of rivers and streams by the removal of invasive alien acacias. The results from this study would provide knowledge on clearing practices and concerns related to invasive N₂-fixing plants to aid decision makers when prioritizing catchments and areas planned for clearing programmes.

1.7 AIM, RESEARCH OBJECTIVES, KEY QUESTIONS AND HYPOTHESES

1.7.1 Aim

The aim of this study was to quantify the impacts of invasive alien *Acacia mearnsii* and clearing thereof on relative sediment loads and the nutrients dynamics (C, N and P) in two mountain streams in the Western Cape, South Africa. In order to achieve this aim, we compared the

relative sediment loads and the nutrients trapped in these sediments, draining *Acacia*-invaded, *Acacia* cleared, and near-pristine fynbos riparian areas, of two streams in different river systems of the Cape Floristic Region.

There are two components of this study:

1. Quantification of relative sediment loads in two *Acacia mearnsii* infested mountain streams of the Cape Floristic Region, South Africa
2. Quantifying the impacts of invasive N₂-fixing *Acacia mearnsii* on sediment nutrient dynamics in mountain streams of the Cape Floristic Region, South Africa

1.7.2.1 Quantification of relative sediment loads in two *Acacia mearnsii* infested mountain streams of the Cape Floristic Region, South Africa

This part of the study's objectives are:

1. To investigate the temporal and spatial variability of relative sediment loads of mountain streams in the CFR;
2. To compare the relative sediment loads to river discharge data and;
3. To quantify the relative sediment contributions from near-pristine fynbos, *Acacia*-invaded and *Acacia*-cleared sections of the rivers in the CFR

The following key questions will be addressed and hypotheses tested in this part of the study:

1. How do the different invasion statuses (near pristine fynbos, *Acacia*-invaded and *Acacia* cleared) influence relative sediment loads in streams?

Hypothesis: The cleared and invaded ecotones trap more sediment compared to near-pristine reaches.

2. What are the seasonal differences in the relative sediment loads for near pristine, *Acacia*-invaded and cleared stream reaches?

Hypothesis: Relative sediment loads will be lower during high-energy events compared to low energy events.

1.7.2.2 Quantifying the impacts of invasive N₂-fixing *Acacia mearnsii* on sediment nutrient dynamics in mountain streams of the Cape Floristic Region, South Africa

This study's objectives are:

1. To determine the temporal and spatial variation in nutrient concentrations (N, P and C) of fine sediments (< 63 µm);
2. To determine the influence of particle-size distributions (i.e. clay percentage) on sediment-associated nutrient concentrations (C, N and P) and;
3. To quantify the nutrient (C, N and P) contributions from near-pristine fynbos, *Acacia mearnsii* invaded, and *Acacia* cleared riparian sections.

The following key questions will be answered in this study:

1. How do the nutrient concentrations (C, N and P) in fine sediments (< 63 µm) differ depending on invasion status (near pristine fynbos, *Acacia*-invaded and cleared)?
Hypothesis: Nutrient concentrations in the fine sediments found in invaded and cleared reaches are higher than those found in nutrient poor, near pristine fynbos reaches.
2. How do changes in flow conditions influence nutrient concentration?
Hypothesis: Nutrient concentration will be lower during high flow conditions compared to low flow conditions as nutrients will accumulate and concentrate in the riverbed sediments.

1.8 Thesis structure

The thesis is divided into five chapters. **Chapter 1** (this chapter) introduces the work by outlining the background, with particular focus on invasive *Acacia* spp. and their impacts on sediment availability in the catchment. **Chapter 2** describes general field and laboratory methods, which were chosen to address the two main objectives of this dissertation. Furthermore, a brief outline of the study area is provided, which consists of the climate, lithology, research design, geographical location and descriptions for each river system and study site. The data of this dissertation are presented in **Chapters 3** and **4**, which address the two main objectives. **Chapter 3** aims to compare the relative sediment loads from upstream to downstream in mountain streams of the CFR and determine the potential impacts of different invasion statuses on the relative sediment loads. It should be noted that the methods used in this chapter measured suspended sediment and sediment loads respectively. **Chapter**

4 aims to elucidate the possible contribution by invasive N₂-fixing *A. mearnsii* and clearing thereof to stream nutrient dynamics (C, N and P), by focusing on the nutrient concentrations in fine sediment (< 63 µm) and surface water in streams. **Chapter 5** provides brief discussions on the major findings from chapters three and four, including some of the management implications and recommendations. The limitations encountered during the research programme and future research are presented in Chapter 5. It should be noted that the methods described in chapter two are repeated in short in the two data chapters (3 and 4) as they were written up as draft publications.

Chapter 2

Research Methodology

2.1 Research design and site selection criteria

The research design required three sites on each river, consisting of one upper stream near pristine (fynbos) (with no or low densities of woody alien invasive plants, and free of human disturbance), a mid-stream cleared riparian site (a site that was previously invaded by *Acacia mearnsii* and cleared ± 10 years ago) and invaded (invaded primarily by *A. mearnsii* for at least ten years). Due to the lack of near-pristine fynbos systems (reaches) in similar longitudinal zones of river systems of the Western Cape, and the extensive transformations and developments in most of these systems, fieldwork was carried out in a combination of mountain stream and mountain stream transitional zones of two perennial rivers within the Western Cape, South Africa. The surface water resource (WR90) quaternary catchments (sub catchments) were delineated in 1990 for South Africa's principal water management areas (Midgley et al. 1994). The sub catchments which were suitable for inclusion in the study were selected using five basic criteria. First, rivers should have the following treatment sequence from upstream to downstream: a) near pristine (fynbos), b) cleared and c) *A. mearnsii* invaded. Second, candidate catchments had to be free of land use (i.e. agricultural, industrial and residential) from the recent past in order to compare documented changes in vegetation structure composition with any estimated changes in the relative sediment and nutrients contributions derived from individual source types. Third, no significant tributaries originating in other catchments were to be entering the main stream at or near the sites used, but fitted the criteria if these entered the main stem a significant distance above or below the sites. Fourth, sites had to be reasonably similar in physical characteristics (e.g. same or similar geology, gradients and topology). Fifth, sites should be in close proximity to Stellenbosch University and accessible. The study sites and their respective characteristics are summarized in Table 2.1.

a) Near pristine fynbos sites (Reference sites)

Near pristine sites (fynbos) are often used to provide structural and background information for restoration efforts (Blanchard and Holmes, 2008). In this study, near pristine sites were selected in order to determine the stream sediment regimes and nutrient concentrations without invasion. This was to serve as a benchmark for understanding changes in community structure with invasion and after clearing. Near pristine sites were selected based on the following criteria: riparian species composition mainly indigenous to the Cape Floristic Region

(CFR) with no or less than 5% invasive species density; no impact by any anthropogenic disturbances; and on the same stream, in close proximity to the cleared or invaded sites (Table 2.1). The fynbos sites selected for this study were in different zones of their quaternary catchments, one a mountain stream and one in the mountain stream transitional zone with gradients of 0.039 - 0.115, respectively (Rowntree et al. 2000), within the boundaries of protected areas, and no previous history of invasion.

b) Cleared sites

Cleared sites had to have been previously invaded by *Acacia mearnsii* species prior to clearing. All cleared sites were cleared either between ten (2002-03) and twenty (1996-97) years ago. Cleared sites had the additional criterion that follow-up treatment should have occurred in the last 5 years. This would allow indigenous vegetation at the cleared sites sufficient time to re-establish. The cleared sites were both in mountain stream transitional reaches, with gradients ranging from 0.038 - 0.102 (Rowntree et al. 2000). Cleared sites of both quaternary catchments were within protected areas.

c) Invaded sites

The main selection criteria for the invaded sites were based on the alien invasive species composition. The dominant species present at the invaded sites were nitrogen fixing *Acacia mearnsii*. Aerial cover of *A. mearnsii* had to be fifty (50%) percent or more to be selected as an invaded site. The invaded sites were in mountain stream transitional zones of their respective quaternary catchments (Rowntree et al. 2000). The gradients ranged from 0.036 - 0.093. Both sites were in nature reserves.

2.2 Catchment and study site descriptions

2.2.1 Breede River catchment

The river systems represented fall predominantly in the Breede Water Management Area (WMA). The WMA is the southern-most WMA in Western Cape Province of South Africa (Figure 2.1a). Most of the rivers in the Breede WMA have been invaded by alien invasive plants (*A. mearnsii*, *A. saligna* and *Eucalyptus camaldulensis*) with the exception of the upper reaches of rivers that occur in protected areas (Forsyth et al. 2009). Based on the five selection criteria, the following river systems were chosen for this study: Du Toit's and Wit River. The Du Toit's River system is located in the Riviersonderend sub catchment, while the Wit River system is in the upper Breede sub-catchment (DWAF, 2004).

The Breede catchment is characterized by a Mediterranean type climate, with a winter-rainfall which occurs mostly between April and September, while the summer months are notably warm and dry (Deacon et al. 1992; Sieben, 2003). The mean annual precipitation (MAP) in the catchment varies considerably with values exceeding 1800 mm in the mountainous regions to 300 mm in the lower eastern parts (Western Cape IWRM Action Plan, 2011). The mean annual temperature (MAT) for the Breede catchment area is 12°C on higher-lying areas and 19°C on lower-lying areas. The mean annual evaporation rate is relatively high and varies from 1200 mm south to 1700 mm north of the WMA (DWAF, 2004). The dominant lithology in the Breede WMA consists of Quartzitic sandstone of the Table Mountain Group (TMG; Rebelo et al. 2006). Quartzitic Table Mountain Sandstone weathers slowly, producing coarse textured soils with relatively low silt and clay concentrations (Le Roux, 2011). The erosivity is classified as low to medium, with high erosion rates occurring on a few areas (DEAT 1997). The Cape Fold Belt (CFB) is the dominant structural feature in the Breede catchment area and characterises the geomorphology of the area (Prins et al. 2004; Partridge et al. 2010). The Breede WMA consists of 69 quaternary catchments. The five most important water yielding quaternary catchments (H10E, H60B, H60A, H10D and H10K) of the Breede WMA are found in the mountainous head waters (Forsyth et al. 2009).

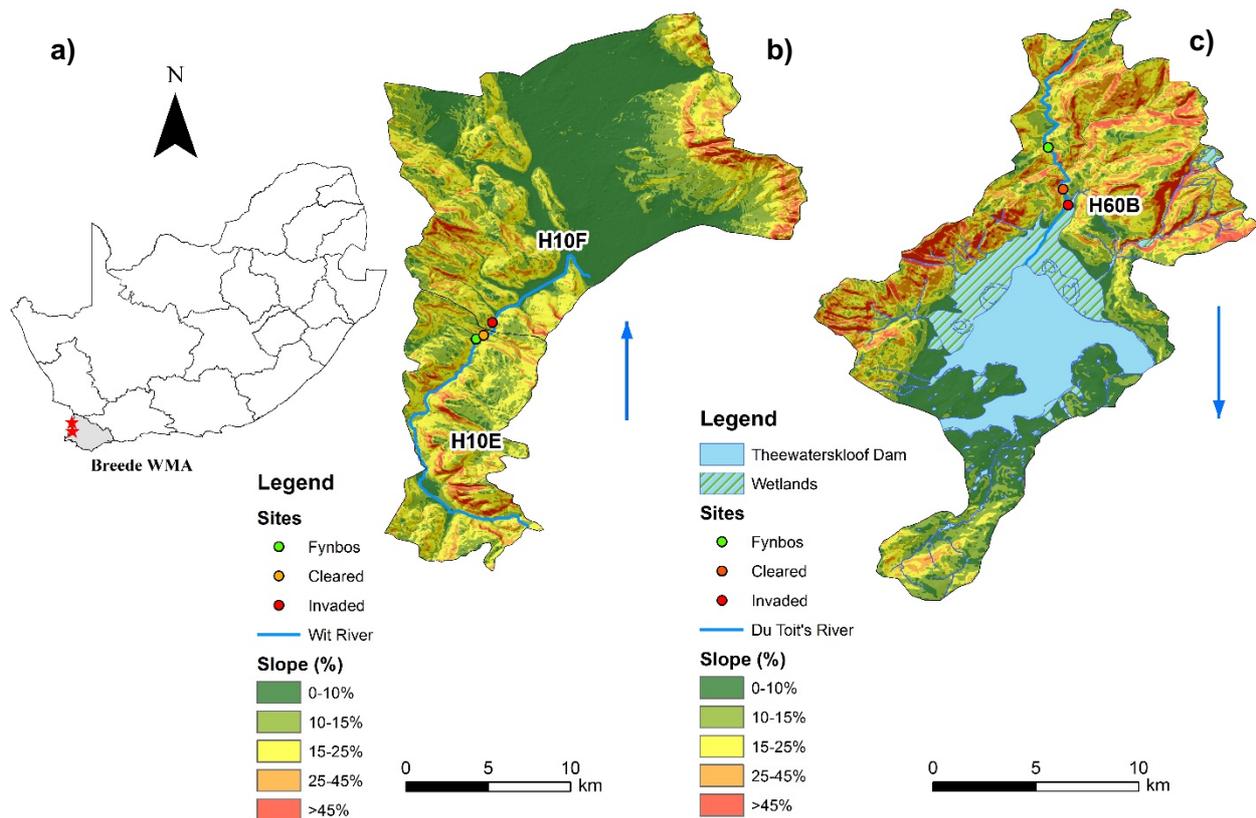


Figure 2.1. Location of the Breede WMA and study sites. (a) Inset map showing the location of Breede WMA in the Western Cape, South Africa. (b) Map of quaternary sub catchments H10E and H10F with the Wit River and sites. (c) Map of quaternary catchment H60B with the Du Toit's River and sites. Near pristine (fynbos) sites are indicated with green symbols, cleared sites with orange symbols and invaded sites with red symbols. The blue arrows indicate river flow directions.

2.3.1.1 Du Toit's River and sites

The Du Toit's River system and sites falls predominantly within the WR90 quaternary sub catchment H60B, and receives a MAP of approximately 1445 mm a^{-1} (DWA, 1997). The Du Toit's River is flanked by very steep catchment slope (average: $\sim 41\%$) and drains a catchment area of approximately 210 km^2 . The river rises in the Franschoek and Middagkrans Mountains and has an estimated length of 13.4 km that flows through the Franschoek Pass valley and into the Theewaterskloof Dam (Tharme, 2010). The upper section of the Du Toit's River has steep gradients ($\sim 12\%$) and shows characteristics similar to south-western Cape mountain streams. The river transforms into a mountain stream transitional below the weir at Purgatory Kloof with gentler gradients ($\sim 11.6\%$). The upper sections in the sub catchment is situated within the protected Mont Rochelle Nature Reserve with the lower reaches largely encompassed by the Hottentots Holland Nature Reserve. As a result, the sub catchment is

largely undeveloped, with land-use primarily limited to near pristine fynbos vegetation, *A. mearnsii* invasions and recreational activities.

The geology of the Du Toit's River is very complex because of the large faults and vertical displacements. The river system is predominantly on quartzitic sandstone of Table Mountain Group with both the Peninsula and Skurweberg formations represented. They are separated by the Cedarberg shale band and conglomerate and Pakhuis mudstone or sandstone (Wu, 2005; Figure 2.2b).

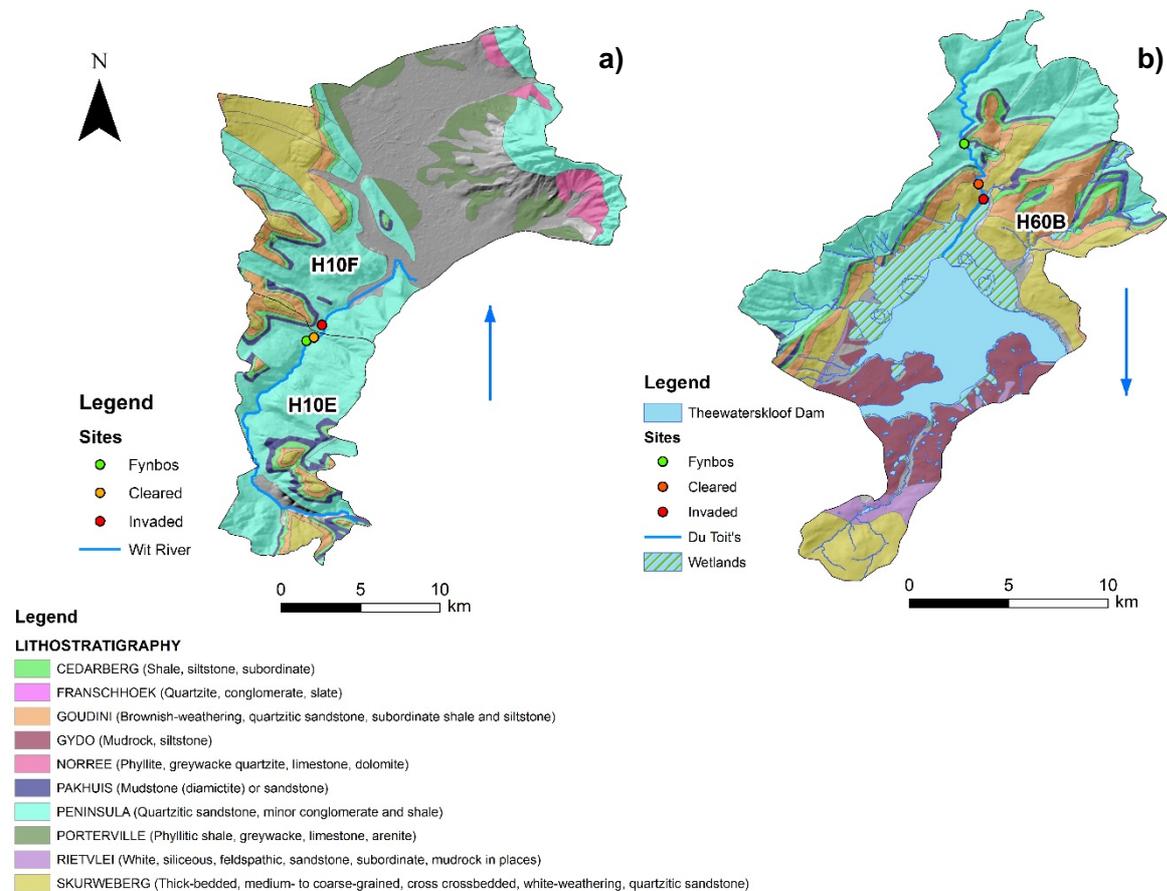


Figure 2.2. Descriptions of lithology and geology for (a) Wit River system and sites; (b) Du Toit's River system and sites.

The near-pristine fynbos reach consists mainly of an open riparian canopy along its length (Figure 2.3 a, b). The dominant riparian species identified in Tharme and Brown (1994) comprises *Metrosideros angustifolia*, *Brabejum stellatifolium*, *Podalyria calyptrate*, *Euryops* spp., *Myrica serrate* and *Erica caffra*. There were also visible signs of exotic species encroachment with a density of < 5%, particularly *A. longifolia*. The river bed substratum is mainly composed of round to sub angular cobbles, in addition to gravel with interspersed sand deposits.



Figure 2.3. Lower near pristine site **(a)** and upper near pristine site at the Du Toit's River **(b)**.

The invaded site is located in the mountain stream transitional zone, below the bridge. The right bank is covered with closed dense invasions comprising *A. mearnsii* while the left bank has scattered stands of *A. mearnsii*, located on high riverbanks and terraces (Figure 2.4a). At the cleared site, there are visible signs of encroachment of exotic species. Several scattered patches of alien vegetation, including *Acacia mearnsii* and *A. longifolia*, with signs of indigenous fynbos vegetation recovery (Figure 2.4b). The initial clearing method used was 'Fell and Remove', a method by which slash was stacked out of the riparian zone more than ten years ago (Blanchard, 2008). Working for Water records indicate that numerous *Pinus* spp. were also present in the mountain catchment area with a density greater than 25%, estimated as 3.5 km² in 1993 (DWAF, 1997) but was successfully eradicated by the WfW (*E. Davids, pers. comm.*).



Figure 2.4. Lower invaded site **(a)** and lower cleared site at the Du Toit's River **(b)**.

2.3.1.2 The Wit River and sites

The Wit River system and sites falls predominantly in WR90 sub catchment H10E, with the lower portion and most downstream site extending into sub catchment H10F (Figure 2.1b). The catchment, which has an area of 333 km², is generally steep sided with average catchment slope of ~37.7%. The Wit River is a perennial river that forms part of the Upper Breede system, and is an important tributary of the Breede River. The river has its source in the Hawequas mountain range (Slanghoek and the Obiekwa Mountains) and has an approximate length of 12 km, flowing through the Bainskloof Pass valley east of the town of Wellington (Brown et al. 2004). According to Brown et al. (2004) the Wit River system comprises predominantly quartzitic sandstone of the Peninsula formation with minor shale and conglomerate (part of the Table Mountain Group; Figure 2.2a). The river bed of the Wit River is dominated by boulder-cobble layers, with interspersed gravel and sand deposits. The Wit River exhibit a gentle gradient (~3.6%; Table 2). The channel banks comprised a combination of boulders and patches of unconsolidated sand deposits. The predominant land-use is natural fynbos vegetation with extensive *A. mearnsii* invasions and agricultural activities lower down in the catchment.

The near pristine (fynbos) site is located above the campsite at Tweede Tol. The near pristine (fynbos) study site is in the Wit River's mountain stream (transitional) zone. The riverbed substratum consists mainly of large boulders, and cobbles with interspersed gravel and fine sand deposits (Brown et al. 2004). The common riparian vegetation community includes *B. stellatifolium*, *M. angustifolia*, *E. capensis*, *E. caffra* and *Prionium serratum* (Campbell, 1985; Figure 2.5 a, b).



Figure 2.5. Upper near pristine fynbos site (a) and Lower near pristine fynbos site at the Wit River (b).

At the invaded site, both the left and right banks are densely invaded by *A. mearnsii* trees (Figure 2.6a), with *B. stellatifolium* and *M. angustifolia* individuals present underneath the *A. mearnsii* canopies. The site is located below the weir and the confluence of the Drosterskloof stream. Clearing has been carried out in section and is on-going. Initial clearing (Fell and Remove) was carried out by the Waterval Cape Nature team more than ten years ago. The site was located above the Drosterskloof weir below the Steenboks Nature Reserve (Figure 2.6b).



Figure 2.6. Upper invaded site (a) and Cleared site at the Wit River (b).

Table 2.1. Geomorphological site characteristics and specific information on each of the selected sites within the Western Cape region

Sites	Co-ordinates (dd.ddddddd)	Altitude (m.a.s.l)	Invasion Status	History of Invasion	Longitudinal zone	Dominant Geology	Distance from source (m)	Dominant substrate	Stream gradient (%)
Wit River (WR)									
WR F	-33.583296 19.134988	279	Near-pristine fynbos	None	Mountain Stream (Transitional)	Quarzitic Table Mountain Sandstone	16604	Bedrock, large boulders	3.9
WR C	-33.569920 19.129420	277	Cleared	<i>A. mearnsi</i> / <i>A. longifolia</i>	Mountain Stream (Transitional)	Quarzitic Table Mountain Sandstone	17174	Bedrock, large boulders	3.8
WR I	-33.551780 19.160930	272	Invaded	Invaded more than 15 years with <i>A. mearnsii</i>	Mountain Stream (Transitional)	Quarzitic Table Mountain Sandstone	18310	Large boulders, stones	3.6
Du Toit's River (DT)									
DT F	-33.916610 19.162592	520	Near-pristine fynbos	None	Mountain stream	Quarzitic Table Mountain Sandstone	6896	Bedrock, large boulders	11.6
DT C	-33.946636 19.168288	355	Cleared	<i>A. mearnsii</i> / <i>A. longifolia</i>	Mountain Stream (Transitional)	Quarzitic Table Mountain Sandstone	9343	Large boulders, stones	10.1
DT I	-33.954028 19.171662	345	Invaded	<i>A. mearnsii</i> more than 10 years predominantly with <i>A. mearnsii</i>	Mountain Stream (Transitional)	Quarzitic Table Mountain Sandstone	10131	Sand, stones	9.5

2.3 Field methods

2.3.1 Sample Collection

a) Surveys of riverbed and channel characteristics

Channel and riparian morphological characteristics were assessed through river cross-profile surveys on five pre-defined transects along each study reach. The channel cross-profiles were surveyed with an electronic Total Station (Top-Shot GP – I22L), with a standard reflecting prism fixed onto a staff and measuring tape.

Channel bed slope (gradient) was calculated by measuring the difference in height over the distance of the sample reaches using the Total Station. Five height measurements were taken from the thalweg of each of the pre-defined transects over a ± 100 m distance at each sample reach and calculated by the following equation (Eq. 2.1). This was ultimately used to calculate the specific stream power of each reach expressed by equation 2.1 and 2.2.

Channel bed slope (S ; m m^{-1})

$$S = \frac{h_2 - h_1}{L} \quad \text{Eq. 2.1}$$

Where S is slope in m m^{-1} , $(h_2 - h_1)$ is the difference in height (m) and L is distance (m) (Gordon et al. 2004).

Total stream power (TSP , W m^{-1}):

$$TSP = \gamma QS, \quad \text{Eq.2.2}$$

Specific stream power (SSP , W m^{-2}):

$$SSP = \frac{TSP}{W}, \quad \text{Eq.2.3}$$

where γ is the specific weight of water (980.7 kg m^{-3}), Q is discharge ($\text{m}^3 \text{ s}^{-1}$) and S is bed slope (m m^{-1}), w is the channel bankfull width (m) (Bagnold, 1966, 1977).

b) Time-integrated suspended sediment sampler

Time-integrated suspended sediment (SS) samplers design was based on Phillips et al. (2000). The SS samplers were constructed from one meter long Polyvinyl Chloride (PVC) pipes, with a diameter of 110 mm (Figure 2.7b). The samplers were sealed with a plastic traffic cone at the inlet to make the sampler streamlined, thereby minimizing the flow resistance, and

covered with PVC threaded end cap at the outlet. A PVC pipe with a reduced diameter of 4 mm was inserted through the traffic cone and the end cap, extending 20 mm into the main cylinder, to serve as an inlet and outlet. The samplers were positioned parallel to the riverbed, at approximately 60% of the flow depth, and mounted with hose clamps to two metal rebar uprights driven into the riverbed. The samplers were installed in the central, quickest flowing part towards the most downstream end of each reach, with the inlet tube pointing towards the upstream end of the reach. The flow velocity is reduced as the water passes through the main cylinder due to the difference in cross-sectional area in comparison with inlet pipe, and thus allowing the sediment to settle out (Phillips et al. 2000).

The efficiency of the samplers in representing suspended sediment loads have been tested by Perks et al (2014). It has been showed that the SS load were underestimated ranging between 66% and 99%, and thus demonstrates that the SS samplers are not suitable to quantify the absolute sediment loads (Perks et al. 2014). However, Perks et al. (2014) showed that the relative efficiency of the samplers produced a significantly strong relationship with the reference sediment load. Therefore, the samplers can provide a useful and inexpensive means to assess relative sediment loads in ungauged catchments. In addition to assessing the relative sediment loads, SS samplers are also used in geochemical analysis studies.

These samplers were placed in-stream at the respective study reaches and were monitored over 12 months. At the end of each sampling interval (month), the contents of each sampler were washed out in-field using distilled water, transferred into sealable plastic buckets (10 L) and transported to the laboratory for further processing. The buckets were disinfected using ammonia and phosphorus-free cleaning detergents, to avoid contamination samples, before each sampling event. Samplers were then returned to their rebar anchorages and depth was adjusted for the change in monthly water level. The content in the buckets were allowed to settle over (24 h) and the overlying water was removed by means of a siphon. The remaining sediment was oven dried at 105°C (24 h) and weighed to calculate the relative suspended sediment load.

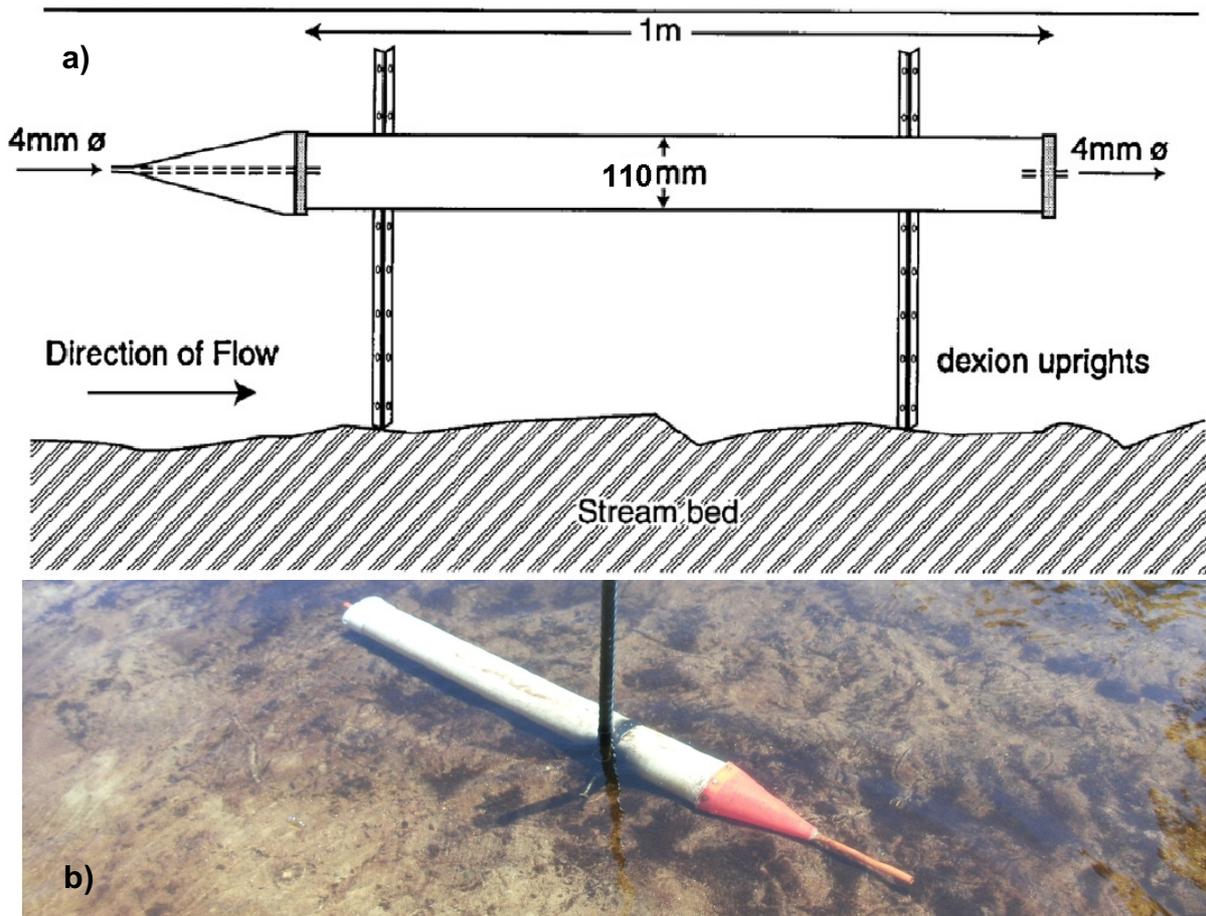


Figure 2.7. (a) Schematic of the time-integrated suspended sediment sampler designed by Phillips et al. (2000) and (b) time-integrated suspended sediment sampler used in this study.

c) Bed-load sediment baskets to measure sediment transport and accumulation in the river bed

Fine sediment on the river bed was determined with sediment basket traps similar to Bond (2002) and Schindler Wildhaber et al. (2012b). Bond (2002) evaluated the efficiency of the traps and they were found to trap 71 and 31% of inorganic and organic suspended material, respectively, that were transported over a defined patch on the river bed where the traps were buried.

A total of fifty-four traps (twenty-seven per river) were buried in the riverbed at three different locations during the sampling period. The locations of traps in each river were chosen in a way that the different vegetation conditions (ecotones) i.e. *Acacia*-invaded, cleared and near pristine fynbos were represented. This was done to allow a comparison between invaded, cleared and fynbos sections of the rivers studied. The baskets were made of a 50×50 mm squared wire mesh that had the following dimensions; 200×150×150 mm (Figure 2.8). The top of the traps was installed such that the top was approximately flush with the riverbed and filled

with clean gravel (grain size >50 mm). This allows cobbles to roll over the basket while fine sediment can infiltrate in between the pore spaces of the pebbles and cobbles into the basket. In order to retrieve the sample without sediment sample loss, a customized bag with handles, made of polyethylene material was placed around the baskets (Figure 2.8b). The bag was retracted to the bottom of the trap before it was buried, to enable water to flow through the trap while it was buried. The bag was pulled over the basket during removal to avoid the sediment loss during sampling. The traps were buried *in situ* in a hole that was dug in the riverbed. The traps were placed far enough apart and in such a manner that there was no disturbance to the sample when a trap was placed or removed in its vicinity. Each study site was equipped with nine sediment infiltration baskets that were equally spaced across the stream channel. The first row of three was placed towards the upstream section of the reach and the next two towards the downstream end. The baskets were sampled at monthly intervals to determine the monthly-infiltrated sediment rates and nutrient concentrations for the study sites. The baskets' sediments were sieved in field through a 4 mm sieve and filled with the same gravel throughout the monitoring period. The < 4 mm sediment was transferred into 25 L buckets and transported to the laboratory for further sieving, nutrient and particle size analyses.



Figure 2.8. (a) Sediment Infiltration baskets filled with gravel used in capturing fine sediment and; (b) embedded basket in the riverbed.

d) Surface water sampling

Water samples were collected by hand in 250 ml plastic bottles that were pre rinsed (x3) with stream water at each sampling site. Samples were collected from three locations at each study reach site, which were representative of up, mid and downstream. The water was filtered in the field through a 0.45- μm Milliex-HP syringe filter units (Whatman, Piscataway, New Jersey,

U.S.A.). The samples immediately placed on ice in a cooler box to minimise changes in water quality due to chemical and biological processes and transported to the laboratory and submitted to CSIR, Stellenbosch. The following water quality parameters were analysed: ammonium (NH_4^+), nitrite (NO_2^-) + nitrate (NO_3^-), nitrite (NO_2^-) and orthophosphate (PO_3^{4-}) using a segmented flow auto analyser. All samples were analysed by the CSIR laboratories, Durban, KwaZulu-Natal. Water samples were collected quarterly, to represent different seasons (autumn, winter, spring and summer) and different flow conditions (intermediate, high and low flows).

e) River Discharge

Each site was equipped with an automated pressure logger (Onset HOBO U20L-02, Onset Computer Corporation, Pocasset, Massachusetts, USA). Continuous stream water level data were recorded at 30 minute intervals with automated water level loggers installed at each of the six sites. The data loggers were retrieved on a seasonal basis to perform data retrieval and routine maintenance and inspection of the device. Rating curves were developed based on on-site discharge measurement for a specified stream cross-section at different stream water levels at each site. Water depth and flow velocities were measured at approximately ten points across one of the predefined cross sections Du Toit's River with an electromagnetic sensor (Flo-Mate, Marsh-McBirney, model 2000, SEBA FlowSens) and calculated instantaneous stream discharge (Q ; $\text{m}^3 \text{s}^{-1}$) from the pre-defined stream cross-sections. The cross-sectional discharge was calculated using the velocity-area method described in Gordon et al. (1992). The discharge of the river was used to establish the temporal relationships between in suspended sediment, mean monthly sediment infiltration rates and mean monthly discharge.

2.3.2 Laboratory preparation procedures

The collected sediment samples were oven dried at 105°C and then disaggregated using a mortar and pestle. Particle size distributions have a significant effect on the sorption capacity, with fine particles ($< 63 \mu\text{m}$) which have a higher sorption capacity (larger surface area to volume ratio, and surface charges) (Bilotta and Brazier, 2008). Elemental concentrations in sediments have been recognized by various authors as being significantly dependent on the particle size distribution in sediments (Horowitz, 1991; Walling et al. 1999). This potentially introduces challenges when comparing the chemical properties of sediment from different sites and of different particle size distributions (Haley, 2010). Horowitz and Elrick (1987)

recommended that sediment samples should be sieved through a $< 63 \mu\text{m}$ sieve to reduce the effect of particle size composition. This facilitated the direct comparison of sediment samples from the upstream sites with the often finer sediment samples from the downstream sites (Wang and Chen, 2000; Whiting et al. 2005; Stutter et al. 2009). The possibility for comparing sediment samples of different sites were assessed for each river using a non-parametric statistical test (Spearman's Rank Correlation Coefficient) to determine the relationship between the clay percentage of samples and their associated nutrient concentrations. The non-parametric test was used because the residuals of the nutrient concentrations in infiltrated sediments were not normally distributed and were tested with a Shapiro-Wilks statistical test. In cases where significant correlations were found between clay percentage and nutrient concentrations across all sites, nutrients were corrected for clay percentage. Where the correlation between nutrient concentrations and clay was not significant across all three sites, particle size correction was considered unsuitable (Haley, 2010).

2.4 Laboratory analytical procedures

The samples that were collected by the sediment baskets and suspended samples were assessed on the following parameters:

- Monthly: Total Nitrogen (TN) and Total Carbon (TC); particle size distribution;
- Seasonal: Total Phosphorus (TP) and Iron (Fe)
- Biannually: Aluminium (Al), potential hydrogen (pH) and electrical conductivity (EC).

Total Carbon and Nitrogen

Prior to analysis, between 30 - 50 mg of the dried sediment sample ($< 63 \mu\text{m}$) was accurately weighed in a tin boat on a Sartorius Super micro balance. The tin cup was carefully sealed using a micro spatula and tweezers to ensure to prevent sample loss and placed in a well plate (transport block). Sediment total C and total N was analysed by the dry combustion CN elemental analyser method (EU EA Analyser: Department of Soil Science and Department of Geology, Stellenbosch University) from which sediment total C, total N and C:N ratios were computed.

Total phosphorus and heavy metals

Between 0.5 g and 1.0 g of sediment sample was accurately weighed and submitted to the CSIR laboratory, Stellenbosch. The technique used involved microwave assisted acid digestion of the dried sediment sample with 1 ml concentrated H_2O_2 , 5ml concentrated HNO_3 , and 1 ml concentrated HClO_4 . The resulting digest was cooled overnight and made up to 50 ml with acidified MilliQ water. The final volume was filtered and subsequently analysed on a Thermo iCap 6500 Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES). Total phosphorus was quantified at a wavelength of 214.91 nm.

Particle size distributions

Particle size distributions of grains with a diameter $< 63 \mu\text{m}$ were measured with a Saturn Digitizer 5200 (Micrometrics, Process Engineering, Stellenbosch University). Particle sizes were categorized according to Udden Wentworth classification systems (Wentworth, 1922): Silt: $63 \mu\text{m} - 3.9 \mu\text{m}$ and clay: $< 3.9 \mu\text{m}$ and used to determine the percentage of clay in each sediment sample.

Prior to analysis, between 1 g and 2 g of the previously dried and sieved ($< 63 \mu\text{m}$) sample material was put into a clean, dry glass beaker. 10 ml of distilled water was then added to each sample, followed by 5 ml of hydrogen peroxide (H_2O_2). The samples were then observed for a period of around 5 minutes for any signs of over-active effervescence. After 2 hours, providing any frothing had ceased, an additional 5 ml of H_2O_2 was added and the sample was left to stand overnight at room temperature. Watch glasses were then placed on the beakers to minimize evaporation and they were warmed on a water bath, starting at 80°C and steadily increasing the temperature up to 100°C . This warming process continued until the reaction was complete and there was a clear supernatant over the sample.

Chapter 3

Quantification of relative sediment loads in two *Acacia mearnsii* infested mountain streams of the Cape Floristic Region, South Africa

3.1 Abstract

Riparian zones in the CFR are heavily invaded by invasive alien woody plants such as Australian *Acacia* spp. Invasive acacias have severely transformed and degrade terrestrial ecosystems by altering vegetation cover, river geomorphology and local soil erosion. No studies have examined the impacts of invasion and clearing on relative sediment loads. My study's aim was to quantify relative sediment loads in two fynbos mountain streams and the sediment contribution of *A. mearnsii* invaded and *Acacia* cleared riparian ecotones. I hypothesised that *A. mearnsii* invaded and *Acacia* cleared sites would have significantly higher sediment loads than fynbos sites. Research was conducted in a combination of mountain stream and mountain stream transitional zones of two perennial rivers in the CFR. Temporal and spatial relative sediment loads were monitored using sediment infiltration baskets and time-integrated mass sediment sampling devices, at three sites associated with 1) near pristine fynbos riparian; 2) cleared riparian and 3) invaded riparian ecotones. Sediment traps were sampled on a monthly basis through the course of a one-year monitoring period.

My results suggest that relative sediment loads in all sites are supply limited and subject to depletion, except for the invaded site on the Wit River. During the wet season up to 84% of the relative sediment load was transported. Analysis of the hysteresis loops showed that the dominant sources of sediment at the Du Toit's River was from in-channel sources, primarily stored sediment on the riverbed and bank erosion. Additionally, relative sediment loads on the Du Toit's River were found to be related to catchment contributing areas and channel characteristics. In contrast, the dominant sources of sediment on the Wit River are from a combination of in-channel and catchment sources. Relative sediment loads in the Wit River were found to be inversely correlated to the catchment contributing area, indicating that invaded and cleared ecotones enhanced sediment delivery to the river. The relative sediment load increase in the invaded site was up to 260% compared to fynbos site, with only a 60% increase at the cleared site. The excessive sediment contributions associated with these invaded and cleared sites can make significant contributions to stream nutrient stocks and fluxes. In contrast, invasion and clearing appeared to have no effect on sediment loads at the Du Toit's River. This suggests that the channel and riparian zone characteristics might reduce immediate connectivity to the channel, thus limiting the effect of invasion and clearing on relative sediment loads. However, additional research is required to develop insight of how

different channels and catchment units respond to disturbances along the stream continuum. This could provide meaningful information to catchment management and restoration projects.

3.2 Introduction

Freshwater resources of the Cape Floristic Region (CFR) are under constant and increasing threat from the establishment and spread of invasive alien plants (IAPs; Richardson and van Wilgen, 2004). IAPs are specialised in establishing themselves in nutrient-poor and disturbed environments, as is characteristic for riparian zones of the fynbos biome in the CFR (Versveld et al. 1998; Richardson et al. 2007). Several invasive *Acacia* spp. (*A. saligna*, *A. longifolia* and *A. mearnsii*) are widespread in the CFR, especially the Berg and Breede River catchments of the Western Cape, with the Breede catchment having a densely invaded area of 84,398 hectares (Versveld et al. 1998). The primary invader is *A. mearnsii* (Brown et al. 2004), which tends to form a dense evergreen monoculture or near monoculture stands that severely transform and degrade riparian ecosystems and often render the invaded area unsuitable for the survival of indigenous plants (Richardson et al. 1997; Le Maitre et al. 2011).

Comprehensive research has shown that dense IAP stands, especially along watercourses in South Africa, use more water than indigenous vegetation such as plant communities in the fynbos and grassland biomes because of the increased aboveground biomass and evapotranspiration rates of the invaders (Le Maitre et al. 1996; Dye and Jarman, 2004). This reduces the hydrological functioning of water resources (Görgens and van Wilgen, 2004) and put pressure on water-dependent ecosystems (i.e. wetlands, estuaries and springs; Ehrenfeld, 2003, van Wilgen et al. 2012). This, in turn, can lead to reduced dilution capacity and increased sediment accumulations, nutrient concentrations, pollution (Chamier et al. 2012).

Given the increasing spread and threat of IAPs on South Africa's limited water resources, the Working for Water (WfW) Programme was introduced in 1995 to eradicate and control IAPs. The programme was initiated to control woody IAPs to protect and restore the hydrological functioning of streams and conserve biodiversity, while enhancing ecological integrity and creating job opportunities for disadvantage communities (van Wilgen et al. 1998). Through this programme, extensive areas of land are being cleared and left bare without additional restoration interventions (Marais et al. 2004). According to Ruwanda et al. (2013), natural vegetation recovery is slow process and restoration of cleared sites does not always occur after the removal of IAPs. Despite the success in terms of hydrological functioning through these clearing initiatives, extensive areas are being exposed and have become

vulnerable to erosion (Richardson and van Wilgen, 2004). The dense vegetation overgrowth and increased aboveground biomass associated with IAPs, often results in a shift from fynbos (shrubland) to tree-dominated stands and a loss of ground cover and development of understory vegetation. This increases soil erodibility, de-stabilization and erosion rates, especially during storm events (Rowntree, 1991; Versveld et al. 1995; van Waal et al. 2009). Thus, the invasion of IAPs, such as *A. mearnsii* and clearing are the main cause of change in vegetation and land cover in the headwater catchments of the Western Cape. Vegetation cover has been shown to exert a first-order control on sediment yield (Molina et al. 2008).

Soil erosion is a major challenge confronting land resource managers in South Africa (Le Roux et al. 2007) with 70 percent of the country's surface area affected by varying types of soil erosion (Garland et al. 2000). Scott et al. (1998) found that fynbos catchments have typically very low sediment yields ($< \text{one t ha}^{-1} \text{ yr}^{-1}$). Van der Waal (2009) quantified soil loss associated with *Acacia*-invaded, cleared and natural (control) fynbos slopes on the Kouga Mountains of the Eastern Cape province of South Africa and found that erosion rates increased on *Acacia*, and cleared slopes. Increased soil erosion not only involves reduced soil productivity through loss of fertile topsoil, but also further extends off-site to increased sediment delivery to watercourses and suspended sediment loads transported by streams and rivers (Walling, 1983, 2005; Trimble and Crosson, 2000; de Vente et al. 2007).

The suspended sediment load in rivers is often used as a measure of catchment-level disturbance that is driven by internal (e.g. changes in vegetation cover or soil erodibility) and external (e.g. climate or land use change) factors (Foster et al. 2014). For example, Grenfell and Ellery (2009) found that suspended sediment concentrations in the Mfolozi River (Kwa-Zulu Natal Province) were poorly correlated to discharge. Instead, the availability of sediment in the catchment is likely the result of a reduction in vegetation cover through vegetation dieback during dry months, and the capacity of overland flow to transport available sediment into the river channel during the rainy season. Although all streams carry sediments under natural conditions (Ryan, 1991), excessive sediment delivery can pose a serious threat to river channels through sedimentation and siltation that have adverse effects on channel morphology and aquatic ecosystems (Walling, 2008; Vigiak et al. 2016). However, this aspect has received little attention in South Africa.

Estimating the relative sediment loads in rivers draining areas invaded by woody IAPs and land cleared thereof will provide a good estimate and a more complete understanding of the level of disturbances these changes in vegetation structure have on sediment delivery to these river catchments. This research is among the first to quantify sediment loads in fynbos mountain streams and determine whether invasive *A. mearnsii* and clearing thereof alter the rate and timing of sediment transfer from the catchments to the stream. The key research

objectives of this study were to (i) investigate the temporal and spatial variability in relative sediment loads of two mountain streams in the CFR, (ii) compare relative sediment loads to river discharge data, and (iii) quantify the relative sediment contributions from near-pristine fynbos, *Acacia* cleared and *A. mearnsii* invaded riparian sections.

3.3 Methods

3.3.1 Study areas and experimental design

The mean annual temperature in the Breede WMA is 12 °C, with a mean annual rainfall of 1800 mm in high lying areas where the rivers are situated (Western Cape IWRM Action Plan, 2011). Quartzite Table Mountain Group sandstones dominate the geology of the two rivers (Rebelo et al. 2006). However, a fault system at the Du Toit's River is responsible for a mixture of rock types throughout the course of the river system. The Upper Du Toit's is dominated by sandstone where the lower sections consist of a combination of sandstone, shale, mudstone and siltstone. The hard impermeable quartzite sandstone produces coarse textured, leached and well-drained soils with relatively low silt and clay concentrations, thus accounting for clear waters (Le Roux, 2011). The land use in the study catchments are dominated by woody invasive alien plants predominantly *A. mearnsii*. The riparian zones of upper reaches of the rivers that occur in protective areas are still mostly in a near-pristine state, with fynbos shrubland occurring.

Measurements were set up in two perennial rivers of the Cape Floristic Region, namely the Wit River (Bain's Kloof Pass) and Du Toit's river (Franschhoek Pass). Each river had three experimental reaches namely A) near pristine fynbos, B) cleared and C) *Acacia*-invaded (from up-to downstream; Fig. 3.1) at different elevations of above sea level and longitudinal zones (Table 3.1). The reaches were relatively straight to ensure that discharge and sediment transport were distributed uniformly across the channel. Each site was equipped with nine bed-load sediment traps that were equally spaced across the channel. The first row of three was placed towards the upstream section of the reach and the next two towards the downstream end. One time-integrated suspended sediment sampler was installed towards the downstream end of each treatment. Data was assessed from February 2016 to January 2017 to cover both high and low flow events.

3.3.2 Channel and riverbed surveys

Channel and riparian morphological characteristics were assessed through river cross-profile surveys on five pre-defined transects along each study reach. The channel cross-profiles were surveyed with an electronic Total Station (Top-Shot GP – I22L), with a standard reflecting prism fixed onto a staff and measuring tape.

The river channel slope was defined by measuring elevations along the channel thalweg using the Total Station. This was ultimately used to calculate the specific stream power of each reach expressed by the following equation:

Total stream power (TSP , $W m^{-1}$):

$$TSP = \gamma QS, \quad \text{Eq. 3.1}$$

Specific stream power (SSP , $W m^{-2}$):

$$SSP = \frac{TSP}{W}, \quad \text{Eq. 3.2}$$

where γ is the specific weight of water (980.7 kgm^{-3}), Q is discharge ($\text{m}^3 \text{ s}^{-1}$) and S is bed slope (%), w is the channel bankfull width (m) (Bagnold, 1966, 1977).

3.3.3 Time integrated suspended sediment samplers

Time-integrated suspended sediment (SS) samplers were designed based on Phillips et al. (2000). The SS samplers were constructed from a one-meter long Polyvinyl Chloride (PVC) pipes, with a diameter of 110 mm. The samplers were then sealed with a traffic cone at the inlet to make it stream line to minimize the flow resistance, and covered with a threaded PVC end cap at the outlet. A smaller PVC pipe with a diameter of 4 mm was inserted through the traffic cone and end cap, extending 20 mm into the main cylinder, to serve as the inlet and outlet. The samplers were positioned parallel to the riverbed at approximately 60% of the flow depth and mounted with hose clamps to the metal rebar uprights that were driven into the bed of the river. The samplers were installed in the quickest flowing part towards the most downstream end of the respective study reaches with the inlet pointing towards the upstream end of the reach. The flow velocity is reduced as the water passes through the main cylinder due to the difference in cross-sectional area in comparison with inlet pipe, and thus allowing the sediment to settle out (Phillips et al. 2000).

These samplers were placed in-stream at the respective study reaches and was monitored over one water year. At the end of each sampling interval (month), the contents of each sampler were washed out in field using distilled water, transferred into sealable plastic buckets (10 L) and transported to the laboratory for further processing. Samplers were then returned to their rebar anchorages and depth was adjusted for the change in monthly water level. The contents collected by the samplers were allowed to settle out (usually over 24 h) in the buckets and the overlying water was removed by means of a siphon. The remaining sediment was oven dried at 105°C (24 h) and weighed to calculate the relative suspended sediment load. The load was standardized by dividing the duration of the sampling period by the load collected in the samplers to give values in g d^{-1} .

3.3.4 Sediment infiltration baskets and sediment transport in the river

Fine sediment transported along the riverbed and settles out of the water column (sediment infiltration) was determined with sediment basket traps following Bond (2002) and Schindler Wildhaber et al. (2012b). Fifty-four traps (27 per river) were buried in the riverbed at three different locations and sampled monthly over one water year. The locations of the traps in each river were chosen in a way that the different vegetation conditions (i.e. *Acacia*-invaded, cleared and near-pristine fynbos) were represented. This experimental setup allowed a comparison between invaded, cleared and fynbos sections of the rivers studied.

The baskets were made of a 50 × 50 mm squared wire mesh (50 mm mesh) that had the following dimensions: length (200 mm), width and depth of 160 mm (200x160x160). The traps were installed so that the top was flush with the riverbed and were filled with clean gravel from the study reach (grain size > 50 mm). This setup allows cobbles to roll over the basket while fine sediment (< 2 mm) can move through the basket and cobbles. A customized polyethylene bag was retracted to the bottom of the baskets before it was buried. This allowed for water and sediment to move freely through the basket and to prevent sediment loss upon retrieval. The traps were buried *in situ* in a hole that was dug in the riverbed. The traps were placed far enough apart and in such a manner that there was no disturbance to the sample when a trap was placed or removed within its vicinity. Each study reach was equipped with nine sediment infiltration baskets. The first row of three was placed towards the upstream section of the reach and the next two towards the downstream end. The baskets were equally spaced at each of the three predefined transects. The baskets were sampled at monthly basis to quantify the monthly sediment infiltration rates and sediment transport. The baskets' sediments were sieved in-field with a 4 mm sieve and refilled with the same gravel throughout the course of the monitoring period. The < 4 mm sediment was transferred into 25 L buckets

and moved to the laboratory for drying, sieving and particle size analyses. The monthly-infiltrated sediment (< 2 mm) for each basket was weighed. This was ultimately used to calculate the sediment infiltration rates (IS) and relative specific suspended sediment yields for each site expressed by the following equations:

Sediment infiltration rate (IS; kg m⁻² d⁻¹):

$$IS = M S_A^{-1} d^{-1} \quad \text{Eq.3.3}$$

where M is mass of dried sediment in the baskets (< 2 mm; kg), S_A is the surface area of the baskets (m²) and d is the length of the sampling period in days.

Relative suspended sediment yield (SSY; kg km⁻² d⁻¹):

$$SSY = M C_A^{-1} d^{-1} \quad \text{Eq.3.4}$$

where C_A represents catchment area.

3.3.5 River discharge

Discharge in the Wit River was monitored at high frequencies at the Drosterskloof weir. However, in the Du Toit's River, the flow was not gauged and therefore the development of a stage-discharge rating curve was necessary. The temporal variation of the water level was measured at 30 minute intervals at each of the six sites using capacitive pressure loggers (HOBO - U20L-02). The pressure was later converted to stage (h) then transformed into discharge (Q) by using the corresponding h/Q rating curve. At each site on the Du Toit's River, h/Q rating curves were developed from direct velocity measurements. Water depth and flow velocities were measured at approximately ten points across one of the predefined cross sections Du Toit's River with an electromagnetic sensor (Flo-Mate, Marsh-McBirney, model 2000, SEBA FlowSens). The cross-sectional discharge was calculated using the velocity-area method as described in Gordon et al. (1992). A large volume of water passes through the boulder beds of these river systems, below the surface, so that low flows are potentially underestimated. The discharge of the river was used to establish the temporal relationships between mean monthly sediment infiltration rates and mean monthly discharge to assess temporal variability in sediment input. This technique has been well documented in sediment transfer literature, and often shows that sediment transfer regimes are largely characterised

by hysteresis effects whereby peak sediment loads precede the peak in river discharge and sediment exhaustion before decrease in discharge (Salant et al. 2008; Grenfell and Ellery, 2009; Smith and Dragovich, 2009).

3.3.6 Sample analyses

Particle size distributions of grains with a diameter < 63 μm were analysed with a Saturn Digisizer 5200 (Micrometrics). Particle-size fractions were classified according to the Udden Wentworth (Wentworth, 1922): Sand: 2 mm - 63 μm , silt: 63 μm - 3.9 μm and clay: < 3.9 μm and used to compare the particle size distributions between sites.

3.3.7 Statistical analyses

For the temporal variation in sediment infiltration rates, a repeated measures analysis of variance (RMANOVA) was computed to test for significant temporal variations over the course of the sampling period. RMANOVA were used to monitor changes in sediment infiltration rates that were collected at the same sites over the course of the sampling period. One-way ANOVAS were computed on all available to determine if the spatial variation in sediment infiltration rates across sites were significantly different. When the sediment contributions from the different invasion statuses (sites) were quantified, one-way ANOVAs were computed on all suspended sediment yield data, regardless of the months. If the residuals were not normally distributed in the one-way, a Kruskal-Wallis test (non-parametric) were used for the confirmation of the parametric one-way ANOVA results. The Spearman's rank correlation coefficients ρ (rho) were computed individually to analyse the relationships between mean sediment infiltration rates and discharge, relative suspended sediment load with discharge. Non-normal distribution of the residuals for sediment infiltration rates and suspended sediment loads justified the use of Spearman's non-parametric test. Fisher's LSD post-hoc test was used for means separations at $p < 0.05$ level of significance. Normal probability plots were used to check if the residuals of all data are normally distributed with Shapiro-Wilks tests. All statistical analyses in this chapter were conducted using the Statistica 13 software package (Dell Inc., 2015).

3.4 Results

3.4.1 Channel morphology and hydrology

In general, the channel and catchment characteristics show considerable variation along the length of the Du Toit's River. Bankfull width of the river cross-sections at the Du Toit's River increased progressively with distance downstream, ranging from 7.6 m at the upper section (DT-F) to 17.5 - 18.4 m in the lower sections (DT-C and DT-I; Table 3.1). The active channel at the DT-I site was deeply incised into the macro-channel with relatively high and steep banks of unconsolidated material (Figure 3.1i). The DT-F site had mean bank heights less than 1.0 m (Table 3.1). The channel banks were highest at the DT-C and DT-F sites, where mean bank heights were consistently greater than 3.0 m (Table 3.1). The riparian zone of DT-F channel is generally steep sided and the channel is less incised. This implies that the channel banks were more stable. Cross-sections WR-F and WR-I, located in the Wit River, were not distinctively different in terms of the mean channel width, mean riverbank height and mean depth (Table 3.1). Thus, channel and catchment characteristics were similar along the length of the Wit River. The bankfull channel width and mean depth ranged between approximately 17.6 – 21.3 m and 0.6 – 0.9 m respectively. The highest mean channel bank heights were observed at the WR-I site (1.4m), followed by the DT-F site (1.0m) and DT-C site (0.8m; Table 3.1).

Site DT-F had the highest stream gradient, at 5.1 % (Table 3.1). This was followed by the site DT-C at 3.4% and Site DT-I at 2.0 %. Site WR-I had the highest stream gradient of 2.9% at the Wit River followed by WR-F at 1.5 % and WR-C at 0.8% (Table 3.1). The specific stream power between the DT-F and DT-I sites is roughly comparable with 1066.1 and 1052.3 $W m^{-2}$, respectively (Table 3.1). Specific stream power in the DT-C site, at 847.3 $W m^{-2}$, is around 0.25 times lower than at the DT-F and DT-I sites. This indicates that the channel at the DT-F and DT-I site has a higher sediment transport capacity compared to the DT-C site. Specific stream power ($W m^{-2}$) is different across the three sites at the Wit River, with specific stream power in the WR-F site approximately 3.5 times higher than in the WR-I site and 0.9 times higher than the WR-C site. This indicates that the channel at the DT-F site has a higher sediment transport capacity as compared to DT-C and DT-I.

Table 3.1. Morphometric characteristics of the Du Toit's and Wit River. Stream power data are given as bankfull flow conditions. Abbreviations: f = Fynbos, c = Cleared, i = Invaded.

River Site	Du Toit's			Wit		
	F	C	I	F	C	I
Catchment elevation (m.a.s.l)	982	947	924	958	919	901
Drainage Area (km ²)	15.6	46.7	49.6	68.5	79.6	86.0
Mean catchment Slope (%) ^a	37	44	43	38	37	38
River bed slope (%) ^b	5.1	3.4	2.0	1.5	0.8	2.9
Average channel width (m)	7.6	18.4	17.5	18.7	17.6	21.3
Average channel depth (m)	0.9	0.6	1.0	0.6	0.8	0.9
Average channel bank height (m)	1.3	3.7	4.0	1.0	0.8	1.4
Cross sectional area (m ²)	6.8	11.0	17.5	11.2	14.1	18.9
Specific stream power (W m ⁻²)	1066.1	847.3	1052.3	1407.9	774.7	313.3
Average stream temperature (°C)	14.4	15.6	16.1	17.7	17.8	17.7

a Catchments slope were based on the value of each pixel were derived from a 5 m digital elevation model in ArcGIS.

b Slope value based on calculation from a Total station survey of the channel bed (Eq. 2.1).

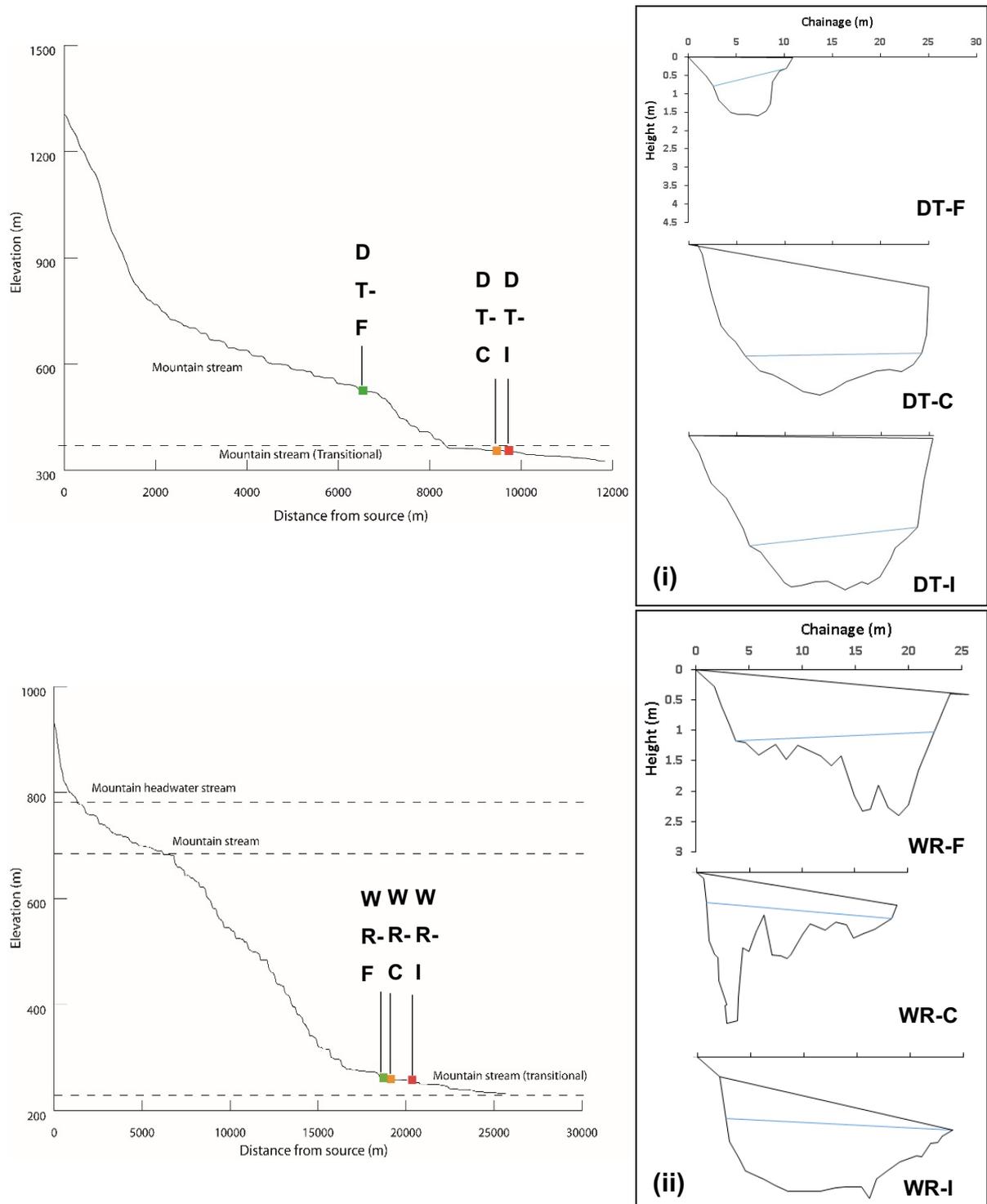


Figure 3.1. Longitudinal profiles of the (a) Du Toit's (DT) and (b) Wit River (WR). Locations and channel cross sections of the three sites in their respective longitudinal zones are given. Dotted line indicates the transition from one zone to the next based on slope values calculated from 5m DEMs. Inset (i) and (ii) shows the channel cross sectional surveys at each site with the blue line indicating water level at bankfull and black line the macro channel. Abbreviations: F = Fynbos, C = Cleared, I = invaded.

3.4.2 Relative suspended sediment load

Only a small number of SS samplers withstood the high-energy flows, consequently, a reduced data set across the study period was available: Wit River (WR) (WR-F: $n = 9$, WR-C: $n = 10$, WR-I: $n = 12$) and Du Toit's River (DT) (DT-F: $n = 11$, DT-C: $n = 11$, DT-I: $n = 12$). This made statistical analyses and comparison between sites difficult. Thus, only general statements concerning the suspended sediment loads were possible. Generally, SS load at the Du Toit's increased from site DT-F to site DT-I, i.e. upstream to downstream. The maximum suspended sediment collected by the SS samplers at the Du Toit's River increased along the river from 4.34 g d^{-1} at site DT-F and 13.43 g d^{-1} at the DT-C site to 15.41 g d^{-1} at the DT-I site (Figure 3.2c and d). The maximum SS measured at the Wit River was 4.5 g d^{-1} (Figure 3.2a).

Spearman rank analyses between SS samplers and sediment infiltration baskets were only feasible at the most downstream sites (WR-I and DT-I; Figure 3.2b, d) of the respective rivers. Within and between seasons, strong monthly fluctuations in the suspended sediment loads occurred, which are largely a function of the mean monthly discharge and rainfall. There is an observable structure in the timing of variations in suspended sediment (SS) load, which is demonstrated in Figure 3.2. The seasonal SS loads are also quite varied throughout the monitoring period with individual months contributing to the annual load. The spring and summer months contribute the least to the annual load with 0.16 g d^{-1} and 0.99 g d^{-1} respectively. The periods of highest SS transport occur in autumn and winter.

The relationship between monthly discharge and SS load reveals periods of the year where there is an abundance of suspended sediment available for transport, followed by depletion. These periods can be visualized and analysed through hysteresis loops (Figure 3.2a, c). At the invaded site on the Wit River, the hysteresis loop is a figure-eight shape. The figure-eight hysteresis loop can be interpreted as a counter clockwise loop during low rainfall events and a positive clockwise loop during high rainfall events (Figure 3.2b). Figure-eight hysteresis suggests that sediment was derived from remote sources. During summer and spring, SS transport was relatively low, which yielded only 0.04 and 0.24 g d^{-1} respectively. The highest SS transport was during autumn and winter, which yielded 1.18 and 5.52 g d^{-1} .

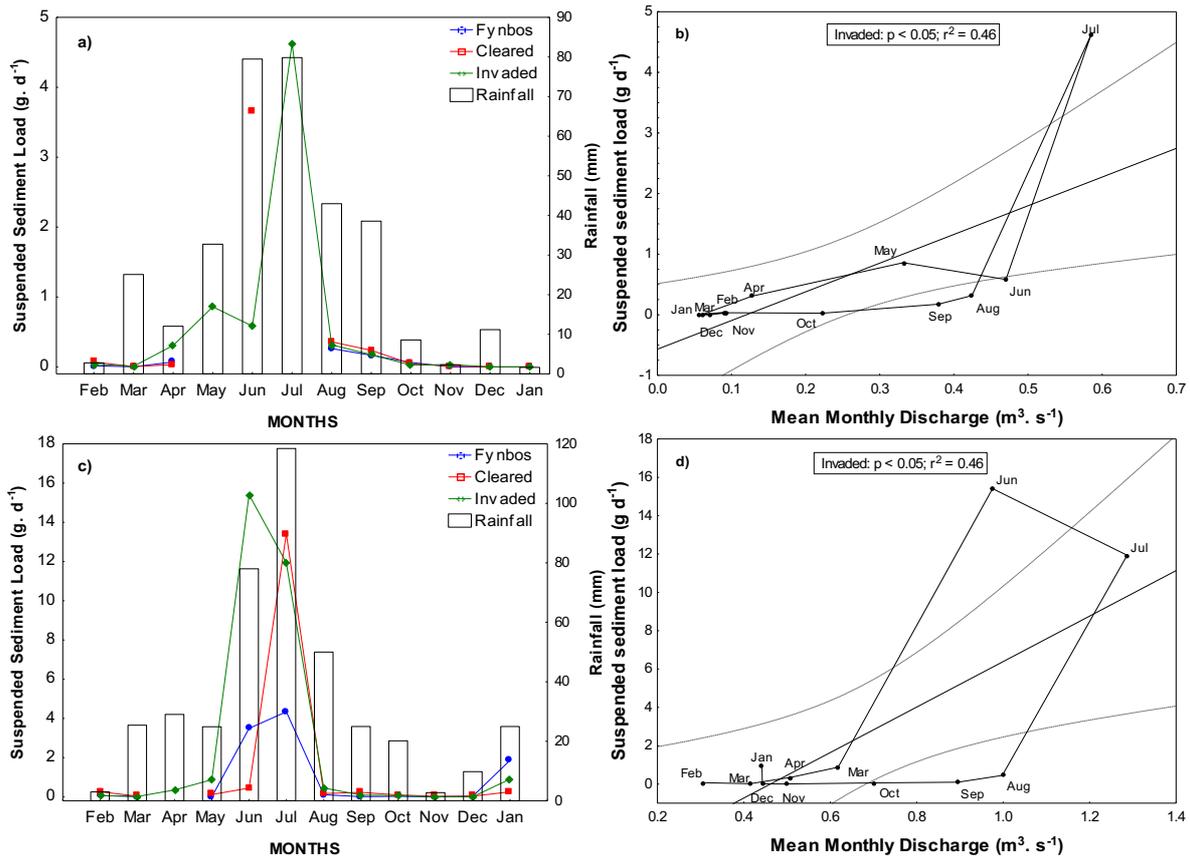


Figure 3.2. Monthly suspended sediment load collected by suspended sediment samplers in comparison to rainfall at the three sites for (a) Wit and (c) Du Toit's River. Note that the months of April at the Du Toit's (Fynbos and Cleared) and May -July at the Wit River (Fynbos and Cleared) are not represented in the record. Fig 3.2 (b) and (d) represents SS load in relation to discharge at the invaded sites on the Wit and Du Toit's River, respectively. Note the different scales on the suspended sediment load axes.

In the Du Toit's River, the relationship between SS and discharge shows a positive clockwise hysteresis loop (Figure 3.2 d). As shown in this figure, SS increased rapidly and peaked during the first high intensity rainfall event in June, before peak rainfall in July. After SS reached its peak, SS decreased considerably. The periods of highest sediment transfer occurred in autumn and winter, which yielded 1.25 and $27.80 \text{ g} \cdot \text{d}^{-1}$ of SS respectively. The dry season (spring and summer) contributed the least SS with 0.16 and $0.99 \text{ g} \cdot \text{d}^{-1}$ respectively. Although there is still relative depletion in January, there is a greater availability of SS compared to the levels in August and September. In interpretation of positive clockwise hysteresis loops, sediments are derived from readily available sources. At the invaded site on the Du Toit's River, sediment transport peaks before rainfall and discharge (Figure 3.4 d). The invaded site is conversely transport limited at the beginning of the wet season. From this data, it is clear that more sediment is available for erosion and transport at the invaded site. In addition, the cleared site is producing more sediment than the fynbos site.

3.4.3 Temporal patterns in sediment infiltration

Significant overall temporal variation was observed for sediment infiltration at the Du Toit's River (ANOVA: $F_{[10, 240]} = 64.10$, $p < 0.001$) with rates between $0.001 \text{ kg m}^{-2} \text{ d}^{-1}$ during low flows and $2.10 \text{ kg m}^{-2} \text{ d}^{-1}$ during high flows (Table 3.2). Mean sediment infiltration rates were significantly lower during the dry months (Oct – Mar) at all sites ($0.08 \pm 0.09 \text{ kg m}^{-2} \text{ day}^{-1}$), compared to the wet months (April - Sept) ($0.48 \pm 0.26 \text{ kg m}^{-2} \text{ d}^{-1}$; ANOVA: $F_{[10, 240]} = 64.10$, $p < 0.001$; Figure 3.5a). Similarly, significant temporal variation was observed at the Wit River with sediment infiltration rates ranging of $0.001 \text{ kg m}^{-2} \text{ d}^{-1}$ and $0.90 \text{ kg m}^{-2} \text{ d}^{-1}$. The mean sediment infiltration rate at the Wit River was $0.02 \pm 0.04 \text{ kg m}^{-2} \text{ d}^{-1}$ during the dry months (Oct - Apr) compared to $0.29 \pm 0.16 \text{ kg m}^{-2} \text{ d}^{-1}$ during the wet months (May to Sept; ANOVA: $F_{[9, 216]} = 68.16$, $p < 0.001$; Figure 3.5b).

A linear regression model was applied to examine the relationships between sediment infiltration and discharge. The Spearman's rank correlation coefficients (ρ) for the Wit River were all above 0.70 and reached up to 0.94 at the most downstream site (Figure 3.4 a-c). These results suggest a strong relationship between sediment infiltration and water discharge ($r^2 > 0.70$, $p < 0.01$). Sediment infiltration rates show considerable temporal variability in the mean sediment infiltration rates for a given average monthly discharge, resulting in the generation of clockwise hysteresis loops at all sites of the respective rivers. In February, at the start of the sampling period, discharge and sediment infiltration rates were both low, at $0.09 \text{ m}^3 \text{ s}^{-1}$ and $0.05 \text{ kg m}^{-2} \text{ d}^{-1}$ respectively. In May, the discharge increases substantially to a value comparable to that seen in September ($0.33 \text{ m}^3 \text{ s}^{-1}$). At the sites WR-F and WR-C, the increase in sediment load is not of the same magnitude, with total mass of 0.07 and $0.13 \text{ kg m}^{-2} \text{ d}^{-1}$ being transported at the respective sites. These values are over 50% smaller than the mass flux during May 2016 (0.17 and $0.31 \text{ kg m}^{-2} \text{ d}^{-1}$). These findings suggest that between June and September, there seems to be a depletion of available sediment sources. However, this depletion never occurred at the WR-I site.

Similar patterns were observed at the sites of the Du Toit's River (Figure 3.4 d-f). However, a lower coefficient ($r^2 < 0.50$) was observed at all the sites (Figure 3.4 d-f). In general, relative sediment loads were low at the end of the summer dry season (February and March). The progressive increase in discharge produced a rapid rise in sediment infiltration rates, with high loads on the rising limb acting to produce strong clockwise hysteresis loops. At sites DT-F and DT-C, sediment infiltration rates and discharge produced a peak in August, whereas site DT-I produced a peak in May prior to discharge peak. Throughout the remainder of the monitoring period, sediment loads were dramatically reduced compared to those with

comparable discharge. This was an indication of sediment depletion over successive high flow events at all sites due to the flushing of readily available sediment accumulation in the channel.

Figure 3.6 shows the relationship between average particle size distributions of the finer sediment fractions ($< 63 \mu\text{m}$) and mean sediment infiltration rates ($< 2 \text{ mm}$) that infiltrated the samplers. The particle size analyses showed a decrease of silt and clay in relative values) with increasing sediment infiltration rates (Figure 3.6). Coarser grain sizes (2 mm and $> 63 \mu\text{m}$) had a tendency to be enriched during periods of higher discharge. Conversely, lowest concentrations of these size classes were found during low flows, particularly over the summer months. The finer grain-size categories ($< 63 \mu\text{m}$), showed the opposite pattern; tending to be enriched during low flow periods and depleted during winter peaks flows. During periods with low sediment infiltration rates and low flow events, silt and clay percentage increased up to 0.97% of the sediment consisted of sediment $< 63 \mu\text{m}$ compared to the 0.62% during high flows (Figure 3.6).



Figure 3.3. Photos showing the distinct visual difference between (a) base flow and (b) high flow conditions at the invaded site on the Wit River.

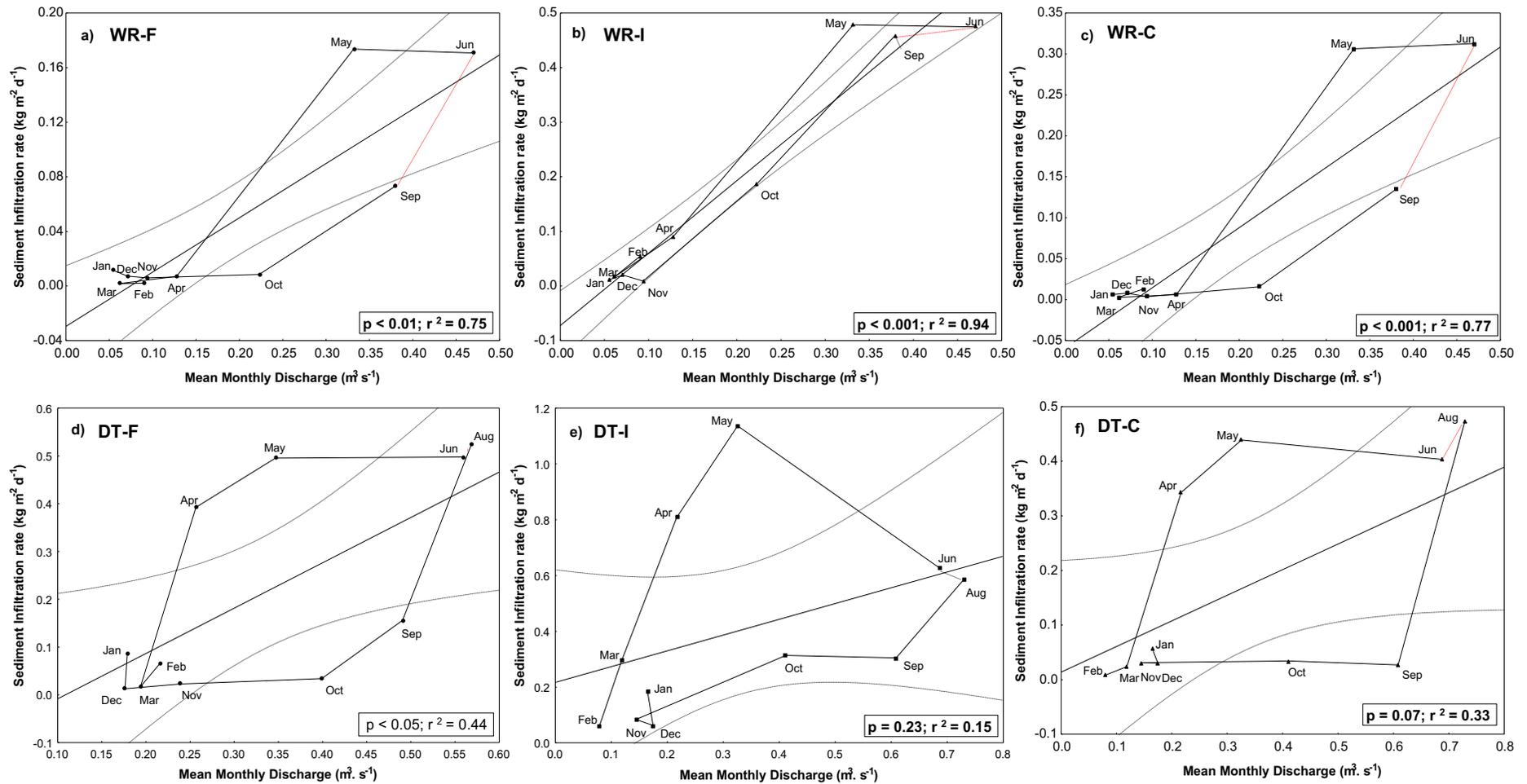


Figure 3.4. Sediment infiltration rate in relation to the mean discharge at the sites during the measurement month at (a – c) at Wit and (d – f) at Du Toit’s River. The relationships at the sites are described by clockwise hysteresis loops. Note that the months of July at the Du Toit’s and July – Aug at the Wit River are not represented in the record (indicated by the red dotted line), which may hide further evidence of monthly hysteresis in the catchment. Dashed lines are the 95% confidence intervals.

3.4.4 Spatial pattern in sediment infiltration

Significant interaction between sediment infiltration rates and the three sites were noted at the Wit River (two-way ANOVA: $F_{[2, 24]} = 22.92$, $p < 0.001$; Figure 3.7b). The highest values for sediment infiltration were noted at the most downstream site (WR-I; one-way ANOVA, $p < 0.001$). In general, sediment infiltration rates on the Wit River increased from upstream to downstream (Figure 3.5b). The averaged sediment infiltration was significantly greater in both WR-C ($0.08 \text{ kg m}^{-2} \text{ d}^{-1}$; $p < 0.001$) and WR-I ($0.18 \text{ kg m}^{-2} \text{ d}^{-1}$; $p < 0.001$) than in WR-F ($0.05 \text{ kg m}^{-2} \text{ d}^{-1}$) (Table 3.2), especially during period of high discharge (May to September; Figure 3.7b). The overall relationship between mean sediment infiltration rates and specific stream power for sites WR-F, WR-C, and WR-I are shown in Figure 3.5.

Similarly, significant overall interaction was evident between sediment infiltration rates and sites at the Du Toit's River ($F_{[2, 24]} = 46.74$, $p < 0.001$). The mean sediment infiltration rates observed at the Du Toit's river ranged from $0.21 \text{ kg m}^{-2} \text{ d}^{-1}$ at the DT-F and $0.17 \text{ kg m}^{-2} \text{ d}^{-1}$ DT-C sites to $0.41 \text{ kg m}^{-2} \text{ d}^{-1}$ at the DT-I site (Table 3.2). Sediment infiltration rates were significantly greater in DT-I ($0.41 \text{ kg m}^{-2} \text{ d}^{-1}$) compared to DT-C ($0.18 \text{ kg m}^{-2} \text{ d}^{-1}$; $p < 0.001$) and DT-F ($0.21 \text{ kg m}^{-2} \text{ d}^{-1}$; $p < 0.001$), with no differences difference between DT-F and DT-C (ANOVA, $p = 0.054$, Figure 3.6a).

Table 3.2. Mean values and ranges of sediment (< 2mm) infiltration rate (IS) and of the coefficient of variation (CV) of the monthly values between the nine baskets of the two rivers at the three sites. The different letters ^(a, b, c) that is vertically distributed represent significant differences between mean sediment infiltration rates at the different sites on each river.

Rivers	Du Toit's		Wit		
	Site	IS ($\text{kg m}^{-2} \text{ d}^{-1}$)	CV (%)	IS ($\text{kg m}^{-2} \text{ d}^{-1}$)	CV (%)
Fynbos		$0.21 (0.001 - 0.75)^a$	76 (13.3 - 187.8)	$0.05 (0.001 - 0.50)^a$	83 (3 - 156)
Cleared		$0.18 (0.001 - 0.82)^a$	103 (25.6 - 208.8)	$0.08 (0.001 - 0.55)^b$	83 (14.7 - 118)
Invaded		$0.41 (0.002 - 2.10)^b$	73 (15.7 - 168.2)	$0.18 (0.002 - 0.90)^c$	102 (22.3 - 184)

Sediment infiltration rates at the Wit River sites appear to be roughly inversely related to specific stream power. For the traps situated at the WR-F sites, low rates of sediment infiltration were found at specific stream power greater than 1000 W m^{-2} , whereas much greater rates were found in the WR-I site for specific stream power less than 1000 W m^{-2} (Figure 3.5). The higher rates of sediment infiltration in these areas reflect enhanced sediment availability and delivery in the WR-I site. Conversely, on the Du Toit's River, specific stream power is roughly comparable between the DT-F and DT-I sites, however, the DT-I site exhibit significantly higher sediment infiltration rates than DT-F (Figure 3.5).

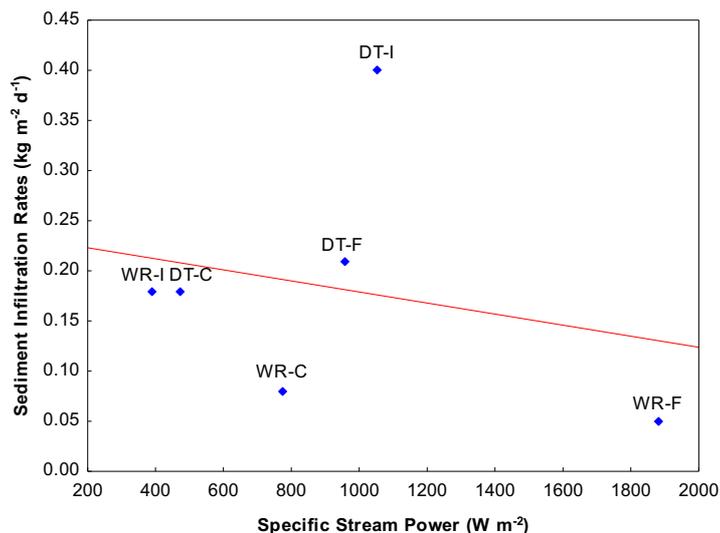


Figure 3.5. Channel classification, criteria and description of the geomorphic processes based on specific stream power and average sediment infiltration rates. A1 = $SSP > 1000 \text{ W.m}^{-2}$ and A2 = $SSP \text{ } 300 - 1000 \text{ W m}^{-2}$. Floodplain and channel classifications are based on the work of Nanson and Croke, (1992).

The cross-sectional variation in sediment infiltration rates between infiltration baskets at the respective sites as expressed by the coefficient of variation (CV) were high, with CV up to 184% on the Wit River and 208.8 % on the Du Toit’s River (Table 3.2). The silt and clay fractions ($< 63 \mu\text{m}$) in the mean captured infiltrated sediment was generally highest at sites DT-C and DT-I of the Du Toit’s River and WR-C and WR-I of the Wit River. The fraction of the sediment smaller $63 \mu\text{m}$ decreased with higher averaged sediment infiltration rates (Spearman-rank correlation, $p < 0.05$; Figure 3.6).

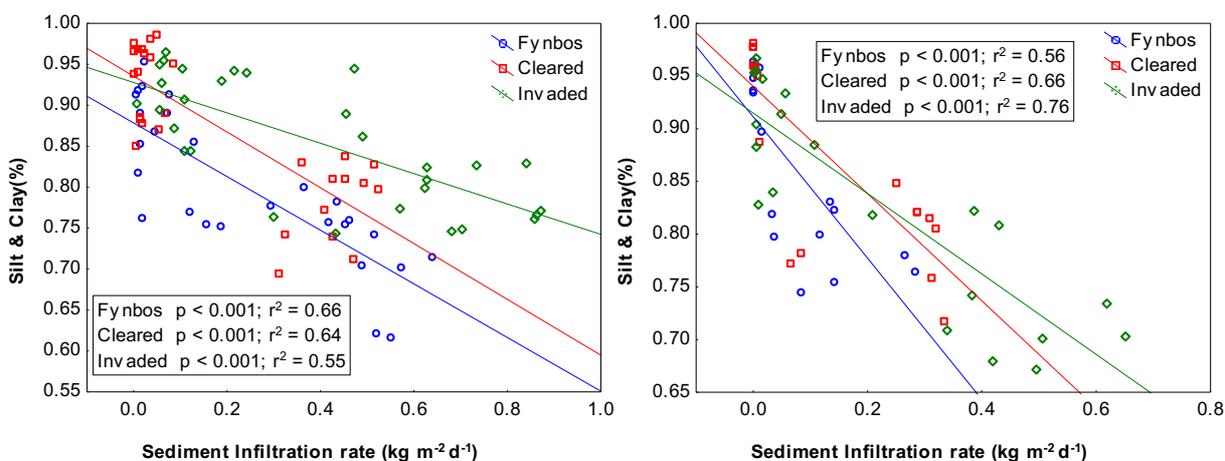


Figure 3.6. Monthly relative silt and clay infiltration at (left) Du Toit’s and (right) Wit River in relation to the mean monthly sediment infiltration rate.

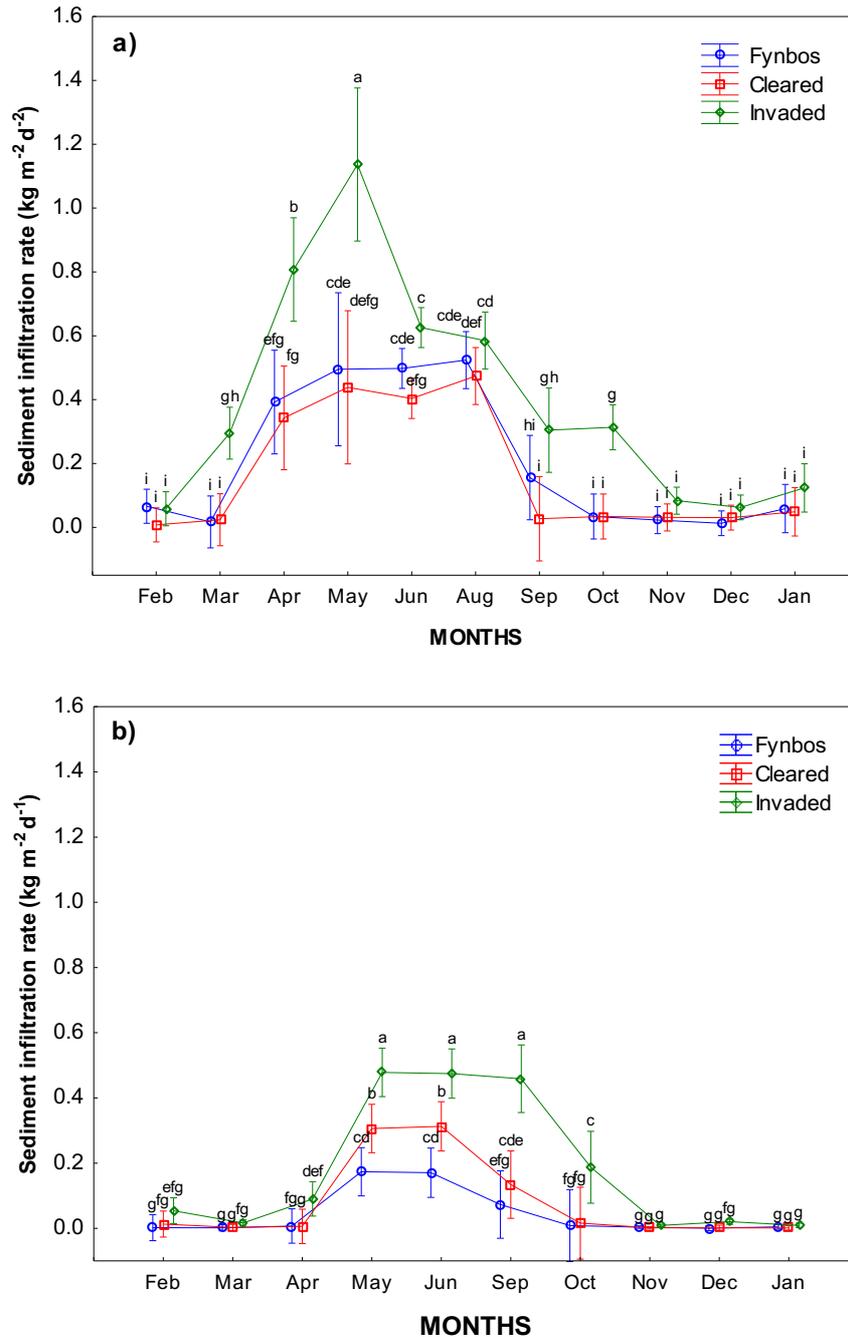


Figure 3.7. Temporal and spatial variability in fine sediment infiltration rates at **(a)** Du Toit's River and **(b)** Wit River across months. The different point symbols represent the mean values and whiskers indicate \pm 95% confidence interval. Interaction between months and sites based on two-way repeated measures ANOVAs with different letters indicate significant differences (LSD tests, $p < 0.05$).

3.4.5 Catchment suspended sediment yield

When the monthly sediment infiltration rates are recalculated to represent the specific suspended sediment yield (SSY; $\text{kg km}^{-2} \text{yr}^{-1}$), a different pattern of sediment production is apparent with the smaller sub-catchment of the Du Toit's River generating a greater load of suspended sediment per-unit area than the larger sub-catchments of the Du Toit's River

(Figure 3.8a). Significant interaction was noted between sites and suspended sediment yield (SSY) ($F_{[2, 24]} = 59.80$, $p < 0.001$, Figure 3.8a). Suspended sediment yield differed significantly between the upstream site, DT-F, and the most downstream site DT-I at the Du Toit's River (one-way ANOVA: $F_{[2, 24]} = 55.65$, $p < 0.001$). The mean annual suspended sediment yield decreased significantly from DT-F ($0.002 \text{ kg km}^{-2} \text{ d}^{-1}$) to DT-I ($0.001 \text{ kg km}^{-2} \text{ d}^{-1}$; ANOVA, $p < 0.001$) and DT-C ($0.0004 \text{ kg km}^{-2} \text{ d}^{-1}$; ANOVA, $p < 0.001$) at the Du Toit's River (Figure 3.8a). The general decrease in suspended sediment yield from upstream to downstream broadly indicates a general trend that as contributing area increase, suspended sediment yield decrease resulting in an inverse relationship (Figure 3.9).

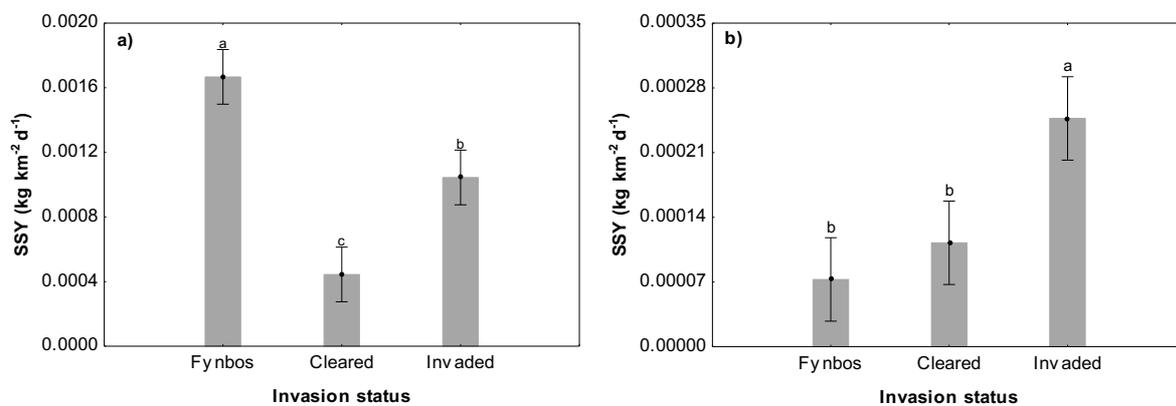


Figure 3.8. Catchment suspended sediment yield for invasion statuses at (a) Du Toit's and (b) Wit River. The bars represent the mean values and whiskers the \pm 95% confidence interval. Interactions between invasion statuses are based on one-way ANOVAs with the different letters on the bars indicate the significant differences (LSD tests, $p < 0.05$). Note the different scales on the suspended sediment yield axes.

Overall, significant interaction was observed between suspended sediment yield and sites at the Wit River ($F_{[2, 24]} = 17.43$; $p < 0.001$). Suspended sediment yield differed significantly between WR-F and WR-I (ANOVA, $p < 0.001$). Mean suspended sediment yield increase from $0.00010 \text{ kg km}^{-2} \text{ d}^{-1}$ at sample site WR-F to $0.00028 \text{ kg km}^{-2} \text{ d}^{-1}$ at sample site WR-I (Figure 3.8b). The increase in suspended sediment yields from upstream to downstream may indicate a positive relationship between catchment contributing area and suspended sediment yield (Figure 3.9).

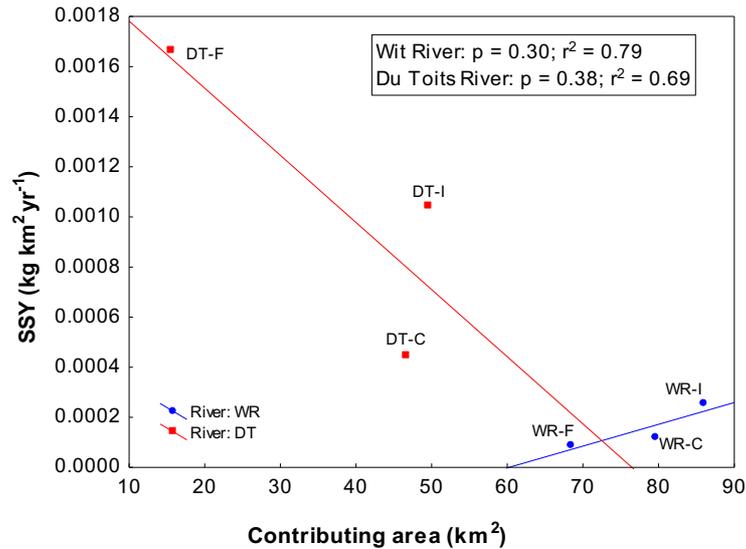


Figure 3.9. Relationship between catchment contributing area and suspended sediment yield (SSY). Sites on the Du Toit's River are coloured red and Wit River is indicated by blue.

3.5 Discussion

3.5.1 Temporal variability in sediment infiltration

There was a consistent trend of higher rates of sediment infiltration associated with the higher flow events, particularly during the wet season (autumn and winter), contributing up to 84% of the relative sediment load. Conversely, lowest infiltration rates were associated with the summer low flow period, contributing about 16% of the relative sediment load. The clear association between sediment infiltration rates and discharge, with greater sediment infiltration rates being associated with high winter flows is a feature supported by other investigations (Greig et al. 2005; Maturana et al. 2014; Milan, 2017).

The association is shown more clearly in Figure 3.4 (a-f), where sediment infiltration rate is plotted against discharge. An analysis of the relationship shows that sediment infiltration rates can be characterized by clockwise hysteresis loops at all sites (Figure 3.4 a-f). The initial peak in SS occurs with marginal increases in discharge until sediment depletion of readily available sources results in a subsequent decline in SS. The early flushing and exhaustion of readily available sediment often drives clockwise hysteresis loops. This phenomenon is well documented in sediment transfer literature, whereby peak sediment concentrations precede the peak in river discharge and depletion results in a subsequent decline in sediment load before decrease in discharge (Scott et al. 1998; Salant et al. 2008; Grenfell and Ellery, 2009; Smith and Dragovich, 2009). This also implies that the available sediment is easy to mobilise and quickly entrained, and thus it is most probably from within the channel and sources proximal to the river channel (Rodríguez-Blanco et al. 2010; Perks, 2013). These sediment sources

include sediment deposits on the riverbed (Arnborg et al. 1967; Bogen, 1980), channel bank material (Seeger et al. 2004; Lefrançois et al. 2007) and floodplains (Mano et al. 2009).

The assessment of the hysteresis loops on the Du Toit's River separately highlights the control of sediment availability within and proximal to the river channel. The relationship between monthly discharge and sediment infiltration rates also reveals periods of the year where there is a relative abundance and depletion of sediment available for transport. At the invaded site on the Du Toit's River, sediment transport peaks before discharge. This suggests that in this part of the catchment, sediment has become available for transport throughout the dry season, and as soon as water is available to transport it, sediment is displaced. The latter feature indicates that the invaded site is transport limited at the beginning of the wet season. In contrast, in the cleared and fynbos sites, sediment transport peaks at the same time as discharge. This suggests that less sediment has become available, and that the system tends towards being supply limited. Furthermore, the results indicate that twice as much sediment is available for transport at the invaded site. In addition, the fynbos site is producing more sediment than the cleared site. The actual amount of sediment available for transport is also a major control on sediment transport rates, as described in Grenfell and Ellery (2009). This is illustrated in Figure 3.4 (d-f), where sediment infiltration rates vary considerably during discharge of similar intensity and magnitude. Although discharge is frequently considered the dominant control of temporal variation in SS (Maturana et al. 2014), variability may also occur due to sediment availability in the catchment as is evident in the Du Toit's River. The invaded site on the Wit River shows a relatively strong relationship between sediment infiltration rates and discharge compared to the other sites. This finding has often been associated with high sediment availability for transport, with a lack of depletion (Wood, 1977; Smith and Dragovich, 2009). Sediments tend to accumulate in the catchment during low magnitude flows governed by low transport capacity (transport limited) whereas during peak transport events, erosional processes and sediment availability become the dominant control (supply limited) (Grenfell and Ellery, 2009; Perks, 2013). On the contrary, at the cleared and fynbos sites sediment infiltration peaks with discharge and shows evidence of sediment exhaustion during later stages of the wet season. Again, this suggests that in these sections, less sediment becomes available, and the system tends to be more supply limited.

Temporal variations in the particle size distribution in the sediment infiltration baskets also reflects changes in flow conditions in the channel in this study, higher winter flows transported greater quantities of varying particle sizes. This result supports the finding that increases in flow facilitate the transport of larger-sized particles (Horowitz, 1991; Horowitz, 2008). However, finer fractions of suspended sediment ($< 63 \mu\text{m}$) are more prevalent during lower energy flows. A similar finding was reported by Schindler Wildhaber et al. (2012b), where

the dominant particle size fraction during low flows was less than < 0.25 mm while particles greater than > 0.25 mm was found during high flows.

3.5.2 Spatial variability in sediment infiltration

Sediment infiltration rates were highly variable between sites. Significantly higher sediment infiltration rates tended to occur in the traps situated in the downstream/invaded reaches of the respective rivers. To develop insight whether this was related to river geomorphological processes or changes in vegetation structure, the specific stream power was calculated to classify the reaches according to Nanson and Croke's genetic classification of floodplains. Specific stream power has been widely used as a tool to assess sediment movement and geomorphic behaviour of river channels (e.g. Nanson and Croke, 1992; Ferguson, 2005 and Bizi and Lerner, 2015).

Based on the commonly used floodplain classification system of Nanson and Croke (1992) the floodplains of all the sites were characterized as high-energy non-cohesive floodplains, where specific stream power exceeded 300 W m^{-2} . The invaded and fynbos sites on the Du Toit's River were classified as A1 orders where specific stream power is greater than 1000 W m^{-2} . This indicates that in these sections, the streams have an extremely high capacity to erode and transport sediment. Despite the comparable specific stream power, sediment infiltration rates in the DT-I site on the Du Toit's were around two times higher than in the DT-F site. This could be an indication of enhanced sediment availability and transport in the DT-I site on the Du Toit's River. This is thought to be primarily related to sediment input from the high and exposed riverbanks made up of unconsolidated sand. The bank material could be weakened by the high specific stream power during consecutive high flows events, resulting in material being eroded and temporally deposited in stream when the flow capacity is reduced. These materials are available and quickly entrained when the stream capacity to transport these sediment increase (Ashbridge, 1995; Bull, 1997). The low riverbanks at the fynbos site are well protected by the dense undergrowth and ground cover with its fine root system that stabilise bank material and reduce erosion (Pietersen, 2009). The DT-C site on the Du Toit's River, suspended sediment load was almost 1.3 times less than in the DT-I site with no significant difference compared to the fynbos site. The lower specific stream power at the DT-C could explain the reduced sediment loads in this section, as the capacity of the stream to transport and do geomorphic work is lower than in the DT-F and DT-I sites.

On the Wit River, the invaded site was classified as an A2 order; the stream's capacity to do work is almost four times lower than in the fynbos site. Regardless of the lower stream power in the WR-I site, suspended sediment load was almost 2.6 times higher than in the

fynbos site. These high sediment infiltration rates in the WR-I site could reflect enhanced sediment input from within-channel sources and *A. mearnsii* invaded riparian zones in the downstream sections of this rivers. The sediment infiltration rates in the cleared site of the Wit River are two times higher in the fynbos site, irrespective of the lower specific stream power. The clearing of IAPs dramatically reduced the protective vegetation cover consequently exposing soil surface to increased erosion. Thus the high sediment infiltration rates in the cleared site are likely explained by high sediments inputs from previously cleared ecotones. The slow recovery of indigenous fynbos vegetation in the previously invaded site can further contribute to the increased sediment loads in these sections.

3.5.3 Catchment scale suspended sediment yield (SSY)

Suspended sediment yields are often reduced with increasing catchment area (Lane et al. 1997; de Vente and Poesen, 2005). This could be a consequence of a decrease in local slope and the presence of wide floodplains with distance downstream, creating sediment sinks (Walling et al. 1999; Syvitski et al. 2005; Birkinshaw and Bathurst, 2006). When the monthly sediment infiltration rates are transformed to represent the area-specific suspended sediment yield (SSY-A; $\text{kg km}^{-2} \text{d}^{-1}$), a different pattern of sediment production is apparent in the Du Toit's River. The study reach in the smaller sub-catchment generate greater amounts of suspended sediment loads per-unit area than the reaches in the larger sub-catchments.

Suspended sediment yield generally increased with an increase in catchment contributing area in the Du Toit's River, thereby resulting in an inverse SSY-A relationship, although sediment infiltration rates increase. This suggests that suspended sediment originated from and was transported from within-channel sources and proximal to the channel (Walling and Webb, 1996). Similarly, Dedkov (2004) reported that a positive relationship between suspended sediment yields and drainage basin area often results when channel erosion (bank and bed) is the dominant source of suspended sediment. This may occur in headwater areas with good vegetation cover and resistant rocks, where erosion rates will increase downstream in association with greater transport capacity and entrainment (Walling and Webb, 1996) or where large volumes of unconsolidated sediment in the channel and channel banks are available for erosion (Church and Slaymaker, 1989). Therefore, the increase in contributing area and changes in local channel characteristics (slope and riverbank height) within the Du Toit's are clearly important controls and may be the reason for substantial spatial variation in sediment load downstream. Relatively steep valley sides with a narrow channel characterize the upstream site, thus sediment have a higher potential energy with which it reaches the stream. The channel and adjacent areas have a high degree of

connectivity but the riverbanks and riparian area are stabilised by the indigenous fynbos vegetation, resulting in low specific suspended sediment yields (Scott et al. 1998; van der Waal and Rowntree, 2015). The wide channel and high stream banks with relatively flat and wide riparian zones in the cleared and invaded site could potentially reduce the connectivity between the stream and adjacent areas. This result suggests that catchment sources contribute a small amount to the sediment load in the invaded and cleared sites on the Du Toit's River.

Conversely, the spatial pattern of suspended sediment yields on the Wit River does not conform to the conventional model of inverse relationship between catchment contributing area and suspended sediment yields. Suspended sediment yields per unit area increase with increase in contributing area, resulting in a positive (indirect) SSY-A relationship. This suggests that the invaded and cleared ecotones on Wit River are subject to high erosion rates and high catchment to river connectivity, indicating a dominance of catchment sources. This could be a consequence of poor vegetation cover and high surface erosion rates of channel banks and/or areas proximal to the channel (Walling and Webb, 1996; Dedkov, 2004; de Vente et al. 2007), consequently, increases the catchment sediment contributions to the river network. Therefore, the higher surface and bank erosion rates associated with the change from fynbos shrubs to invasive woody trees, might contribute significantly to increased suspended sediment yields in lower sections of the Wit River. Furthermore, results from this section provides evidence that the reduction of vegetation cover through clearing of IAPs and slow recovery of native vegetation also contribute to significantly higher suspended sediment yields relative to the areas with intact indigenous fynbos vegetation but to a lesser extent than *A. mearnsii* invaded site.

3.5.4 Impacts of different vegetation conditions on suspended sediment loads

At the near pristine fynbos sites, fynbos shrubland vegetation appeared to make low contributions to the relative sediment loads of the Wit and Du Toit's Rivers with total contributions of 15% and 27%, respectively. The dense undergrowth and fine root systems in undisturbed natural fynbos vegetation provides protection against soil loss through its ability to bind soil and increase bank shear strength and consequently restrict sediment availability and delivery to streams. The geology and soils are also considered as important controlling variables of soil erosion rate, with hard quartzite sandstone having a relatively low erodibility (van Wyk, 1981; Scott et al. 1998). Scott et al. (1998) showed that soil erosion rates in an undisturbed fynbos mountain catchment in the Western Cape, with underlying sandstone, are naturally low ($< 1 \text{ t ha yr}^{-1}$).

The invaded site on the Wit River appeared to have increased the total annual relative sediment loads up to 260% during the course of monitoring period. The total contribution of the invaded site on the Wit River is almost three times higher than that at the fynbos sites. The change in vegetation structure and dense overgrowth of these woody invasive trees inhabits vegetation undergrowth and groundcover, which leaves soil unprotected and vulnerable to soil erosion (Cowling et al. 1976; Rowntree, 1991; van der Waal, 2009). The lack of undergrowth also reduces the trapping and filtering capacity of riparian zones and, thus, could lead to increase sediment delivery to streams. In addition, Scott et al. (1998) found a strong increase in water repellency in soils under *Eucalyptus* and *A. mearnsii* plantations, compared to soils under fynbos that could impede water infiltration and percolation into the soil, which can result in increased generation of overland flow and subsequent soil erosion. The doubling of erosion rates found on *Acacia*-invaded slopes in a study by van der Waal (2009) partially supports this finding. Similarly, Kosmas et al. (1997) recorded increased erosion rates under woody *Eucalyptus* ($23.8 \text{ t km}^{-2} \text{ yr}^{-1}$) invasions compared to shrubland ($6.7 \text{ t km}^{-2} \text{ yr}^{-1}$) in a Mediterranean region. This was mainly attributed to the loss of native species richness, in particular groundcover and understory vegetation that no longer binds and protects the soil. Several field investigations also reported that invasions of riparian zones by alien vegetation such as *A. mearnsii* have an adverse impact on bank resistance and stability (Rowntree, 1991; Versveld, 1995; Esau, 2005; Smith-Adao and Scheepers, 2007; Pietersen, 2009). The observed effects of *A. mearnsii* were channel widening as result of the lack of protected groundcover and a dense root system that makes the banks highly vulnerable to undercutting and erosion. Petersen (2009) estimated that the channels width of streams that were densely invaded by *A. mearnsii* increased by 40% in comparison with the those with indigenous fynbos vegetation in small upland rivers in the Kouga Mountains of the Eastern Cape, South Africa.

Relative sediment loads were significantly increased at the site that has been previously cleared of woody invasive alien trees, but to a lesser extent than at the invaded site. The total annual relative sediment load increased 60% compared to the site surrounded by natural fynbos vegetation. The clearing of IAPs dramatically reduce the protective vegetation cover which, in turn, expose soil surface to increase erosion. Many authors reported that after the removal of IAPs, native species have the chance to re-establish (Galatowitch and Richardson, 2005; Blanchard and Holmes, 2008; Reinecke et al. 2008). In their study along the Berg River in the Western Cape, Ruwanza et al. (2013a) found increased richness and cover of native vegetation four years after clearing of *Eucalyptus* spp. However, the rate at which this occurs depends on several factors such as invasion duration and severity before clearing (Holmes et al. 2005). Thus, the unexpected response in the cleared site on the Wit River could be related to the rate of native vegetation recovery. Furthermore, it can be

expected that suspended sediment load at the cleared would return to uninvaded levels once native riparian vegetation has fully re-established and streams has flushed out the deposited sediments. For example, in a riparian revegetation project in an east Queensland catchment, Marsh et al. (2004) found that suspended sediment yield increased in a stream that has been cleared and revegetated to approximately double that of the forested stream (control). They concluded that suspended sediment yield in the revegetated stream increased significantly as a result of the slow growth rates of the planted trees and expect that the suspended sediment yield will decrease once the riparian plants are fully established. However, streams in Cape Floristic Region are predicted to become drier overall, and rivers in the area may lose significant stream flow in the near future (New 2002) through change in rainfall patterns (i.e. reduced frequency and intensity). These changes in stream flow behaviour in catchments could result in a marked decrease in streams capacity to flush excessive sediments through the stream network, and thus pose a significant threat to in-stream habitats and communities. Fire in invaded areas is another complicating factor, particular in mountain catchments and riparian zones of the fynbos biome. The intense fires that occur in IAP invaded fynbos areas can dramatically alter soils and leave soil susceptible and vulnerable to excessive soil erosion (Scott et al. 1998; Euston-Brown, 2000; van Wilgen and Scott, 2001). The reduced vegetation cover by fire can negate the filtering and trapping capacity thereby increasing sediment delivery to streams, negatively affecting its quality. Smith et al. (2011), for example, found that sediment transport in streams had increased up to 1459 times following a wildfire. However, the impacts of fires in alien plant cleared and alien plant invaded riparian systems on sediment transport to rivers in fynbos mountain catchments are less well understood and still needs to be quantified.

The relative sediment contribution of the invaded site on the Du Toit's river is almost twice as high as fynbos site; however, it was primarily related to the increase in contributing area decrease in slope from up to downstream. Similarly, the clearing had no significant impacts on the suspended sediment loads on the Du Toit's River. The heterogeneity between sites in terms of riparian and stream geomorphological characteristics, such as stream width, river bank height and slope of the riparian areas could explain the unexpected results observed at the Du Toit's River. The wide channel and high riverbanks with relatively flat and wide riparian zones at the cleared and invaded site could potentially reduce the sediment connectivity between the stream and adjacent areas. This results in catchment sources having lesser potential energy with which material are transported to the stream (Walling, 1983; Warburton et al.2003). These attributes could be considered as important controlling variables to mitigate the impacts *A. mearsnii* invasions and clearing thereof on sediment loads.

3.6 Conclusions

There is a dearth of information about the temporal, spatial and transfer dynamics of suspended sediment through the rivers that drain fynbos mountain catchment areas. The findings clearly demonstrate the link between IAPs and clearing on sediment loads. However, our results showed that the impacts of IAPs were indeed highly context-dependent, as patterns of sediment loads did not respond in a linear fashion for the two rivers. Although the patterns are inconsistent, the study provided a number of insights into the sediment dynamics in fynbos streams and the extent to which invasive *A. mearnsii* and clearing may alter sediment regimes. The sediment load in the invaded site on the Wit River increased up 260% relative to sites surrounded by natural fynbos, with only a 60% increase at the cleared site. The spatial variability in sediment availability and delivery are primarily influenced by the catchment contributing area and channel characteristics such as riverbank height and riparian slope. Changes in vegetation cover is a second factor affecting sediment loads, primarily through influence of invasive alien woody trees and subsequent clearing (the extent of which requires further research). These affect the sediment availability of the contributing catchment and channel bank stability, which ultimately affect the sediment delivery to streams.

In general, relative sediment loads in fynbos mountain streams may be characterised as complex and predominantly supply-limited systems. The local hydraulic properties and site characteristics such as channel morphology, vegetation cover, and topography are clearly important controls and produce substantial variations in the quantity of suspended sediments. Relative sediment loads are relatively low, especially when considering the steep terrain and high transport capacity of these streams. Sediment transport was in particular sensitive to high flows, when ~84% of the sediment is flushed through the rivers. This study has provided important benchmark data, furthering the understanding of fluvial geomorphological processes in Western Cape Rivers. Additionally, the findings of this study, especially with respect to sediment loads of mountain streams in the CFR, may facilitate future research and highlight problem areas (natural or human-induced changes) at the local catchment scale. Further research needs to be undertaken to enable better understanding of the behaviour of suspended sediments in these river systems and to determine the holistic impacts of IAPs, especially with regards to clearing and rehabilitation efforts on fluvial geomorphological processes. The findings also provide insights into novel suspended sediment sampling techniques that are commonly used abroad to characterise and quantify relative sediment loads in river systems.

Chapter 4

Quantifying the impacts of invasive N₂-fixing *Acacia mearnsii* sediment nutrient dynamics in mountain streams of the Cape Floristic Region, South Africa

4.1 Abstract

Invasive nitrogen fixing plant species are widespread along streams in the mountain catchments of the Cape Floristic Region. The Australian specie *Acacia mearnsii*, is the most successful and damaging invader in fynbos riparian areas, strongly influencing nutrient cycling in surface soils. Invasive N₂-fixing plant species has been shown to affect stream nutrient concentrations. However, the impacts of N₂-fixing plants on stream nutrient concentrations remain poorly understood and no studies have examined this aspect in South African streams. The aim of this chapter was to quantify the amount of nutrients that invasive N₂-fixing *A. mearnsii* and clearing thereof add to two fynbos mountain streams. It was hypothesized that the stream sections draining *A. mearnsii* invaded riparian ecotones, and those cleared of invasive plants would significantly alter in-stream nutrient concentrations. The research was conducted in a combination of mountain stream and mountain stream transitional zones of two perennial rivers within the Western Cape, South Africa. Monthly suspended sediment, infiltrated sediment (IS) and seasonal surface water were sampled at three sites associated with near pristine (fynbos), *Acacia* cleared and *A. mearnsii* invaded.

My results show that exotic *A. mearnsii* and clearing to some extent, increased total nitrogen and total phosphorus in infiltrated sediments of the Wit River. Mean total nitrogen content in infiltrated sediments of invaded sites were up to 30% and total phosphorus were up to 43% higher than in fynbos sites. Furthermore, a long lasting biogeochemical effect of invasion after clearing was detected at one of the two sites with regards total phosphorus concentrations in IS. The mean total phosphorus concentration was 40% higher in the cleared site compared to the fynbos site on the Wit River. This could be a result of the strong relationship found between phosphorus contents in sediments and iron, reducing its bioavailability in this system, resulting in the accumulation of P in this system. This also holds for the invaded site. The high phosphorus and nitrogen content in sediments of the invaded and cleared sites can accumulate over time and contribute to eutrophication processes. In contrast, invasion and clearing had no significant effects on surface water quality in this study. Phosphate concentrations in surface water were not significantly different between invasion statuses on both rivers. Similarly, NH₄⁺ and NO₃⁻+NO₂⁻ concentrations in surface water were similar between invasion statuses. This may be attributed to the combination of the wide

stream channels and higher stream temperatures that enhance in-stream nutrient cycling processes. It also appeared that the proximity of the invasive trees to the stream channel and channel morphological characteristics may be important controlling variables in the transfer of nutrient-rich leaf litter and sediments between riparian zones and streams. Thus, improved understanding of how different channels and catchments route nutrients through the catchment could be essential for stream management and restoration strategies.

4.2 Introduction

Riparian ecosystems in South Africa and particularly the Western Cape are constantly under threat from invasion by woody N₂-fixing invasive alien plants (Richardson and van Wilgen, 2004). Due to these threats, the interaction between invasive plants and indigenous ecosystems has been a major focus in the field of restoration ecology (Ehrenfeld, 2003). The introduction of invasive species into natural ecosystems is mainly a result of human activities (Milton, 2004; van Kleunen et al. 2015). One of the key aspects thus far has been the impacts of invasive plants on the indigenous vegetation and the attributes of natural ecosystems that make it susceptible for invasion (Rejmánek and Richardson, 1996). Many invasive species have the ability to transform the landscape, and negatively impact on ecosystem processes (Richardson and van Wilgen, 2004). The invasion of non-indigenous species such as Australian *Acacia* spp. in fynbos riparian zones caused a reduction in native plant species diversity (Richardson et al. 2007), reductions in hydrological functioning and changes to nutrient cycling (Stock et al. 1995).

The fynbos region of the Western Cape is dominated by oligotrophic soils, and characterized by sclerophyllous shrubs (Cowling et al. 2004). In general, fynbos soils are nitrogen poor and the nitrogen cycle appears to be very slow (Stock et al. 1995). Very few fynbos plants have the ability to fix nitrogen (Cocks and Stock, 2001). Nitrogen fixing fynbos plants (e.g. *Aspalathus* spp. *Indigofera* spp. and *Cyclopia* spp.) are short-lived and they only contribute 0.5% of the nitrogen stocks to the ecosystem (Cocks and Stock, 2001). In contrast, invasive species such as *Acacia* spp. are effective nitrogen-fixers and alter nitrogen and carbon cycles (Richardson and Cowling, 1992; Ehrenfeld, 2003). The invasion by non-indigenous plants often cause an increase in nutrient stocks, as well as an increase in the productivity of many ecosystems regardless of the decrease in plant diversity (Stock et al. 1995; Tye and Drake, 2012). Le Maitre et al. (1996) modelled the predicted impacts of *Acacia* spp. on water resources. Their models suggested that if no management plan was being done that the catchments that supply Cape Town could lose up to 30 % (87 million cubic meters) of

water per year to invasive species. The Working for Water Programme (WfW) was initiated in 1995 to control and eradicate woody invasive alien plants to preserve stream flow in South African catchments (van Wilgen et al. 1998). Nitrogen fixing alien invasive plants (e.g. *Acacia saligna*, *A. mearnsii* and *A. cyclops*.) is primarily targeted by the WfW. The clearing of IAPs are largely successful in restoring the hydrological functioning of surface and groundwater systems, however concerns has been raised that residual nitrogen in soils will be mobilised, causing a flux of nitrogen into streams (Chamier et al. 2012; Jacobs et al. 2012). It is unclear how the clearing of invasive N₂-fixing trees in the riparian zones will affect water quality and other aspects of the aquatic ecosystems. Indeed, little attention has been paid to quantify the amount of nutrients that nitrogen fixing invasive plants may be adding to aquatic systems in South Africa.

Research on the changing levels of nitrogen in riparian soils associated with N₂-fixing IAPs has been documented in Central Japan (Akamatsu et al. 2011), New Mexico (De Cant, 2008) and more recently in South Africa (Jacobs et al. 2012). Soils under invasive *Acacia* spp. were found to have a two-fold increase in soil nutrient concentrations compared to adjacent Fynbos soils in terrestrial and riparian environments (Musil and Midgley, 1990; Witkowski, 1991; Naude, 2012). This enrichment has been attributed to the high annual litterfall rates of invasive *Acacia* spp. (Milton 1981; Mitchell et al. 1986; Musil and Midgley, 1990), higher foliar litter N and P concentrations (Milton, 1981), and rapid litter decomposition rates (Milton, 1981; Mitchell et al. 1986). Changes in riparian soil biogeochemistry (N and P) have the potential to significantly impact on in-stream nutrient cycling processes and can adversely alter surface water quality (McClain et al. 2003; Vidon et al. 2010). For instance, the woody N-fixer Russian Olive (*Elaeagnus angustifolia*) that invades riparian ecosystems in the western United States alters stream nitrogen dynamics (Mineau et al. 2011), while *Falcataria moluccana* can increase stream nitrogen levels up to six-fold (Wiegner et al. 2013). However, exactly how nitrogen-fixing invasive plants increase nitrogen concentrations in streams is poorly known, although several mechanisms have been proposed (Wiegner et al. 2013). Thus, further research is required to determine which mechanisms, if any, are primarily responsible for the elevated nutrient concentrations in streams.

Stream ecosystems have the ability to transform and retain nutrients in association with sediments, which can lead to the increase in residence and exposure time in aquatic environments (Petersen et al. 2001; Dent et al. 2007). While much attention has been focused on riparian soil biogeochemistry and water quality changes in streams as a function of invasive nitrogen fixing plants, much less is known about the role of suspended sediment (SS) as nutrient carriers. Nutrients, in particular P, have a relatively higher bonding-affinity for clay particles in soils and sediments compared to the low bonding-affinity to water molecules

(Steegeen et al. 2001; Verstraeten and Poesen, 2002; Makarova et al. 2004). The SS can have direct and indirect effects on the primary productivity of streams by directly affecting photosynthesis by blocking sunlight, and indirectly by affecting water chemistry by releasing and adsorbing nutrients. Permanently suspended sediments and re-suspension of suspended sediment have the capacity to absorb and regulate the release of different forms of nutrients into the water column (Ding and Henrichs, 2002; Tappin et al. 2010).

The aim of this chapter was to quantify the impacts of invasive N₂-fixing *Acacia mearnsii* and clearing thereof on nutrient dynamics in SS of mountain streams in the Cape Floristic Region. Estimating the amount of nutrients attached to sediment in the rivers draining areas invaded by woody IAPs and land cleared thereof will provide good insight and improve understanding of how invasive N₂-fixing species changes the nutrient contents and delivery to these river systems. This study was among the first to quantify nutrient concentrations in sediments of fynbos mountain streams and the effects of invasive *A. mearnsii* and restoration activities in fynbos riparian ecosystems. The key research objectives of this study were to (i) determine the temporal and spatial variability in nutrient concentrations in sediments of two mountain streams in the CFR, (ii) compare the results to physical and other chemical properties data, and (iii) quantify the nutrient contributions from near-pristine fynbos, *Acacia mearnsii* invaded and *Acacia* cleared riparian sections.

4.3 Methods

4.3.1 Study areas and experimental design

The rivers selected for this study falls in the Breede Water Management Area (WMA) in the Western Cape Province, South Africa. The rivers are situated in largely mountainous upper sub catchments and are a designated Mountain Catchment Area (MCA); much of it is located within protected areas. As a result, it is largely undeveloped with no anthropogenic disturbances, with land use primarily limited to natural, recreation and alien invasive plants. The experimental design required three sites on each river system. One up-stream near pristine fynbos site (reference) (with no or marginal invasive alien plants and free of anthropogenic disturbance), a mid-stream cleared riparian site (a previously *A. mearnsii* invaded site that has been cleared \pm 10 years ago) and invaded (with *A. mearnsii* as the dominant invader for \pm 10 years). The sites are hereafter referred to as fynbos, invaded and cleared.

The dominant lithology in the catchment is Quarzitic Table Mountain Sandstone of the Peninsula formation (Rebello et al. 2006). The hard impermeable quarzitic sandstone

produces shallow, coarse textured soils and is characterized as nutrient poor, leached and acidic (CapeNature, 2017). The dominant terrestrial vegetation types within the catchments consist mainly of fynbos in the higher lying areas with Succulent Karoo occurring in lowland areas (Rebello et al. 2006; Mucina and Rutherford, 2006). The Breede WMA is characterized by a Mediterranean type climate, with rainfall occurring predominantly in winter, while the summer months are notably warm and dry (Deacon et al. 1992; Sieben, 2003). The mean annual precipitation in the catchment varies considerably with values exceeding 1800 mm in the mountainous regions to 300 mm in the lower eastern parts (Western Cape IWRM Action Plan, 2011). The mean annual temperature (MAT) for the Breede WMA is 12°C on higher-lying areas and 19°C on lower-lying areas (Western Cape IWRM Action Plan, 2011). The MAT measured at the study areas for the monitoring period, Wit River (19.3°C) and Du Toit's River (17.8°C). The hydrological regime of streams is driven by precipitation; discharge in the rivers is high in winter with major peaks in May through to August, and lowest between December and March.

4.3.2 Sediment collection and analysis

Suspended sediment was collected with a time-integrated suspended sediment sampler based on the design described in Phillips et al. (2000). These samplers were positioned parallel to the riverbed at approximately 60% of the flow depth towards the downstream end of the study reaches. Sediment infiltration (IS) baskets were used to collect sediment transported along the riverbed and sediment that settles out of the water column. A total of 54 (9 per site) sediment baskets were buried *in situ* in a hole that was dug in the riverbed. The first row of three was placed towards the upstream section of the reach and the next two towards the downstream end. The baskets were equally spaced at each of the three predefined traverses. The samples were collected on a monthly basis over one water year. The baskets' sediments were put through a 4 mm sieve in field and filled with the same the gravel throughout the course of sampling period. The contents of the samplers were transferred into sealable buckets and taken to the laboratory. The content in the buckets were allowed to settle over (24 h) and the overlying water was removed by means of a siphon. Sediment samples were oven dried at 105°C for 24 hours and then disaggregated and homogenised using a mortar and pestle. All samples were sieved through a 63 µm mesh prior to nutrient analysis to minimise the effect of particle size as recommended by Horowitz and Elrick (1987).

Water samples were collected by hand in 250 ml plastic bottles that were pre rinsed with stream water at the sampling sites. Samples were collected from three locations at each study

reach site, which were representative of up, mid and downstream. The water was filtered in the field through a 0.45- μm Millieux-HP syringe filter units (Whatman, Piscataway, New Jersey, U.S.A.). The samples were immediately placed in a cooler box with ice and submitted to the CSIR. Water sample collection was done quarterly during periods of low, moderate, intermediate and high flow.

The samples that were collected by the sediment baskets and suspended sediment samplers' were measured monthly for the following parameters: total Nitrogen (TN) and total Carbon (TC); particle size distribution; seasonally: total Phosphorus (TP) and iron (Fe); biannually aluminium, pH and electrical conductivity (EC). Total carbon and nitrogen of the IS and SS was measured by a dry combustion followed by a CN elemental analyser (Euro EA Analyser) using standard processing techniques (Sollins et al. 1999) at the Central Analytic Facility (CAF), Stellenbosch University (Stellenbosch, Western Cape, South Africa). Total phosphorus and iron in IS were analysed using a Thermo iCap 6500 Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES), after sample preparation by microwave digestion at CSIR laboratories, Stellenbosch. Water samples were analysed for ammonium, nitrite + nitrate, and orthophosphate by the CSIR laboratories, Durban, KZN.

4.3.3 Particle size corrections

Particle size distributions of the previously dried and sieved sediment samples ($< 63 \mu\text{m}$) from IS were assessed with a high definition digital laser particle size analyser (Micromeritics Saturn DigiSizer 5200) at the Department of Process Engineering, Stellenbosch University. Particle size fractions were classified according to Wentworth Udden: Silt: $63 \mu\text{m} - 3.9 \mu\text{m}$ and clay: $< 3.9 \mu\text{m}$; the classification was used to determine the percentage of clay in each sediment sample. Previous studies have established that particle-size can exert a strong influence on associated nutrient concentrations (Horowitz, 1991; Motha et al. 2002; Chapman et al. 2005).

C, N and P properties were tested for correlation between sample clay percentage and property concentration within the individual sites. The non-parametric Spearman's rank correlation coefficients (ρ) test was used as both nutrient concentration data and clay fraction data were not normally distributed. Previous studies have commonly based particle size correction factors on specific surface areas (SSA) ratios, however, due to time and resource constraints, the correction factor of this study was based on clay percentage to further minimize the effect of particle size.

In cases where a significant correlation was found between clay percentage and nutrient concentrations across all sites on the respective rivers, nutrient concentrations were

corrected to account for differences in particle size distributions, thus allowing the direct comparison between sites. However, where the correlation test indicated that nutrient concentrations were not significantly correlated with clay percentage across all sites, then particle size correction was considered inappropriate (Haley, 2010). The clay percentage and nutrient concentrations of each sample were then used to make corrections for differences in clay percentage between samples and sites using Equation 4.1:

$$C_c = C / C_p \quad \text{Eq. 4.1}$$

where C_c the nutrient concentration corrected for clay percentage, C is the measured nutrient concentration and C_p is the clay percentage in infiltrated sediments.

4.3.4 Statistical analyses

The null hypotheses that *Acacia mearnsii* invasion and clearing does affect TC, TN, and TP in sediment and surface water were tested using one-way analysis of variance (ANOVA) on all data irrespective of months and seasons, and significance was determined at $p < 0.05$. Kruskal-Wallis tests were used in cases where data were non-normally distributed to confirm the results of the one-way ANOVA. Shapiro-Wilks test were computed to test if the residuals were normally distributed. For temporal variation in nutrient concentrations (TC, TN, C:N and TP) in infiltrated sediment, a repeated measures analysis of variance was used to test for significant interaction between invasion statuses across seasons. The RMANOVA was used because the data was collected at the exactly the same sites over the course of the sampling period. Fisher's LSD test was used for *post-hoc* means separations on all statistical analyses. Non-normal distributions of residuals and unequal variance of nutrient concentrations in sediments justified the use of Spearman's non-parametric tests. Spearman rank correlation coefficient ρ (rho) were computed individually on all data, irrespective of sites and invasion statuses, to test for significant relationships at $p < 0.05$, between sediment properties (chemical and physical). All statistical analyses in this chapter were performed on Statistica 13 software package (Dell Inc., 2015).

4.4 Results

4.4.1 Temporal variation in TC and TN in sediments

The total N in infiltrated sediments (IS) at the Wit River showed significant interaction between seasons (two-way ANOVA: $F_{[9, 63]} = 29.42$, $p < 0.001$; Figure 4.1c). TN had significantly higher mean concentrations during the autumn months (mean = $1.09 \pm 0.34\%$) relative to winter ($0.54 \pm 0.07\%$), spring ($0.60 \pm 0.12\%$) and summer ($0.77 \pm 0.27\%$). The TC concentrations in IS showed a similar temporal pattern to TN in IS with a significant overall seasonal trends (two-way ANOVA: $F_{[9, 63]} = 61.82$, $p < 0.001$; Figure 4.1b), with higher overall mean concentrations in autumn months ($21.03 \pm 5.07\%$) and significantly lower concentrations during the winter ($11.56 \pm 1.42\%$), spring ($13.47 \pm 3.77\%$) and summer (14.19 ± 7.9) months. The mean TN and TC concentrations were typically lowest during the wet season (with high flows and sediment infiltration rates) and peak concentrations during low flows in the autumn months. The effect of season on C:N ratio showed a significant overall interaction (two-way ANOVA: $F_{[9, 63]} = 23.90$, $p < 0.001$). Overall, highest C:N ratios was measured during winter (mean = 17.45) and lowest ratios during summer (mean = 22.14) (Figure 4.1d). TN and TC concentrations in SS showed similar temporal patterns as TN and TC concentration in the IS. The smallest concentrations were assessed during the winter months with a mean of (TN = 0.6 % and TC = 14.13%). TN and TC concentrations in SS were significantly higher in summer with means of 1.47 and 23.95 %, respectively (Figure 4.1b, c).

In the Du Toit's River, significant seasonal variability was observed between total N in IS and seasons (two-way ANOVA: $F_{[10, 80]} = 11.32$, $p < 0.001$; Figure 4.1c). The highest mean TN concentrations were observed during the summer months (0.68 ± 0.14) relative to autumn ($0.43 \pm 0.19\%$), winter ($0.40 \pm 0.15\%$) and spring ($0.55 \pm 0.09\%$). The seasonal pattern in TC showed a comparable trend to TN, with significantly higher average TC concentrations during the summer months ($13.51 \pm 3.12\%$) and spring ($12.78 \pm 2.49\%$) compared to winter ($9.37 \pm 3.37\%$) and autumn ($8.72 \pm 4.69\%$). The lowest C:N ratios were measured during summer months (19.91) and highest ratios during winter months (23.59).

Overall, TN concentrations showed significant inverse relationships with the total amount of SS and averaged IS (Figure 4.2 a, b). A significant positive correlation was found between clay percentage and TN in IS on both river systems irrespective of sites (Figure 4.2c). Similarly, significant positive correlation exists between TN and TC concentrations in IS ($r^2 = 0.81$, $p < 0.001$) and SS ($r^2 = 0.74$, $p < 0.001$) (Figure 4.2d).

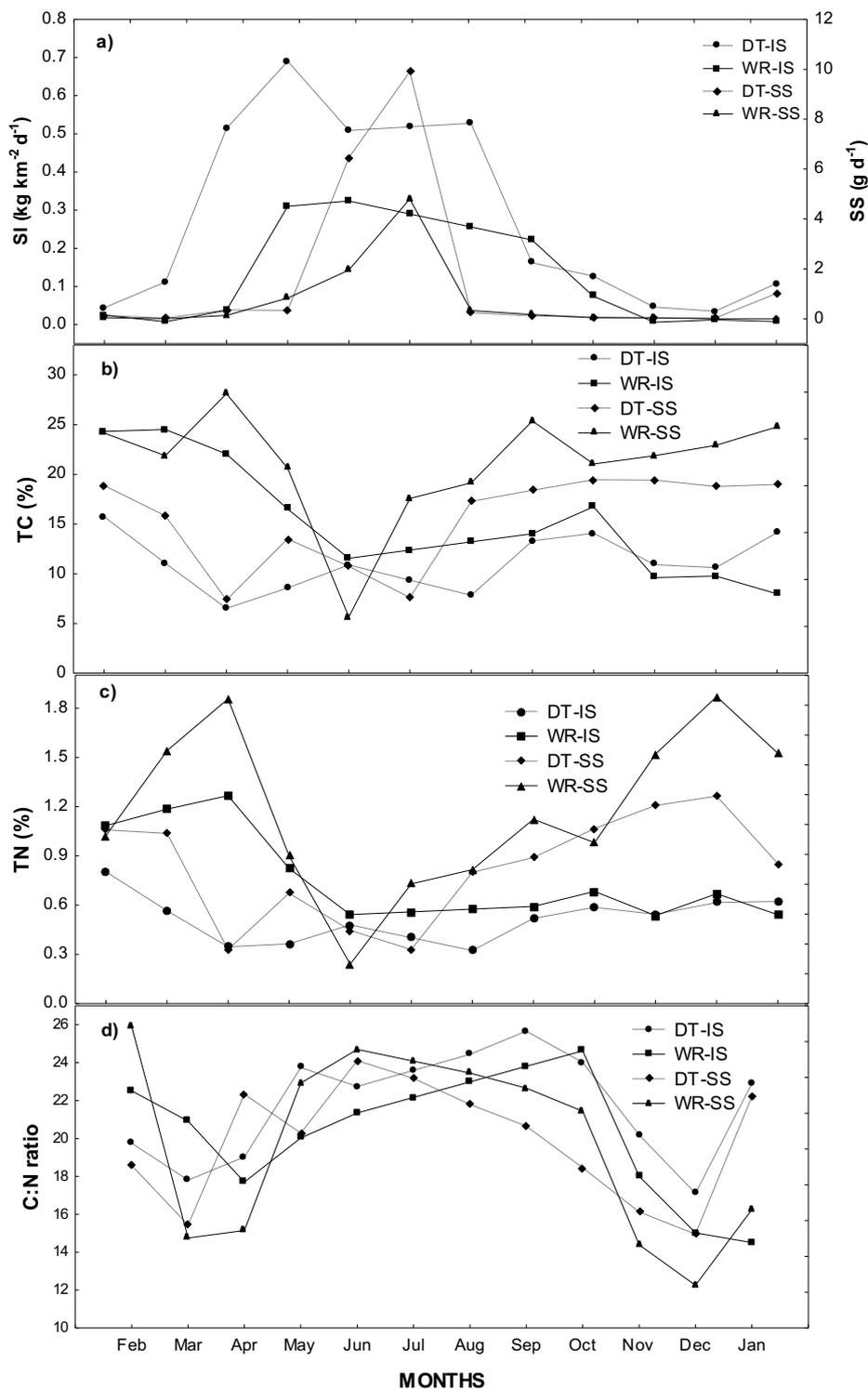


Figure 4.1. Temporal variation in sediment and nutrient dynamics during the sampling period. **(a)** Mean monthly sediment infiltration rates (IS; < 2mm) and the mean suspended sediment (SS; g d^{-1}); **(b)** total C of IS and SS; **(c)** total N of the monthly IS and SS; **(d)** C:N ratios of IS and SS. It should be noted that the temporal patterns of IS, SS, TN, TC and C:N ratios depicted in Figure 4.1 a, b, c and d were consistent for the sites on both the Wit (WR) and Du Toit's (DT) River. See APPENDIX A, B, C for temporal variation across sites.

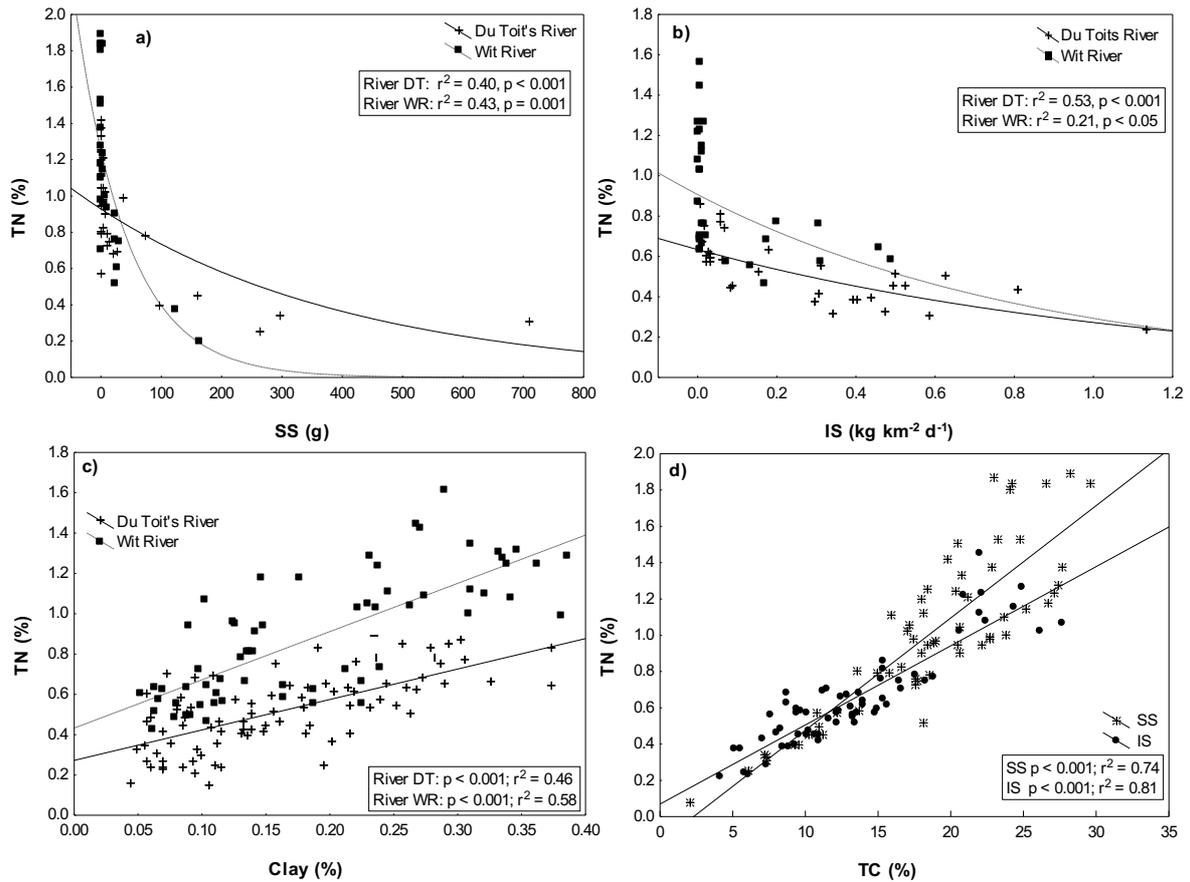


Figure 4.2. Relationship between total N and (a) total SS (< 2mm); (b) average IS; (c) clay percentage of IS and (d) total C. Solid squares and dotted lines: Wit River (WR); crosses and solid lines: Du Toit's River (DT).

4.4.2 Spatial variation in TC and TN in IS and SS

The interaction between invasion statuses were not significant for total N in IS at the Wit River (two-way ANOVA: $F_{[2, 6]} = 4.796, p = 0.001$; Figure 4.3a). However, a general trend towards enhanced TN concentrations was noted in the invaded site when TN concentrations were averaged over the course of the sampling period. The highest mean was found at the invaded site (mean = $0.83 \pm 0.23\%$) compared to cleared ($0.78 \pm 0.30\%$) and fynbos sites ($0.77 \pm 0.37\%$) (Table 4.1). Significant interaction was observed between total C and invasion statuses (two-way ANOVA: $F_{[2, 6]} = 32.96, p < 0.001$; Figure 4.3a). TC increased in a way that is similar to TN from fynbos (mean = 14.31 ± 6.19) to the cleared ($15.18 \pm 6.44\%$) and invaded sites ($17.81 \pm 6.31\%$). The C:N ratios were higher in the invaded site (mean: 21.28) compared to cleared and fynbos sites with means of 19.50 and 18.95, respectively (Table 4.1).

Table 4.1. Comparison of average values (\pm standard deviation) at the three sites sampled of TC, TN and TP in infiltrated sediment (IS) and suspended sediment (SS). Abbreviations: F = Fynbos, C = Cleared, I = invaded. All concentrations are reported in % except for TP, which is reported as mg g^{-1} and C:N which is unitless.

Rivers	Wit			Du Toit's		
	F	C	I	F	C	I
TC IS	14.30 \pm 6.20	15.20 \pm 6.40	17.80 \pm 6.30	12.90 \pm 3.50	10.90 \pm 4.50	10.00 \pm 3.60
TN IS	0.77 \pm 0.37	0.78 \pm 0.30	0.83 \pm 0.23	0.56 \pm 0.15	0.53 \pm 0.21	0.48 \pm 0.17
TC SS	24.66 \pm 4.09	23 \pm 3.78	23.36 \pm 3.68	16.67 \pm 4.75	16.98 \pm 3.78	15.15 \pm 5.77
TN SS	1.33 \pm 0.43	1.11 \pm 0.52	1.14 \pm 0.48	0.85 \pm 0.31	0.93 \pm 0.29	0.80 \pm 0.42
TP IS	0.47 \pm 0.33	0.88 \pm 0.69	0.78 \pm 0.51	0.41 \pm 0.22	0.59 \pm 0.27	0.59 \pm 0.31
C:N in IS	18.95 \pm 0.64	19.50 \pm 0.39	21.30 \pm 0.58	22.98 \pm 0.82	21.08 \pm 0.41	21.11 \pm 0.83
C:N in SS	19.49 \pm 3.94	21.58 \pm 7.30	19.51 \pm 4.29	20.14 \pm 3.37	18.78 \pm 2.75	19.73 \pm 3.88

In the Du Toit's River, TN concentration in IS differed significantly between invasion statuses (two-way ANOVA: $F_{[2, 6]} = 14.68$, $p < 0.01$; Figure 4.3b). Mean total N concentrations were significantly lower in the invaded site (mean = $0.48 \pm 0.17\%$) than in the cleared ($0.53 \pm 0.21\%$, $p < 0.05$) and fynbos sites ($0.56 \pm 0.15\%$, $p < 0.01$). No significant differences were apparent between fynbos and cleared ($p = 0.07$; Table 4.1). Similarly, the highest mean TC content was found at the fynbos site ($12.87 \pm 3.53\%$) and was significantly higher than cleared ($10.87 \pm 4.52\%$, $p < 0.05$) and invaded sites (10.02 ± 3.64 , $p < 0.01$) (Figure 4.3b). The fynbos site had the highest C:N ratios (mean: 22.98) compared to the invaded (mean: 21.11, $p < 0.05$) and cleared sites (mean: 21.08, Table 4.1). The total N concentration in SS showed a different spatial pattern as TN concentration of the IS (Table 4.1). The highest mean total N concentrations were found at the cleared site with a mean of 0.94%; fynbos and invaded sites had lower concentrations with means of 0.85 and 0.80%, respectively (Table 4.1). TC in SS increased similar to TN from the cleared site (16.98 ± 3.78) to the fynbos (16.67 ± 4.75) and invaded (15.15 ± 5.77) sites. TC concentrations in SS generally increase from invaded site with a mean of 15.15% relative to the fynbos and cleared sites with means of 16.67% and 16.98, respectively. In general, mean TN and TC concentrations in SS was 0.34% and 4.8%, respectively, higher than in IS. The clay percentage in IS was significantly correlated with TN ($r^2 = 0.43$, $p < 0.01$). Overall, C:N ratios showed a significant and inverse correlation with Fe concentrations ($r^2 = -0.71$; $p < 0.01$), clay percentage ($r^2 = -0.29$; $p < 0.01$) and stream temperature ($r^2 = -0.36$, $p < 0.01$; Table 4.2).

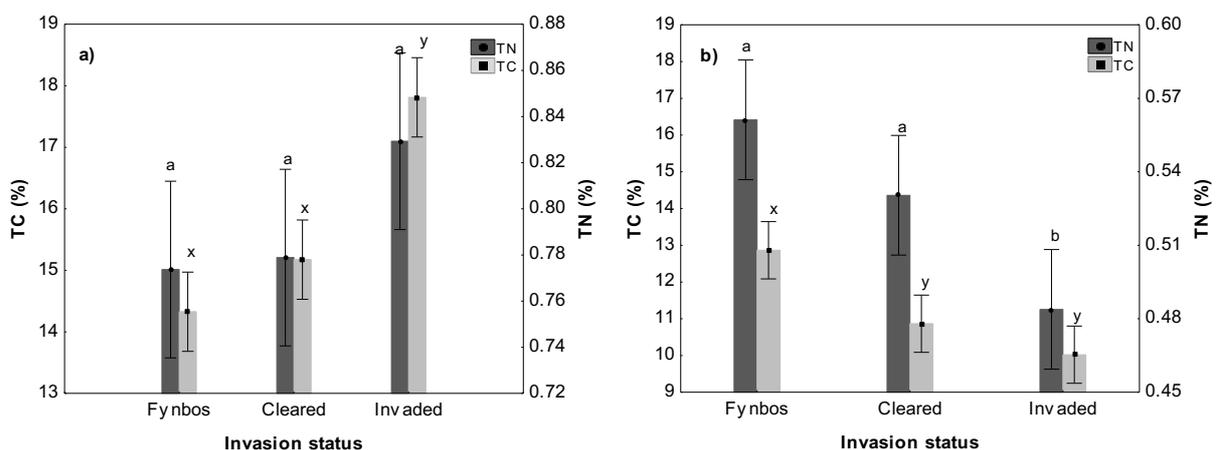


Figure 4.3. Total N and C in IS (< 63 μm) for the different invasion statuses at (a) Wit and (b) Du Toit's River. Mean values are indicated by bars and whiskers indicate $\pm 95\%$ confidence interval. The interaction between invasion statuses are based on one-way ANOVAs and letters on the bars indicate significant differences (LSD tests, $p < 0.05$).

Table 4.2. Spearman's rank correlation coefficient ρ (rho) for selected chemical and physical properties of IS. Correlation values were calculated from all available data and show only instances where comparisons could be made. Significant correlations are indicated at $p < 0.05$ (*) and $p < 0.01$ (**).

	TP (mg g ⁻¹)	TN (%)	TC (%)	C:N	Fe (mg g ⁻¹)	Al (mg g ⁻¹)	pH (H ₂ O)	Clay (%)
Temperature in water (°C)	0.71**	0.25**	0.08*	-0.36**	0.92**	N/A	-0.64**	0.66**
TP (mg g ⁻¹)		0.10**	-0.01	-0.45**	0.83**	0.32*	-0.07	0.79**
TN (%)		-	0.81**	N/A	0.15**	N/A	N/A	0.43**
TC (%)			-	N/A	-0.20**	N/A	N/A	0.25**
C:N ratio				-	-0.71**	0.03	N/A	-0.29**
Fe (mg g ⁻¹)					-	0.29**	-0.03	0.90**
Al (mg g ⁻¹)						-	-0.05	0.33*
pH (H ₂ O)							-	-0.12

4.4.3 Temporal and spatial dynamics of TP in infiltrated sediments

Significant seasonal trends were noted for TP in infiltrated sediments (IS) on the Wit River ($F_{[3, 18]} = 290.38$, $p < 0.001$; Figure 4.4a) with significantly higher concentrations in summer (1.63 ± 0.39 mg g⁻¹) relative to autumn (0.50 ± 0.11 mg g⁻¹), winter (0.32 ± 0.11 mg g⁻¹) and spring (0.52 ± 0.18 mg g⁻¹). TP differed significantly between invasion statuses (two-way ANOVA: $F_{[2, 6]} = 18.07$, $p < 0.01$; Figure 4.4b). Average over the sampling period, invaded sites had significantly higher total P concentrations (mean = 0.78 ± 0.51 mg g⁻¹) than fynbos sites (0.47 ± 0.33 mg g⁻¹, $p < 0.05$), but no significant difference was apparent between invaded and cleared sites (0.88 ± 0.69 mg g⁻¹, $p = 0.37$).

Similar spatial trends were observed at the Du Toit's River, with significant differences between invasion statuses ($F_{[2, 6]} = 6.04$, $p < 0.05$; Figure 4.4d). Fynbos sites had significant lower mean total P concentration than invaded ($p < 0.01$) and cleared sites ($p < 0.01$) with no significant differences between cleared and invaded sites ($p = 0.98$). TP concentrations differed significantly between invasion statuses with lowest concentration in the fynbos sites (0.41 ± 0.22 mg g⁻¹), followed by higher concentrations in the cleared (0.59 ± 0.27 mg g⁻¹) and invaded sites (0.59 ± 0.31 mg g⁻¹). Significant seasonal trends were observed for TP in IS ($F_{[3, 18]} = 46.10$, $p < 0.001$; Figure 4.4c) with significantly higher mean concentration in summer (0.98 ± 0.25 mg g⁻¹) compared to spring (0.46 ± 0.1 mg g⁻¹), autumn (0.40 ± 0.16 mg g⁻¹) and winter (0.41 ± 0.08 mg g⁻¹).

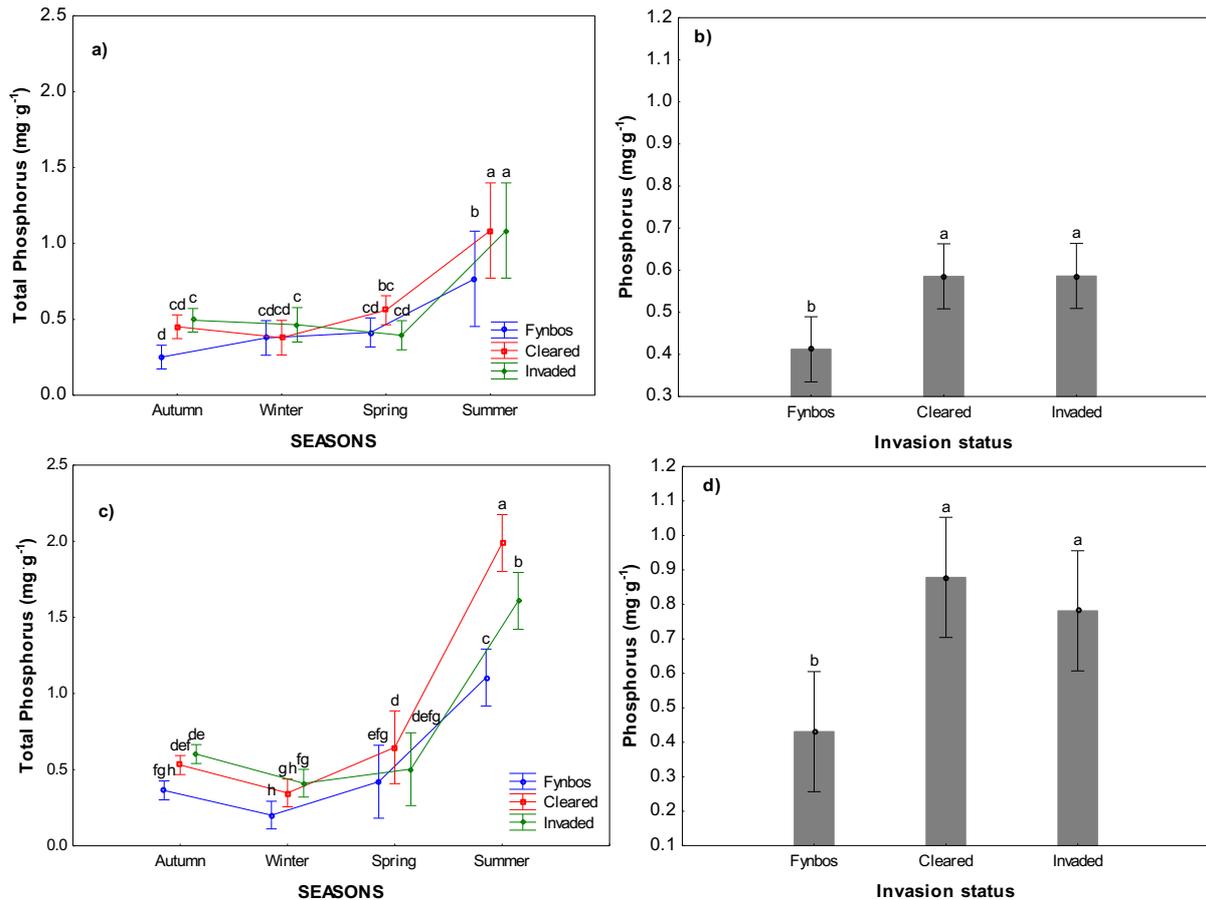


Figure 4.4. Total P concentrations in IS (< 63 μm). Mean total P for invasion statuses across seasons are depicted in Fig 4.4a at the Du Toit's River and Fig 4.4c at the Wit River. Mean total P concentration for the different invasion statuses are depicted in the bar graphs for the Du Toit's River (Fig 4.4b) and Wit River (Fig 4.4d). Different point symbols (Fig 4.4 a, b) and bars (Fig 4.4 band d) indicate mean values and whiskers the ± 95% confidence intervals. The interaction between seasons and invasion statuses were based on repeated measures ANOVAs and letters indicate significant differences (LSD tests, $p < 0.05$). In Fig. 4.4 b and d the letters on the bars represent significant differences (LSD tests, $p < 0.05$) based on one-way ANOVAs.

Total P (TP) concentrations in IS was significantly correlated with clay percentage, iron (Fe) and aluminum (Al) concentrations, regardless of invasion status (Table 4.2). TP in IS showed a significant inverse relationship with C:N ratios ($r^2 = -0.45$, $p < 0.01$; Table 4.2). TP concentrations had significant relationship with total N ($r^2 = 0.10$, $p < 0.01$), while pH (H₂O) had no effect on total P ($r^2 = -0.08$; $p = 0.11$; Table 4.2).

4.4.4 Spatial variation in nutrient content between invasion statuses

In this section, nutrient concentration measured in section 4.4.2 and 4.4.3 has been corrected for differences in clay percentage between sites. The method accounts for variations in particle size composition via the clay percentage of the samples. The general trend is that an increase

in nutrient concentration is associated with a decrease in particle size. This equates to an increase in specific surface area and the number of potential sorption sites with decreasing particle size (Bilotta and Brazier, 2008). It was found that particle size composition differed both spatially and temporally between the invasion statuses on the two rivers (see section 3.4.4, Figure 3.6 for illustration). Therefore, a particle size correction factor was applied to further minimize the influence of particle size variations on nutrient concentrations and to allow the direct comparisons between invasion statuses (Wang and Chen, 2000; Whiting et al. 2005; Stutter et al. 2009).

Particle-size effect correction test

The correlation analysis of the association between clay percentage and N, P and Fe concentrations for the individual sites from the respective rivers within this study indicated that particle size influence was highly significant across the nutrient concentrations (TN, TP and Fe) and sites concerned (Table 4.3). Therefore, particle size correction was considered appropriate. The correlations between TC and clay percentage for the individual sites from the rivers were variable across sites resulting in a non-linear relationship (Table 4.3). Thus, it was considered inappropriate to apply particle size correction to the TC concentrations.

Table 4.3. Spearman's rank correlation coefficients ρ (rho) and significance for clay percentage versus selected nutrient concentrations in IS (< 63 μm) from fynbos (F), cleared (C) and invaded (I) sites on the rivers. Significant correlations are indicated for $p < 0.05$ (*) and $p < 0.01$ (**).

Rivers	Du Toit's			Wit		
	F	C	I	F	C	I
TP (mgg^{-1})	0.63**	0.90**	0.47**	0.53*	0.97**	0.98**
TN (%)	0.62**	0.64**	0.26**	0.64**	0.79**	0.47**
TC (%)	0.36**	0.30**	0.08	0.17*	0.00	0.08
Fe (mgg^{-1})	1.00**	0.92**	0.90**	0.73*	0.99**	1.00**

Between site variations in Total N content

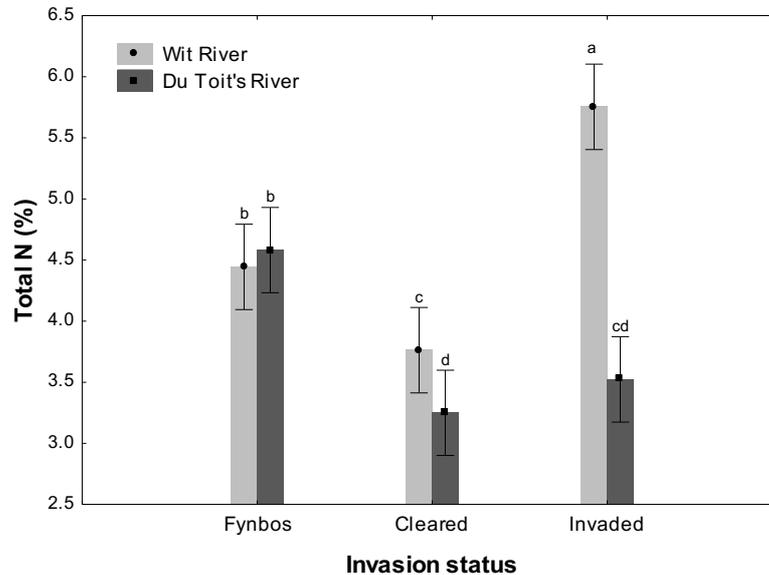


Figure 4.5. Mean total N in IS for invasion statuses across rivers, after total N concentrations were particle size corrected. Bars represent means and whiskers represent \pm 95% confidence intervals for TN. Significance levels (LSD post hoc test; $p < 0.05$) are indicated by different letters on the bars based on two-way ANOVAs.

Significant interaction between invasion statuses and rivers was observed (two-way ANOVA: $F_{[2, 12]} = 29.11$, $p < 0.01$; Figure 4.5). The invaded site on the Wit River had significantly higher TN content (mean = 5.75%) than the fynbos sites on the Wit (4.44, $p < 0.001$) and Du Toit's River (4.58%, $p < 0.001$), with no differences between the fynbos sites ($p = 0.56$). Conversely, the invaded site on the Du Toit's River had significantly lower TN content (3.52%) than the Fynbos sites on both the Du Toit's ($p < 0.001$) and Wit River ($p < 0.01$; Figure 4.5). Mean TN content for the cleared sites on the Wit and Du Toit's Rivers, 3.25 and 3.76 % respectively, was significantly lower than both fynbos sites ($p < 0.001$) and the invaded site on the Wit River ($p < 0.001$). The differences between both cleared sites and the invaded site on Du Toit's River was not significant for TN ($p > 0.05$) whereas significant differences were noted between the cleared sites ($p > 0.05$; Figure 4.5).

Between site variations in Total P content

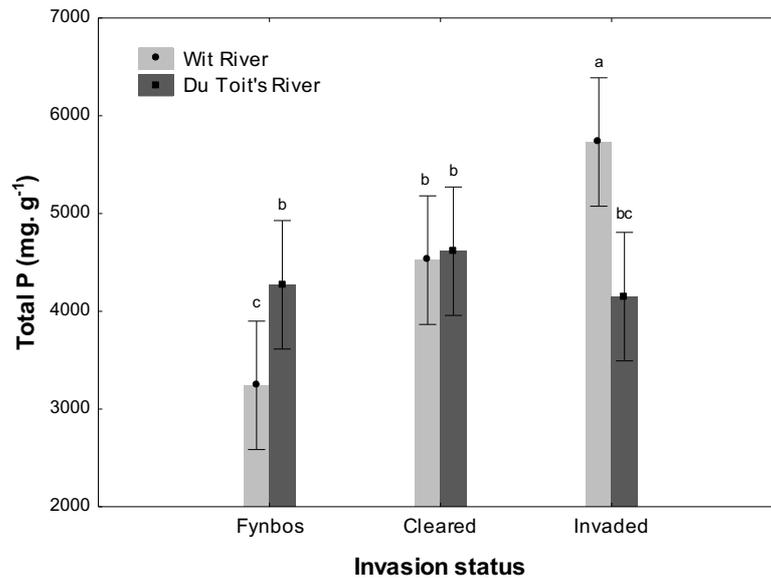


Figure 4.6. Mean total P in IS for invasion statuses across rivers, after total P concentrations were particle size corrected. Bars represent means and whiskers represent \pm 95% confidence intervals for TP. Significance levels (LSD post hoc test; $p < 0.05$) are indicated by different letters on the bars based on two-way ANOVAs.

Interaction between invasion statuses and rivers for total P in IS were significant (two-way ANOVA: $F_{[2, 12]} = 9.60$, $p < 0.01$; Figure 4.6). The invaded site on the Wit River had a significantly higher TP content (mean = $5731.73 \text{ mg g}^{-1}$) relative to the fynbos site on the Wit ($p < 0.001$) and the Du Toit's River ($p < 0.01$) with mean values of $3242.73 \text{ mg g}^{-1}$ and $4271.05 \text{ mg g}^{-1}$, respectively. The invaded site on the Du Toit's River had significantly lower TP content ($4150.01 \text{ mg g}^{-1}$) than the invaded site on the Wit River ($p < 0.01$; Figure 4.6). The mean TP content did not differ between the invaded site on the Du Toit's and the cleared sites on both the Du Toit's and Wit River ($p > 0.05$). However, TP content appeared to be significantly enhanced in the cleared site relative to the fynbos site on the Wit River ($p < 0.01$). Overall, no significant differences were found between invasion statuses on the Du Toit's River ($p > 0.05$; Figure 4.6).

4.4.5 Spatial and temporal variability in surface water nutrient concentrations

Ammonium (NH_4^+) concentrations were below detection limit of $10.0 \mu\text{g l}^{-1}$ in May 2016 at all sites. The highest mean NH_4^+ concentrations on the Wit and Du Toit's River was observed in November with maximum values of $86.0 \mu\text{g l}^{-1}$ and $80.3 \mu\text{g l}^{-1}$, respectively. Averaged over the

course of the sampling period, fynbos sites had the highest NH_4^+ concentrations (mean = $56.9 \mu\text{g l}^{-1}$) relative to the cleared ($45.9 \mu\text{g l}^{-1}$) and invaded sites ($44.0 \mu\text{g l}^{-1}$) on the Du Toit's River, however this was not significant (one-way ANOVA: $F_{[2, 24]} = 0.39$, $p = 0.68$; Table 4.4). The highest mean NH_4^+ concentrations on the Wit River was observed at the cleared site with a mean of $31.2 \mu\text{g l}^{-1}$ compared to the fynbos and invaded sites with means of 31.2 and $26.8 \mu\text{g l}^{-1}$, respectively (Table 4.4). Again, no significance was found between invasion statuses (one-way ANOVA: $F_{[2, 30]} = 0.14$, $p = 0.87$). This is supported by the substantial variation in NH_4^+ concentrations among the sites with different invasion statuses (Table 4.4).

Nitrate plus nitrite ($\text{NO}_3^- + \text{NO}_2^-$) concentrations were below the detection limits of $5.0 \mu\text{g l}^{-1}$ in May 2016, June 2016 and February 2017 at all sites. $\text{NO}_3^- + \text{NO}_2^-$ concentrations peaked in September at $9.0 \mu\text{g l}^{-1}$ at the Du Toit's River and $25.0 \mu\text{g l}^{-1}$ at the Wit River. Averaged over the course of the sampling period, $\text{NO}_3^- + \text{NO}_2^-$ concentrations on the Du Toit's River increased from the fynbos site (mean = $6.7 \mu\text{g l}^{-1}$) to the cleared and invaded sites with means of 7.7 and $8.3 \mu\text{g l}^{-1}$, respectively (Table 4.4). $\text{NO}_3^- + \text{NO}_2^-$ concentrations on the Wit River was highest at the cleared site ($13.7 \mu\text{g l}^{-1}$) compared to the fynbos and invaded sites with means around $7.7 \mu\text{g l}^{-1}$ (Table 4.4). However, too few $\text{NO}_3^- + \text{NO}_2^-$ data points were available for a valid statistical analysis.

Orthophosphate (PO_4^{3-}) concentrations ranged from below detection limit ($4 \mu\text{g l}^{-1}$) to $126.3 \mu\text{g l}^{-1}$ in the Wit River and below detection limit to $67.7 \mu\text{g l}^{-1}$ in the Du Toit's River. The invaded site in the Wit River showed relatively higher orthophosphate concentration with a mean of $26.6 \mu\text{g l}^{-1}$ compared to the cleared and fynbos sites with means of 10.0 and $4.8 \mu\text{g l}^{-1}$, respectively (Table 4.4). Despite the apparent spatial differences across invasion statuses, PO_4^{3-} concentrations were not significantly different (one-way ANOVA: $F_{[2, 21]} = 1.59$, $p = 0.23$). Similarly, the interaction between invasion status and PO_4^{3-} concentrations on the Du Toit's River was not significant (one-way ANOVA: $F_{[2, 12]} = 1.87$, $p = 0.20$). This was mainly due to the high variability among samples in the respective sites (Table 4.4). Nevertheless, mean PO_4^{3-} concentrations in the Du Toit's River were highest at the cleared site with a mean concentration of $32.5 \mu\text{g l}^{-1}$. Mean PO_4^{3-} concentrations in the fynbos and invaded sites was $10.7 \mu\text{g l}^{-1}$ and $22.4 \mu\text{g l}^{-1}$, respectively (Table 4.4). PO_4^{3-} concentrations in the Du Toit's River ranged from below detection limit ($4.0 \mu\text{g l}^{-1}$) to $26.7 \mu\text{g l}^{-1}$ in the fynbos site, $67.7 \mu\text{g l}^{-1}$ in the cleared site, and $33.3 \mu\text{g l}^{-1}$ in the invaded site.

Table 4.4. Comparison of mean values (\pm standard deviation) of nutrient concentrations (ammonium (NH_4^+), nitrate + nitrite ($\text{NO}_3^- + \text{NO}_2^-$), Nitrate (NO_3^-) and phosphate (PO_4^{3-}) between invasion statuses of the respective rivers. Mean and standard deviation are based on pooled data for autumn, winter, spring and summer. All concentrations are reported in $\mu\text{g l}^{-1}$. Note that no significant differences were observed (one-way ANOVA: LSD post hoc test; $p < 0.05$). See APPENDIX D for actual values across seasons.

Rivers	Invasion status	NH_4^+	$\text{NO}_3^- + \text{NO}_2^-$	NO_3^-	PO_4^{3-}
Wit	Fynbos	31.2 \pm 34.7	7.7 \pm 3.5	1.9 \pm 3.5	4.8 \pm 5.6
	Cleared	34.3 \pm 35.1	13.7 \pm 7.5	3.4 \pm 7.5	10.0 \pm 16.2
	Invaded	26.8 \pm 28.5	7.7 \pm 3.5	1.9 \pm 3.5	26.6 \pm 40.8
Du Toit's	Fynbos	56.9 \pm 39.7	6.7 \pm 3.6	1.3 \pm 1.2	10.7 \pm 14.6
	Cleared	45.9 \pm 34.4	7.7 \pm 3.5	0.7 \pm 1.5	32.5 \pm 25.2
	Invaded	44.0 \pm 31.8	8.3 \pm 4.1	1.5 \pm 2.3	22.4 \pm 10.7

4.5 Discussion

4.5.1 Temporal variability in sediment-bound nutrient concentrations

Significant temporal variations for nutrient (TC, TN and TP) concentrations in IS and SS were noted in the two streams that were assessed. The minimum TN and TC concentrations in IS coincided with high discharge and high sediment loads in winter. Schindler Wildhaber et al. (2012) documented a similar phenomenon in the Enziwigger River, Switzerland in which they found that low TN and TC concentrations during high sediment loads and high concentration associated with low sediment loads. This is suggested to be related to the dilution of TC and TN with coarse grained inorganic material (i.e. sand) obtained from sources within or proximal to the channel and lower specific gravity of organic material (e.g. leaf litter) that will remain suspended during high flows (Sear et al. 2008; Zhang et al. 2009; Schindler Wildhaber et al. 2012). While during low flows, organic material will settle out in the riverbed where it is mineralised (Schindler Wildhaber et al. 2012). The significant positive correlation between TN and clay percentage in IS supports this assumption. Nutrient rich clay particles also remain suspended longer than coarse grained sediments and only settle out during low energy flows (Schindl et al. 2005; Schindler Wildhaber et al. 2012).

Furthermore, the supply of materials to streams also varies significantly with the season due to the timing of leaf litterfall and soil nutrient cycling processes in the riparian zones. For example, peak litterfall rates reported for *A. mearnsii* invaded riparian zones in the Western Cape, South Africa coincided with the first seasonal rains in autumn while indigenous fynbos litter fall rates peak in summer (Railoun, 2017, unpublished data). Further, Naude (2012) found that nitrate (NO_3^-) concentrations were highest in autumn with maximum ammonium (NH_4^+)

concentrations observed in summer for riparian soils in the south-western Cape, South Africa. During high rainfall events in the wet season (autumn to spring), higher water levels increase the connectivity between the riparian zone and stream channel, enabling overbank flow to access, leaf litter and nutrient rich soils on the riverbanks and in riparian zones (Bull, 1997). Similarly, increased runoff in the catchment will connect distant sources of sediment and organic matter to the river channel (Walling, 1983; Campbell, 1985). As a result, these materials are delivered to the river channel during high flows and deposited on the riverbed during low flows.

The significant correlation between TC and TN in IS and SS indicate that TN is mainly derived from TC (Onstad et al. 2000; Schindler Wildhaber et al. 2012b). By assessing the C:N ratios in sediments, some inferences can be made concerning the origin of sediment and organic matter in sediments (Gordon and Goni, 2003; Zhang et al. 2007; Ramaswamy et al. 2008). Mean C:N ratios in infiltrated sediments ranged between 18.72 and 21.27 for the sites on the Wit River and between 20.80 and 22.98 for the sites on the Du Toit's River. The C:N ratios found in this study were generally higher compared to the C:N ratios reported for autochthonous material such as algae (< 10; e.g. Meyers, 1994; Schindler Wildhaber et al. 2012b). Consequently, the findings support the notion of a terrestrial origin (allochthonous) of sediments and organic matter in this study, especially during high flows. Therefore, riparian vegetation is considered as an important source of nutrients to these streams through leaf litter and root exudates (Helfield and Naiman, 2002). Furthermore, the significant inverse relationship between stream temperature and C:N ratios in IS provides evidence that the seasonal variation in stream temperature also has a pronounced effect on the rate of carbon mineralisation. Conen et al. (2008) also report that C:N ratios generally decrease with increase decomposition. Thus, the high TN concentration during the summer period could also be as result of a combination of warmer stream temperatures that stimulate microbial activity and the decomposition of large stocks of fynbos and *A. mearnsii* leaf litter in the riverbed.

TP concentrations in IS has been found to remain relatively constant throughout the high flow period (autumn to spring). The P concentrations during the period of low flow in summer were significantly higher than that during high flows. The difference in P concentrations during the high flows compared to the low flow period suggest that P sources in the rivers were replenished during the wet season and accumulated in the channel during the summer low flow conditions (Petersen et al. 2017). Phosphorus, particularly phosphates (PO_4^{3-}), are highly reactive ions and are readily absorbed onto negatively charged clay particles in soils and sediments (Makarova et al. 2004) through chemical bonding to positively charged particles such as iron and aluminium complexes (Addiscott, 1991; Tate et al. 1995; Gross, 2000) or a

substitution of phosphates for silicates in the clay structure (Harper et al. 2008). This often results in the accumulation of P in surface soils in riparian areas so that movement relies on the mobilization of soils by soil erosion (Stanley and Doyle, 2002; Chattopadhyay et al. 2005). In this study, no significant relationship was found between P and TC in infiltrated sediments, which suggests that P was most likely not derived from TC in these systems. Therefore, it is suggested that these systems receive most of their P through the mobilisation of sediment from catchment sources (Correl, 1998; Walling et al. 2008). Soil erosion rates increase during consecutive storm events due to high rainfall and overland flow, which may result in increased sediment delivery to streams (Hicks et al. 2000). Thus, the elevated P retained in riparian soils during summer and autumn as reported by Naude (2012) can readily be flushed to the stream through soil erosion during winter storm events (van der Perk, 2006; Petersen et al. 2017). However, these fine clay particles will remain in suspension and mix with inorganic mineral matter remobilised from the riverbed until conditions are favourable for it to settle out of the water column. The significantly higher P during summer may be attributable to the accumulation of enriched clay sized particles under low to zero flows conditions (Schindl et al. 2005; Bilotta and Brazier, 2008).

4.5.2 Spatial variability in sediment bound nutrient concentration

The two rivers in this study are predominantly underlain by Peninsula Formation quartzite sandstone. Soils in the fynbos biome derived from sandstone are generally characterized as acidic and nutrient poor (Rebelo et al. 2006; Soderberg and Compton, 2007; Cramer, 2010). Therefore, they contribute very little nutrients, specifically P, to the ecosystem. The mean P content of the soils in the fynbos range between 0.003 and 0.02 mg P g⁻¹ (Hawkins et al. 2005), with mean TN concentrations of 0.09% (Cowling and Holmes, 1992). Generally, total P concentration in infiltrated sediments (IS) increased from the upstream fynbos sites to the two downstream cleared and invaded sites on both rivers. The total N concentrations in IS increased similarly to total P, from upstream to downstream on the Wit River. The increase of total P and TN in IS from upstream to downstream appear to be related to two factors. First, nutrient concentrations, especially P, in sediment is largely influenced by particle size distributions in sediment, with finer particles (i.e. silts and clays; < 63 µm) tending to have large surface areas and higher sorption capacities (Horowitz, 1991; Peterson et al. 1996; Bilotta and Brazier, 2008). The particle size analysis showed a general trend of increased silt and clay percentage (< 63 µm) from the fynbos sites to invaded sites i.e., from upstream to downstream. Thus, the potential significance of the variation in the total P and total N concentrations in IS across sites could be accounted for by the difference in the proportion of

finer fractions (silts and clay). Secondly, the invasion of N₂-fixing and clearing at the two downstream sites probably resulted in changes in the P concentration observed. Previous studies have shown that N₂-fixing species such as *A. mearnsii* can enhance nutrient availability (N and P) in riparian surface soils (DeCant, 2008; Akamatsu et al. 2011; Naude, 2012) and input into streams (Volk et al. 2003; Compton et al. 2003; Mineau et al. 2011). Therefore, it is possible that the increase in N and P observed in the downstream sites could have resulted from the mineralisation of large stocks of *A. mearnsii* leaf litter in the riverbed and/or nutrient rich sediment export from the riparian zones. The conflicting results obtained from the Du Toit's River, where total N and TC decrease from up to downstream, were likely from the heterogeneity between sites in terms of channel and riparian zone characteristics. It can be deduced that the high riverbanks and wide stream channel with wide, gentle sloping riparian areas observed at the invaded and cleared sites on the Du Toit's River could have reduced the connectivity between riparian sources and the stream channel, consequently trapping N-rich leaf litter in the riparian zone. Thus, the mineralisation of N-rich leaf-litter stocks and nutrient transformation processes occurs in the riparian zone rather than in stream. The riparian zone at the Du Toit's River may have played a significant role as a buffer to N entering the stream.

In general, the average total N concentrations in IS and SS are relatively high with mean concentration of 0.70% in the Du Toit's River and 0.98% in the Wit River. Schindler Wildhaber et al. (2012) reported mean total N concentrations of 0.20% and 0.47%, respectively, with a mean of 0.33% in the Enziwigger River, which is impacted by different land uses (incl. forest, pasture and arable farmland) in Switzerland. The mean N concentrations reported for mountain fynbos soil were more than five times lower than the N concentrations observed in this study (Cowling and Holmes, 1992). Similarly, mean P values for the respective invasion statuses exceeds that of the background soil concentrations provided by Hawkins et al. (2005) and Stock and Verboom (2012). Likewise, the maximum P concentrations in riparian sediments along the highly impacted Berg River in the Western Cape, South Africa reported by Struyf et al. (2012) were found to be almost ten times lower than that found in the invaded sites of this study. The high total N and P concentrations reported in this study suggest that invasive N₂-fixing *A. mearnsii* and clearing may have played a significant role in sediment nutrient concentrations in these river systems.

4.5.3 Impacts of N₂-fixing *A. mearnsii* on nutrient contents in fynbos streams

Effects on sediment-bound nutrient content in rivers

In order to further characterise and quantify the impacts of invasion statuses, the particle-size correction method was used to minimize the influence of different particle size distributions among sites. This facilitated the direct comparison between samples from upstream sites with the often finer sediment samples from the downstream sites.

Total N and total P content in IS were significantly higher in the Wit River site invaded with *A. mearnsii* compared to sections surrounded by native fynbos vegetation. This was expected due to the higher input of nutrient-rich leaf litter from *A. mearnsii* trees in the invaded sites compared to the fynbos sites (see cf. Naude, 2012). The annual litterfall of invasive *A. mearnsii* was about 297.29 g m⁻², which was three times greater than the indigenous mountain fynbos riparian vegetation (68.88 g m⁻²) of the Western Cape, South Africa (Railoun, 2017, unpublished data). The foliar phosphorus and nitrogen content of the several *Acacia* spp. also exceeds that of litter from the indigenous vegetation (Witkowski, 1991; Stock et al. 1995). Large stocks of nutrient-rich (N and P) leaf litter from *A. mearnsii* break down faster in fynbos mountain streams, as was observed with leaf litter from fynbos species (Witkowski, 1991; Railoun, 2017, unpublished data). The close proximity of invasive trees and the low riverbanks resulted in a high degree of lateral connectivity between riparian area and stream channel at the invaded site on the Wit River. This could facilitate direct leaf litter input and lateral transport through surface runoff into streams, where it is stored and mineralized in the riverbed and consequently releasing nutrients under favourable conditions.

In addition, the impacts of invasive N₂-fixing on soil N stocks in riparian zones have been well documented in Central Japan (Akamatsu et al. 2011), in the semi-arid New Mexico (DeCant, 2008) and South Africa (Jacobs et al. 2012). In their study, Jacobs et al. (2012) found that soil N and P stocks in *Acacia*-invaded riparian zones were significantly enhanced compared to natural fynbos soils in riparian zones of the south-western Cape, South Africa. This has been mainly ascribed to the large accumulations of N-rich leaf litter with rapid mineralization rates that promotes enhanced nutrient concentration in surface soils of *Acacia*-invaded riparian zones (Naude, 2012). Similarly, in New Mexico, USA at the Rio Grande River, DeCant (2008) found elevated total nitrogen and available nitrogen in soils under N₂-fixing *Elaeagnus angustifolia* as opposed to soils under *Populus deltoides*, however, there was little focus on P. Total N and total P in sediment caught in the sediment infiltration baskets were positive and strongly correlated with clay particles found in this study. Therefore, substantial amounts of N and P may build up in *A. mearnsii* invaded riparian soils before being eroded

through surface runoff during consecutive high-intensity rainfall events into the invaded site on the Wit River. However, the impacts of invasion on riparian soil available P concentrations are less consistent compared to soil N. Several studies have shown that soils in *Acacia* invaded ecosystems tend to have higher available P concentrations than soils in indigenous fynbos systems (Witkowski, 1991; Yelenik et al. 2004). Witkowski and Mitchell, (1987) and Witkowski, (1991) found significantly higher available P in soils under *A. cyclops* and *A. saligna* stands compared to native fynbos, whereas Naude, (2012) did not find any significant difference between riparian soils under *Acacia* spp. and native fynbos, however, higher phosphatase levels. Therefore, this needs to be investigated to understand the impacts of *Acacia* invasions on phosphorus cycling and processes in nutrient poor soils of fynbos riparian areas and their holistic effects on in-stream biogeochemistry.

There is also evidence of a long-lasting biogeochemical effect with respect to total P content ~10 years after the removal of exotic N₂-fixing plants such as *A. mearnsii* on the Wit River. The mean values for total phosphorus content in infiltrated sediments were up to 40% higher than that in the fynbos site. Phosphorus in freshwaters is mostly available at a slightly acidic pH of 6-7 (Naiman and Bilby, 2001). In headwater streams with lower pH (< 6), phosphorus combines readily with metals (aluminium and iron) (Tate et al. 1995; Gross, 2000; Tessler et al. 2014) while at a higher pH it becomes associated with calcium. The correlation analysis used to examine the relationships between total P concentrations and related metals (Fe and Al) and clay fraction (< 3.9 µm) in IS, showed that there were significant and strong relationships between total P and Fe with clay. The strong correlation between iron (Fe) and total P concentrations in IS and the low pH values (< 6) in these systems suggests that P forms a less soluble complex with iron ions (hydroxides) and adheres to clay particles. Therefore, the strong absorptive reaction between P and fine sediment such as iron oxides on clay particles represents a substantial accumulating sink of P (Peterson, 1996; Lambers et al. 2006). The availability of P is therefore closely related with the capacity of iron to release or adsorb phosphorus (Søndergaard, 2007; Talib et al. 2016).

Unexpectedly, no differences were found in total N and total P content of IS between invasion statuses on the Du Toit's River. The contradicting results might be related to the between site variation in stream characteristics (e.g. stream width, size of streambed, stream temperature and riverbank height and riparian topography (see Chapter 3; Table 3.1). In contrast to the Wit River, a relatively wide stream channel with wide, gentle sloping riparian zones and high riverbanks characterizes the cleared and invaded site on the Du Toit's River. These stream characteristics could reduce the connectivity (supply-delivery) between the riparian zone and stream channel in these sections and consequently leading to efficient

trapping of *A. mearnsii* litter and nutrient-rich sediment in these areas (riparian zone and floodplains). This most likely leads to mobilisation (mineralisation) of nutrients through elevated soil processes (e.g. nitrification, denitrification and assimilation) to occur in the floodplain and riparian zones rather than in stream and, thus, preventing nutrients from entering the stream (Jacobs et al. 2007; Navoza, 2015). Moreover, the proximity of N₂-fixing invasive trees to the stream channel, abundance and density could also be considered as important controlling variables in the transfer of N-rich leaf litter and nutrients between riparian zones and streams. Therefore, channel characteristics and the proximity of the invasive *A. mearnsii* trees to the stream channel could potentially regulate stream nutrient fluxes.

Effects on nutrient concentrations in surface water

The increase in NO₃⁻ + NO₂⁻ concentrations in surface water is the most commonly documented effect of N₂-fixing invasive plants in streams on water quality (Compton et al. 2003; Mineau et al. 2011; Wiegner et al. 2013). In our study, there was a marginal increase in NO₃⁻ + NO₂⁻ concentrations in the cleared sites of both rivers. The greatest concentration increase was in cleared site on Wit River, where NO₃⁻ + NO₂⁻ increased by ~5 µg l⁻¹. The effects of N₂-fixing *A. mearnsii* trees on stream NH₄⁺ concentrations were similar to that reported for NO₃⁻ + NO₂⁻. In this study, small increases in NH₄⁺ concentrations were observed in the fynbos sites than to the invaded sites; however, these were not significant. Similarly, Compton et al. (2003) and Wiegner et al. (2013) reported that riparian areas invaded by N₂-fixing trees had no effects on surface water NH₄⁺ concentrations. Invasion statuses had less effect on stream water concentrations of NH₄⁺ and NO₃⁻ + NO₂⁻ than on fine sediment total N concentrations in this study. The high average stream temperature and the loss of N through denitrification and biological uptake likely explain the low available nitrogen concentrations between indigenous fynbos and invaded areas.

The effects of invasive N₂-fixing trees on P concentrations in surface water have received little attention and the effects that have been reported are equivocal. In this study, *A. mearnsii* invaded sections showed much higher concentrations of PO₄³⁻ than near pristine fynbos sections, but the difference was not significant due to the high variability among samples. The greatest concentration increase was in the *A. mearnsii* invaded site on the Wit River, where PO₄³⁻ increased by ~450 percent and at the cleared site on the Du Toit's River (~200%). Similar to my findings, Wiegner et al. (2013) reported no significant change in total dissolved phosphorus (TDP) concentrations in Hawaiian streams below sites invaded by *Falcataria moluccana* stands. A similar result was found in two western United States streams

draining *Elaeagnus angustifolia* invaded riparian zones, where PO_4^{3-} concentrations are unaffected. In contrast, Volk et al. (2003) reports significantly higher total phosphorus concentration in surface water of streams draining N_2 -fixing *Alnus rubra* infested riparian areas compared to streams surrounded by old growth coniferous forests in the Pacific Northwest.

In general, it is worth noting that surface water nutrient concentrations showed no significant longitudinal trends. Thus, this provides some evidence that surface water chemistry may not be affected by the invasion of *A. mearnsii* to the same degree as suspended sediment in this study. The high average stream temperature and stream size may govern the rate of in stream nutrient processes such as denitrification and P assimilation, and stimulate microbial activity, increasing nutrient turnover rates and their availability. The capacity of clay particles to bind and trap nutrients (Steege et al. 2001; Verstraeten and Poesen, 2002), enables sediments in rivers to become a substantial sink for nutrients, regulating its release to pore water and overlying surface water.

4.6 Conclusion

This study offered insight into the nutrient dynamics of fynbos mountain streams and the extent to which these nutrients are altered by *Acacia mearnsii* infestations and clearing. The data supports the hypothesis that invasion by N_2 -fixing *A. mearnsii* changes fine sediment nutrient content. The analysis of the temporal variation showed that nutrient concentrations in IS were predominantly influenced by timing of litterfall and variation in discharge sediment transport. The nutrient content (N and P) from the invaded site on the Wit River differed significantly from the two near pristine fynbos sites. The excessive amount of leaf litter with its high nutrient content from *A. mearnsii* might explain the increase total N and total P in the sediments. The total N content in sediments of the cleared site showed a trajectory towards uninvaded levels on the Wit River. However, the P content in cleared site on the Wit River was significantly higher than in the fynbos site suggesting that the clearing of IAPs could lead to a long-term effect on nutrient content in fine sediments. The high phosphorus and nitrogen content in sediments of the invaded and clearing sites can accumulate over time and contribute to eutrophication processes. Conversely, due to the difference in stream characteristics (i.e. riverbank height and stream width) at the invaded site on the Du Toit's River, invasive species had a lesser impact on nutrient content. Dissolved inorganic nutrients in surface water was similar in *A. mearnsii* invaded, cleared and fynbos reaches, suggesting that the largest effect was on sediment-bound nutrient contents in these fynbos streams. This result could potentially be attributed to the high average stream temperature and stream size that may promote in

stream nutrient processes to counterbalance additional nutrient inputs. Furthermore, this study adds to the growing knowledge of in-stream nutrient biogeochemistry by showing that different systems might respond differently to riparian disturbances. The rainfall frequency and intensity are constantly declining in the Western Cape, which could result in reduced stream flow and a lower dilution capacity of streams. Thus, we expect that the residence time of sediment-bound nutrients in these rivers to further increase.

Chapter 5

General discussion, Recommendations and Future research

5.1 Key findings

This study has investigated the temporal and spatial variability of relative sediment loads and associated nutrient properties in relation to *A. mearnsii* invaded and cleared ecotones in two fynbos headwater streams. Chapter 3 investigated temporal and spatial patterns in relative sediments loads. In general, the results indicate high intra-annual variability in the relative sediment loads, highlighting the complex control over sediment availability, transfer and depletion at varying times of the year. Periods of highest sediment transfer occurred during the autumn and winter months. These periods are responsible for the transferring on average around 84% of annual sediment load through the river systems. Sediment availability is at its minimum under low flow conditions, with the spring and summer months only contributing around 16% of the annual sediment load. Clockwise hysteresis loops were the most common pattern observed, indicating rapid delivery of sediment during moderate-high discharge and sediment exhaustion effects towards the end of the wet season, except in the invaded site of the Wit River.

The results suggest that the fynbos sites specifically can be characterised by having a restrictive sediment transfer regime with low relative sediment loads. Near-pristine reaches (fynbos) of the rivers are surrounded by dense indigenous fynbos vegetation cover, which effectively stabilise the riparian zones and riverbanks. In the Du Toit's River, sediment is predominantly derived from bank erosion and in-channel sediment deposits. This assumption is supported by the inverse (direct) correlation between catchment contributing area and suspended sediment yield. This can be attributed to the wide riparian zones and high riverbanks in the lower Du Toit's that could potentially reduce the sediment connectivity between river channel and catchment sediment sources, and thus minimize the impacts of invasive *A. mearnsii* and clearing on relative sediment loads. These patterns observed are the first for these rivers, and among the first using these sampling approaches for fynbos rivers.

The invaded site on the Wit River appeared to have increased the total annual relative sediment loads up to 260% during the course of the monitoring period relative to the fynbos site. Relative sediment loads were also increased at the site that has been previously cleared of woody invasive alien trees, but to a lesser extent than at the invaded site. The total annual relative sediment load increased up to 60% compared to the site surrounded by natural fynbos vegetation. This unexpected response in the cleared site on the Wit River could be related to

the slow rate of native vegetation and soil recovery in the riparian zone. Furthermore, it can be expected that sediments loads at the cleared site would return to uninvaded levels once native riparian vegetation has fully re-established and streams has flushed out the deposited sediments. It is clear that sediment dynamics should be taken into account when planning around clearing and restoration of invaded riparian zones takes place. The excessive sediment loads associated with the invaded and cleared sites can make a significant contribution to in stream nutrient stocks and fluxes.

Chapter 4 presented the nutrient concentrations in fine sediments ($< 63 \mu\text{m}$) associated with near pristine fynbos, *A. mearnsii* invaded and cleared sections of the two river systems. This chapter adds significantly to our knowledge of the biogeochemical behaviour in fynbos mountain streams. The analysis of temporal variation in nutrient concentrations reveals that C, N and P were significantly lower during the winter high flows compared to summer low flows. This suggests that during high flows, TC and TN are diluted with inorganic material associated with high sediment loads. Furthermore, this is accompanied by flushing of nutrient rich litter and sediment bound nutrients that have accumulated in riparian zones into the river systems. Total nitrogen content in infiltrated sediments of the invaded site on the Wit River were up to 30% and total phosphorus were up to 43% higher than in fynbos sites. This is consistent with Naude (2012), which reported that the phosphatase activity and N inputs appears to be altered in invaded riparian soils, though this was not evident in available phosphorus in soil. Additionally, the increase in TN and TP concentrations observed at the invaded site could be a result of a combination of nutrient availability in large stocks of leaf litter and the rapid mineralization rates of *A. mearnsii* leaf litter stored in the stream bed. Furthermore, a long lasting biogeochemical effect, or legacy effect, of invasion (even after clearing) was detected on the cleared site on the Wit River with regards to TP concentrations in infiltrated sediments. Mean total phosphorus concentration was 43% higher in the cleared site compared to the fynbos site on the Wit River. Total P concentrations also showed a significant relationship with iron and clay percentage in sediments. This might introduce a long lasting effect through the capacity of clay particles and iron ions to bind and trap nutrients in sediments of these rivers and become a substantial sink for P which can accumulate over time. The strong absorptive reaction between P and fine sediment such as iron oxides on clay particles may decrease the availability of P for plant uptake (Peterson, 1996; Lambers et al. 2006) and therefore depends largely on the capacity of iron to release P into overlying surface water and pore water (Søndergaard, 2007; Talib et al. 2016). In addition, understanding the coherent mechanisms between total P and Fe, could prove to be essential in understanding the potential contribution of P to eutrophication processes in these river systems. Sediment

nutrient stock size appears to be considerable, and should be taken into consideration when accounting for catchment nutrient models and management.

However, in the Du Toit's River, channel characteristics, and to some extent catchments characteristics, exerts a greater influence on the concentration of sediment-associated nutrients. The channel and catchment characteristics are primary influences over spatial variability in nutrient transport and delivery to streams as seen in the Du Toit's River. The invaded and cleared sites on the Du Toit's River were characterized by high terraces with wide, gentle sloping riparian zones which proved to exercise some control and regulation on stream nutrient content. These characteristics were found to exert a greater influence on the concentration of sediment-associated nutrients and showed that the impacts of IAP's and clearing might be a highly context-dependent response and depend on geomorphology and other factors. Therefore, general conclusions cannot be applied to other river systems without considering the catchment and channel geomorphological characteristics. Nevertheless, these results are useful when considering the whole-ecosystem impact of invasion of riparian ecosystems by N₂-fixing invasive trees and subsequent clearing, and in this respect is novel, and should be followed up by other similar studies.

5.2 Management implications and recommendations

This study was among the first to examine the temporal and spatial dynamics of sediments loads and associated nutrient concentrations in fynbos mountain streams, specifically with the context on invasive *A. mearnsii* and restoration activities in fynbos riparian ecosystems. The changes in sediment loads and nutrient concentrations in sediment as a result of invasion and clearing emphasise the need for consideration of abiotic changes with invasion and clearing, in addition to biotic modification. Active restoration and rehabilitation after the removal of the N₂-fixing alien plants may need to include nutrient and sediment management. These include mechanisms to mitigate sediment and nutrient export downstream. The improved understanding of how different catchments and channels route sediments and associated nutrients can provide insights into the application of site characteristics as a tool to mitigate and reduce the downstream export of sediment and nutrients.

Furthermore, the recent changes in the hydrological cycle (precipitation and stream flow regimes) may also raise concerns as it is predicted that rivers in the south-western Cape will have reduced stream flow (New, 2002). These changes in stream flow behaviour in catchments could result in a reduction of the rivers' transport and dilution capacity. Therefore, we can expect that the amount of sediment and nutrients that actually reaches the river

channel, relative to the dilution and transport capacity is likely to further enhance the residence and exposure time. This can ultimately have widespread impacts on aquatic ecosystems, negatively affecting water quality and in-stream aquatic biota (e.g. algal and macro invertebrates communities). Therefore, the flushing of sediments as recommended in Brown et al. (2004) needs to be considered as a management tool in riverine ecosystems where clearing activities are planned.

5.3 Limitations and future research

Generally, there is a dearth of information concerning the nutrients and sediment export to the hydrological networks that drain N₂-fixing invaded and cleared riparian ecosystems. Additionally, there is also a lack of understanding of the sediment and nutrient dynamics in fynbos mountain streams as a whole. These knowledge gaps have led to calls for the development of fluvial geomorphological frameworks that characterises temporal and spatial variability in sediment loads in these river systems. Therefore, it is recommended that further research needs to be undertaken to monitor the sediment regimes, combined with geomorphological surveys to assign processes responsible for erosion and sediment transfer.

An addition to this study would be to relate sediment loads to immediate clearing of IAPs to show whether the two are explicitly linked and to monitor whether the situation worsen or improves. The sediment sampling techniques (e.g. time-integrated samplers) used in this study proved to be an effective and low cost method to monitor the spatial and temporal extent of sediment loads. Therefore, continuing the use of time-integrated samplers over an extended period would provide stronger spatial trends and long-term trends in sediment transport rates. This will provide valuable information on how sediment supply and transport processes operate and change over time. Additional research could focus on a range of channel types (unimpacted and impacted) and may use sediment tracing or fingerprinting techniques to elucidate potential sediment sources (e.g. floodplain, bank, catchment sources). The application of sediment fingerprinting techniques is widely used and can be useful in providing an informed view of the disturbances in catchments (natural or human induced). The variety of the channel types will also shed light on how different channels types route nutrients through the catchment and the contribution of different land uses to nutrients in streams. The understanding of how different catchments and channels route their pollutants can provide insight on how use of site characteristics as a tool to mitigate and reduce the downstream export of sediment and nutrients. Another aspect that needs to be addressed is the behaviour of geochemical coupling between total P and Fe found in this study, in order to understand

the fate and release of phosphorus in the river systems and the potential contributions to eutrophication processes.

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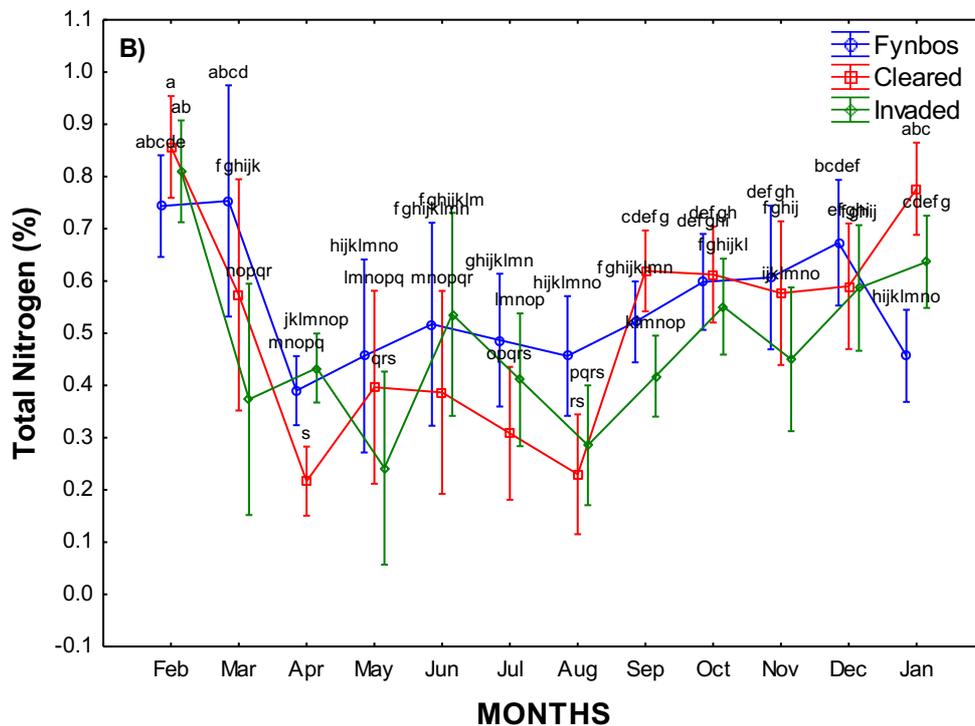
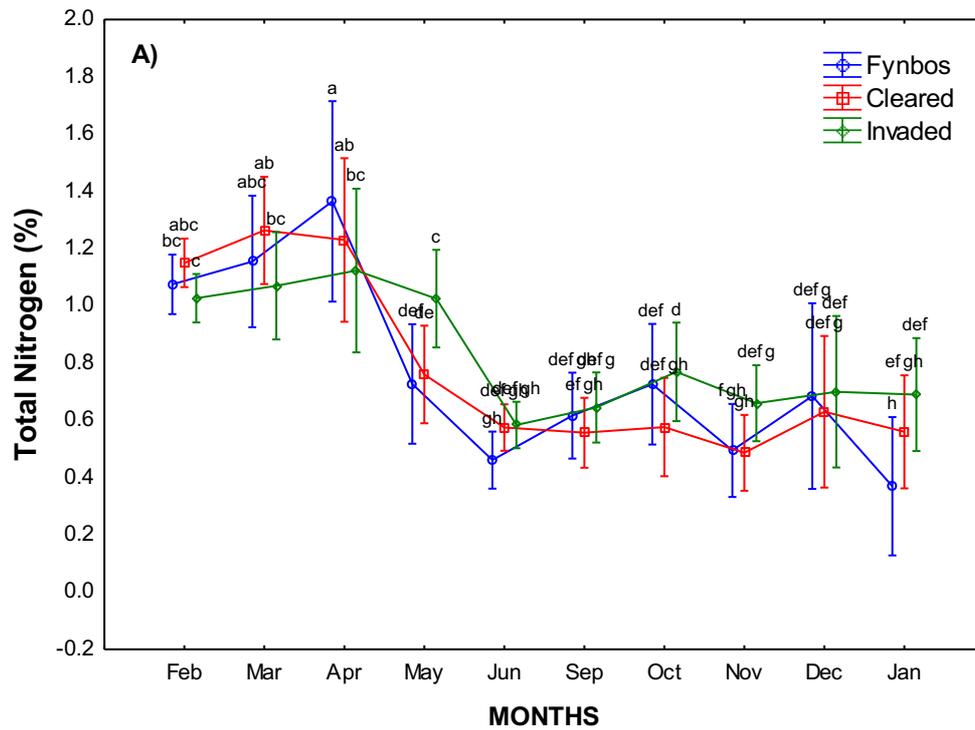
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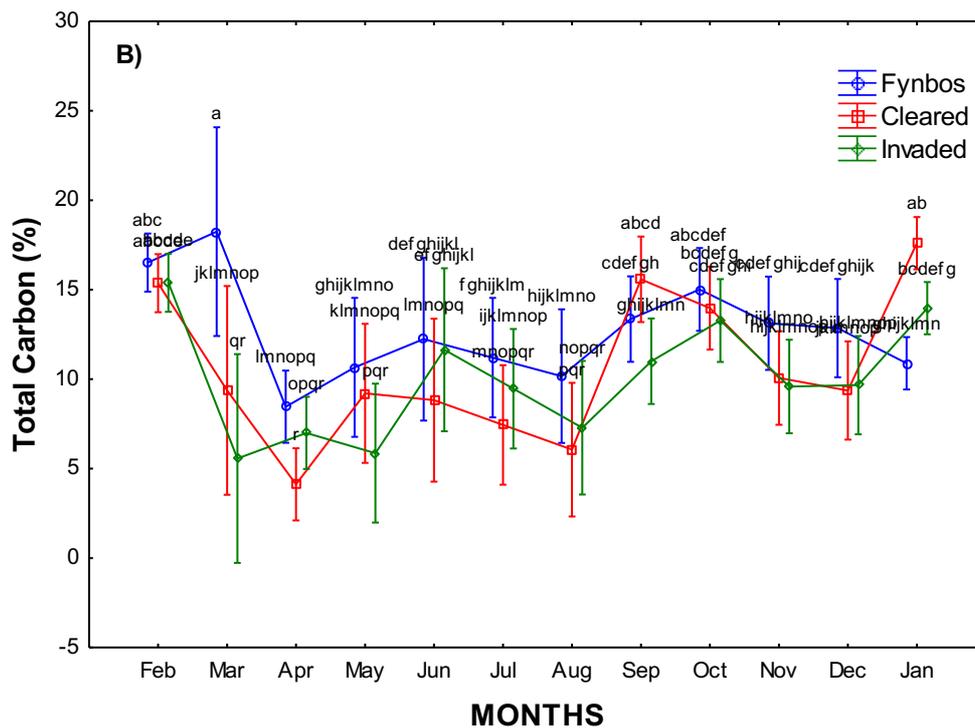
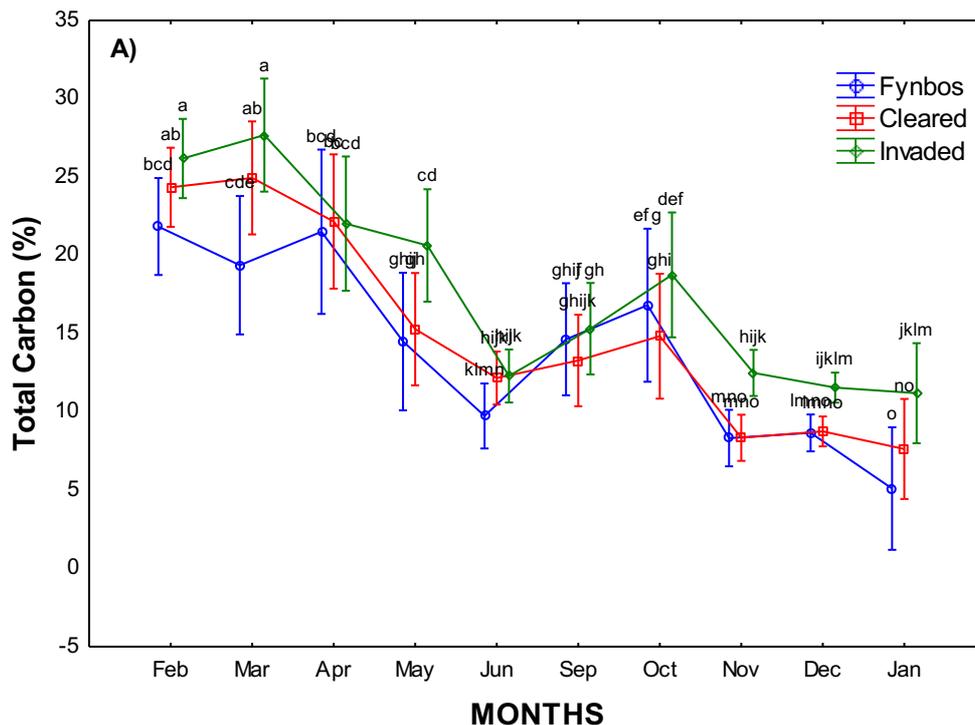
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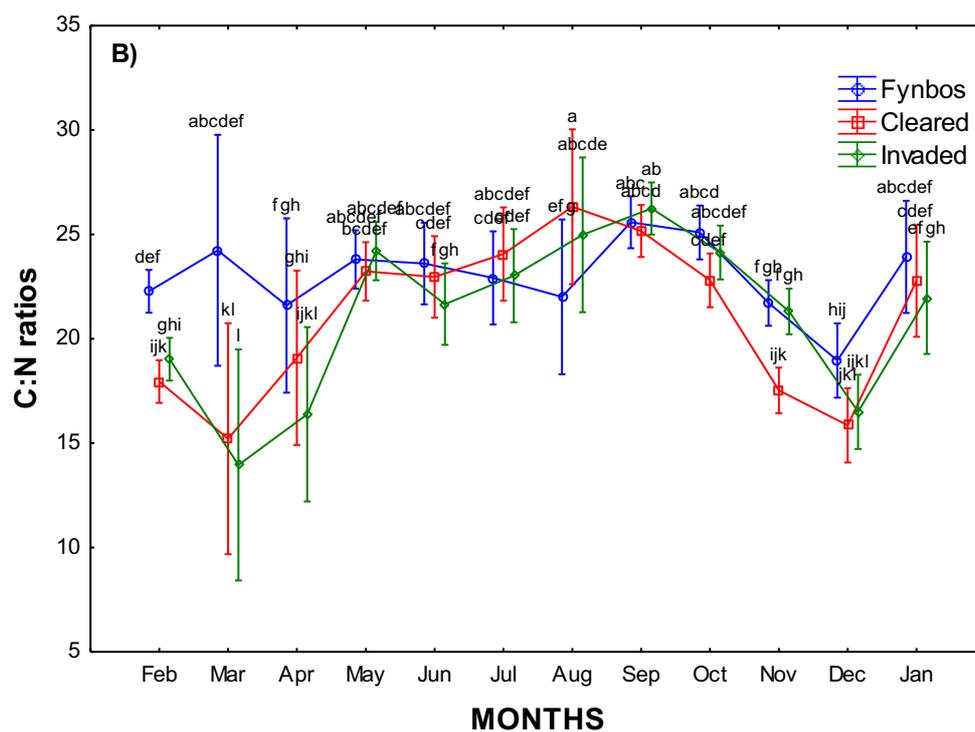
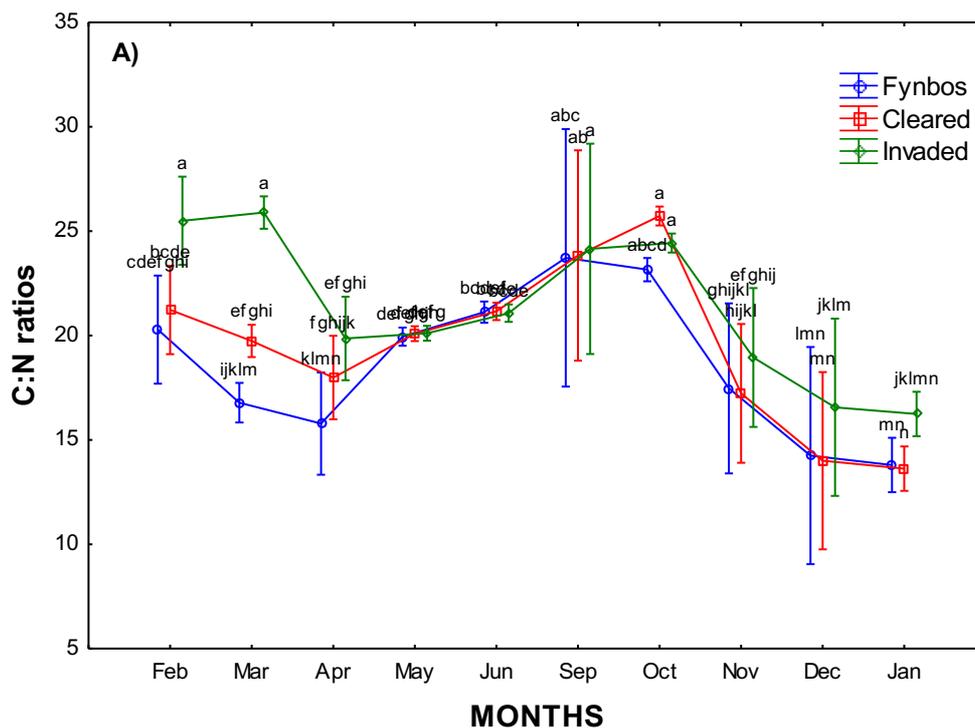
APPENDICES



Appendix A: Total nitrogen concentration in IS for each of the Rivers. Mean values indicated by different point symbols, and whiskers \pm 95% confidence interval. Letters denote significant differences between sites based on a one-way ANOVA for **A)** Wit River ($F_{[18, 45]} = 1.82, p=0.05$) and **B)** Du Toit's River ($F_{[22, 66]} = 3.43, p < 0.001$). Invasion statuses are represented by different colours: fynbos (blue), invaded (green), and cleared (red).



Appendix B: Total carbon concentrations in IS for each of the Rivers. Mean values indicated by different point symbols, and whiskers \pm 95% confidence interval. Letters denote significant differences between sites based on a one-way ANOVA for **A)** Wit River ($F_{[18, 45]} = 1.34, p = 0.21$) and **B)** Du Toit's River ($F_{[22, 66]} = 3.21, p < 0.001$). Invasion statuses are represented by different colours: fynbos (blue), invaded (green), and cleared (red).



Appendix C: C:N ratios in IS for each of the Rivers. Mean values indicated by different point symbols, and whiskers \pm 95% confidence interval. Letters denote significant differences between sites based on a one-way ANOVA for **A)** Wit River ($F_{[18, 45]} = 2.12, p > 0.05$) and **B)** Du Toit's River ($F_{[22, 66]} = 3.64, p < 0.001$). Invasion statuses are represented by different colours: fynbos (blue), invaded (green), and cleared (red).

Appendix D: Chemical properties in IS for invasion status (fynbos, invaded and cleared) on the (i) Du Toit's River and (ii) Wit River across seasons. The data reflect means \pm standard deviations.

Sediment properties	Season	Fynbos	Invaded	Cleared
TC (%)	Autumn	12.45 \pm 5.30	6.14 \pm 1.92	7.57 \pm 3.88
	Winter	11.20 \pm 2.55	9.47 \pm 3.70	7.44 \pm 3.16
	Spring	13.83 \pm 2.19	11.29 \pm 2.10	13.21 \pm 2.64
	Summer	13.42 \pm 3.04	13.01 \pm 2.71	14.11 \pm 3.80
TN (%)	Autumn	0.53 \pm 0.21	0.35 \pm 0.11	0.40 \pm 0.19
	Winter	0.49 \pm 0.08	0.41 \pm 0.17	0.31 \pm 0.15
	Spring	0.58 \pm 0.10	0.47 \pm 0.08	0.60 \pm 0.05
	Summer	0.62 \pm 0.15	0.68 \pm 0.11	0.74 \pm 0.14
TP (mg g ⁻¹)	Autumn	0.25 \pm 0.16	0.49 \pm 0.12	0.45 \pm 0.06
	Winter	0.38 \pm 0.07	0.46 \pm 0.07	0.38 \pm 0.10
	Spring	0.41 \pm 0.03	0.40 \pm 0.11	0.56 \pm 0.01
	Summer	0.77 \pm 0.09	1.09 \pm 0.36	1.09 \pm 0.08
C:N ratios	Autumn	23.21 \pm 1.57	18.18 \pm 5.75	19.18 \pm 4.28
	Winter	22.81 \pm 1.76	23.32 \pm 3.01	24.64 \pm 2.43
	Spring	24.13 \pm 1.96	23.89 \pm 2.33	21.83 \pm 3.43
	Summer	21.72 \pm 2.71	19.16 \pm 2.47	18.86 \pm 3.28
Fe (mg g ⁻¹)	Winter	11.23 \pm 0.61	27.13 \pm 9.11	13.75 \pm 3.58
	Spring	15.50 \pm 4.88	32.53 \pm 10.97	46.18 \pm 4.24
	Summer	109.37 \pm 23.79	204.51 \pm 8.08	223.84 \pm 25.51
Al (mg g ⁻¹)	Winter	5.68 \pm 1.53	8.11 \pm 1.62	6.16 \pm 0.90
	Spring	4.32 \pm 0.57	7.55 \pm 1.01	6.66 \pm 0.46
pH (H ₂ O)	Winter	4.99 \pm 0.08	4.98 \pm 0.14	4.83 \pm 0.06
	Spring	4.87 \pm 0.21	4.60 \pm 0.07	4.58 \pm 0.10
Ec (μ scm ⁻¹)	Winter	72.1 \pm 4.16	73.97 \pm 0.40	72.83 \pm 2.72
	Spring	72.67 \pm 2.19	72.47 \pm 2.03	73.80 \pm 0.61

(i)

Sediment properties	Season	Fynbos	Invaded	Cleared
TC (%)	Autumn	18.85±4.51	23.45±4.19	20.80±4.64
	Winter	10.22±1.08	12.30±1.33	12.17±1.01
	Spring	12.74±3.99	15.52±3.56	12.15±3.21
	Summer	12.72±8.43	16.31±7.62	13.58±8.14
TN (%)	Autumn	0.53±0.37	0.35±0.17	0.40±0.25
	Winter	0.49±0.04	0.41±0.06	0.31±0.05
	Spring	0.58±0.13	0.47±0.12	0.60±0.07
	Summer	0.62±0.35	0.68±0.20	0.74±0.28
TP (mg g ⁻¹)	Autumn	0.36±0.04	0.60±0.06	0.53±0.02
	Winter	0.20±0.04	0.41±0.05	0.35±0.09
	Spring	0.42±0.06	0.50±0.02	0.65±0.29
	Summer	1.10±0.64	1.61±0.15	2.00±0.16
C:N ratios	Autumn	17.42±2.10	22.00±3.13	19.27±1.08
	Winter	21.81±1.19	21.07±0.11	21.16±0.42
	Spring	21.66±3.25	22.51±3.81	22.26±4.00
	Summer	16.62±4.31	19.44±4.60	16.28±3.90
Fe (mg g ⁻¹)	Winter	8.92±1.22	6.77±0.50	18.66±10.63
	Spring	28.93±8.58	13.00±2.18	40.10±25.84
	Summer	286.76±18.38	228.09±40.15	300.45±4.60
Al (mg g ⁻¹)	Winter	2.67±0.42	4.96±0.79	8.01±0.64
	Spring	5.66±0.57	5.88±0.40	6.40±1.32
pH (H ₂ O)	Winter	4.69±0.08	4.57±0.08	4.58±0.14
	Spring	4.82±0.24	4.39±0.17	4.47±0.10
Ec (μscm ⁻¹)	Winter	74.87±0.06	75.10±0.17	75.13±0.06
	Spring	74.70±0.10	74.33±0.12	74.67±0.31

(ii)

Appendix E: Comparison of average \pm standard deviation of nutrient concentrations in surface water for invasion statuses (fynbos, invaded and cleared) on (i) Du Toit's River and (ii) Wit River. All results are reported in $\mu\text{g l}^{-1}$.

Surface water quality	Season	Fynbos	Invaded	Cleared
Ammonium	Autumn	<10	<10	<10
	Winter	67.33 \pm 59.21	42.00 \pm 32.05	51.33 \pm 38.21
	Spring	66.67 \pm 8.50	80.33 \pm 12.06	69.00 \pm 7.81
	Summer	36.67 \pm 23.01	9.67 \pm 2.08	17.33 \pm 4.62
Nitrate + nitrite	Autumn	<5	<5	<5
	Winter	6.00 \pm 0.00	8.00 \pm 3.00	<5
	Spring	7.33 \pm 1.53	7.67 \pm 1.53	7.33 \pm 0.58
	Summer	<5	<5	<5
Nitrate	Autumn	<5	<5	<5
	Winter	4.00 \pm 0.00	5.00 \pm 0.00	4.00 \pm 0.00
	Spring	4.00 \pm 0.00	5.00 \pm 0.00	4.00 \pm 0.00
	Summer	<1	<1	<1
Orthophosphate	Autumn	17.78 \pm 15.40	47.55 \pm 17.90	29.27 \pm 7.03
	Winter	<4	<4	<4
	Spring	<4	<4	<4
	Summer	<4	<4	<4

(i)

Surface water quality	Season	Fynbos	Invaded	Cleared
Ammonium	Autumn	<10	<10	15.36 \pm 1.07
	Winter	16.67 \pm 1.15	18.33 \pm 1.15	17.00 \pm 4.36
	Spring	83.00 \pm 17.09	68.00 \pm 18.36	86.00 \pm 24.33
	Summer	14.67 \pm 0.58	12.00 \pm 2.65	12.33 \pm 1.15
Nitrate + nitrite	Autumn	<5	<5	<5
	Winter	<5	<5	<5
	Spring	7.67 \pm 1.53	7.67 \pm 1.15	13.67 \pm 9.87
	Summer	<5	<5	<5
Nitrate	Autumn	<5	<5	<5
	Winter	<5	<5	<5
	Spring	4.67 \pm 0.58	6.33 \pm 0.58	4.33 \pm 3.21
	Summer	<5	<5	<5
Orthophosphate	Autumn	<4	73.72 \pm 52.57	<4
	Winter	<4	<4	<4
	Spring	<4	10.33 \pm 9.24	<4
	Summer	7.67 \pm 0.58	11.50 \pm 0.50	10.67 \pm 0.58

(ii)