THE IMPACT OF HIGH RAINFALL AND FLOOD EVENTS ON *EUCALYPTUS CAMALDULENSIS* DISTRIBUTION ALONG THE CENTRAL BREEDE RIVER.

by
Gideon Raath

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Supervisor: Mr. N.K. Poona

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DECLARATION

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SUMMARY

Eucalyptus camaldulensis Dehnh., or River Red Gum, is a commercially valuable yet recognised invasive alien plant (IAP) of riparian zones throughout South Africa. The invasive potential of *E. camaldulensis* is widely recognised, with specific regulations aimed at the management of *E. camaldulensis*. *E. camaldulensis* is known to use large amounts of water, reduce biodiversity, change river morphology and impact hydrological regimes of rivers. In the native range throughout Australia, *E. camaldulensis* displays a distinct relationship between rainfall, and flood events, for seed dispersal, germination and establishment, and consequently spatial extent, yet little is known about the relationships in the South African context. The aim of this project was to assess the impact of high rainfall and flood events on the establishment and distribution of *E. camaldulensis* along the Middle Breede River, between Worcester and Swellendam in the Western Cape, by establishing the current spatial extent of *E. camaldulensis* along the river, identifying flood events since 1950 and evaluating the impact rainfall and flood events had on the spatial extent thereof. Aerial imagery, rainfall, discharge and river level data was obtained dating back to 1980, as well as field data comprising of GPS-bounding of *E. camaldulensis* stands. Additionally, density measurements were obtained and interviews conducted with land users. Spatial analysis of aerial imagery, coupled with perimeter (GPS) data and density data were used to conduct spatio-temporal analysis, employing GIS and conventional statistical approaches to address the various objectives. Results indicated *E. camaldulensis* stands had a small overall increase in spatial extent since 1980. Flooding and rainfall events coincided with an increase in occurrence of *E. camaldulensis* with elevated river levels and frequent flooding, while spatial variation of this relationship was observed. The hydrological regime of the Breede River coincides with a slow increase in spatial extent of *E. camaldulensis* stands, but no affirmation of a positive real-world relationship was possible using the available data. Results further suggested, based on the current age class composition, that existing stands originated roughly during 1980, possibly due to commercial forestry related seeding into the river. Reduced fragmentation between stakeholders, educational programmes and improved reporting systems were recommended for improved IAP management within the area.

KEYWORDS: *Eucalyptus camaldulensis,* River Red Gum, Middle Breede River, Invasive alien plant, rainfall, flood, Western Cape, Department of Agriculture.

OPSOMMING

Eucalyptus camaldulensis, of Rooibloekom (RB), is ‘n waardevolle kommersiële, maar erkende indringer plantspesie (IP) wat veral oewersones in Suid-Afrika indring. Die indringerpotensiaal van *E. camaldulensis* is welbekend, en spesifieke regulasies, gemik op die bestuur van RB en ander spesies is reeds aangeneem. *E. camaldulensis* is veral bekend vir sy hoë watergebruik, sy vermindering van biodiversiteit, sy vermoë om riviervorme te verander en sy algehele impak op die hidrologiese patroon van riviere waarmee dit in aanraking kom. In sy oorspronklike verspreidingsgebied in Australië toon *E. camaldulensis* ‘n bepaalde verhouding tussen reënval en vloedgebeurtenisse vir saadverspreiding, ontkieming en vestiging en derhalwe die ruimtelike verspreiding van die spesie; alhoewel hierdie verhouding in die Suid-Afrikaanse konteks steeds redelik onverduidelik bly. Die doelwit van hierdie studie was dus om die impak van hoë reënval en vloedgebeurtenisse op die ruimtelike verspreiding en vestiging van *E. camaldulensis* teenaan die Middel Breëde Rivier, spesifiek tussen Worcester en Swellendam, te evalueer. Hierdie doelwit was bereik deur die historiese ruimtelike verspreiding teenaan die rivier te meet, hoë reënval en vloedgebeurtenisse vanaf 1980 te identifiseer, en die huidige verspreiding en omtrek met GPS te meet. Digtheidafmetings, sowel as onderhoude met belanghebbendes teenaan die rivier was ook opgeneem. Visuele interpretatasie van lugfotos, sowel as omtrek (GPS) en digtheid-data was gebruik om ruimtelike analise uit te voer, deur die gebruik van GIS en konvensionele statistiese metodes, ten einde die doelwitte te evalueer. Resultate dui aan dat *E. camaldulensis* areas ‘n klein algemene groei getoon het sedert 1980. Hoë-reënval en gereëlde vloedgebeurtenisse het ook gepaard gegaan met ‘n groei van *E. camaldulensis* oppervlak, alhoewel hierdie verhouding ruimtelike variasie getoon het, met ‘n algemene groei patroon gemerk oor die volledige studietydperk. Ook geen stimulerende verhouding kon vanuit die beskikbare data bevestig word nie. Addisionele resultate het aangedui dat die verspreiding van *E. camaldulensis* ongeveer 1980 onstaan het, moontlik as gevolg van kommersiële bosbou-aanplanting en verwante saadverspreiding in die rivier vanaf daardie tyd. Aanbevelings ten opsigte van verbeterde indringerbestuur sluit in die beperking van huidige fragmentasie tussen belanghebbendes en betrokke verwyderingsorganisasies, addisionele onderrigprogramme sowel as die verbetering van terugvoersisteme.

TREFWOORDE: *Eucalyptus camaldulensis*, Rooibloekom, Middel Breëde Rivier, Indringer Plant, Reënval, vloede, Wes-Kaap, Departement van Landbou.

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ACRONYMS AND ABBREVIATIONS

Biocontrol	Biological control
BOCMA	Breede-Overberg Catchment Management Agency
CARA	Conservation of Agricultural Resources Act (43 of 1983)
CMA	Catchment management area
DoA	Department of Agriculture
DWA	Department of Water Affairs
ECA	Environmental Conservation Act (73 of 1989)
IAP	Invasive alien plant/s
IAS	Invasive alien species
IAPS	Invasive alien plant species
LW	Large wood
Mamsl	Metres above mean sea level
NEMA	National Environmental Management Act (107 of 1998)
NEMBA	National Environmental Management: Biodiversity Act (10 of 2004)
NEMPAA	National Environmental Management: Protected Areas Act (57 of 2003)
NWA	National Water Act (36 of 1998)
OS	Oxidative stress
RRG	River Red Gum (*Eucalyptus camaldulensis*)
RWC	Relative water content
SA	South Africa
WfW	Working for Water
WMA	Water management area

CHAPTER 1 INTRODUCTION

1.1 BACKGROUND

Globally, *Eucalyptus camaldulensis Denh.*, commonly known as River Red Gum (RRG), is known to establish along riverine environments (Di Stefano 2001), such as the rivers of the Murray-Darling basin in Australia (Butcher, McDonald & Bell 2009) and are known to prefer perennial, seasonal and intermittent watercourses (Butcher, McDonald & Bell 2009; Henderson 2007). In their native range of Australia, they have shown a dependence on flood events to supplement water requirements during drought (Roberts & Marston 2000). Although germination and establishment are not dependent on flooding, certain high rainfall events, or flood events, can be highly beneficial to their germination, establishment, growth and long term survival (Jensen, Walker & Paton 2008; Robertson, Bacon & Heagny 2001; Steinfeld & Kingsford 2008). Flood timing has been shown to influence germination success (Roberts & Marston 2000). Survival of *E. camaldulensis* has also been linked to the inundation caused by infrequent flooding (Steinfeld & Kingsford 2008), and as such the hydrological regime has an impact on the establishment and succession of this species (Roberts & Marston 2011).

E. camaldulensis displays a complementary recruitment strategy, where germination is initiated by rainfall, and seed dispersal is facilitated through flooding (Jensen, Walker & Paton 2008). Propagules are spread and germination promoted via such events. 'Maintenance' recruitment, or recruitment at a rate sufficient to replenish existing population numbers, is often observed with the event of rainfall, but is usually localised (Jensen, Walker & Paton 2008). Peak recruitment, where larger amounts of seedlings become established than is required for maintenance, can be initiated by flooding events which spread seed over longer distances (Jensen, Walker & Paton 2008). For example, the Breede Valley in the Western Cape, which is currently invaded by *E. camaldulensis*, experiences frequent flooding, observed by twelve large flood events between 2003 and 2008 alone (Holloway et al. 2010), and during each year frequent smaller flood events occur during the winter months. Subsequently, the hydrological regime of the Breede catchment may potentially influence the succession of *E. camaldulensis* locally, in much the same manner as observed in Australia.

South African authorities have recognised the threat invasive alien plant species (IAPS) pose to the natural environment and in particular, natural resources such as water (Richardson & Van Wilgen 2004; Van Wilgen et al. 2001). In a country where the national mean annual precipitation is only 480 mm (Van Rensburg et al. 2011), adaptive resource planning is required in order to allow for sufficient water resources (Warburton, Schulze & Jewitt 2011). Invasive alien plant (IAP) management, such as thinning or clearing, has been shown to lead to partial recovery of indigenous vegetation (Ruwanza et al. 2012), and programmes have been put in place to manage such invasions (Turpie, Marais & Blignaut 2008).

Currently, the relationship between *E. camaldulensis* and ecosystem function, river geomorphology and stream flow reductions are poorly understood (Ruwanza et al. 2014). Ecological research conducted has broadly been on the topics of riparian restoration after invasive clearing (Holmes et al. 2008; Holmes, MacDonald & Juriz 1987), catchment prioritisation (Forsyth, Le Maitre & Van Wilgen 2009), fire interactions (Van Wilgen et al. 1992), modelling the spread of IAPs (Higgins, Richardson & Cowling 1996), characteristics of invasiveness (Rejmánek & Richardson 1996), hydrological impact of IAPs (Le Maitre 2004; Le Maitre et al. 2002) as well as the economic impact of IAPS (Van Wilgen et al. 2001).

Despite the large volume of research conducted on IAPS in the local context, there is still much to learn. Ruwanza et al. (2013) points out that little research has been conducted into species dynamics after clearing, for example. Specifically, our understanding of *E. camaldulensis* invasion extent and impacts require further investigation (Tererai et al. 2013). Common themes related to *E. camaldulensis* specifically is the recognition of the species as a riparian invader, particularly in the Western Cape (Forsyth et al. 2004), and subsequently the various detrimental impacts that *E. camaldulensis* has on biodiversity (Tererai et al. 2013) and available water (Van Wilgen, Nel & Rouget 2007).

Clearing organisations such as the Department of Agriculture (DoA) and Working for Water (WfW) need accurate, current, fine-scale spatial information on invasive alien plants, as part of their management objectives (Röscher 2012, Pers com). Understanding the relationship between the hydrological regime and *E. camaldulensis* could allow for greater understanding of the IAP problem in South Africa, leading to better environmental resource management in conserving our environment, and preserving the ecosystem services we gain therefrom.

1.2 PROBLEM STATEMENT

IAPS are regarded as harmful to ecosystems, and pose a threat to the availability of natural resources throughout the country (Shackleton et al. 2007). *E. camaldulensis* is one such species, which is known to invade riparian zones, consume large amounts of water, encroach on agricultural land, change ecosystem dynamics (Forsyth et al. 2004) and reduce biodiversity (Vila et al. 2011). This species has been in the Western Cape since 1880 (Bennett 2014).

While *E. camaldulensis* is a non-native species to South Africa, it has a variety of purposes such as timber wood, wind shelters, construction and habitat (Roberts & Marston 2000). It is however, also a category II declared invader (Henderson 2001) and is found both in commercial and natural environments throughout the country, either through deliberate introduction or by invasive spread (Bennett 2014). The establishment, long term survival and health of *E. camaldulensis* is also known to be varyingly dependent on flood events (Roberts & Marston 2011). A survey of the literature showed little research has been conducted in the South African context. The impact of the local hydrological regime on *E. camaldulensis* is thus still unknown.

It is the evident invasiveness of *E. camaldulensis* in South Africa, combined with its rapid growth (Pinyopusarerk et al. 1996; Thoranisorn, Sahunalu & Yoda 1991), comparatively high water consumption (RHP 2011), prominent distribution along rivers, and ability to change the morphology of river channels (Jacobsen et al. 1999; RHP 2011) make this species particularly problematic in terms of water resource management. Effective management of *E. camaldulensis* requires such an understanding, since floods in the Western Cape, are known to be frequent and often severe in extent.

1.3 AIM AND OBJECTIVES

The aim of this research was to assess the impact of high-rainfall and flood events on the establishment and distribution of *E. camaldulensis* in the Breede River catchment, Western Cape.

In order to achieve the abovementioned aim, the following specific objectives were identified:

i) Establish current spatial extent, density and proportional age of *E. camaldulensis* within the middle Breede River catchment.
ii) Identify high rainfall and flood events in the Breede River between 1980 and 2013.
iii) Determine the spatial distribution of *E. camaldulensis* along the Breede River before and after high rainfall and flood events between 1980 and 2013, using multitemporal high resolution aerial photography.
iv) Establish the impact of high rainfall and flood events on the establishment of *E. camaldulensis* between 1980 and 2013.
v) Provide recommendations regarding the future management of *E. camaldulensis* in the Breede River catchment.

1.4 KEY RESEARCH QUESTIONS

The key questions investigated, in order to achieve the objectives were:

i) What is the current spatial distribution and density of *E. camaldulensis* in the absence of clearing?
ii) What was the *E. camaldulensis* spatial extent prior to, and after high-rainfall and flooding events?
iii) How do flooding and high rainfall events impact on the distribution and establishment of *E. camaldulensis*?
iv) What factors influence the distribution of *E. camaldulensis* within the Breede River catchment?

1.5 METHODOLOGY AND RESEARCH DESIGN

Kothari (2004) defines research methodology as the manner in which a research problem is systematically solved. The research methodology used for this report is centred on four different data sources, comprised of rainfall data (daily measurements in mm), river level data (daily measurements in m), aerial photography and field data (interviews and field measurements), in order to address the various research objectives (discussed in detail in the chapter 3). Using the distinction supplied in Kothari (2004), this research falls within the 'descriptive' and 'fundamental' research types. It is 'descriptive' in the attempts made to assess the current distribution of the target species, as well as 'fundamental' in the attempt to elucidate the relationship between rainfall events and *E. camaldulensis* on a theoretical level.

Furthermore, this research is primarily of a quantitative nature (Kothari 2004). The sampling method employed during this research is categorised under 'purposive sampling' (Kothari 2004), as only *E. camaldulensis* was sampled, and not, for example, all species present at each site. Teddlie & Yu (2007) describe purposive sampling as being based on a specific purpose, as opposed to a random sampling approach, and may include qualitative and quantitative techniques (Tongco 2007). This sampling approach is regarded as being able to produce greater depth of information for a smaller number of cases as compared to probability sampling (Teddlie & Yu 2007). As both quantitative data (area in m^2), and qualitative data (interview responses) were generated during this research, the inquiry can be classified as 'mixed-method', which combines representativeness and information-rich cases (Teddlie & Yu 2007).

1.6 REPORT STRUCTURE

This report is structured with five distinct chapters, each addressing individual components of the research (Figure 1.1). Chapter one is the introduction, which provides the study background, rationale, aim and objectives of the study. Chapter two encapsulates the relevant literature. Chapter three provides details of the research methods employed. Chapter four contains results for hydrology data, patch and island area, and fieldwork data respectively, including the interpretation. Chapter five concludes the report, addressing study strengths, weaknesses, as well as the theoretical and practical contributions of this study.

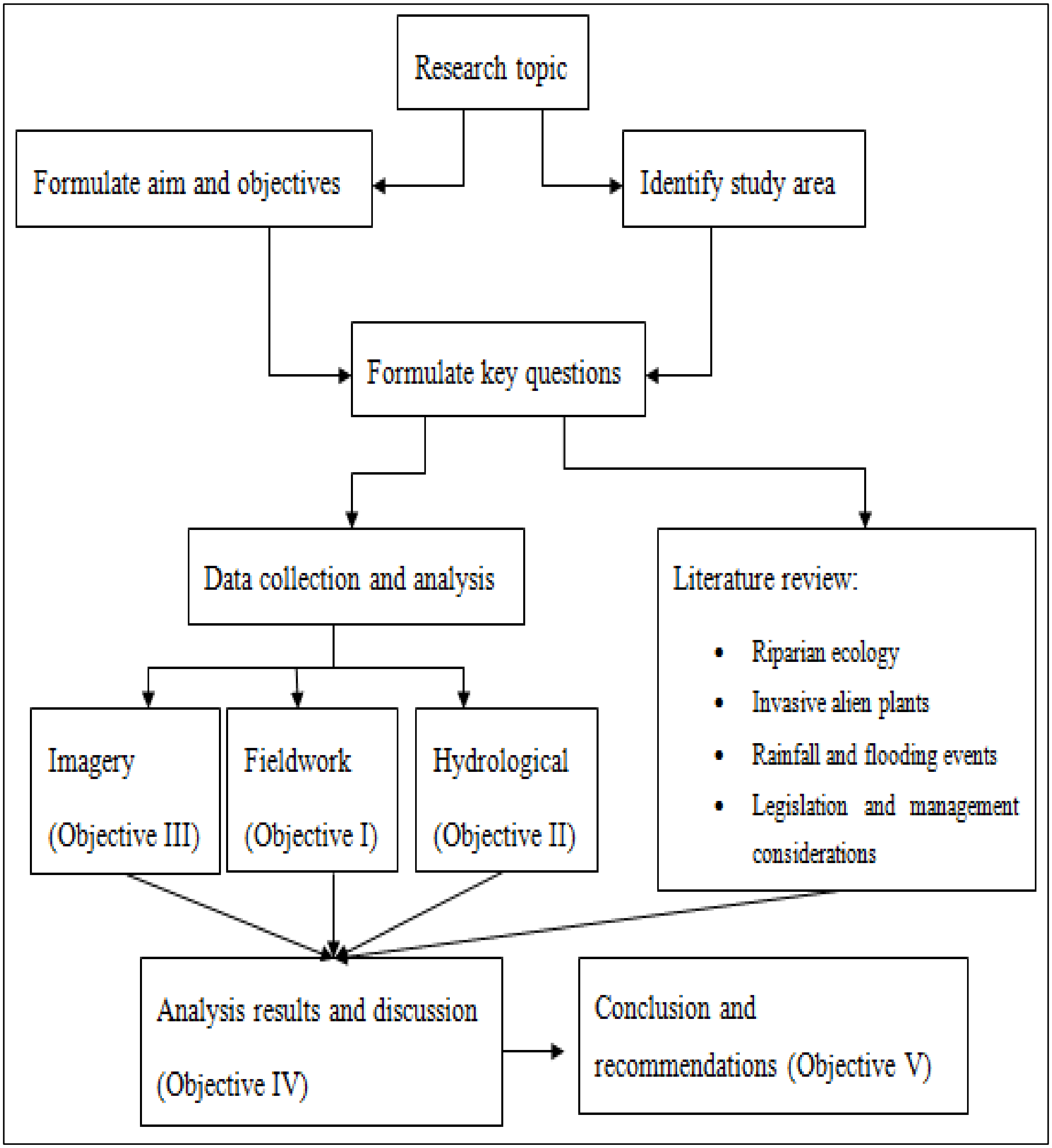


Figure 1.1 Research design depicting various tasks and organisation of this study.

CHAPTER 2 LITERATURE REVIEW

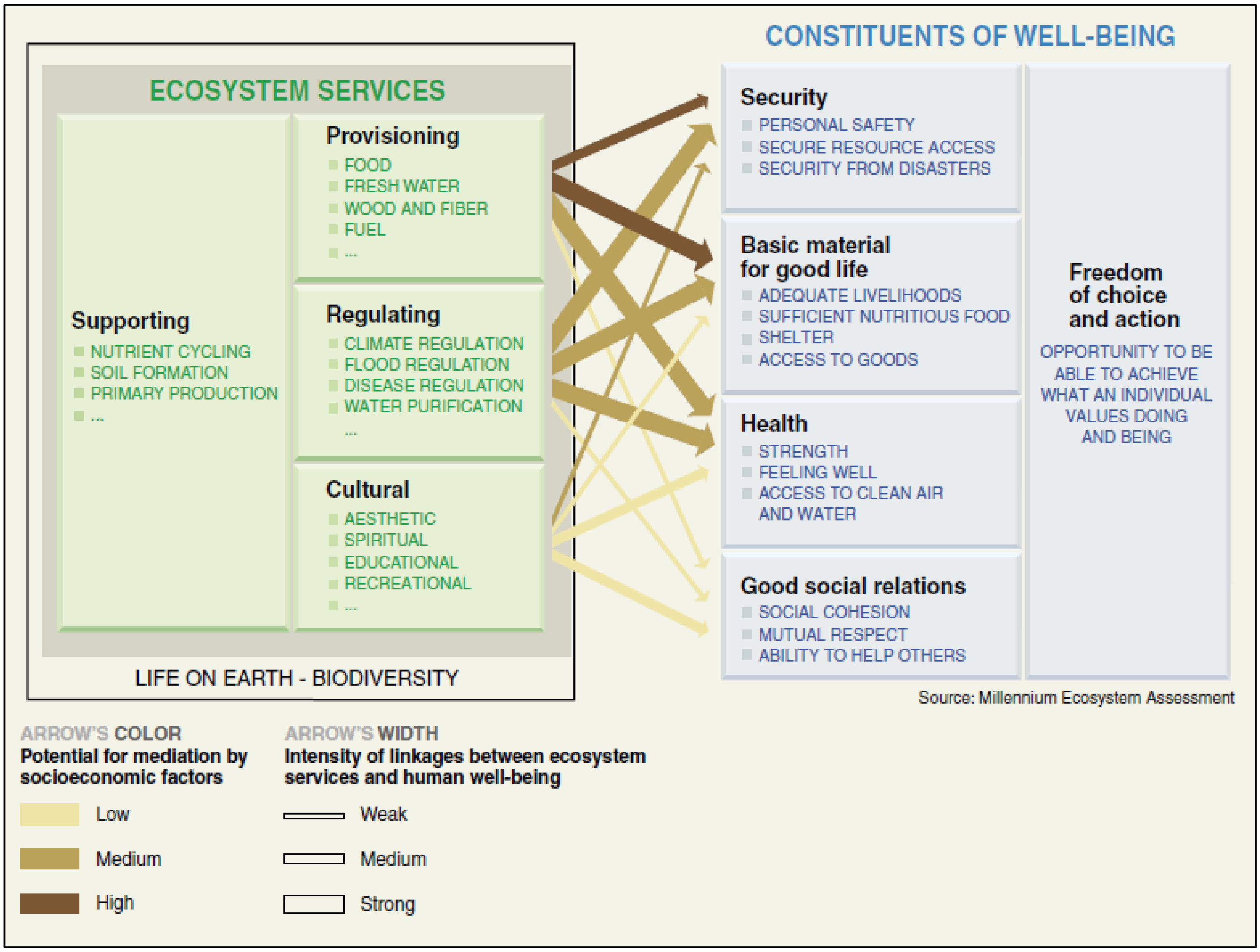
2.1 ECOSYSTEM SERVICES

It is widely accepted that human societies depend daily on a multitude of services supplied by natural systems (Egoh et al. 2009; Salles 2011). Amongst others, biodiversity in ecosystems provide services such as food, fuel, and construction materials along with less tangible benefits such as air purification, water purification, and flood mitigation (Barbier 2007; Liu et al. 2010; Salles 2011).

Humanity has always relied upon these natural benefits for survival and wellbeing (Reid et al. 2005). South Africans also rely on the environment for a myriad of resources such as agricultural products, pharmacological products, fuel and building materials, among others (Van Jaarsveld et al. 2005). In particular, the poor are especially dependant on ecosystem services and are thus often more vulnerable to the loss or absence of such services (Kumar & Yashiro 2013).

Broadly defined, ecosystem services refer to anthropogenic benefits obtained from ecosystems (Liu et al. 2010), or alternately, as the processes by which natural ecosystems, their flora and fauna, sustain and fulfil human life (Srivastava & Vellend 2005). These ecosystem services are categorised by the Millennium Ecosystem Assessment (2005) within four categories, namely those of 'supporting, provisioning, regulating, and cultural'. The various relationships between these services and human wellbeing are shown below in Figure 2.1, and serves to illustrate the highly complex relationship between humanity and the environment.

It is clear that ecosystem services are currently being reduced and degraded by human activities, such as overexploitation of resources and water abstraction (Van Wilgen et al. 2008), as global population and demand for ecosystem services increase (Kumar & Yashiro 2013; Liu et al. 2010). Many of the ecosystem services we depend upon are threatened by the presence of invasive species (De Lange & Van Wilgen 2010; Wolmarans, Robertson & Van Rensburg 2010), often facilitated by anthropogenic factors (Rodríguez-Labajos, Binimelis & Monterroso 2009).



Source: Millennium Ecosystem Assessment (2005: 6)

Figure 2.1 Relationship between ecosystem services and human wellbeing, illustrating humanity's dependence on ecosystem services for basic human needs.

2.2 BIOLOGICAL INVASIONS

Biological invasion is when an organism moves beyond its native or original range (Williamson 1996). IAPs can be defined as non-native species to a particular ecosystem and whose presence cause, or are likely to cause, socio-cultural, economic, environmental or human health harm (FAO 2010). Perkins, Leger & Nowak (2011) identified three categories that influence invasion, namely (i) the attributes of the potential invasive species, (ii) the biotic characteristics of the specific site, and (iii) the environmental conditions of the site. In addition, external influences such as land use change, climate change and nitrogen deposition are regarded as factors influencing invasion (Perkins, Leger & Nowak 2011).

Interest in biological invasions has increased recently, with more focus being placed on the impacts of established exotic species (Levine et al. 2003). Declining diversity is regarded as one such impact (Levine et al. 2003). A general ‘rule of thumb’ is that 10% of introduced non-native species will become established, of which only a further 10% will become problem species (Barbier 2001; Hays & Barry 2008; Richardson & Pyŝek 2006). Indeed, Hays & Barry (2008) found 10% to be conservative, with the successful establishment of non-native species regarded as being higher than the ‘rule’ suggests. Nevertheless, this is a deceptively small incidence of establishment, as even single established species may itself be extremely damaging and financially costly (Barbier 2001).

Scale is an important consideration when assessing the influence of IAPs on biodiversity (Wilson et al. 2007). Globally, IAPs tend to reduce biodiversity, whereas locally established invasive species may change species composition, with uncertain outcomes in the net-species sum thereof (Gaertner et al. 2009). For example, a recent study conducted by Tererai et al. (2013) within the Berg River catchment in the Western Cape found a decrease in richness and diversity of native species with an increase in *E. camaldulensis* invasion. This suggests, at least in the local context, one can assume the presence of *E. camaldulensis* reduces local species diversity. Such impacts may be attributed to the various characteristics of the particular species, many characteristics of which are commonly shared by IAPS globally (Pyšek & Richardson 2007).

2.3 CHARACTERISTICS OF IAP SPECIES

Tree species, in comparison to other growth forms, dominate the invasive species count worldwide (Richardson & Rejmánek 2011). Their widespread use has historically been accounted for through the anthropogenic advantages they supply (Dodet & Collet 2012). IAPs are often faster growing (Richardson 1998) and able to establish on degraded soils where native plants can not, thus successfully competing with native vegetation (Dodet & Collet 2012). Such species, when useful to humans, are often cultivated with little regard to their long term invasive potential (Richardson & Rejmánek 2011). It is frequently the very same traits that are sought after for use as forestry species, that allow for successful establishment and, consequently, their high invasive potential within native ecosystems (Richardson & Rejmánek 2011).

Amongst the various species characteristics related to invasive potential, such as competitive ability, propagule pressure and varying reproduction methods, growth and survival rate are of primary concern for forest species (Richardson 1998). An exotic species with a greater survival and growth rate would commonly exhibit greater competitive ability (Dodet & Collet 2012). These species may also be suited to a variety of environments, allowing it to spread to areas that were not initially intended or anticipated as ecologically suitable (Dodet & Collet 2012).

A further characteristic of invasive species is that of propagule production. Higher survival rates are often due to a high production of propagules, although this is not a fixed 'rule' (Rejmánek & Richardson 2011). Invasive forest species do commonly show increased seed production (Richardson 1998), or alternatively shorter intervals between large amounts of seed production (Rejmánek 1996; Rejmánek 2000; Richardson, Williams & Hobbs 1994), which increases reproductive potential and thus competitive advantage (Blossey & Nötzold 1995). Vegetative reproduction, for example by suckering or resprouting, also contributes to a species' invasive potential, allowing for greater fitness in comparison to native species (Blossey & Nötzold 1995). Additionally, certain invasive species are known to rely on long distance dispersal of their seed, allowing them to spread to greater extents than their native counterparts, further increasing the propagule pressure applied by the presence of such species (Richardson, Williams & Hobbs 1994).

Furthermore, characteristics often ascribed to pioneer plants, such as exhibiting low or intermediate shade tolerance, have also been shown to contribute to a species' invasive potential (Dodet & Collet 2012). Genera such as *Eucalyptus*, *Acacia* and *Pinus* display such characteristics (low to intermediate shade tolerance), which allow for greater competitiveness during initial stages of establishment and succession (Dodet & Collet 2012).

Species occurring within close proximity to each other, that are genetically compatible, can lead to hybridisation in natural environments (Potts & Dungey 2004). Such hybridisation is another means whereby invasive potential of a population may be increased (Ellstrand & Schierenbeck 2000; Huxel 1999). Spontaneous hybridisation is not a rare occurrence - having been noted for two centuries in South Africa (Arnold 1992) and even more recently (Steiner & Cruz 2009). Hybridisation however, may be unpredictable when occurring under field conditions (Dodet & Collet 2012). In South Africa, hybrids of *E. grandis, E. globulus* and *E. camaldulensis* have been specifically created for the forestry sector, and are planted commercially, despite their exotic origin (Coutinho et al. 2002; Potts & Dungey 2004). These hybrids are sought after as forestry species due to their perceived increased vigour, and subsequent commercial benefit (Poke et al. 2005; Potts & Dungey 2004). Hybridisation may allow for a greater competitive advantage through combining various 'desirable' traits of two species which, in practice, often increase the development and survival potential (Poke et al. 2005). However, few natural hybrids formed through hybridisation are viable and able to reproduce on their own (Arnold et al. 1999). Furthermore, fewer offspring are better suited to their environment than their 'parent' plants (Arnold et al. 1999).

Site characteristics and biotic factors also determine the potential for invasive species to colonise an area (Lamarque, Delzon & Lortie 2011). In the context of forestry, whereby species are chosen with their site-specific suitability in mind, such species would be able to successfully colonise an area in the long term (Richardson 1998). Furthermore, the 'enemy release hypothesis', which postulates that absence of natural enemies within the specific region would allow for greater competitive advantage and subsequent invasive potential (Dodet & Collet 2012), would also apply.

The presence of certain 'companion' species may additionally increase the competitive ability of an invasive species (Dodet & Collet 2012). For example, honeybees in areas where *Eucalyptus* is grown act as pollinators (Allsopp & Cherry 2004), and would thus increase the reproductive capacity of a specific group of trees (Dodet & Collet 2012). All the above mentioned characteristics, may contribute to the invasive potential in *E. camaldulensis* in South Africa, allowing for greater impact or influence of *E. camaldulensis* in areas where it is established.

2.4 INVASIVE ALIEN PLANT DISTRIBUTION

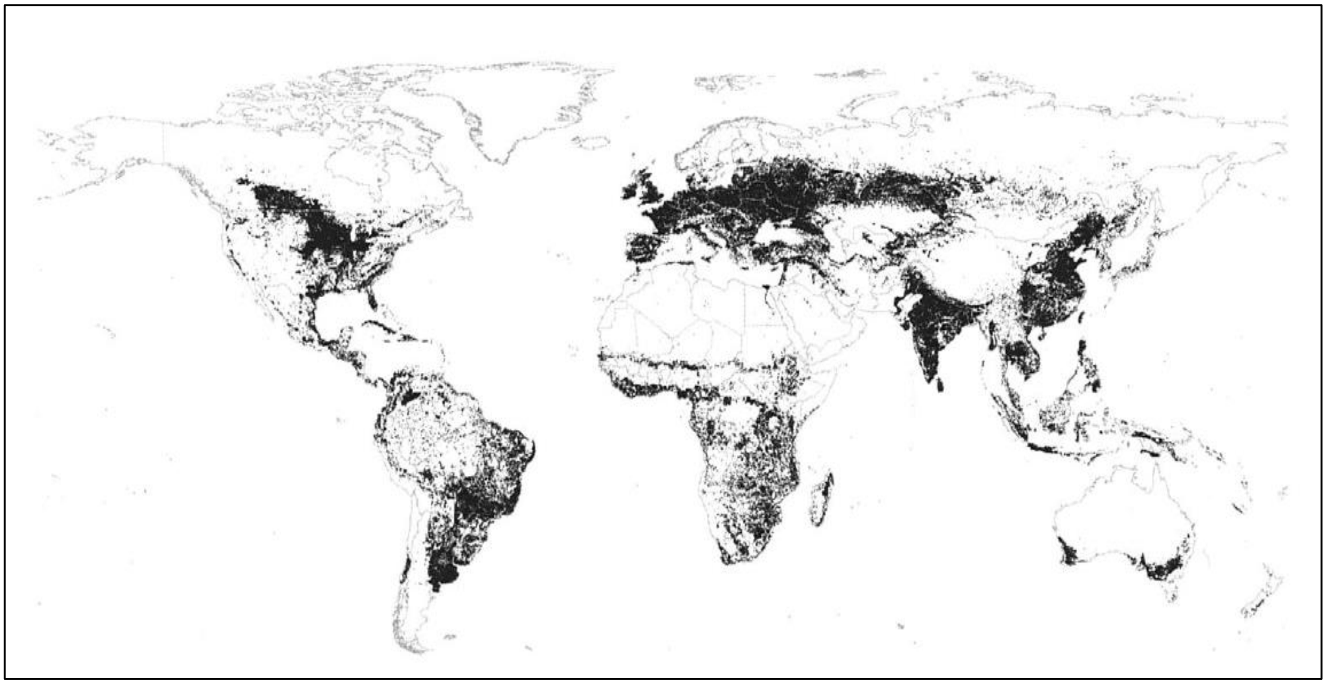
Over the last 60 years the rate of species invasions has increased sharply, due to invasions being related mostly to human movements (García-Llorente et al. 2008), and are now a global concern (Chornesky & Randall 2003; Clout & De Poorter 2005). This occurs through improved transport and increased numbers of traded species, which have allowed partially for their widespread distribution (Chornesky & Randall 2003; Clout & De Poorter 2005; Meyerson & Mooney 2007; Pyšek & Richardson 2008).

Introductions vary in manner, but may be deliberate for example, through direct introduction where the species may have some benefit to humans, or coincidental through stow-away on board a transport vessel (Clout & De Poorter 2005). Further examples of such spread is through transport of IAPS by the timber industry, horticultural industry, internet seed trade and even the landscaping industry (Everett 2000). With an increase in ease of transport, the related risk of invasion is also increasing (Everett 2000), leading to accelerated economic and biodiversity loss via biological invasions (García-Llorente et al. 2008).

Since IAPS are not contained by political or man-made boundaries, distribution and spread is rarely contained to only one political entity, causing the problem to be global and shared in nature and, as such, requiring a global solution (Jenkins & Pimm 2003). Invasion of an exotic species can occur in virtually any region of the Earth, however, they are thought to occur predominantly in disturbed landscapes such as pasture and crop lands (Jenkins & Pimm 2003).

Disturbed landscapes refer to areas which have undergone ecosystem changes (Jenkins & Pimm 2003), and would thus include agricultural land (Tscharntke et al. 2005). Jenkins & Pimm (2003) estimated that approximately 27 million km^2, equivalent to 21% of the total land surface of the Earth, has been converted to disturbed landscapes (Figure 2.2), which may serve as future regions of spread. relatively limited in their extent (Le Maitre et al. 2002), as much as 1 905 000 hectares (ha) of forest in 2005 was invaded by *Eucalyptus* spp. alone (FAO 2010).

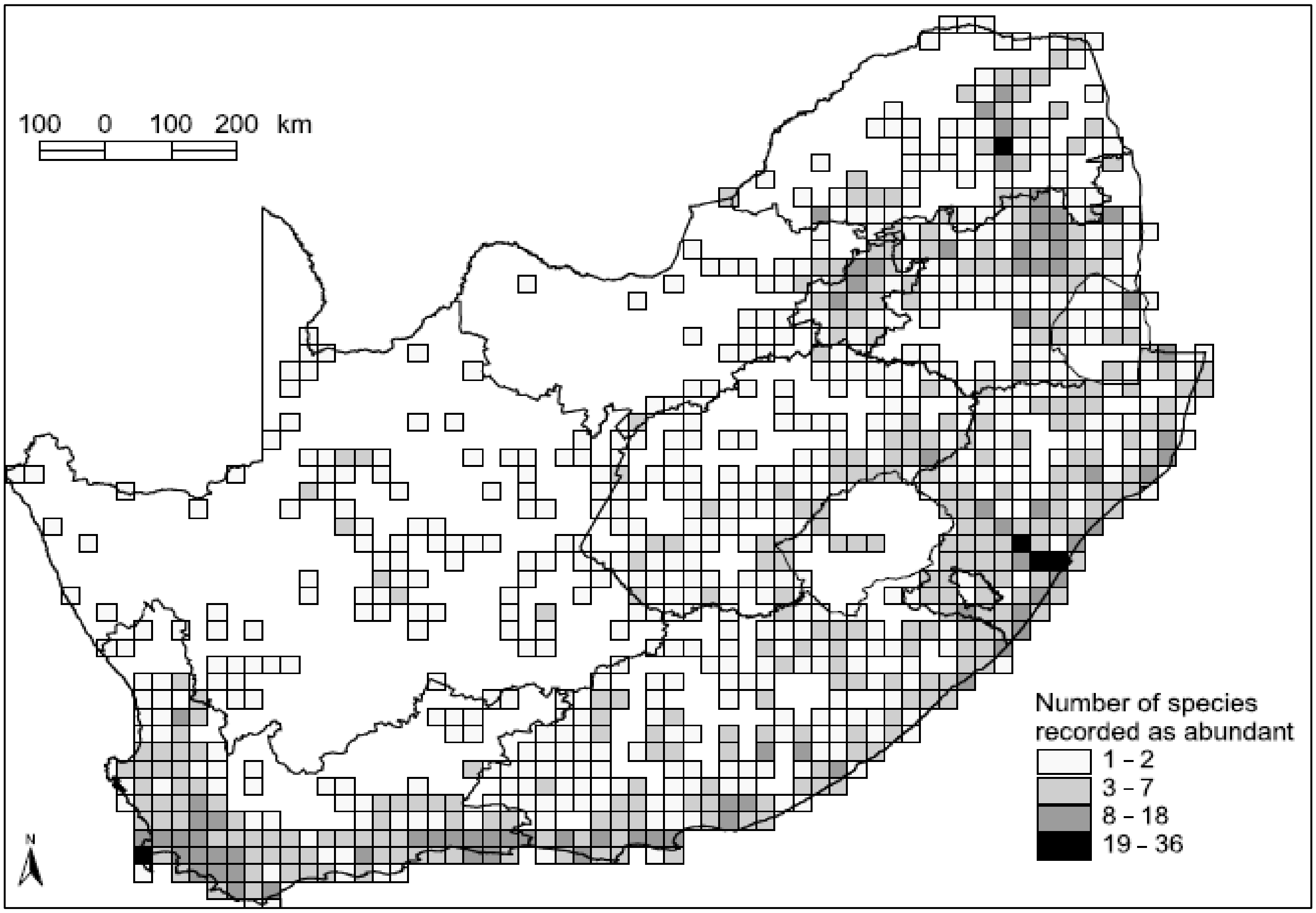
Although Figure 2.2 is an approximation, it serves to illustrate the severity of disturbed landscapes at a global scale. Although forested areas (inclusive of natural and commercial forests) in South Africa are



Source: Jenkins & Pimm (2003: 175)

Figure 2.2 Global map of disturbed areas, including croplands, urban areas, transformed woodlands and forests (shaded in black), which represent their respective disturbance intensity.

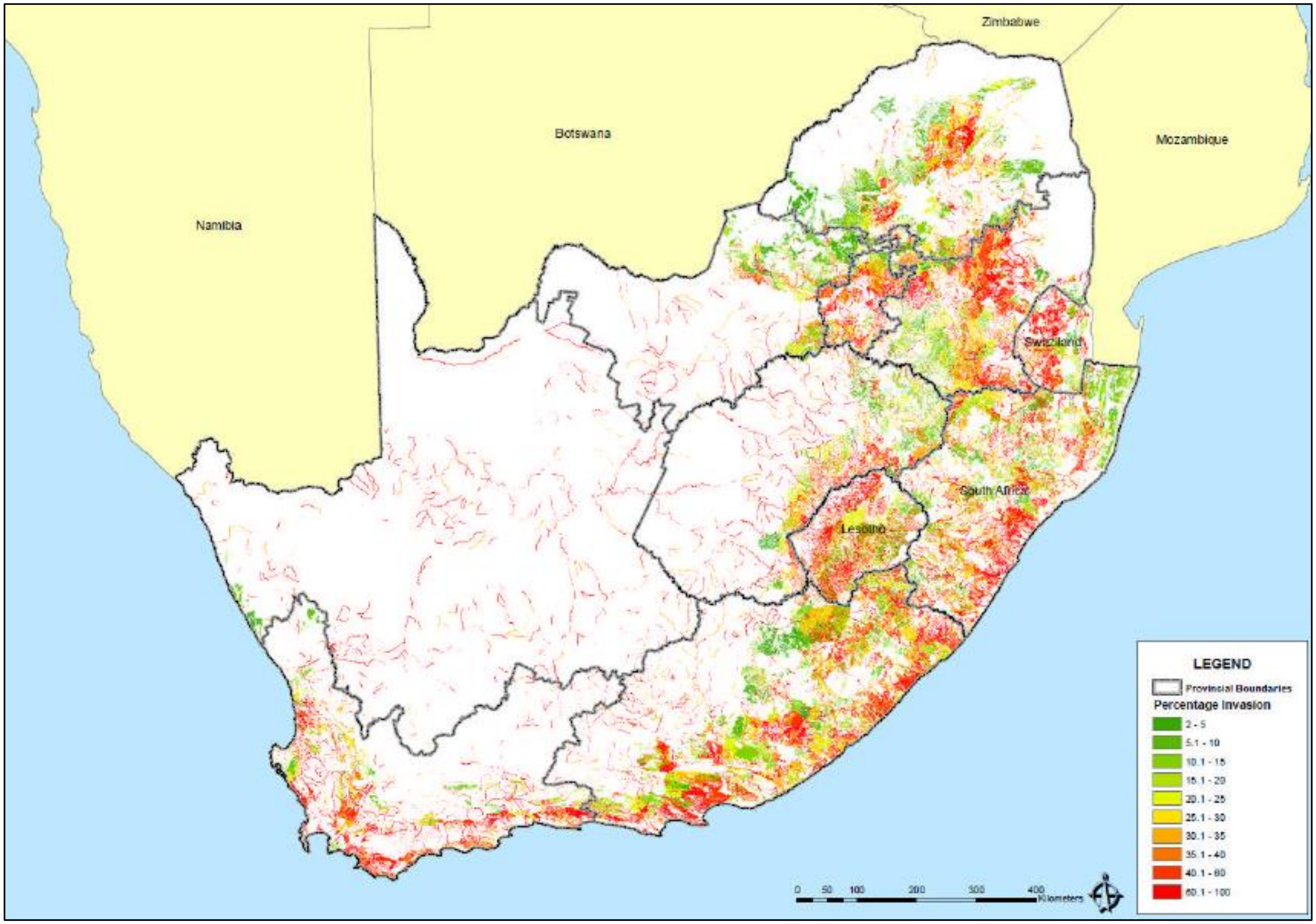
Nel et al. (2004) reported that around 10 million hectares, equivalent to 8.2% of the total surface area of South Africa, was already invaded by IAPS. The majority of the IAP species distribution in South Africa (Figure 2.3) tends to be towards the south-western, southern and eastern coastal regions and adjacent interior, which exhibit the highest rainfall (Henderson 2007). The squares in Figure 2.3 are based on the Universal Transverse Mercator coordinate system (UTM), and equate to the total area enclosed by 15' x 15' units with the grid superimposed on the national boundaries (Larsen et al. 2009). The national IAP survey (Kotze et al. 2010) produced similar results in terms of distribution and abundance, indicating vast areas of the landscape as invaded (Figure 2.4).



Source: Richardson & Van Wilgen (2004: 46)

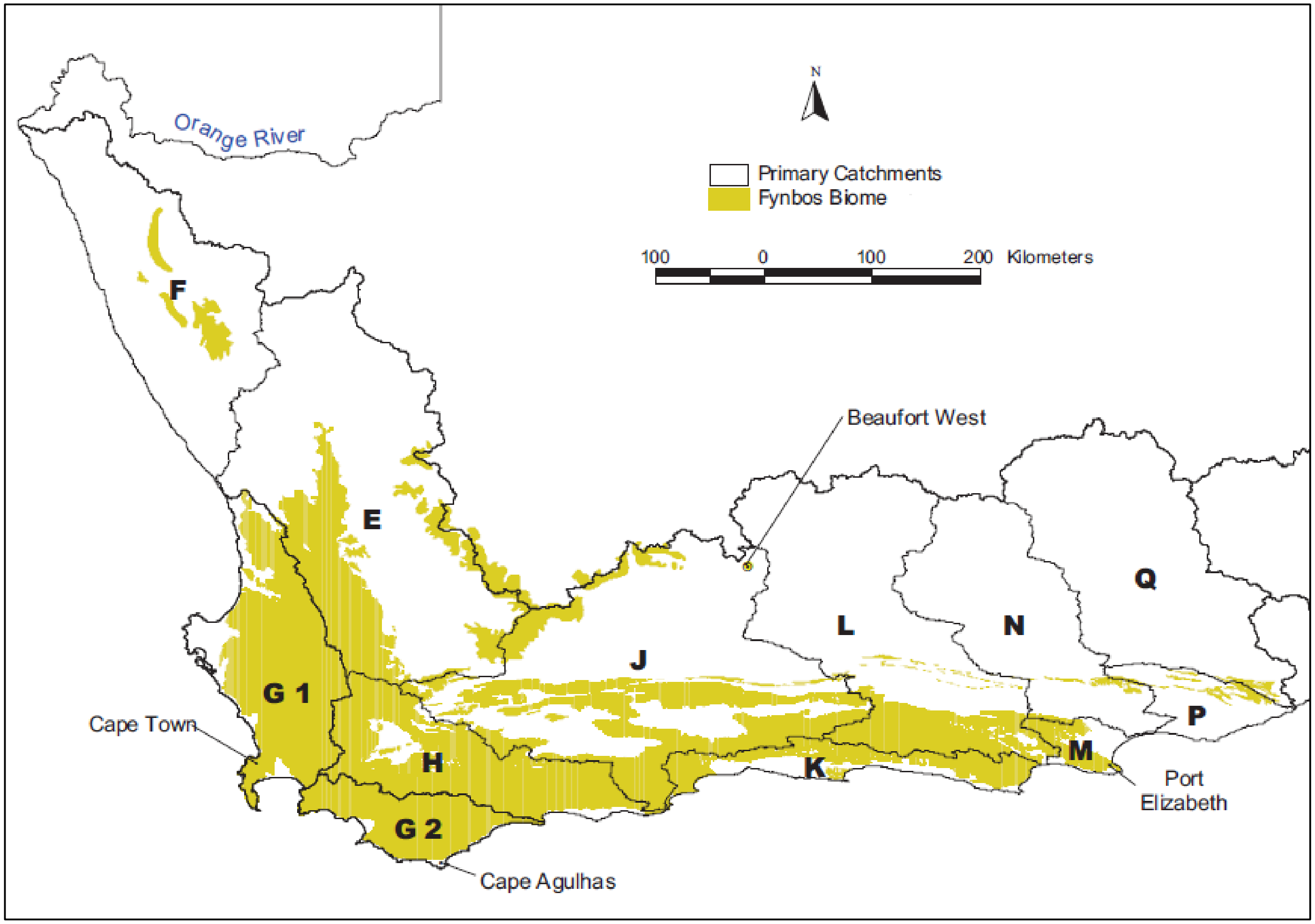
Figure 2.3 IAPS distribution and abundance in South Africa per quarter degree square (15'x15' units).

In the Western Cape, 37 274 km^2, equivalent to 28.82% of the total area with a 16.80% mean canopy cover, has been invaded by exotic species (Van Wilgen et al. 2001). The fynbos biome (Figure 2.5), the low shrubland of the Mediterranean type climate zone of the Western Cape (Moll et al. 1984), which is contained within the Cape Floristic Region (CFL), is particularly threatened by woody IAPS invasion (Roura-Pascual et al. 2009). The two invasive species of highest priority within the fynbos biome are *Acacia mearnsii* and the *Pinus* genera (Van Wilgen, Forsyth & Le Maitre 2008). However, *E. camaldulensis* is also regarded as being a species of major ecological importance within the fynbos biome (Forsyth et al. 2004; Van Wilgen 2009) given the considerable impact *E. camaldulensis* has on water resources, a potential fire hazard, its ability to invade riparian and landscape zones, as well as its widespread distribution (Van Wilgen, Forsyth & Le Maitre 2008).



Source: Kotze et al. (2010: 41)

Figure 2.4 IAPS extent for Southern Africa, Lesotho and Swaziland. Percentage invasion is indicated from 0% (dark green), to 100% (Dark red). Mapping units were 100 m^2.



Source: Van Wilgen, Forsyth & Le Maitre (2008: 10)

Figure 2.5 Fynbos biome spatial extent (indicated in yellow) and primary catchments within the Cape provinces. Catchments E & J are largely confined to the Western Cape. Catchments H & G are entirely confined to the Western Cape and catchments K & L stretch into the Eastern Cape. A small portion of the southern part of catchment F falls within the Western Cape, with the rest within the Northern Cape.

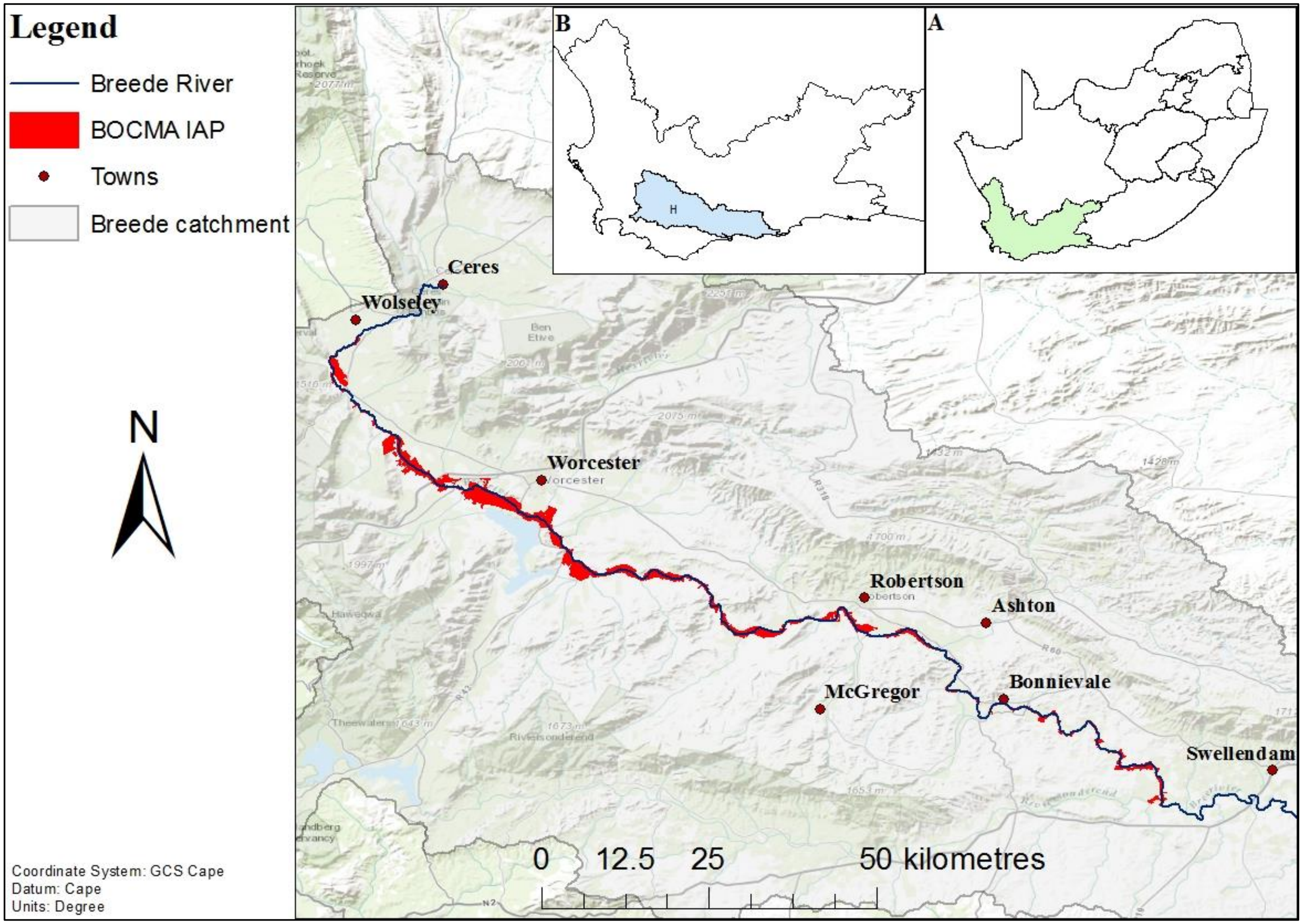


Figure 2.6 BOCMA identified IAPS locations (indicated in red) within the Breede River catchment, planned for future clearing. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014).

Eucalyptus species form part of the priority species, as they occupy large areas, being ranked second in terms of national area coverage between 1995 and 2008 (Van Wilgen et al. 2012), as well as being expensive to clear (Hugo et al. 2012). The Breede-Overberg Catchment Management Agency (BOCMA) has been tasked with the mapping of IAPS along the Breede River, the extent of which can be seen in Figure 2.6. The invasive species extent shown in red includes that of *E. camaldulensis*, as well as other commonly occurring IAPS, such as *Acacia mearnsii*. Species are ranked by combining their unique characteristics and widespread distribution nationally as well as globally, which determine their impacts on the local environment. The presence of these species within the riparian zone of the catchment are certain to impact on the environment in a variety of negative ways.

2.5 EFFECTS OF IAP

The ecological effects of IAPs are varied in severity and type. Invasion by IAPs have negative effects on the function and structure of existing ecosystems (Clout & De Poorter 2005; Higgins, Richardson & Cowling 1996; Jenkins & Pimm 2003; Ordonez, Wright & Olff 2010; Van Wilgen et al. 2012). IAPS are known to threaten biodiversity (Born, Rauschmayer & Brauer 2005; Brown et al. 2004; Levine et al. 2003; Thuillier et al. 2006), able to alter disturbance regimes (natural and human), in addition to competing with other flora for basic resources such as space, light, moisture and nutrients (Gaertner, Richardson & Privett 2011; Yurkonis, Meiners & Wachholder 2005). Such competition may reduce recruitment and survival of the existing native flora, further leading to a homogenous landscape (Olden 2006).

A specific effect of IAPS introduction is the reduction of species richness (Meiners, Pickett & Cadenasso 2001; Yurkonis, Meiners & Wachholder 2005) and composition, as found by Gaertner, Richardson & Privett (2011) after *Eucalyptus* and *Acacia* spp. invasion within the fynbos biome. The flora within the fynbos biome is sensitive to soil-enrichment from nitrogen-fixing invasive plants, which alter soil conditions in such a manner, that a homogenous, low-diversity landscape is created (Gaertner, Richardson & Privett 2011). Furthermore, reducing available light through canopy cover, and allelopathy, has been shown to influence species composition and vigour in communities near *E. camaldulensis*, which is another means by which biodiversity is impacted upon (Tererai et al. 2013).

Further effects include the consumptive use of water by invasive species, leading to reduced water yields (Le Maitre et al. 1996; 2002). In South Africa, mean river flow may have reduced by 6.7% up to 2002 through the consumption of water by IAPs alone (Le Maitre et al. 2002), mostly as a function of invasive species increasing above ground biomass and evapotranspiration, leading to the greater water use (Chamier et al. 2012). Dye & Poulter (1995) found that clearing *Acacia* and *Pinus* species from riparian zones in afforested areas allowed for a significant increase in water yield. Water may also be impacted through the changing of water quality (Le Maitre et al. 1996), water chemistry and the subsequent degradation of aquatic biodiversity (Richardson & Van Wilgen 2004).

Additionally, fire regimes within a given region may also be influenced by the presence of IAPS (Jurskis 2012). Fire regime changes may further indirectly influence nutrient cycling, hydrological regime and energy balances of a region, further impacting biodiversity (Jurskis 2012). Growth of IAPs may reduce soil nitrogen deposits, or alternately frequently burning may promote nitrogen deposition, which depending on the region may be ecologically harmful or beneficial (Richardson & Van Wilgen 2004). Another impact is the promotion of soil erosion (Richardson & Van Wilgen 2004), in addition to impacts such as redistribution of salts in the ecosystem and reducing or increasing litter for a given region (Richardson & Van Wilgen 2004).

These impacts and changes are part of the ‘ecosystem engineers’ concept, whereby the impact of invaders are mostly observed rapidly altering disturbance regimes (Pyšek & Richardson 2008), which have numerous indirect consequences, such as alteration of trophic levels, biogeochemical cycles, physical living space and available resources (Richardson & Van Wilgen 2004).

Trees also provide a habit for other species with hollows, fallen branches and leaf litter and tree canopies being the vehicles for such ‘ecosystem services’ (Roberts & Marston 2011). As a generalisation, an average of 2.1 hollows per tree were observed for *E. camaldulensis* trees (Roberts & Marston 2011), which were utilised by bees and birds, amongst others species. *E. camaldulensis* is additionally regarded as forming part of the healthy functioning of lowland rivers in its native range, mainly through litter fall, carbon and nutrient cycling contributions, and the habitat function mentioned above (Roberts & Marston 2011). The discussion above relates mostly to ecological impacts of IAPS, however, these are by no means the only way in which their presence influence the environment, with other impacts notably being economic or social in nature.

2.6 HUMAN IMPACT

It is well known that the clearing of IAPs can be highly expensive (Krug, Roura-Pascual & Richardson 2010), and is often difficult to manage due to the highly complex nature of ecosystems and invasions, and the socio-political interplays involved (Roura-Pascual et al. 2009). Direct economic impacts of IAPS differ from the loss of potential or realised agricultural output through yield loss to the expenditure on herbicides and staff during clearing actions (CFIA 2008).

Accurate estimates of the financial costs involved are often difficult to establish, as a variety of factors need to be considered, which often require large expenditures of time and money (Frazee et al. 2003). Over the past 15 years, IAP management funds have mainly been allocated to the removal and control of the *Acacia*, *Pinus*, *Eucalyptus* and *Prosopis* taxa nationally, two of which were regarded as prominent in the Western Cape (Van Wilgen et al. 2012). Studies have estimated economic loss from invasions of $US 11.75 billion, or R117.5 billion (R10 to the dollar) to the fynbos biome alone (Van Wilgen et al. 2001), related to invasion of roughly 56% of the surface area of the Cape Floristic Region (CFR). The CFR spans roughly 80 000 km^2 of South Africa (Moran & Hoffmann 2012). Additionally, control cost of mesquite have been in excess of R95 million per year in 2009 for example (Wise, Van Wilgen & Le Maitre 2012).

Cost benefit analyses have highlighted the economic benefit to clearing of IAP species (Van Wilgen, Le Maitre & Cowling 1998). It was in response to such analyses that the WfW programme was initiated in 1995. In 1998, it was estimated that clearing of all IAPs in South Africa would require 20 years and $US 2 billion (Van Wilgen, Le Maitre & Cowling 1998) – or R21.34 billion (using 2008 maximum Rand exchange value of 1USD = R10.67). Ten years later Van Wilgen et al. (2008; 2012) reported government contribution to IAP control since initiation, to be approximately R3.2 billion. WfW receives a large annual budget, as much as $US 67.6 million (or R486.72 million at R7.2 to the dollar in 2011) to address invasions (McConnachie et al. 2013).

Costs from 2008 remain the most up to date, as that was the last year WfW applied a national scale inquiry into the extent of invasion (Van Wilgen et al. 2012), which allowed for operational cost comparisons. Table 2.1 indicates the priority species as identified by Van Wilgen et al. (2012), with *Eucalyptus* species ranked second on the list. According to Van Wilgen et al. (2012) clearing costs accrued by the WfW between 1995 and 2008 was an estimated R237 million, for *Eucalyptus* alone (Table 2.1). It is clear a substantial amount of money was, and is still being spent on the eradication programmes within South Africa, to combat the impact of invasions.

The resultant economic implications for the continued existence of *E. camaldulensis* along water courses (and outside of management areas) are thus extreme, and well worth the investment of money, time, expertise and research effort. Financial cost is also not the only impact of IAPS invasion, as Roura-Pascual et al. (2009) shows social impacts to also be a factor.

The social impacts of invasive clearing programmes are numerous, and include direct impacts to human health (for example allergies and dermatitis), loss of environmental aesthetic appeal, indirect influence on property values (both by adding and decreasing value) and a loss of tourism or employment (CFIA 2008). IAPS may further play a role in the alteration of ecosystems, such as creating environments suitable to mosquito development, indirectly impacting on human health (CFIA 2008).

Table 2.1 WfW clearing cost of prominent IAP taxa of South Africa between 1995 and 2008 in ZAR (2008 equivalent).

Invasive Alien Plant Taxon	**Rank in terms of area occupied (Kotze et al. 2010)**	**Clearing cost (million ZAR)**

Acacia dealbata	1 (grouped with other *Acacia spp.*)	79.3
Acacia mearnsii	1 (grouped with other *Acacia spp.*)	561.9
***Eucalyptus* spp.**	**2**	**237.0**
Acacia cyclops	7	58
Pinus spp.	3	183.5
Chromolaena odorata	4	171.8
Populus spp.	6	42.5
Acacia saligna	8	88.4
Solanum mauritianum	9	121.5
Salix babylonica	10	8.1
Lantana camara	12	69.3
Cereus jamacaru	17	57.5
Caesalpinia decapetala	18	33.2
Arundo donax	24	8.2
Acacia melanoxylon	25	28.0

Source: Adapted from Van Wilgen et al. (2012: 4)

IAPS impacts are not necessarily detrimental, as they may also provide value – notably medicinally and economically, amongst other uses (Poona 2008). Socio-economic benefits obtained are for example, through the adoption of a 'public works programme' to conduct clearing on IAPS throughout the country, which create employment opportunities (McConnachie et al. 2013). Timber derived from IAPS (for example, *Acacia mearnsii*) further contribute to stocks of building material, pulp wood, and wood chips (Moyo & Fatunbi 2010). IAPS are often also used in tannin extraction and in rural settings as fuelwood and building material, wind shelter, manure, or as cattle feed and bee forage (Moyo & Fatunbi 2010). Anthropogenic uses of *E. camaldulensis* include timber (Roberts & Marston 2000), pulp, paper, honey for the apiary industry, amenity planting, charcoal, site remediation, railway sleepers and as direct source of pollen (Roberts & Marston 2011). The timber industry furthermore, aside from public work programmes, employ people nationwide (Van Wilgen et al. 2011). These disadvantages and benefits need to be evaluated against the impacts and influences of *E. camaldulensis* in the planning of any clearing effort.

2.7 INFLUENCE OF *E. CAMALDULENSIS*

2.7.1 Island formation

Of particular interest in this study is the influence of *E. camaldulensis* on stream flow. It has long been accepted that riparian vegetation plays an important part in ecosystem function through prohibiting erosion, as well as obstructing water movement and stabilising sediment, sand, and soil within the river channel (Rowntree & Dollar 1999; Van Der Nat et al. 2003), or by promoting species and habitat diversity via the hollows and fallen branches, which form habitats for other species (Sieben & Reinecke 2008). *E. camaldulensis* wood is highly durable and resistant to decay, requiring as much as 375 years to reach decay of over 95% of the initial mass (Roberts & Marston 2011). Branches that fall from the trees are therefore highly persistent within their environment and serve as obstruction to water flow if dropped into river systems (Roberts & Marston 2011). This frequently occurs as they prefer riverine environments, growing along watercourse margins (Thorburn & Walker 1994). *E. camaldulensis* additionally regulate river water temperature through shading (Roberts & Marston 2011).

The influence of riparian vegetation on river habitats are not specific to one species only, and is attributed to the whole of the riparian zone plant community, regardless of origin (Rowntree & Dollar 1999). Resistance to the flow of water in channels, posed by the presence of riparian vegetation, can be great, with even the smallest changes influencing the roughness measure (known as the 'Mannings n' value) up to ten times the original value (Hickin 1984). Additionally, the strength of the bank may also be increased by the presence of riparian vegetation, usually through the binding action of the root systems of the plants (Hickin 1984). These influences may not outweigh the potential negative impacts associated with IAPS, and thus should not be interpreted as motivation for their introduction.

The presence or absence of riparian vegetation would thus influence the fine-scale shape of the river, by allowing for sediment transport further downstream (Hickin 1984). Another important mechanism through which large wood (i.e. logs with a diameter greater than 0.1 m, and a length greater than 1m (Kail et al. 2007)), such as produced by *E. camaldulensis*, influences flow and morphology of a river, is by acting as a nucleus for bar sedimentation (Hickin 1984). A bar is an active, depositional surface made up of coarse, finely-sorted sand (Merritt & Cooper 2000). Although direct evidence for the causal relationship between large wood and islands, or sand bars, is rare, there is general consensus that large wood does have a role in the production of such bars (Hickin 1984).

Rivers in the Western Cape, being dynamic ecosystems, are highly heterogeneous in terms of species composition, while also being extremely ‘patchy’ in their composition of riparian zone species (Sieben & Reinecke 2008). Moreover, Western Cape rivers have been shown to have a high species turnover between different catchments (Sieben & Reinecke 2008) and are composed typically (in an undisturbed or ‘natural’ state) of ‘closed-scrub fynbos’ (mainly tall scrubs), although riparian vegetation may also range from forest to tall herbland forms (Sieben & Reinecke 2008). Reference sites, such as studied by Sieben & Reinecke (2008), showed considerable variation in riparian vegetation, composed of native fynbos.

However, large areas of vegetation within the riparian zone of the Western Cape, and specifically the Breede catchment is composed of IAPS, mostly medium to large woody species such as *A. mearnsii* (Brown et al. 2004) forming dense stands or forests. As the invasion of IAPS is associated with greater above-ground biomass (Le Maitre et al. 1996) and greater litterfall (Yelenik, Stock & Richardson 2004) when compared to fynbos, potential exists for a greater input of large wood (LW) from the presence of IAPS, in comparison to fynbos input. It is likely that *E. camaldulensis* contributes LW to the system given its abundance in the area.

2.7.2 Water use

Another significant influence of *E. camaldulensis* on the surrounding environment is its water use. Several authors (Chamier et al. 2012; Dye 2013; Whitehead & Beadle 2004) have identified and discussed the relatively high water usage by *Eucalyptus*, and specifically *E. camaldulensis*, for example, comparing *E. camaldulensis* to different species such as *Tectona grandis* and *Pinus caribaea* (Calder & Dye 2001). *E. camaldulensis* in well-watered soils displayed the greatest water usage compared to *E. camaldulensis* stands in drier soils (Morris, Collopy & Mahmood 2006), indicating that the species utilises more water when sufficient quantities are available.

A recent study on water usage of *E. camaldulensis* found water use of 1 160 mm per annum in Pakistan, and 310 mm water usage per annum in Australia (Morris, Collopy & Mahmood 2006). *E. camaldulensis* trees reach a DBH of 30 cm or more, and transpire between 60 and 350 litres of water per day, over 10 years of growth (Qui & Loehr 2002). Using a spacing of 3-4.5 m between individual *E. camaldulensis* trees, and planting roughly 200-400 trees per acre (0.4 hectare), daily water abstraction yields in plantations could reach between 11 300 - 60 000 litres of water per day per acre (Qui & Loehr 2002), which is useful in estimating the quantities of water used by *E. camaldulensis*.

Steep water use gradients are to be expected for riparian communities of *E. camaldulensis* due to variation in the fine-scale environment (O'Grady et al. 2002). Individual *E. camaldulensis* trees are known to transpire water at a rate of between one and three litres per hour depending on soil moisture, tree age, tree size, and time of day (Salama, Bartle & Farrington 1994). A study by White et al. (2002) recorded transpiration rates of *E. camaldulensis* at 1.9 mm per day but also noted rates of up to 3 mm per day in certain areas of Australia.

It is clear from the literature that *E. camaldulensis* is a species that uses much water, especially when compared to that of native fynbos flora. Dye et al. (2001) found respiration of fynbos at a reference site in Jonkershoek, to be 1 332 mm per annum (as a measure of water volume), which was significantly less than the *Acacia* comparison sites used in their particular study. *Acacia* here are used as an example, as, in South Africa, alien trees (Chamier et al. 2012) like *Eucalyptus*, *Pinus and Acacia* (Mallory, Versfeld & Nditwani 2011), reduce stream flow by greater extents than native vegetation does; especially compared to native vegetation of smaller total heights, such as fynbos (Calder & Dye 2001).

This is due to greater biomass rooting depth of IAPs (Albaugh, Dye & King 2013) in comparison to the native vegetation, as well as seasonal dormancy of the native vegetation, such as grasses or fynbos (Dye 2013). Seasonal dormancy is absent in *Eucalyptus* vegetation, and thus greater evapotranspiration rates in comparison are possible (Albaugh, Dye & King 2013; Mallory, Versfeld & Nditwani 2011).

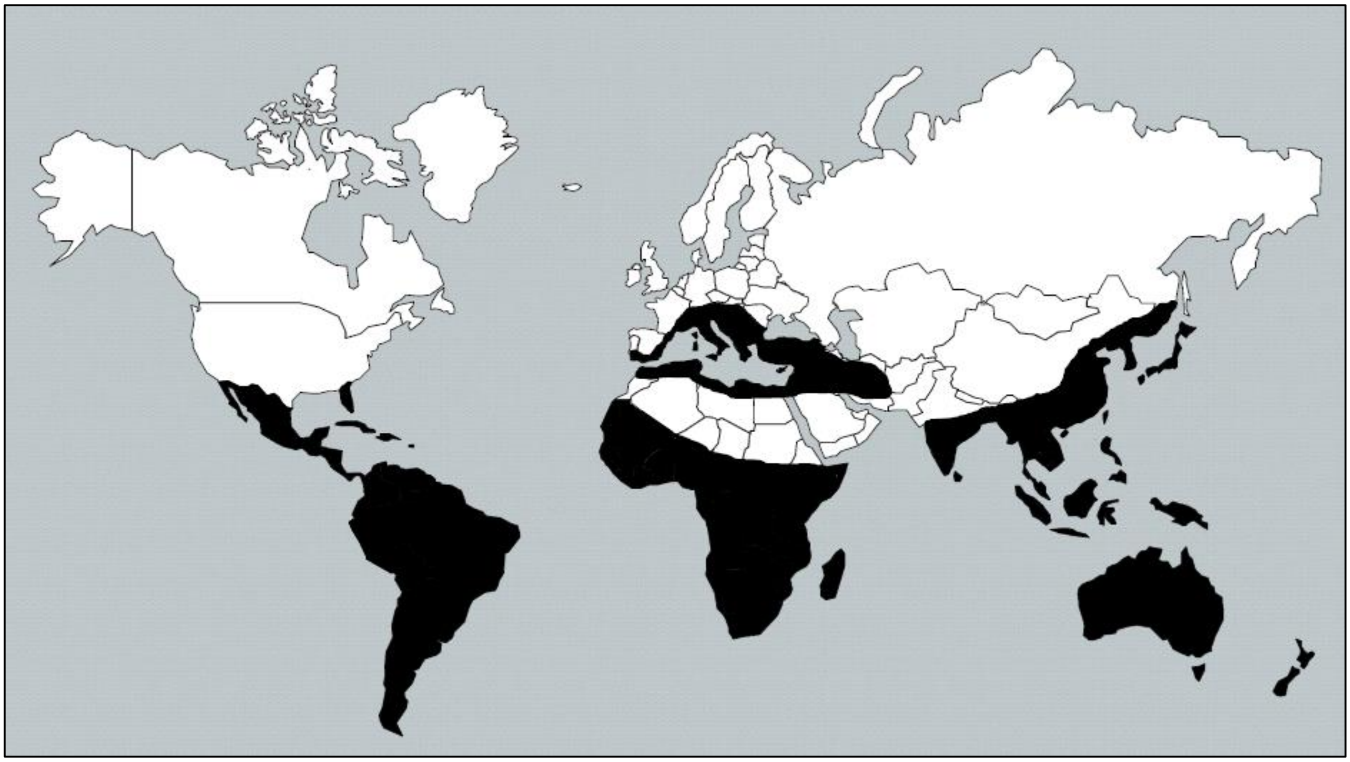
Such greater evapotranspiration lead to reduced stream flow and ground water reserves, reduces a river system's capacity for dilution, allowing for increased pollution, nutrient and salt levels, and consequently a reduced ability to buffer against ecosystem threats (Chamier et al. 2012).

Dye et al. (2001) warns, however, that due to considerable differences in plant communities at any given site (structural and physiological characteristics pre- and post-clear), extrapolation beyond the geographical range of the study should not be conducted, without consideration of the transpiration dynamics unique to the specific region. The relatively high water use of *E. camaldulensis* is an additional motivation for their removal along watercourses, as an estimated half of all the available water management areas are exceeding current water supply (Van Wilgen et al. 2008).

2.8 NATIVE RANGE AND DESCRIPTION OF *E. CAMALDULENSIS*

Eucalyptus is a genus within the *Myrtaceae* family, a large dicotyledonous woody plant species family, which is commonly known as the Myrtle, Clove or Guava family, and contains up to 150 genera making it the eighth largest flowering plant family worldwide (Grattapaglia et al. 2012). The *Myrtaceae* family is found mostly in the southern hemisphere (Figure 2.7), and contains genera with high numbers of individual species (Grattapaglia et al. 2012).

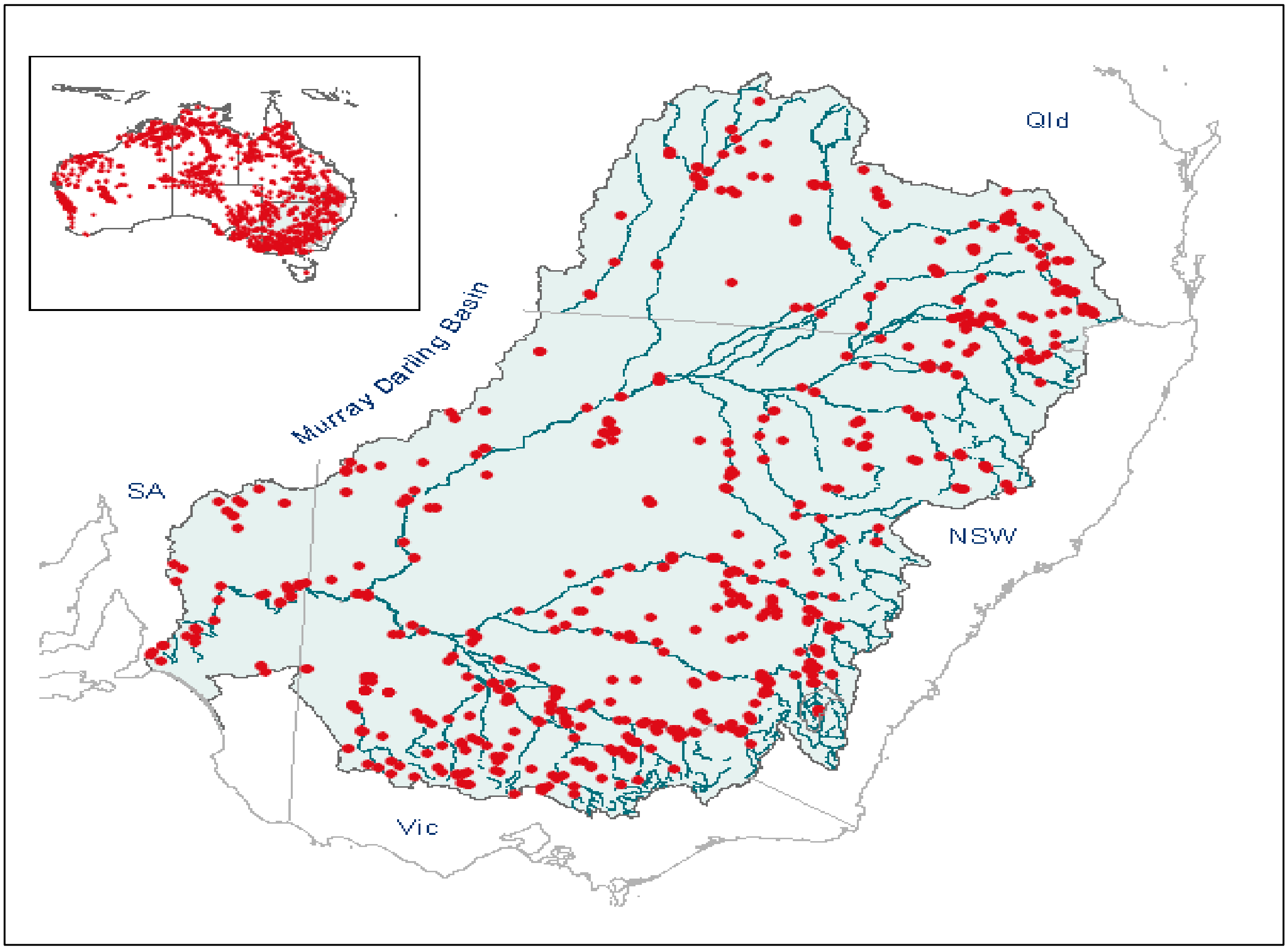
The *Myrtaceae* family is commonly found in a large number of biodiversity hotspots throughout its distribution, and contains species of great ecological and economic importance (Grattapaglia et al. 2012). *E. camaldulensis* in its native range, which stretches from its source in south-eastern New South Wales (36°47'41.31" S; 148°10'18.54" E) along a 2 520 km winding path, dividing the provinces of New South Wales and Victoria, towards the mouth of the Murray River, at the Coorong National Park, just south of Adelaide (35°33'28.10" S; 138°52'55.54" E). It can be found occupying large areas as forests, or along river edges in single rows of trees, and has become an icon of Australian vegetation, particularly along inland rivers (Roberts & Marston 2011).



Source: Grattapaglia et al. (2012: 464)

Figure 2.7 Global distribution of the *Myrtaceae* family, shown in black.

E. camaldulensis is the most widely distributed Eucalypt in Australia (Cunningham et al. s.a.), being found in all of its mainland states (McMahon, George & Hean 2010), although its greatest concentration is within the Murray-Darling basin. It prefers deep, moist subsoils with high clay content (Merchant et al. 2006). Figure 2.8 illustrates the national distribution of *E. camaldulensis* within Australia, with specific focus on the Murray-Darling basin, indicating its widespread distribution. It is important to note that *E. camaldulensis* may also be found in terrestrial environments beyond riparian zones, although it is primarily a riparian species (Roberts & Marston 2011), most commonly found in the aquatic-terrestrial productive transition zone (Ladd, Pepper & Bonser 2010).



Source: CSIRO (2004: 1)

Figure 2.8 *E. camaldulensis* distribution in the Murray-Darling basin, indicated by the red dots. The insert maps shows *E. camaldulensis* distribution throughout Australia.

E. camaldulensis is currently comprised of two variants: var. *camaldulensis* Dehnh. and var. *obtusa* Blakely, as well as one subspecies: subsp. *simulata* Brooker & Kleinig (EUCLID 2012). For the purposes of this report, unless otherwise indicated, *Eucalyptus camaldulensis* is discussed with no discrimination between subspecies or variants.

2.8.1 South African context

Since the first European landing on South African shores, and subsequent change of governance structures and authority in the Cape region, alien vegetation has been imported for a variety of uses (Showers 2010). Mixed, native, non-commercial evergreen forests cover a very small portion of the

South African landscape, with as little as 0.08% of the land surface area (approximately 3 000 km^2 in 1997)(Geldenhuys 1997). Afforestation was initiated on a large scale in 1890 in order to meet timber demands of the time (Geldenhuys 1997). A timber shortage during World War I, along with the need to stabilise dunes and drift sand in the immediate Cape vicinity, stimulated further investment into afforestation and importing of invasive species, many from Australia, such as *Acacia saligna* (Pooley 2010).

Timber plantation area has increased to +/- 1.4 million hectares by 1994 (Le Maitre, Versfeld & Chapman 2000), mostly composed of *Acacia*, *Pinus* and *Eucalyptus* species (Geldenhuys 1997). Australian species, mostly because of their suitability to winter rainfall regions and tolerance to fires and nutrient-poor soil, were particularly successful, but proved to be highly invasive (Bennett 2011; Pooley 2010). *Eucalyptus* species in particular, due to their fast growth and useful wood, were valued as timber species, and cultivated in mountain catchment areas, which were deemed highly suitable (Pooley 2010) and were imported and planted since the 1820s (Bennett 2011). Although little record exists of the spread of *E. camaldulensis* between 1880 and 1950, it is clear that the establishment of this species has not been a recent event.

2.8.2 Mediterranean climatic similarity

The region of origin for *E. camaldulensis* in Australia, is a Mediterranean climate region (Whitworth, Baldwin & Kerr 2012) characterised by cool wet winters, dry summers and schlerophyllus vegetation (Duff, Bell & York 2013). As the Western Cape also falls within a Mediterranean climate region (Roberts, Meadows & Dodson 2001), similarities between vegetation types can be expected even though the regions are large distances apart (Bonada & Resh 2013; Gasith & Resh 1999). Such similarities include rivers with highly variable annual and interannual discharge, predictable seasonal variation and periods of flooding and drought (Bonada et al. 2008; Bonada & Resh 2013), as well as being resource rich regions with water limitations (Stella et al. 2013), containing similar vegetation types (Gasith & Resh 1999). Furthermore, plant species within Mediterranean regions are adapted to fire, seasonal water shortages, and dynamic flooding and sediment regimes (Stella et al. 2013).

E. camaldulensis specifically, is adapted to fire through epicormic growth after burns (Gill 1997), dynamic flooding (Stone & Bacon 1994) and seasonal water shortages (Gindaba, Rozanov & Negash 2004). Furthermore, *E. camaldulensis* is highly adapted to a range of climatic zones, from tropical, to arid and temperate regions (Butcher, McDonald & Bell 2009), and grows well in subtropical and Mediterranean climates (Gomes & Kozlowski 1980). The hydrological and vegetation similarities between different Mediterranean climate regions of the world (Bonada & Resh 2013), in addition to the highly adaptable nature of *E. camaldulensis*, serves to illustrate how native ecological traits of *E. camaldulensis*, such as growth rate, tolerances and allelopathic characteristics of this species, are suitable to the Fynbos biome.

2.8.3 Growth and age

Although *E. camaldulensis* is regarded as possessing remarkable longevity (Roberts & Marston 2000), there are few reliable measures for estimating the age of an individual. In particular, estimates using tree rings are highly variable (Argent et al. 2004) and the use of allometric relationships using characteristics such as height and breast-height diameter (DBH) are not reliable (Roberts & Marston 2011). As such, accurately establishing the age of any given individual is often difficult. A common classification is suggested by grouping trees into different developmental categories based on their canopy characteristics (Roberts & Marston 2011). These characteristics are general for the entire genus (Roberts & Marston 2011). Besides growth characteristics, a general discussion on the ecology of *E. camaldulensis* is required before a complete understanding of the various environmental interactions of this species can be formed.

2.8.4 Ecology

2.8.4.1 General

E. camaldulensis is regarded as capable of exceptional production and growth at suited sites (Zohar & Schiller 1998), as it is a fast growing species (Tanvir, Siddiqui & Shah 2002). In South Africa flowering occurs mainly during late spring to midsummer, or specifically September to January (WIP 2013), though individual time spans may be much shorter (Roberts & Marston 2011). Flowering depends highly on physical location and local conditions, (McMahon, George & Hean 2010) and lasts typically between 4 to 6 weeks (CSIRO 2004). Around 45% of flowers do not mature (CSIRO 2004). Pollination occurs mainly by insects, birds, bats and small mammals (CSIRO 2004), thereby ensuring pollination occurs across relatively long distances, allowing gene flow to be spatially widespread (Roberts & Marston 2011).

E. camaldulensis are furthermore able to survive in both arid and semi-arid environments, due to a high tolerance shown for a range of environmental characteristics (Table 2.2), which coupled with their various economic benefits and uses, lead to the widespread distribution of *E. camaldulensis* (McMahon, George & Hean 2010).

Table 2.2 Native *E. camaldulensis* environmental limits.

Variable	**Minimum**	**Maximum**
Rainfall (mm)	150	2500
Mean monthly temperature (°C)*	1	41
Altitude (m)	20	700

Source: McMahon, George & Hean (2010: 2)

According to McMahon, George & Hean (2010), *E. camaldulensis* is also highly suited for charcoal and firewood production. Other uses of the species is summarised in Table 2.3, indicating the wide range of characteristics *E. camaldulensis* is used for.

Table 2.3 Common environmental services derived from *E. camaldulensis*.

Environmental Services	**Suitability**
Habitat	✔ ✔
Nitrogen fixing	-
Salinity control	✔
Shade/Shelter	✔ ✔

Soil/Water conservation	✔
Windbreak	✔ ✔
✔ =Potentially suitable	✔ ✔ = Very suitable

Source: McMahon, George & Hean (2010: 4)

2.8.4.2 Competition

Competition occurs between organisms when one depletes the supply of an essential resource to the detriment of another (Del Moral & Muller 1970). Such competition may exist above ground, such as in light competition, or below ground, such as for nutrients or water (Bouillet et al. 2013), and is expressed in intraspecific, and interspecific competition. The presence of competition would logically manifest developmental influences on all the species involved. Such influences have been noted with a decrease in size and performance of *E. camaldulensis* seedlings in a recent trial (Ladd, Pepper & Bonser 2010).

- Intraspecific competition

E. camaldulensis exhibit intraspecific competition, whereby individuals of the same species compete for resources and influence other individuals of that same species. The severity of the competition between same-species individuals vary with environmental conditions experienced from stand to stand, but it has been established that intraspecific competition (measured as density or stand structure) is not as strong an influence on tree mortality as, for example, moisture stress would be (Cunningham et al. 2010). One study confirmed that a higher stand density resulted in a reduced mean tree size as a consequence of resource competition (Thoranisorn, Sahunalu & Yoda 1990). Additionally, it was found that self-thinning was initiated earlier on in the life span of the study group trees, in response to increased competition for light (Thoranisorn, Sahunalu & Yoda 1990). However, this is not the only type of competition experienced by *E. camaldulensis*.

- Interspecific competition

In studies evaluating 'natural' *E. camaldulensis* plots and species diversity, it was found that *E. camaldulensis* plots exhibit reduced species diversity (Tyynelä 2001). However, this may not be due to competition inhibition from *E. camaldulensis* alone, as there are often anthropogenic factors that need to be taken into account. *E. camaldulensis* does, however, exert competitive pressure on neighbouring species. In South Africa, and in particular along the Breede River, *E. camaldulensis* are frequently accompanied by invasive *Acacia* species. A recent study conducted indicated that *Eucalyptus* species (specifically *Eucalyptus globulus*) could be combined successfully with *Acacia* seedlings in stands where the relative growth rates between the species are compatible (Bauhus, Van Winden & Nicotra 2004), i.e. where the *Eucalyptus globulus* shade tolerance allows for initial matching grow rates to that of the competitive *Acacia*, resulting in eventual crown dominance of the *Eucalyptus globulus*.

Studies found that *E. camaldulensis* stands may actually benefit from the presence of species such as *Acacia mangium* (Bouillet et al. 2013), or *Acacia mearnsii* (Bauhus, Van Winden & Nicotra 2004), through synergistic effect of combining these species in certain proportions within stands. It is thought that the interactions between such species enhance the production of *E. camaldulensis* through the addition of nitrogen or phosphorous to the soil (Forrester et al. 2006), as well as by reducing intraspecific competition due to a reduced species-specific density and through complementary light requirements of these species (Bauhus, Van Winden & Nicotra 2004; Bouillet et al. 2013).

The apparent favourable competition interactions for the *E. camaldulensis* in such stands could thus be a function of nutrient availability rather than moisture availability. Regardless of which factor is responsible for the synergistic effect, such advantageous net-competition effects for *E. camaldulensis* would conceivably manifest in real-world situations through mixed-species stands within the riparian zone.

2.8.4.3 Tolerances

Abiotic stresses such as high salinity, temperature extremes and drought conditions are often the cause of large scale loss of production, both agriculturally and naturally (Osakabe, Kajita & Osakabe 2011). *E. camaldulensis* displays a variety of responses to environmental disturbances such as drought, flooding and high salinity. The various responses improve the overall fitness of the species for survival, and are discussed in more detail below.

- Salinity

Salinity, although not a specific objective within this study, does impact on the growth and establishment of *E. camaldulensis*. Salinity is controlled in the Breede catchment through dam water regulation, where releases of dam water is used to reduce high salt concentrations within the catchment, which may thus influence *E. camaldulensis* development within the catchment.

E. camaldulensis is regarded as a salt tolerant species (Dale & Dieters 2007; Mensforth et al. 1994; Sun & Dickinson 1995b; Sun, Dickinson & Bragg 1994) through its ability to maintain ion uptake exclusion under stress conditions (Nasim et al. 2009). At high concentrations, a survival rate under various saline conditions of 70-90% (Akhtar et al. 2008), 85-100% (Sun & Dickinson 1995a), or 96.7-100% (Sun & Dickinson 1995b) was found in Australia and Pakistan respectively. *E. camaldulensis* seedlings were further shown to survive concentrations of up to 200 mM NaCl, or 11.689 g/L NaCl in a study conducted by Sun & Dickinson (1993). Elsewhere in the literature, seedlings were shown to tolerate water logging with saline solution equivalents of 1700 mg NaCl (Roberts & Marston 2011).Various other studies showed no *E. camaldulensis* mortality due to higher salt content (Akhtar et al. 2008; Feikema & Baker 2011; Sun & Dickinson 1995b), illustrating the salt tolerance ability of this species further.

Biomass for *E. camaldulensis* was found to be greatest at 50 mM NaCl (compared to biomass measured for 0 mM Nacl solution), where after the biomass declined with a concentration increase

(Sun & Dickinson 1993). This would suggest that *E. camaldulensis* trees develop comparatively better under slightly saline conditions, and not at extremely low salinity conditions that is unlikely to be found in the natural environment. Development, such as height (Feikema & Baker 2011), leaf length and leaf count (Sun & Dickinson 1993) is however, still reduced under highly salinity (Cha-um & Kirdmanee 2010), suggesting mild salinity to be beneficial to *E. camaldulensis* trees development but not extreme salinity.

- Drought

E. camaldulensis display specialised physiological adaptations in order to survive environmental stresses, and has adaptations specific for drought tolerance (Liu et al. 2001) and, as such, is regarded as a drought-tolerant species (Lemcoff et al. 2002). Tolerance is achieved through osmosensing (Hsiao 1973) and delayed stomatal response (Gindaba, Rozanov & Negash 2004). *E. camaldulensis* is most sensitive to water stress as seedlings (Gindaba, Rozanov & Negash 2004), which causes a rapid decline in water potential, photosynthetic rate and stomatal conductance (Gindaba, Rozanov & Negash 2004). Seedlings response is through a greater developmental investment in roots than shoots, known as the ‘chasing water’ response (Roberts & Marston 2011).

The response exhibited in general to drought conditions by *E. camaldulensis*, however, is by a combination of efficient stomatal control of respiration, osmotic adjustment (Cha-um & Kirdmanee 2010), delayed leaf growth, increased turgor pressure, maintaining water uptake and relative water content (RWC) and by varying access to groundwater sources through the deeper taproot (Lemcoff et al. 2002). These mechanisms allow *E. camaldulensis* to survive drought conditions, thus influencing establishment and distribution within its environment. Other stressors, such as waterlogging have also been noted.

- Water logging

E. camaldulensis is further regarded as a submersion tolerant species (Akilan et al. 1997), by regulating ion uptake (Liu et al. 2001), hypertrophy (Gomes & Kozlowski 1980) and increased aerenchyma (Hook 1984) in submerged portions of the stem, and on submerged stems, as well as development of adventitious roots (Gomes & Kozlowski 1980). Seedlings are also highly tolerant of waterlogging, with 14-18 week old seedlings tolerating complete submersion for up to 72 days without mortality, and 50-60cm tall seedlings able to survive 4-6 months of partial inundation with no mortality (Roberts & Marston 2011). Submersion does influence different characteristics of plant development (Akilan et al. 1997), by reduced leaf area responses (Stone & Bacon 1994). The ability of *E. camaldulensis* to tolerate submersion for prolonged periods may contribute to its success outside of its native range.

2.8.5 Hydrological regime and recruitment

In their native range, *E. camaldulensis* requires moisture from flooding or localised rainfall to stimulate germination, with levels required for germination being greater than 5mm in surface coverage (Jensen, Walker & Paton 2008). Germination is possible in the absence of flooding, and may happen in high rainfall years, but is greatly enhanced by the correct flood conditions (Roberts & Marston 2000; 2011). The duration of flooding is also an important consideration in terms of the effect on *E. camaldulensis* establishment (Roberts & Marston 2000). Soil moisture from short duration flood events were found to return to previous levels within a short period (40 days) after the event and, as such, short term benefits only were obtained from short term flooding events (Roberts & Marston 2000). Furthermore highly saline or mildly dry soil conditions could lead to short term moisture stress which, if continued for longer periods, may inhibit tree development (Roberts & Marston 2011).

Sites experience relief through flood water which reduces localised salinity levels and replenishes soil moisture. Greater soil water recharge would be of greater benefit to *E. camaldulensis* in terms of growth and development, and vice versa (CSIRO 2004). According to Roberts & Marston (2000) the survival of *E. camaldulensis* during drought conditions is dependant more on the availability of soil moisture and reduction in canopy water loss, than through physiological tolerances of water stress. *E.*

camaldulensis trees are able to rely on groundwater through the use of their deep 'sinkers' (CSIRO 2004), or taproots which allow for water extraction from depths in excess of 10 m (Roberts & Marston 2000). Although surface water is used by *E. camaldulensis* to varying degrees across various locations, groundwater is a considerable and constant source of water (between 20-50%), regardless of the availability of nearby fresh surface water (Roberts & Marston 2000).

Sites with infrequent flooding have thus been shown to support less growth of *E. camaldulensis* (Roberts & Marston 2011) and, as such, added moisture through flooding or rainfall can be seen to contribute to localised maintenance and development (CSIRO 2004). Such rainfall also assists in avoiding severe drought and dieback, and may even lead to growth pulses (Roberts & Marston 2011). The maintenance of *E. camaldulensis* is thus specifically linked to the flooding regime (CSIRO 2004).

Seeds will germinate readily under conditions of sufficient available moisture, with low temperature and light being the principle restraints in germination (Roberts & Marston 2011). Seeds will float on water, which can be for as long as 10 days under laboratory conditions, after which they will sink and germinate, often aggregating where water levels have started receding (Roberts & Marston 2011). *E. camaldulensis* seedlings and young juveniles exhibit similar morphological responses to submersion and flooding from anywhere between 6 and 22 months of age (Roberts & Marston 2011).

It is the seedlings growth rate characteristics, in particular the speed of growth and resource allocation that determine the invasive potential of *E. camaldulensis* (Roberts & Marston 2011). Seedlings are known also to germinate without flooding conditions (Roberts & Marston 2000). For seedlings, duration and depth of inundation are the two most important factors influencing germination and production, as stress by lack of O_2 if submerged was observed (Roberts & Marston 2011). A summary of the water regime characteristics are described in Figure 2.9 below, indicating the various intricacies of the influences of the hydrological regime on *E. camaldulensis* establishment and development.

WATER REGIME FOR VIGOROUS GROWTH

Maintenance

Frequency of flooding:	About every one to three years for forests, and about every two to four years for woodlands. Site-specific studies are worthwhile for large areas.
Depth of flooding:	Not critical.
Duration of flooding:	About five to seven months for forests, and about two to four months for woodlands. Individual floods may be longer or shorter without major consequence; some variability around the mean is encouraged.
Timing of flooding:	Start of flooding is not critical for River Red Gums, but more growth achieved if flooded during spring–summer. Timing is important for understorey vegetation, plant communities in wetlands and other biota.

Regeneration

Timing of flooding:	Flood recession in spring, possibly even later, so as to provide warm and moist conditions for germination and early seedling growth.
Seedling establishment:	• **Follow-up flood**: Follow-up flood to recharge soil moisture for seedlings is desirable. This may be in the same year as germination or in the following year. • **Depth of flooding**: Shallow flooding, in the order of 20 to 30 cm, preferable to avoid over-topping seedlings in first year. This may not be easily achieved over wide areas. If depth not easily managed, then focus on duration. • **Duration of flooding**: Flooding for four to six weeks adequate, but longer can be tolerated, depending on seedling age and whether totally submerged.

Critical interval

Does not form a seed bank, hence is important to maintain trees in good condition so that a good supply of seed is available for regeneration when needed. Reflood after about three years for forests—and five to seven years for woodlands—if trees are to retain vigour. Longer intervals may be tolerated periodically, but if these become routine then tree condition is likely to deteriorate in the long term.

Source: Roberts & Marston (2011: 49)

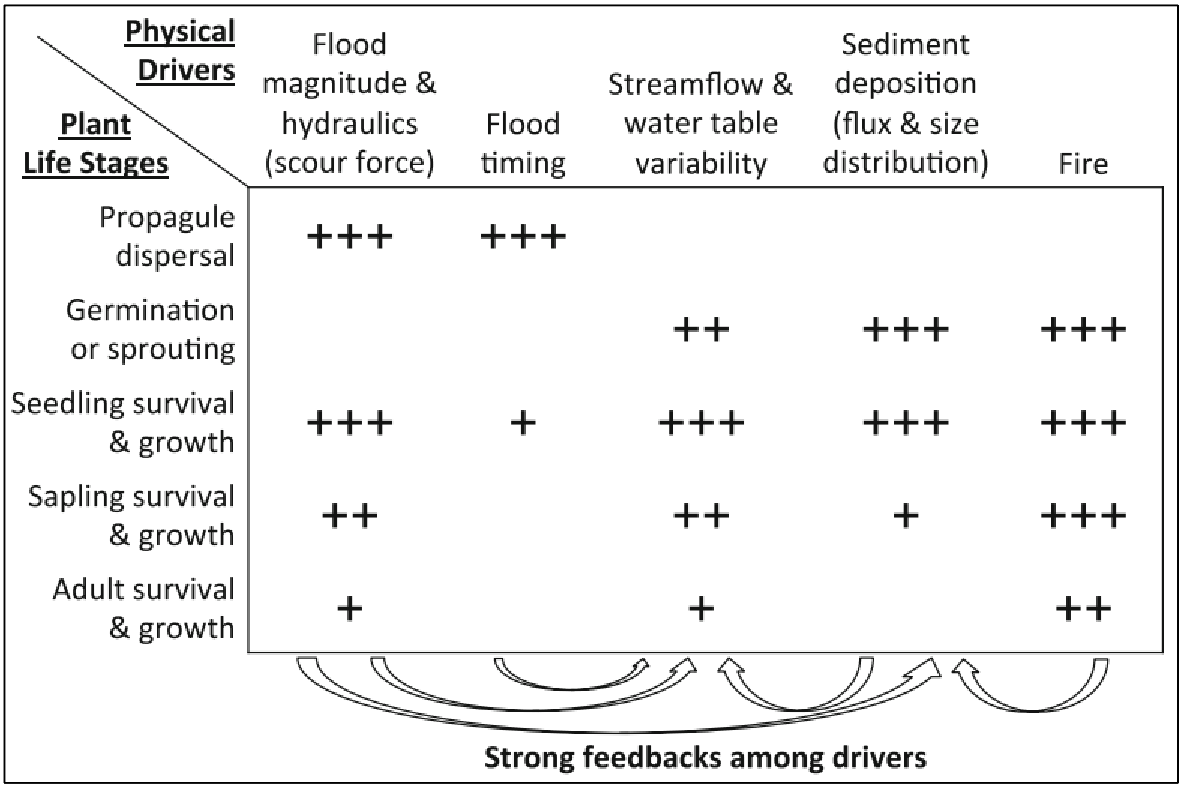
Figure 2.9 Summary of *E. camaldulensis* water regime for vigorous growth.

Winter flooding and flood recession are regarded as being less favourable for germination, as seed and seedlings may be exposed to unfavourable water and/or temperature conditions in the following months (Roberts & Marston 2000). Spring and summer flooding, followed by summer flood recession is regarded as optimal for establishment (Roberts & Marston 2000). The Western Cape experiences predominantly winter rainfall, and thus winter flooding. Cool or rainy summers are able to alleviate the water and temperature stresses experienced by seedlings germinated during winter, thus overcoming the lessened suitability of flooding during the winter periods (Roberts & Marston 2000). Germination may still occur readily during both winter and summer, with the establishment of seedlings being influenced by the absence or presence of sufficient soil moisture experienced in the next few seasons (Roberts & Marston 2000).

Young seedlings are sensitive to frost and desiccation predominantly (Roberts & Marston 2011), which is the cause of mortality during the initial stages in the life cycle. Seedling establishment, rather than seed production or germination is regarded as the critical life history phase for this species (Roberts & Marston 2011). ‘Filling frequency’ (i.e. the frequency of inundation) and duration of flooding are the more influential factors to be considered at the catchment scale (Roberts & Marston 2011).

It is then clear, that *E. camaldulensis* is not dependent on the occurrence of floods to enable reproduction and allow for species survival, but depending on the local conditions, may require the replenishing of groundwater experienced during flood events to relieve potentially fatal water stress (Roberts & Marston 2011). However, flooding and rainfall are still major contributing influences on the spread, survival and distribution of this species (Roberts & Marston 2011).

In their native range, where rainfall is relatively scarce and the climate is regarded as semi-arid, flooding plays a major part in the distribution of *E. camaldulensis* through replenishing groundwater and relieving water stress (Roberts & Marston 2011). Figure 2.10 illustrates the physical influences experienced by woody riparian plants (such as *E. camaldulensis*) in Mediterranean regions, and the relationship of the hydrological regime of a region on the succession of riparian wood species.



Source: Stella et al. (2013: 299)

Figure 2.10 Physical influences on woody riparian plants in Mediterranean regions. Number of symbols indicate magnitude of influence, and the arrows indicate reinforcing or lessening relationships between drivers and plant life stages.

The question remains as to what extent local flooding regimes are similar to that described above, and if the local *E. camaldulensis* communities display the same relationship between the hydrological regime and distribution. One would intuitively expect the alien *E. camaldulensis* to show the same relationship with rainfall and flooding as in nature. Many of the short-duration, high-intensity flooding events experienced during cut-off lows over the Western Cape, are expected to provide sufficient moisture for the germination of *E. camaldulensis* seedlings. Little is currently known about the relationship between the hydrological regime of the Breede River and the presence of *E. camaldulensis*.

2.9 LEGAL FRAMEWORK

Environmental management of invasive species is a complex topic, with various legal implications and ethical considerations. This section focuses on the general legal framework to which IAP management relates, specifically to landowners such as those interviewed in this report.

2.9.1 Chapter II of the Constitution of the Republic of South Africa (No. 108 of 1996)

The bill of rights (South Africa 2006) contains essential rights of each South-African citizen, ensuring democratic legislation and approaches to all concerns of governance. Although IAPs are not specifically addressed therein, the foundation of all environmental regulation follows from section 24 (Environment), in which the right to prevention of pollution and ecological degradation, in addition to the promotion of conservation are enshrined for all citizens (Paterson 2006; South Africa 2006). Environmental legislation giving effect to these principles, such as the National Environmental Management Act (NEMA) 107 of 1998 (South Africa 1998a), therefore adheres to the core rights expressed therein (Van Der Linde 2006).

2.9.2 Common law

Common law relates to law developed through the court decisions as passed down from Roman-Dutch legal tradition, as opposed to statutory law, which represent regulations and legislatures (Schoeman-Malan 2007). The common law is generally not codified, and is guided by court decisions and individual statutes (Schoeman-Malan 2007). Though not specific to the environment, the common law does provide a basis for certain environmental regulation (Van Der Linde 2006). In relation to IAPs, the ‘principle of common nuisances’ apply to the impact neighbouring properties have on one another (Nabileyo 2009). This may be through fire hazard posed by stands of IAPs, or through encroachment of vegetation onto a neighbour’s property (Nature Conservation Corporation 2006). Fire hazard in this context may also relate to the Veld and Forest Fire, Act 101 of 1998 (South Africa 1998c), which require landowners to prevent wildfire spread. Civil claims may arise from a neglect to fulfil such obligation (Nature Conservation Corporation 2006).

2.9.3 National Environmental Management Act (Act 107 of 1998)

The National Environmental Management Act 107 of 1998 (South Africa 1998a) regulations require every person polluting to a significant level take reasonable measures to prevent and mitigate such degradation (Strydom & King 2009). This provision includes control and prevention actions taken to cease or reduce environmental degradation. In the context of IAPs, this would require landowners to ensure mitigation of IAP impacts, and removal on offending properties (Paterson 2006). Training of staff in order to correctly manage IAP instances may also be required. As with the Conservation of Agricultural Act (South Africa 1984) regulations, ministerial authority may be delegated to local authorities to hold responsible offending landowners to remove, rehabilitate and recover environmental damage sustained during their tenure, with costs being incurred and recovered on behalf of - and from the landowner (Paterson 2006).

2.9.4 National Environmental Management: Protected Areas Act (No. 57 of 2003)

Although not specifically addressing IAP presence, the National Environmental Management: Protected Areas Act 57 of 2003 (South Africa 2004a) regulations require protected areas, or the designation of future protected areas, to be managed with an approved management plan, conducted by a management authority (Paterson 2006). Such plans are required to include IAP removal programmes with compliance and monitoring made mandatory in the case of approval, failing which management mandates may be withdrawn in the event of poor implementation (Paterson 2006).

2.9.5 National Environmental Management: Biodiversity Act (No. 10 of 2004)

Unlike other regulations, the National Environmental Management: Biodiversity Act (NEMBA) 10 of 2004 (South Africa 2004b) legislates directly for the IAS presence in the country, and distinction is made between ‘alien species’ and ‘invasive species’ (Paterson 2006). In the case of listed invasive species, such as *E. camaldulensis*, certain ‘restricted activities’ are prohibited unless a permit has been issued (Strydom & King 2009). Risk assessments are required in order to issue permits, allowing for greater control over ‘restricted activities’. NEMBA also includes regulation on the manner in which IAP species are removed, as the use of incorrect methods may be more harmful than the IAP species themselves (Paterson 2006). In 2014, a draft IAP listing was published detailing changes to categories initially proposed under the Conservation of Agriculture Resources Act 43 of 1983 (CARA), to include subcategories of 1a and 1b. Category 1a species are prohibited in terms of 71A(1), which are required to be combatted or eradicated (South Africa 2014b). Category 1b species also relate to sections 71A(1), and are required to be controlled (South Africa 2014b). If accepted, *E. camaldulensis* will be classified under category 1b, within all protected areas, riparian areas, listed ecosystems, or within certain biomes, including the fynbos biome (South Africa 2014a). As such, being in control of or having control over any specimen will be exempted, but causing any spread of the specimen will be prohibited, unless a permit was obtained (South Africa 2014a). This places a greater restriction on *E. camaldulensis* in general, but importantly elevates riparian occurrence to that of category 1b.

2.9.6 Conservation of Agricultural Resources Act (No. 43 of 1983)

The Conservation of Agricultural Resources Act (CARA) 43 of 1983 (South Africa 1984), which is administered by the DoA nationally, categorise IAS into three classes based on their respective impacts and invasive potential. *E. camaldulensis* is a category II species, which are allowed only within specifically demarcated areas, designated by the executive officer, and for which a permit was obtained. These areas are regarded as ‘water use areas’, which broadly defined, means any stream flow reducing activity (Strydom & King 2009), and is subject to further legislation under the National Water Act, no. 36 of 1998 (Department of Agriculture 2013). Specimens outside demarcated areas, and specifically within 30m of the 1:50 year flood line of any water course or wetland, must be controlled, unless exempted or authorised by the executive officer (Department of Agriculture 2013). Demarcation requires the stands to serve either commercial or utilitarian purposes (Nature Conservation Corporation 2006).

Implementation of CARA regulations place landowners under obligation to remove IAPs on their property, as it is a criminal offence to tolerate their occurrence (Department of Agriculture 2013). Clearing may be initiated on behalf of the landowner as directed by the DoA, with costs being recuperated from the landowner or registered against the title deed if required (Nature Conservation Corporation 2006). Further, under the CARA regulation, appropriate removal methods are to be used for each species, with the use of bulldozing and fire for removal requiring permission from the DoA prior to implementation (Department of Agriculture 2013).

2.9.7 National Water Act (No. 36 of 1998)

Under the stipulation of the National Water Act (NWA) 36 of 1998 (South Africa 1998b) any activity reducing stream flow or altering its course, bank or stream characteristics are all regarded as 'water use' activities (Strydom & King 2009). These activities are regulated while considering, amongst others, the current and future water needs of users, reduction and protection of water resources, as well as the degradation of water sources (Strydom & King 2009). Further, under the NWA, any action by an individual detrimentally affecting a water resource is regulated, and is subject to a criminal offence (Nature Conservation Corporation 2006). Although no direct action may be taken by a landowner, in conjunction with the CARA such applications may include the removal of IAPs on such properties.

2.9.8 Environmental Conservation Act (No. 73 of 1989)

The Environmental Conservation Act (ECA) 73 of 1989 (South Africa 1989) regulations apply mainly to two different applications surrounding IAP species in South Africa, the first being commercial applications, and the second applying to all persons in general. Similar to the regulations under NEMA, landowners can be held responsible to control or eradicate IAP under the ECA (Paterson 2006).

2.10 REMOTE SENSING APPLICATION

Since the first use of photography to gather data of the earth's surface in 1858 (Campbell 2002), remote sensing, defined as observation of earth surfaces through reflected or emitted electromagnetic energy (Campbell 2002), has been used for a variety of applications, from climate change studies, desertification studies, weather investigation and ozone depletion inquiries (Barrett & Curtis 2004). Improvements in technology since then have shaped the various uses thereof even further, and have created the Earth observation sciences with which many are familiar in modern times (Gupta 2003).

The particular usefulness of remote sensing applications is evident in resource management (Sawaya et al. 2003), biodiversity management (Turner et al. 2003), change detection (Green et al. 1996) and riparian vegetation mapping (Stella et al. 2013). Ecological studies are increasingly relying on the use of remote sensing (Matson & Ustin 1991). Its use in weed management is particularly noteworthy in the context of this study, especially as remote sensing has been used for environmental monitoring, land use evaluation, vegetation stress inquiries, IAP mapping (Stella et al. 2013) and change detection in the past (Shaw 2005).

In particular aerial photographs, have also been used in vegetation mapping (Harvey & Hill 2001; Huang & Asner 2009). The use of aerial photographs over satellite imagery has proven advantageous in certain situations, taking into consideration the limitations inherent to satellite imagery (Ozesmi & Bauer 2002). Indeed, aerial photographs are the most common remote sensing technique applied to the detection of plant species (Müllerová et al. 2005). Harvey & Hill (2001) used aerial photographs to map Australian vegetation in 2001, by georeferencing scanned aerial imagery, mosaicking datasets and thereafter ‘heads-up digitising’ features using the tone and texture of vegetation features as the main discriminants to define boundaries. Harvey & Hill (2001) found the use of aerial photography to be superior to the use of satellite imagery in their study, due to a greater accuracy obtained, facilitated by the interpretation cues of context and texture, which aided the identification of vegetation communities to greater accuracy.

Additionally, visual interpretation of data was regarded as capable of producing greater accuracy with a higher degree of precision in comparison to automated processing, and this method is preferred for applications of land cover evaluation (Panigrahy et al. 2010). If based on sound ground-knowledge, or real-world knowledge of the study area, this method may reduce the inherent risk factors associated with digital interpretation techniques, as greater control over any observed change may be exerted (Panigrahy et al. 2010).

Mapping of invasive species was also done using aerial photography (Underwood, Ustin & DiPietro 2003), in particular, the mapping of *E. camaldulensis* for stand condition (Cunningham et al. 2009; Johnston & Barson 1993). In the South African context, aerial photographs have also been used to map other invasive species, such as *Acacia cyclops* and *Pinus pinaster* (Higgins, Richardson & Cowling 2001).

The use of aerial photography for mapping island bar formation and change over time has also been demonstrated (Camenen, Jodeau & Jaballah 2013; Dexter & Cleur 1999; Kusimi 2008). Accurate evaluation of fine scale interactions were possible with the use of low-flown helicopter and drone imagery (Camenen, Jodeau & Jaballah 2013).

Spatial analysis, which refers to a subset of analysis methods, specifically with the object of resolving a scientific or decision making process, dependant on certain spatial characteristics, have long been in use (Goodchild & Longley 1999). These methods can be classified as confirmatory or exploratory (Goodchild & Longley 1999), and GIS-based spatial analysis has been a distinctive area of activity in the past, but is increasingly being used for scientific applications (Goodchild & Longley 1999).

GIS-based spatial analysis has specifically been used for vegetation analysis, with raster models, scanned maps, digitising and vectorising raster representations, comprising the four main means of representing vegetation maps in a GIS digitally (Goodchild 1994). One particularly useful application of spatial analysis is the estimation of area (as a measure of surface coverage) of a given feature of interest (Goodchild 1994). Goodchild (1994) argues the unification of classification (such as digitising features according to a specific criteria) and traditional cartographic techniques may be useful in addressing traditional limitations to vegetation mapping.

It is clear the use of aerial photography and remote sensing has multiple applications and advantages for the mapping of vegetation and river morphology. As the use of aerial photography is relative cost-effective (Barrett & Curtis 2004), its use has become invaluable for modern resource managers.

2.11 CURRENT STRATEGIES

2.11.1 South African IAP clearing

South African authorities have long recognised the harmful potential of IAPS nationally (Richardson & Van Wilgen 2004; Van Wilgen et al. 2001). The predominant strategy for the past 15 years to address their presence has been through the national Working for Water (WfW) programme (Van Wilgen et al. 2012). The programme utilised both mechanical and chemical control, with job creation and poverty alleviation as one major objective (Van Wilgen et al. 2012). The efforts of the WfW programme have been supplemented by appropriate biocontrol agents, the promulgation of legislation necessitating landowner involvement and the encouragement of a payment for ecosystems approach (Van Wilgen et al. 2012).

Poona (2008) identified three broad management approaches employed to address the IAP problem, namely that of (i) environmental education, (ii) legislative or policy approach and (iii) a technology centred approach. Van Wilgen et al. (2011) further states five outcomes used in creation of management strategies, namely that of (i) prevention, (ii) eradication, (iii) containment, (iv) impact reduction and (v) value addition. These outcomes may be achieved through a combination of approaches, such as control methods (mechanical, chemical, biological), risk assessments, early detection and rapid response, sterile cultivar development, harvesting of IAPS, development of the 'payment for ecosystem services' approach, environmental education, legislation and spatial prioritisation (Van Wilgen et al. 2011).

The WfW programme has made significant impact on the control of IAPS nationally, and has spent more than R3.2 billion while creating temporary employment for more than 30 000 persons per annum (Marais & Wannenburgh 2008). Other organisations, such as state agencies and government departments (Poona 2008), amongst others, notably the South African National Biodiversity Institute (Wilson et al. 2013), have additionally adopted the common goal of addressing the impacts of IAPS, assisting indirectly in the managerial goals of the WfW programme.

2.11.2 *E. camaldulensis* studies and clearing

Beyond the clearing conducted by various agencies and organisations nationally, much research has been conducted into the ecology, impacts and distribution of *E. camaldulensis* locally. Forsyth et al. (2012) identified *E. camaldulensis* as a priority invader in the Western Cape specifically, while Tererai et al. (2013) found its presence to reduce native species richness and diversity, specifically in riparian ecosystems. The fast growth of *E. camaldulensis* (Pinyopusarerk et al. 1996; Thoranisorn, Sahunalu & Yoda 1991), its comparatively high water consumption (RHP 2011), and its ability to change the morphology of river channels (Jacobsen et al. 1999; RHP 2011) have also been identified. In addition, the Breede catchment was also recognised as being one of the worst IAPS invaded region in the country (Brown et al. 2004), and was identified as being under invasion by *E. camaldulensis* specifically (Forsyth et al. 2004). Studies have been conducted evaluating the potential of active and passive recovery after *E. camaldulensis* clearing (Ruwanza et al. 2013), however, our understanding of the extent and impacts of such invasion is limited (Tererai et al. 2013).

Clearing is currently (2012) being conducted by the DoA along the Breede catchment for all riparian invaders, including *E. camaldulensis* (Röscher 2012, Pers com). Figure 2.11 illustrates the various areas within the catchment in which clearing was conducted (2011-2013) by the DoA and WfW, respectively. Of particular interest is the programme focussed on the eastern tributaries of the Breede River in Figure 2.12 (RHP 2011). Clearing conducted along the eastern reaches of the Breede catchment may allow for greater flow within the tributaries, increasing the water load downstream of the confluence and assisting with salinity within those sections. In addition, any removal of adult IAPs would reduce the amount of viable seed spread downstream, thereby potentially reducing the seedling recruitment of IAPs below Swellendam.

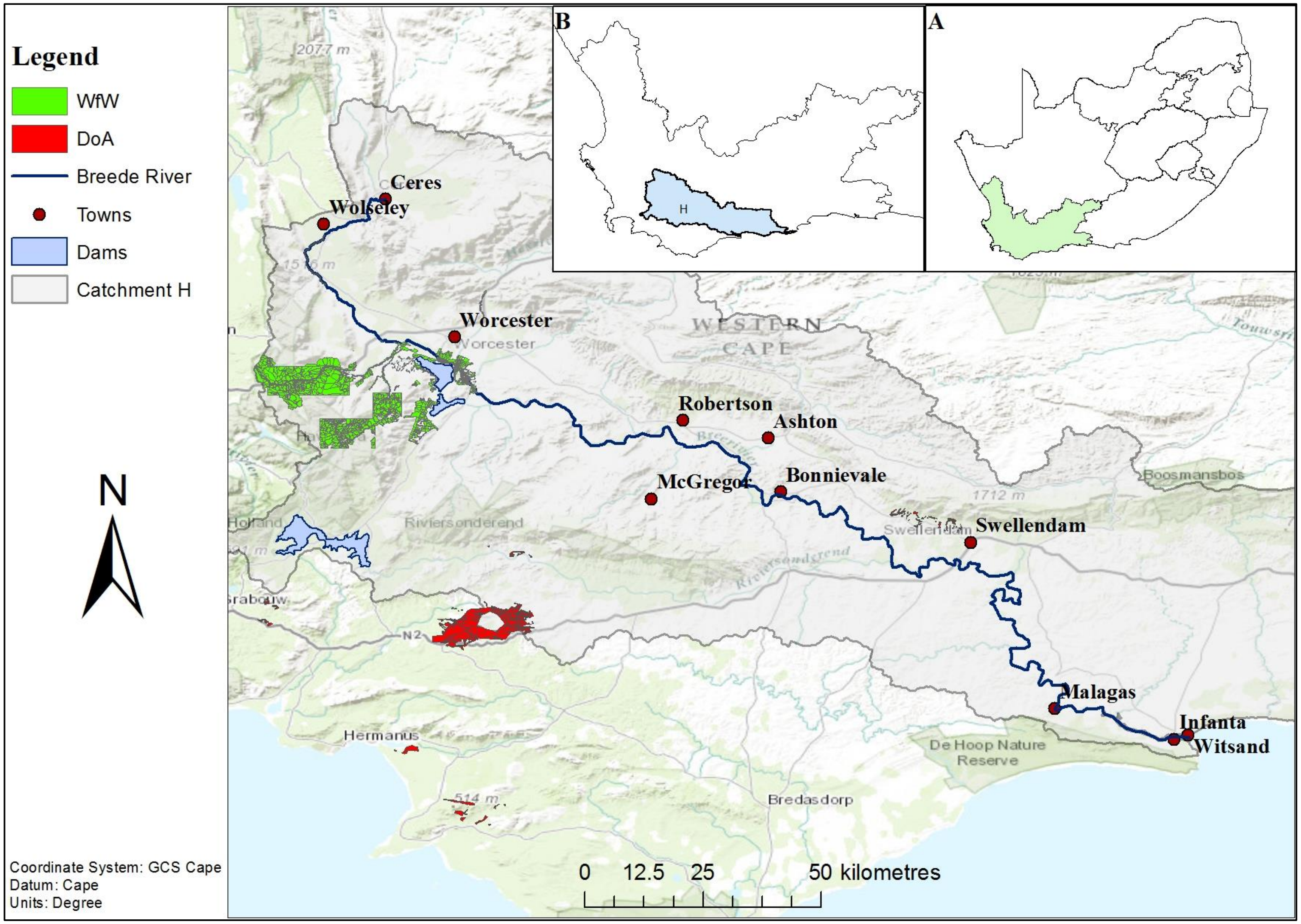
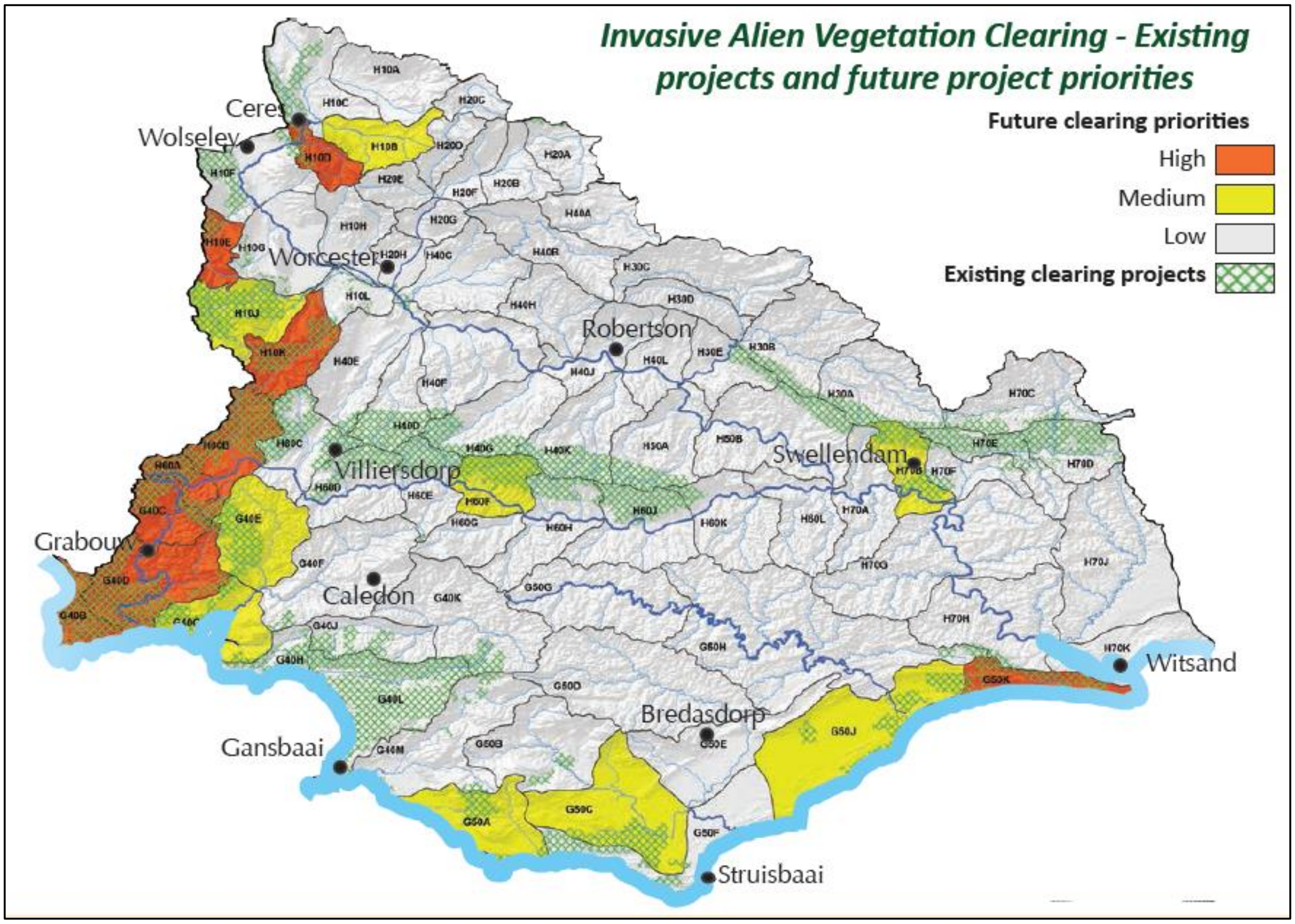


Figure 2.11 Areas within the Breede catchment cleared of aliens in general by the DoA and WfW between 2011 and 2013. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014).



Source: RHP (2011: 69)

Figure 2.12 Areas prioritised for general alien clearing by WfW in 2011.

The WfW website furthermore provide details on the appropriate clearing methods and herbicide for the control of *E. camaldulensis* (WfW 2014), provided freely to the public for own use, and as such serves as an indication of the wider approach utilised by WfW to have the public and land owners assist in their management of this species. Indeed, as indicated in the discussion of the relevant legislation earlier in this report, numerous regulations place an onus on landowners to maintain and eradicate IAPS on their property. Other agencies, such as the Breede-Overberg Catchment Management Agency, although not specific in function to the control of IAPS, have the aim of preserving the environment to assist in water quality within the catchment, and ultimately to improve livelihoods for their respective constituents (RHP 2011). This includes assisting in the management of IAPS in their respective catchment (RHP 2011).

CHAPTER 3 RESEARCH METHODS

This chapter describes the study area, target species, data collection, preparation and analysis, concluding with a discussion on the experimental design.

3.1 STUDY AREA

3.1.1 Breede River and tributaries

The study area was the riparian zone section of the Breede River between Worcester and Swellendam, in the Western-Cape Province (Figure 3.1). The catchment is characterised by the Du Toit's Mountains to the west and the Langeberg Mountains to the east (Figure 3.1). The river originates near Ceres (33° 22' 09.72" S; 19° 18' 41.72" E) at the Skurweberg mountains, after which it flows south-east through Worcester towards the east coast of South Africa, reaching the ocean at Witsand/Port Beaufort (34◦ 23'48.24" S; 20◦ 51' 04.83" E). The channel was studied in the middle section of the river, the upper section being the section from the river source near Ceres, to the town of Worcester. The middle section starts at Worcester, and ends at Swellendam, and the lower section represented by the remainder of the channel from Swellendam to the river mouth at Witsand.

Clearing was planned by Department of Agriculture (DoA) staff members for the upper section of the river, nullifying the study of IAPS therein. The lower section was furthermore found during pilot surveys, for unexplained reasons, to exhibit a very small amount of *E. camaldulensis* populations (two populations, consisting of no more than 30 individuals combined), thus eliminating it as a study section. The remaining middle section was therefore identified as the study area used in this study, due to the absence of clearing programmes, occurrence of annual flooding, and the abundant presence of the target species.

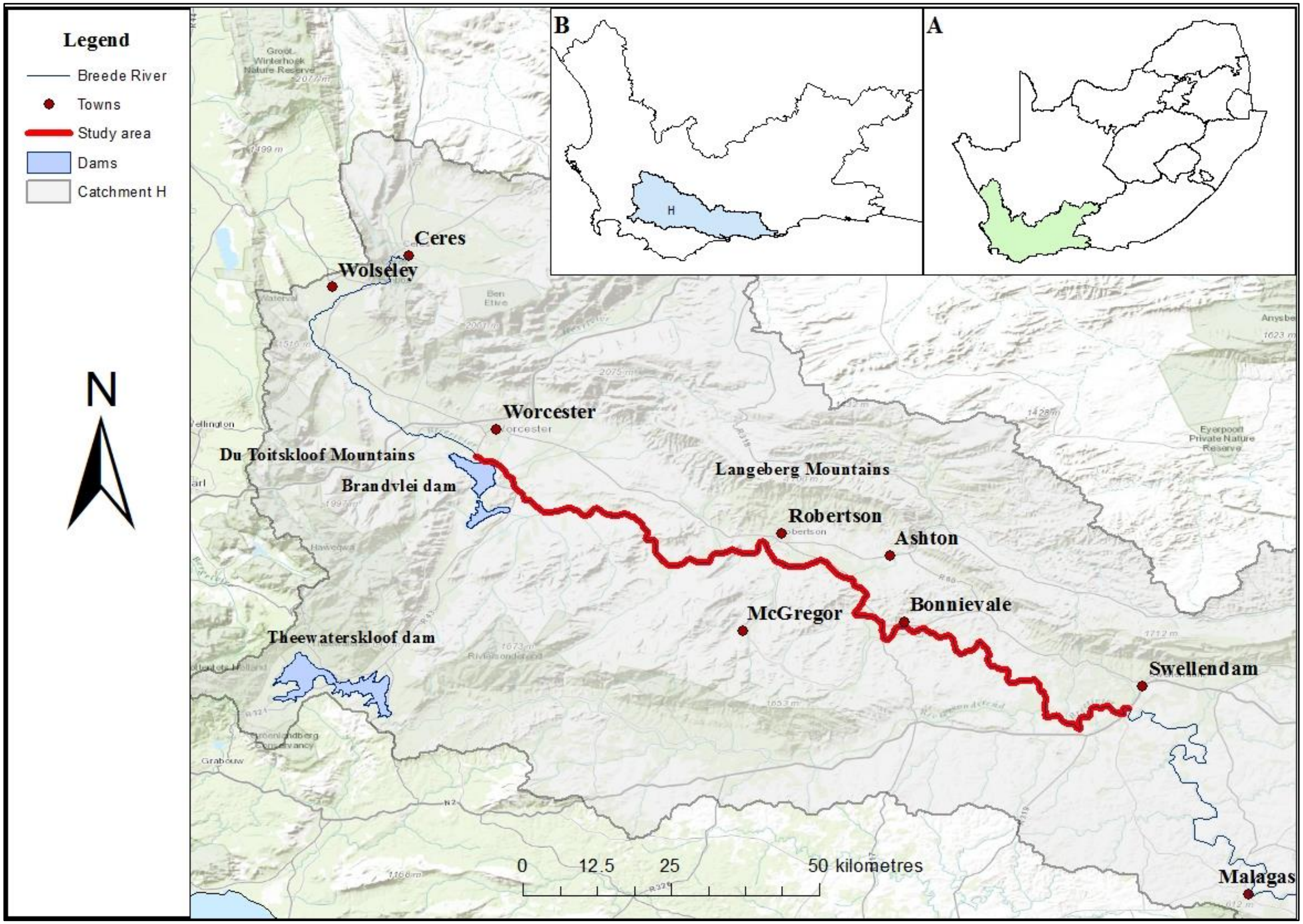


Figure 3.1 Study area, depicted in red. Labels indicate location of dams, mountain ranges and towns. Inset (A) shows the province of interest, and inset (B) depicts the catchment location within the Western Cape. Topographic base map provided by ESRI (2014).

The Breede River is approximately 322 km long, and drains an estimated 12 600 km^2 within the Western Cape (McCord & Lamberth 2009; RHP 2011; Steynor, Hewitson & Tadross 2009). The catchment has several dams with the two most notable being the Greater Brandvlei Dam near Worcester, and the Theewaterskloof Dam near Villiersdorp (Figure 3.1), in which water is stored in winter, for distribution and use during summer (Steynor, Hewitson & Tadross 2009).

The Breede River catchment has little undisturbed or fallow land along the riparian zone, with close proximity to the river channel being advantageous for many industries which rely upon direct water abstraction from the river, in addition to the river being a popular holiday or weekend retreat. Industry along the river consists mainly of commercial orchards and vineyards, with natural vegetation representing the other major land use (Warburton, Schulze & Jewitt 2011). Importantly, both the fruit and wine industry in the Western Cape is dependent on irrigation (Van Rensburg et al. 2011), which is locally sourced.

The largest impoundment within the river is that of the Greater Brandvlei Dam (Figure 3.1), which supplies water to the surrounding farmers, which is used for releases into the Breede River to regulate salinisation. The dam was constructed in 1949 (Brandvlei Dam) and was extended through the construction of the Kwaggaskloof Dam in 1972 to the current size. Construction was initiated mainly to stabilise irrigation water supply to the Breede Valley, as increased demand had outstripped supply, particularly during the dry summer months (Kirchner et al. 1997). The dam itself is not within the main river channel, rather it is filled by diversions to the Holsloots and Smalblaar Rivers (Brown & Le Roux s.a.). Water is provided for agricultural use via an organised system of dedicated canals which run close to, and generally parallel to, the main channel (Brown & Le Roux s.a).

Due to its length (110 km), the study area was subdivided into three subsections, to incorporate expected variation in hydrological characteristics between the various sections. Table 3.1 details the chosen sections, indicating the length of each section and the GPS locations of the divisions. These sections are referred to later during the results and discussed where applicable.

Table 3.1 Study sections subdividing the study area (Worcester-Swellendam).

Section Name	**Start-End**	**Riparian Distance (km)**	**Euclidian Distance (km)**	**Beginning (Lat; Long)**	**Finish (Lat; Long)**
Worcester	R34 bridge to Robertson bridge	47.00	36.25	-33.736226° ; 19.487574°	-33.823185° ; 19.864175°
Robertson	Robertson bridge to'Pony bridge'	63.85	37.28	-33.823185° ; 19.864175°	-34.002163° ; 20.205861°
Confluence	'Pony bridge'to Swellendam bridge	33.87	20.53	-34.002163° ; 20.205861°	-34.067062° ; 20.414006°

3.1.2 Study site selection

Initially, potential patch sites were identified at equal distances from each other between Worcester and Swellendam. Patch sites had to conform to the following criteria in order to be eligible for selection:

(i) The site had to contain *E. camaldulensis* (in order to provide suitable data).

(ii) The site had to be on the riparian fringe of the Breede River (in order to be suitable in terms of the study aims and objectives).

(iii) The site had to be accessible (in order for field data to be captured and *E. camaldulensis* presence verified).

(iv) Entry permission had to be available for the site (as most patch sites were located on private property). If permission was refused or unobtainable, the nearest adjacent property conforming to the above criteria were used.

Due to historical comparison between sites being required during analysis, island sites selection also necessitated specific criteria. The criteria for suitability and selection was:

(i) Island sites were regarded as suitable for this study only when visible across all time stamps for which aerial imagery was available. This was necessary to allow for the measurement of island site area, which was used to evaluate change over time in the data analysis.

(ii) Sites were additionally required to be accessible. Due to only areas with *E. camaldulensis* present being included in this study, the presence or absence of *E. camaldulensis* would have had to be verified during fieldwork. As such, islands sites near patch sites were often suitable, as the researcher was able to come in close contact with the river banks and thus visually confirm the presence of the target species on the island. Sites were thus also regarded as potential island sites if they were visible from bridges or roads, as visual confirmation could be made in such instances.

Taking into consideration the above criteria, the final number of 14 suitable islands sites were used in data analysis, and a total of 31 patch sites.

3.1.3 Breede River climate

The study area lies within a Mediterranean climate zone (Roberts, Meadows & Dodson 2001) and receives a mixture of winter rainfall and yearlong rainfall (Brown et al. 2004). The region receives most of its annual rainfall in the months of April through to September, with considerable spatial variation (Steynor, Hewitson & Tadross 2009). The Breede River valley, which is regarded as being semi-arid, experiences a mean annual rainfall of 290 mm in the Robertson Area (Hunter & Bonnardot 2011). Precipitation along the southern and western coasts is as a result of temperate frontal systems derived from the Westerlies (Chase & Meadows 2007), commonly known as ‘cut-off lows’ or ‘mid-latitude cyclones’ (RHP 2011).

The study area in particular, lies within a narrow band of year-round rainfall, which is an area of convergence between the purely summer rainfall zones of the north and the purely winter rainfall zones of the south (Chase & Meadows 2007). The north-west part of the River (near Wolseley and Ceres) receives significantly more rainfall than further downstream (near Robertson or Swellendam), and subsequently, rainfall between the upper and lower sections differ markedly, and can occur at various times throughout the year (RHP 2011). Figure 3.2 illustrates the mean annual precipitation for the study area. Figure 3.3 illustrates the rainfall seasonality of the region, showing both year-round rainfall and winter rainfall regions within the study area

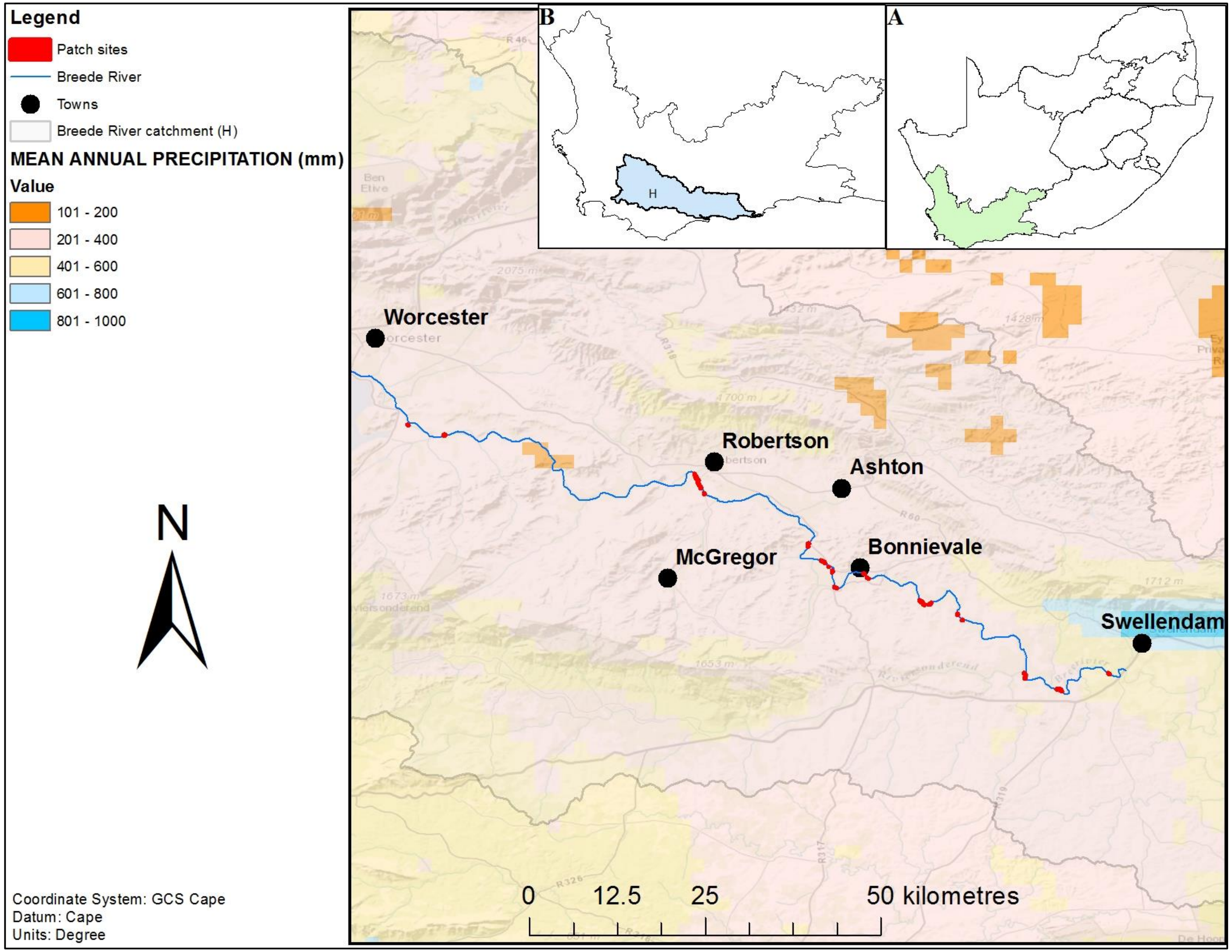


Figure 3.2 Mean annual precipitation (mm) across the study area. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014). Precipitation data supplied by Schulze & Maharaj (2006a).

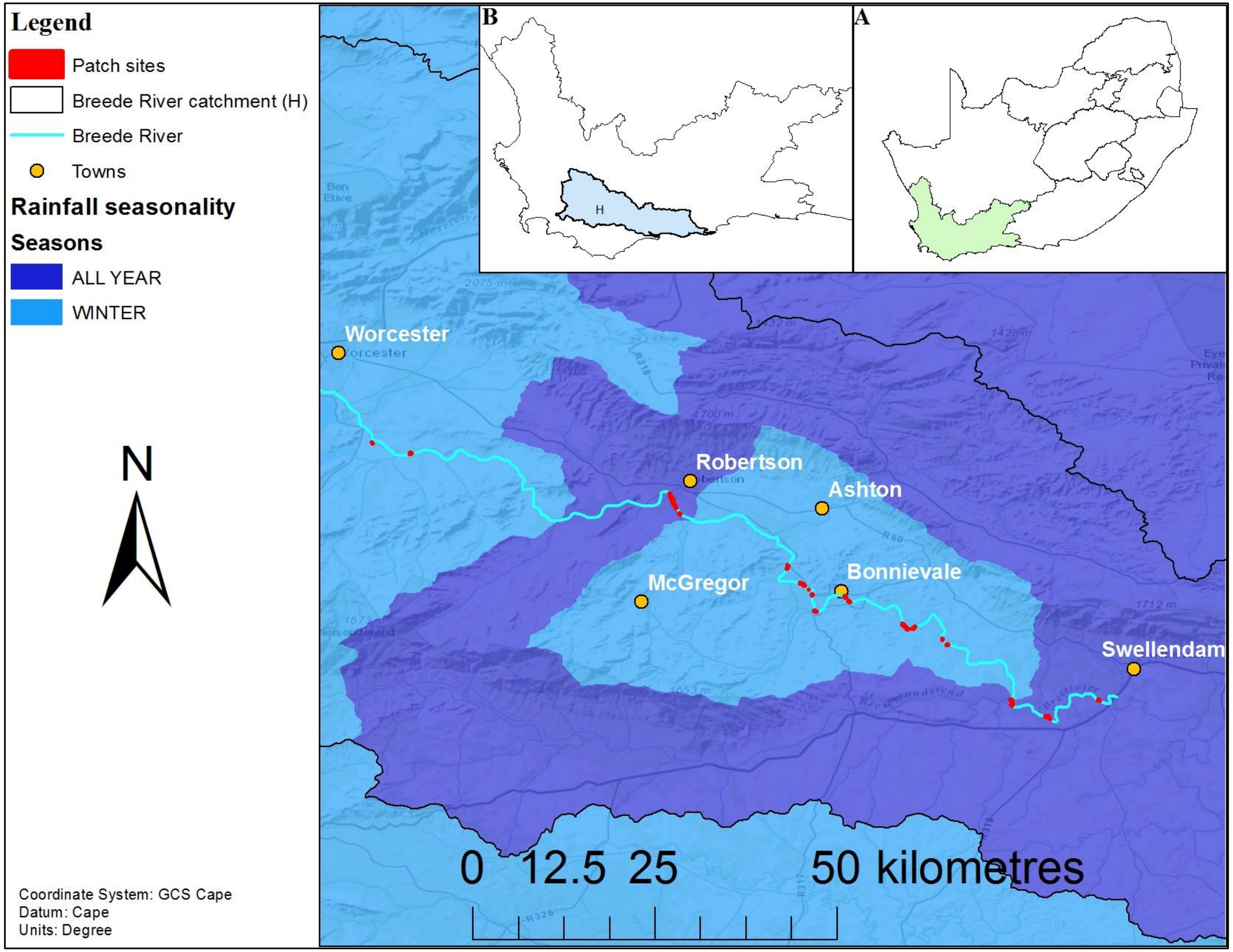


Figure 3.3 Rainfall zones across the study area. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014). Seasonality data supplied by Schulze & Maharaj (2006b).

3.1.4 Breede River hydrology

The Greater Brandvlei Dam currently has a 342 million m^3 capacity, with a firm yield of 155 million m^3 per annum (RHP 2011). Water is transferred between the Breede and Berg River water management area (WMA) through the 'Riviersonderend-Berg-Breede River government water scheme' (DWAF 2004). Water transfer from the Breede River WMA into that of the Berg River WMA is a net annual average of around 161 million m^3 per annum, whereas the Breede River WMA received roughly 2.5 million m^3 per annum from the Berg WMA (DWAF 2004). Water transferred into the Berg WMA forms part of the municipal supply to the city of Cape Town (DWAF 2004) with the majority of the water supply (approximately 95%) being used for agricultural irrigation (DWAF 2004).

Additionally, alteration of the flow regime such as mentioned above, are mainly through the construction of impoundments (dams) within the tributary rivers. For example, the construction of two dams within the Poesjenels River, coupled with the construction of the Keerom Dam and over-abstraction of water, specifically in the lower reaches of the rivers, have markedly influenced the in-stream flow regime and habitat integrity of the Breede River (RHP 2011). In particular, flow during the summer months (which are historically those with the least precipitation) has been dramatically reduced since the advent of the abovementioned dams (RHP 2011). Such management of the water resource of the Breede River thus impacts the quality and volume of water used in the region, directly impacting the salinity levels of the river towards the river mouth, which in turn impact the riparian vegetation cover found.

3.1.5 Breede River agricultural impacts

The Middle Breede is influenced mainly through the addition of nitrogen (N) and phosphorus (P) into the river channel via urban effluent or agricultural sources, principally through fertilisers (RHP 2011). Such conditions often lead to eutrophication of certain sections of the river, particularly during the summer months when flow levels are relatively low (RHP 2011). According to the River Health Programme, the major agricultural impacts along the middle Breede River include a loss of indigenous riparian vegetation due to the establishment of vineyards or orchards, decreased water flow in the river (especially during summer months), and an increase in algal blooms and nutrient levels within the river (RHP 2011). Specifically, the pesticides and fertilisers of the grape, citrus and deciduous fruit industries are often introduced into the river through runoff, spills, spray or vapour which impact upon pollution-sensitive aquatic species (RHP 2011). Further impacts include the replacement of indigenous vegetation with IAP and moderate riparian soil erosion due to livestock grazing on the riverbanks (RHP 2011).

3.2 TARGET SPECIES

E. camaldulensis, generally grows to 20 m tall, with some observed growing to 50 m, with stem diameter varying up to two metres (Orwa et al. 2009). Tree form varies between a 'forest' and 'woodland' form, with the former generally being taller with branches further from the ground, and the latter being shorter with branches located closer to the ground (Roberts & Marston 2011).

Trees have a smooth bark, with the exception of a rough bark zone occupying the first 1-2 metres of the stem (Orwa et al. 2009), shedding in thin long strips revealing whitish, greyish or reddish patches underneath (Henderson 2001; Van Wyk & Van Wyk 2011). Bark in the colder areas of South Africa, tend to exhibit a red tint (Henderson 2001). Bark colour also varies highly between white, grey, yellow-green and grey-green (Orwa et al. 2009). Trunks of *E. camaldulensis* are single and usually straight (Harrington & Fownes 1993), depending on the environmental conditions during the tree's lifetime.

E. camaldulensis leaves are between 90 and 300 mm long (Van Wyk & Van Wyk 2011) and 7-20 mm in width, typically drooping, hairless and aromatic when bruised or broken (Van Wyk, Van Wyk & Van Wyk 2011). Juvenile leaves are smaller and broader, with red petioles and twigs (Henderson 2001) and are 130-260 mm long and 480 mm wide (Van Wyk, Van Wyk & Van Wyk 2011). The leaves (Figure 3.4) also contain essential oils, which are the source of the typical '*Eucalyptus*' smell associated with this species.

Figure 3.4 Abaxial (left) and adaxial (right) leaf surface of *E. camaldulensis*, showing uniform colouration.

Fruits are brown, to reddish brown capsules (Henderson 2001), subglobose, up to 8 mm long with prominent rims and protruding valves; usually four (Van Wyk & Van Wyk 2011). Fruit of *E. camaldulensis* are illustrated in Figure 3.5 below, showing the distinct valves prior to opening, and once opened (Figure 3.6). Seeds are stored in capsules (fruit) on the branches (Roberts & Marston 2011), and may occur on trees from as young as three years of age (McMahon, George & Hean 2010).

Seeds are cuboid, smooth with two seed coats, 1–1.5 mm long, and brownish yellow (McMahon, George & Hean 2010). When grown commercially, seed require no pre-sowing treatment, and are between 230 and 1 160 viable seeds/g (McMahon, George & Hean 2010). Seeds are furthermore lightweight, weighing (0.5-1.4 mg per seed) and are dispersed by barochory, anemochory and hydrochory (Roberts & Marston 2011). Seed fall is evident throughout the year, but varies in intensity, peaking in winter rainfall regions between March and September (McMahon, George & Hean 2010). Seed, once fallen to the ground, is subject to predation, such as from ants (Roberts & Marston 2011).

Figure 3.7 shows the adult *E. camaldulensis* trees as is typically seen along the Breede River. Of note is the grey-blue colour of foliage, which differs slightly to that of *Eucalyptus cladocalyx* (Sugar Gum) which is also commonly found in this area, and are distinct in their glossy leaves and tree shape (‘clumped’ leaves at the end of the branches mostly) from that of *E. camaldulensis*. Flowers are stalked, axillary, composed of clusters of 7-11, white to cream in colour and flower heavily every two or three years (McMahon, George & Hean 2010; Van Wyk & Van Wyk 2011). Buds are spherical, with a hemispherical lid which narrows into a slender tip (Van Wyk & Van Wyk 2011).

Figure 3.5 *E. camaldulensis* fruit, illustrating the four valves prior to opening, and the red-brown hue observed on the fruit (not to scale).

Figure 3.6 Open *E. camaldulensis* fruit (left) and scattered seed (right).

Figure 3.7 Adult *Eucalyptus camaldulensis* trees, as typically seen on the Breede River, illustrating general habit and growth form of the species.

3.3 DATA COLLECTION

This research was conducted making use of three different data sources, namely spatial, hydrological and field measurements. The data collection for all three sources are detailed below.

3.3.1 Field data

3.3.1.1 Interviews

Formal interviews (Appendix A) were conducted with a representative of the property, most often the property owners but in a few instances the farm manager. Representatives were regarded as suitable if:

(i) They possessed the authority to allow access to the property and be interviewed, or (ii) were managers by title.

(ii) They had been working on the particular farm for at least two years and were knowledgeable concerning the IAPs on the property. This was required to ensure respondents understood the context of the study, and were able to provide sufficient responses for inclusion in the analysis.

Respondents were contacted prior to the researcher's arrival, met on the property, appraised of the study particulars and requested to sign the 'prior informed consent form'. Interviews were only conducted with respondents who had signed the form and consented to participation in the study.

3.3.1.2 Patch sites

Farm portion boundaries were established using a municipal delineation and the contact details obtained for the relevant property owners. Contact details were obtained from BolandEnviro (an environmental consultancy company conducting work along the Breede River at the time), who had compiled a riparian property owners database during their public participation phase for a project they managed. Any additional contact details that were required, which were not in the initial database, were obtained through the Central Breede Water Users Association (the organisation established within the central Breede to administer certain water services), and the different farmers spoken to during the course of fieldwork.

Due to permission being required for each individual site, the final selection of sites depended purely on said permission having been obtained. As such the sampling method employed resembled the 'purposive sampling' method as discussed by Kothari (2004).

After consultation with the Department of Agriculture (DoA) specific variables were decided upon and included in a field data sheet (Appendix B), which was captured during field visits. Variables included were location (for spatial comparison), land use (for context), general density (native versus exotic) to quantify patches, target age class and proportion (to detail target species composition), underfoot conditions, obstructive vegetation present, slope and perimeter walk time (all used by DoA). These variables were identified prior to data gathering, in order for the correct data to be captured, so as to make data sharing useful and possible.

In order to evaluate the spatial extent of patch sites, GPS measurements were established by capturing the current spatial extent of the *E. camaldulensis* stand in a walk-around method. Boundaries of patch sites were distinguished by the presence or absence of *E. camaldulensis* based on the overall density class of the given patch. This was done in accordance with the WfW alien polygon mapping standards, as discussed by Gibson & Low (2003). Patch sites were thus regarded as two distinct sites when the maximum spatial error (Table 3.2) was exceeded to the next *E. camaldulensis* occurrence, for the most represented age class (i.e. dominant age class within the polygon, based on stem density), from any given waypoint. Table 3.2 illustrates the maximum spatial error and density definitions used by Gibson & Low (2003). All spatial output files were required to have vertex lengths no longer than 5 m except for straight line boundaries, where a length of 10 m was acceptable (Gibson & low 2003). The density classes were defined as an average for the target species within an area.

Table 3.2 WfW mapping standards for 2003, illustrating cover classes and maximum allowable spatial error during GPS capturing.

Density class	**Percentage cover of class (%)**	**Maximum spatial error (m)**
Rare	<0.01	50
Occasional	>0.01-1	50

Very scattered	1-5	50
Scattered	>5-25	10
Medium	>25-50	5
Dense	>50-75	2
Closed	>75	2

Source: Adapted from Gibson & Low (2003: 21)

As WfW had since 2003 migrated to using numbered values (1-100%) for specifying density of a polygon, and were not using density classes anymore (Malan 2012, Pers com), these classes were used as guideline only during patch site GPS data capturing. Density measurements used in this report were in accordance with the numbered values for each polygon method, as used by WfW, i.e. value (%) of target species density per age class for each patch sites.

Additionally, height classes were required for patch site analysis. Field estimates were made using standard height classes, i.e. seedling (<30 cm), young (30 cm – 5 m), adult (5 m – 15 m), mature (15 m – 30 m) and very mature (> 30 m). Field estimates were then verified through the use of a hand-held Haglöf Vertex IV and T3 transponder (Haglöf 2014), or for seedling and young classes, simply using diameter tape. Measurements for the patch sites were obtained concurrently with the capturing of GPS waypoints.

GPS point data was captured using the Garmin eTrex hand-held GPS (Garmin 2014), using automatic point averaging obtain maximum spatial accuracy, and was incorporated into ArcMap (ESRI 2013) as a point shape-file. GPS points collected were in no instance further than 5m apart. The patch sites identified and captured in the above manner numbered 31 at the completion of the data collection, and were used as the 2013 time stamp for the comparison over years (conducted using the aerial imagery).

3.3.1.3 Island sites

Islands were defined as being areas of land above water level within the river channel, but which were not connected to the riverbanks. Island sites were identified by initially using 2010 aerial photographs of the study area, and visually locating sites within the river displaying islands.

Due to island sites not being physically accessible, no field measurements in the form of GPS waypoints were possible. Thus, only islands that were visually verified during fieldwork were included in the analysis. The suitable sites were then manually digitised on the 2010 aerial photographs, and then evaluated against the previously mentioned suitability criteria.

3.3.2 Hydrological data

Hydrological data used in this study consisted of rainfall and river level data, for the period of 1980 to 2012. Additional data, in the form of discharge values was obtained for the period of 1980 to 2012, depending on the available data for that period. The rainfall data was provided for several stations in the Breede Valley (Table 3.3). Rainfall data, which was provided as daily rainfall measurements in millimetres was obtained from the South African Weather Service (SAWS). River level data and discharge data, which was provided as water level in metres and cubic metres per second respectively, were obtained from the Department of Water Affairs (DWA). Figure 3.8 illustrates the locations of meteorological and weir stations from which data were obtained.

Table 3.3 Meteorological stations for which data were obtained and analysed in this study.

Station name	**Identifier code**	**Date**	**Number of observations**	**Location coordinates (Latitude, longitude)**
Kwaggaskloof Dam	0022825 0	1980-2012	11656	-33.7600, 19.4740
Robertson	0023678 2	1980-1986	10034	-33.8000, 19.8830
Rhebokskraal	0023629 8	1980-1999	11843	-33.9940, 19.8320
Ashton	0024110 X	1980-2012	11649	-33.8330, 20.0670
Marloth	0008751 7	1980-2012	10939	-34.0050, 20.4400
Swellendam	0008813 X	1980-2012	11842	-34.0550, 20.4720

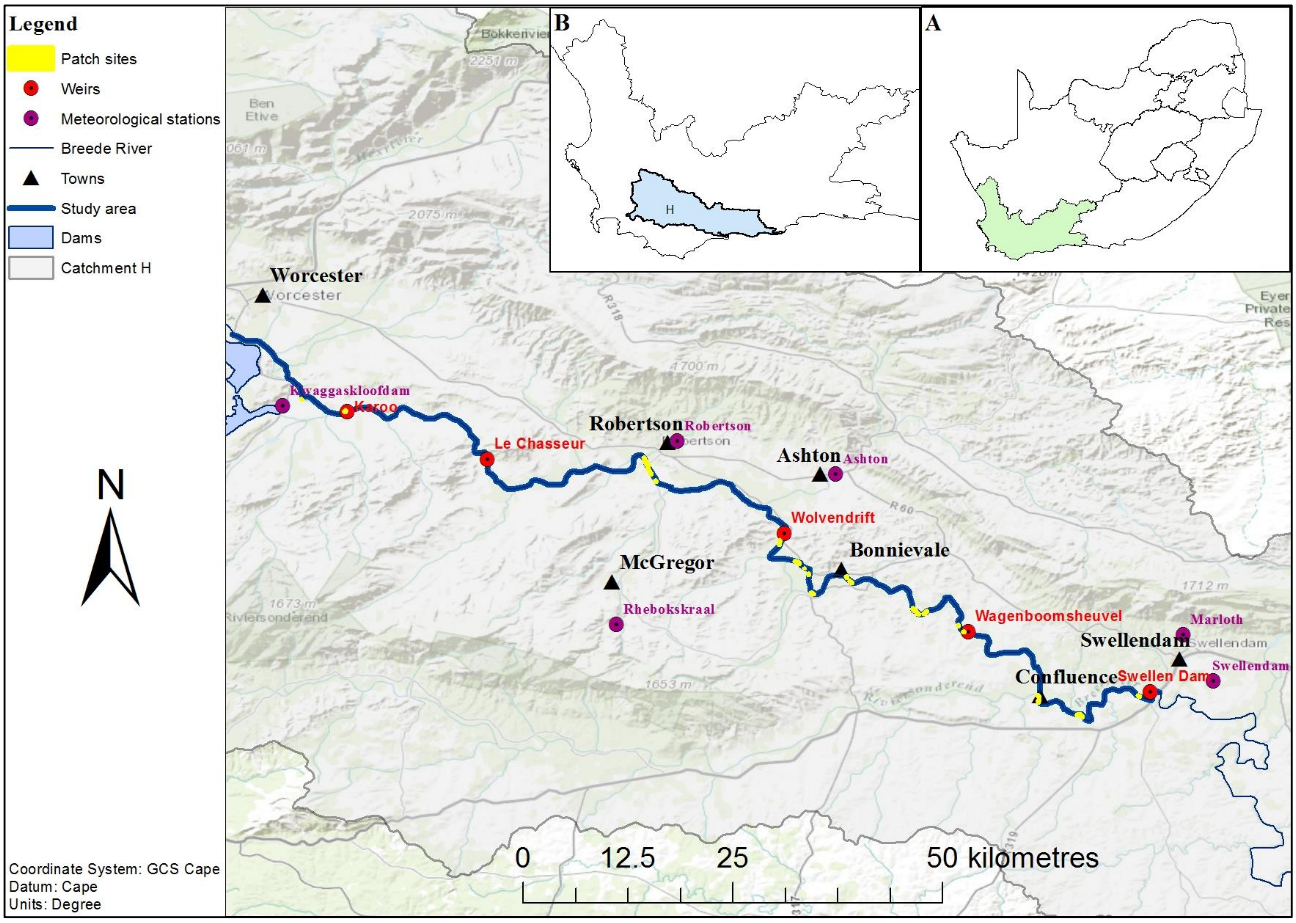


Figure 3.8 Meteorological and weir station locations within the study area. Label and symbol colours match to allow for distinction. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014).

Table 3.4 details the rainfall data obtained for the period of study. Rainfall and discharge stations pairings are further shown below in Table 3.4, illustrating the data periods available and the different pairings between weir and rainfall stations chosen. Of note is the difference in data periods available. This was due to discharge measurements not being available for certain dates. Pairwise deletion was applied prior to data analysis to produce a complete dataset.

Table 3.4 Meteorological and weir station pairings with available discharge data, indicating the data period and the number of pairwise observations available in the dataset.

Weir station	**Meteorological stations**	**Available paired data period**	**Number of observations**
Karoo	Kwaggaskloof	1980-1992	4306

Le Chasseur	Kwaggaskloof	2000-2012	4260
Wolvendrift	Robertson	2000-2010	3197
Wagenboomsheuvel	Ashton	1980-1991	4085
Swellendam	Marloth	1980-2012	7233

3.3.3 Aerial photographs

Multi-temporal aerial photographs were obtained from the Chief Directorate: National Geo-spatial Information (CD:NGI). Table 3.5 indicates the different images obtained, along with their resolution and post-processing level. In addition to the imagery data obtained from the CD:NGI, a 30 m resolution ASTER digital elevation model[1] (DEM) was obtained through the United States Geological Survey (USGS) EarthExplorer (2009).

Table 3.5 Details of aerial photographs used in this study.

Date	Image count	Resolution (m)	Bands	Processing
1987*	5	2.5	Panchromatic	raw
1989*	1	2.5	Panchromatic	raw
2003	6	0.5	Panchromatic	raw
2007	38	0.5	RGB	Orthorectified
2010	38	0.5	RGB+NIR	Orthorectified

*Consisted of a single flight plan of two year duration, thus analysed and discussed as a single period.

3.4 DATA PREPARATION

3.4.1 Field data

GPS waypoints were imported using DNRGarmin 10 software (DNR 2008) into ArcMap 9.3 (ESRI 2013) in shape-file format. Vertices were then connected to complete polygon boundaries, thereby creating the final patch sites. Polygons created were then overlaid onto layers digitised from the historical aerial photographs, and spatial extent of sites compared for each time stamp. The polygons created from GPS waypoints captured during fieldwork were used as the 2013 patch site

[1] Note(s): ASTER GDEM is a product of METI and NASA.

extent during analysis, and formed the most recent spatial boundaries for the various patch sites. Tree height and class measurements obtained for patch sites, were further captured in digital format in Microsoft Excel 2013 (Excel 2013), sorted, and coded for analysis. Coding of qualitative categories were required in order to derive percentages (proportions) during analysis. Numerical data were sorted for further analysis.

Furthermore, interview responses were captured digitally in Microsoft Excel 2013 (Excel 2013), sorted and coded in the case of qualitative responses. Qualitative data were categorised by common content, i.e. grouping similar answers together and providing numeral categories for each. Proportions were then calculated from the various categories obtained, using Microsoft Excel 2013 cell calculations (Excel 2013). Numerical data were sorted for further analysis.

3.4.2 Hydrological data

Both the rainfall, river level and discharge data were exported, sorted and compiled for statistical analysis, and analysed using the open-source statistical package ‘R’ version 2.15 (R Core Team 2012). Table 3.6 indicates the various packages used during statistical analysis. Values were ordered in meaningful columns using Microsoft Excel 2013 (Excel 2013), to complete a full dataset of values for the study period for each appropriate rainfall station and weir. Values within the datasets that were unreliable (due to instrument failure or human error), such as derived or corrected values, or missing values, were excluded from final analysis (by pairwise deletion where required).

Table 3.6 ‘R’ software packages used during statistical testing of data used in this report. Not all packages were required for testing included in this report, and are indicated with an asterisk (*) in the table.

Package	**Author/Reference**	**Use**
psych	Revelle (2013)	Descriptive statistics
GenABEL	GenABEL Project Developers (2013)	Rank transformation*
ez	Lawrence (2013)	ANOVA testing*
pastecs	Ibanez, Grosjean & Etienne (2013)	Descriptive summaries
nortest	Gross & Ligges (2012)	Normality testing

fBasics	Wuertz (2012)	Normality testing

*Used for statistical comparison but not in analysis; not included in report.

The total number of observations was much less after the removal of zero values (days where there was no rainfall), and only the ‘rainy’ days (days on which there was rainfall) were used for the analysis, as extreme rainfall and flooding dates were of primary importance in this study.

3.4.3 Aerial photographs

In order to overlay the imagery from the different years, as was necessary in this study, raw imagery had to first be orthorectified. Orthorectification is the processing of imagery to geometrically remove distortions, creating an image with consistent scale across all parts (ArcGIS Resource Centre 2014). The images were orthorectified using the ASTER 30 m DEM and the OrthoEngine tool in PCI Geomatica software (PCI 2013). Three images (from 1987 imagery) were not orthorectified, but rather georeferenced as their output error was less using the automatic tie-point creation function in ENVI 5 (Exelis 2013).

The images were all georeferenced with a RMSE of less than two pixels. Sites were digitised by creating polygons as close as possible to the waterline of such islands, or in the case of patch sites, as close as possible to the tree line (edge of the canopy).

3.5 DATA ANALYSIS

3.5.1 Statistical tests used

In this section the various test applied during analysis are briefly described and shown for which analysis it was used, to provide context for the results obtained. For all test conducted, unless otherwise specified, the alpha (α) level of 0.05 was used to evaluate significance.

3.5.1.1 Proportions

In this report, proportions (percentages) were used in instances where further statistical testing was not required, or where sample size was insufficient to validate the use of certain statistical tests.

Percentage was calculated as the proportion of the whole sample in all cases. Proportions were used for interview analysis.

3.5.1.2 Descriptive statistics

Descriptive statistics were also explored during analysis, as a means to communicate the general data parameters, specifically to indicate general distribution. The descriptive statistics mean, standard deviation, standard error, range, minimum and maximum, as well as skewness (a measure of 'lean' of the data) and kurtosis (a measure of the 'peakedness' of data (Kim 2013)) values are provided where necessary. Skewness was measured using the adjusted Fisher-Pearson coefficient of skewness, and kurtosis Pearson's measure of kurtosis. These terms are not explained further, as it is assumed the reader is familiar with descriptive statistics. Descriptive statistics were used for all analysis.

3.5.1.3 Kolmogorov-Smirnov normality test

Testing the distribution of experimental data, such as for the 'normal' distribution, is an important statistical exercise to perform prior to further data testing, such as difference in means testing, regression or t-tests (Drezner, Turel & Zerom 2010). One of the most widely known tests for the normal distribution is that of the Kolmogorov–Smirnov Test for Normality, which is one of many 'goodness-of-fit' tests (Drezner, Turel & Zerom 2010). This test evaluates whether a sample is derived from a completely specified continuous distribution, making use of the sample mean and variance as parameters of the normal distribution (Drezner, Turel & Zerom 2010).

The Kolmogorov–Smirnov Test is based on the largest 'vertical' difference between the hypothesised and empirical distribution, and assigns greater weight to tails within the data (Razali & Wah 2011). Mean and standard deviation are required parameters for calculating the statistic, without which the test power becomes excessively conservative (Henderson 2006). Results are reported using the statistic 'D'. Normality of the sample (H_0) is rejected when the 'D' value does not exceed the chosen critical value, which for the common 0.05 significance point is 0.895 (Henderson 2006). The probability (p-value) may also be used to ascertain the test outcome. This test was applied for the rainfall data.

3.5.1.4 Two sample Kolmogorov-Smirnov test

The two sample Kolmogorov-Smirnov test is a non-parametric ‘goodness of fit’ test to evaluate two independent samples differ from each other in terms of their distribution (Lin, Wu & Watada 2010). It is comparable to the Mann-Whitney test or the parametric t-test, although certain advantages are provided by the use of the two sample Kolmogorov-Smirnov to the prior tests mentioned above, such as additional sensitivity to variances for example (Lin, Wu & Watada 2010). The two sample Kolmogorov-Smirnov test thus evaluates whether two independent samples have been drawn from the same population (Lin, Wu & Watada 2010). The test statistic is the ‘D’ value, for which critical value tables are available. The probability value obtained from the results can be used to evaluate significant differences in distribution between the two samples. This test was applied to rainfall and river level data.

3.5.1.5 Anderson-Darling normality test

The Anderson-Darling normality test is a ‘goodness of fit’ type tests, derived from the Cramér-von Mises test (Razali & Wah 2011), which evaluates whether a random sample comes from a continuous population of completely specified distribution function (Scholz & Stephens 1987). The test is not limited by sample size, requires no table of coefficients or critical values (Lesaffre 1983) and calculates the ‘A’ value (Henderson 2006). The weighting of the test allows for more influence on the distribution tails than does the Cramér-von Mises test, and is regarded as performing well for general applications (Henderson 2006).

This test is regarded as being powerful in comparison to other statistical tests of normality (Pettitt 1977). The H_0 of the Anderson-Darling test is that the data follows a specified distribution. The interpretation of the results rest on the critical value of ‘A’, as well as the probability value obtained. Rejection of the H_0 is the conclusion for values of ‘A’ above the critical values for the particular distribution tested against, with probability values below 0.05 (at α level of 0.05) being an alternative indicator. This test was applied for patch sites analysis.

3.5.1.6 Shapiro-Wilk normality test

The Shapiro-Wilk normality test is based on calculations derived from the Pearson correlation coefficient calculated and compared for ordered sample and Gaussian sample values (Henderson 2006) and although initially restricted to sample sizes below 50, further transformations allowed for a sample limit of 5000 more recently (Henderson 2006). It is widely employed due to its good power in testing for normality (Razali & Wah 2011). The computed statistic is the 'W' value, which ranges from 0-1 (Razali & Wah 2011). A value close to one indicates the sample conforms well to the normal distribution, whereas values closer to zero indicate departure from the normal distribution (Henderson 2006; Razali & Wah 2011). This test was applied for island site analysis, due to its preferred power for smaller sample sizes.

3.5.1.7 Wilcoxon signed rank test

The Wilcoxon signed rank test, is a non-parametric difference in median test for paired or related samples (Taheri & Hesamian 2013), equivalent to the paired t-test for parametric samples (McDonald 2014). Unlike the t-test however, where the mean is evaluated, the Wilcoxon signed rank test evaluates the median of the samples (Taheri & Hesamian 2013). The test is well known for its high power in detecting differences between appropriate samples (Vexler, Gurevich & Hutson 2013). Interpretation of the statistic 'W' and the probability value indicate the extent of location shift (i.e. similarity of samples) and the significance of the results. A low value of 'W' (or 'V' as reported in 'R'), closer to zero, would indicate a higher degree of similar medians (McDonald 2014). At significance level 0.05, the probability value below 0.05 would indicate significant difference in medians between the two samples. This test was applied for both patch and island sites.

3.5.1.8 Friedman rank sum test

The Friedman rank sum test is a non-parametric test to evaluate the difference in medians (St. Laurent & Turk 2013) for related samples (Röhmel 1997), for which the parametric equivalent is the one way repeated measures ANOVA test. The H_0 is that there is no difference in mean between the samples, with the H_a of significant difference in mean between samples (Röhmel 1997). The test does not, however, evaluate which groups are responsible for the acceptance or rejection of the H_0 (Röhmel 1997). The test relies on ranked data (O'Gorman 2001). The statistic evaluated is the 'Q' statistic, for which critical value tables are available for small sample sizes (St. Laurent & Turk 2013). Significant difference in median between groups can be evaluated using the probability value provided by the results. This test was applied for both patch and island sites.

3.5.1.9 Holm-Bonferroni post-hoc correction

The Holm-Bonferroni post-hoc correction refers to a correction applied to pairwise comparison results (the probability value) obtained from non-parametric difference in median testing, such as the Friedman rank sum test. As the Friedman rank sum test merely indicates whether a statistically significant difference in median was observed for three or more samples, further comparison is required to elucidate which samples differed from another. Pairwise comparisons can be conducted to that end, using tests such as the Wilcoxon signed rank test.

A common statistical problem results from conducting such pairwise comparisons across a number of samples; is that of inflating the alpha (α) level (Abdi 2010). Known as multiplicity, inflation of the alpha level refers to the increasing chance of committing a Type I error due to the increasing probability of finding a statistically significant result through application of large numbers of statistical tests (Abdi 2010). Various post-hoc tests are available for the correction of such results, in order to limit the probability of a Type I error during interpretation. One such test is the Holm-Bonferroni post-hoc test, which was used in this report.

The test makes use of ranking the probability values obtained from difference in median testing, and calculating critical probability values based on the Bonferroni correction. A stepwise comparison and evaluation of the significance of the different samples are then estimated using the Holm-Bonferroni logic (Abdi 2010). The Holm-Šídák correction is used for independent samples, whereas the Holm-Bonferroni post-hoc test is used for dependant samples (applicable in this case). This method has greater statistical power than the commonly applied Bonferroni post-hoc test, while limiting inflation of Type I error (Abdi 2010). This test was applied for both patch and island sites.

3.5.1.10 Spearman rank correlation

In this report, the Spearman rank correlation was preferred over that of the Pearson product-moment correlation, firstly, as data normality was a rare instance in the analysis, and secondly due to the known sensitivity of the Pearson product-moment correlation to non-normality and specifically outliers in the data (Xu et al. 2013). The Spearman rank correlation is a non-parametric evaluation of the relationship between two variables (Schmid & Schmidt 2007), by calculating the Pearson product-moment correlation between variables using ranked data in place of scores (Zimmerman 1994). The resulting rho value provides a means of quantifying the strength and direction of any potential relationship between the variables (Prion & Haerling 2014). Use of the Spearman rank correlation is commonplace in non-normally distributed correlation analysis, specifically in hydro-meteorological time series evaluations (Yue, Pilon & Cavadias 2002).

The 'rho' statistic is scaled between -1 and +1, where zero values denote no relationship exists (Prion & Haerling 2014). A positive value indicates both variables 'move in the same direction' (Prion & Haerling 2014), i.e. increase in one as the other increases concurrently. A negative 'rho' value indicates opposite direction in movement between the variables (Prion & Haerling 2014), where an increase in one corresponds to a decrease in the other. The magnitude or strength of the relationship is depicted by the proximity of the 'rho' value to either of the extreme poles of the scale. This test was applied for river and rainfall data correlation.

3.5.1.11 Simple lagged cross correlation

Cross correlation is a statistical tool to quantify temporal relationships between variables (Olden & Neff 2001), to assess time delays and to quantify the magnitude of the mechanisms involved (Runge, Petoukhov & Kurths 2014). Typically, the correlation is calculated for two variables adjusting for large quantities of lag periods, and the strongest correlation is then used as the 'correct' lag time between the given variables (Olden & Neff 2001).

Various authors state limitations inherent to this approach, one of multiplicity (Olden & Neff 2001), other of insufficient causal evidence (Rogosa 1980), measurement interval limitations and reciprocal causality (Clegg, Jackson & Wall 1977). The use of cross correlation is applied in a limited capacity in this study (merely to assess the most likely lag times associated with rainfall and river level at the scale of one week, used as a guide to the data arrangement of flood events at the scale of one year intervals). In addition, the limitations inherent to the technique are addressed by using large sample sizes for the two variables (thus reducing the probability of type I error due to multiplicity) and the use of known causal variables (rainfall being a major input into the river water levels) as well as the known direction of influence being one way, i.e. rainfall influencing river levels and not *vice versa*.

This test was deemed sufficient for use in generalising lag periods in the context of the aims and objectives, as the general limitation of this technique had been addressed, in combination with the limited scope and dependence of use within this study. This method was thus applied to the lag period analysis between the hydrological data and the different measurement stations.

3.5.2 Interviews

Interview data compiled during the data preparation phase were only analysed for proportions (%) and compared in terms of descriptive statistics (mean, minimum and maximum), as the total number of interviews did not support probability testing. Proportions were used further to illustrate the current status of clearing efforts as they pertain to the different stakeholders within the study area.

3.5.3 Field data

Patch and island sites were analysed by first testing for normality of the data using the Shapiro-Wilk and Anderson-Darling normality tests. Raw values were then compared in terms of proportional (%) and actual change of area (m^2) across time stamps. Data were further analysed for difference in median using the Wilcoxon rank sum test for two samples (i.e. comparing two time stamps) and the Friedman rank test for three or more samples (i.e. across all time stamps). Post-hoc testing was applied to the results obtained to elucidate significant differences between time stamps.

Patch and island sites were further analysed by plotting mean annual river level across the study time frame for each weir station, against total area per time stamp for the different sites related to the weir, which was then compared across time. Flood dates as identified during hydrological data analysis were also included in the graphs to evaluate any apparent influence or relationship between flooding and area (m^2/km^2) of a given study site.

3.5.4 Hydrological data

Descriptive statistics (mean, median, minimum, maximum, range, standard deviation, skewness, kurtosis, standard error, coefficient of variation, quartile 50 and 99.5) were calculated for both the rainfall and river level data. Descriptive statistics, graphical illustrations, and tests such as the Shapiro-Wilk and the Kolmogorov-Smirnov tests were used to gain an understanding of the general data shape by normality testing. The 'high rainfall' and 'flood events' were identified using outlier values above the bankfull threshold values identified in Table 3.7.

The high rainfall and high river level measurements contained within the data were initially isolated by using only the outlier values, i.e. any value above the 99.5th percentile in the data. However, using this distinction proved to be too coarse a method, providing an unfeasible amount of focus values to be used in further analysis, necessitating further refinement. The criteria for selection was subsequently tightened and values reduced by only using values that were identified using the bankfull method (i.e. using weir schematics to establish bankfull thresholds and subsequent overland flow).

Bankfull river levels (or bankfull threshold values), which indicated the river bank height as well as the weir depth, were established through examination of the weir design plans (Adonis 2013, Pers com). Weir design plans provided the length, calibration type, width and height used in the initial design and construction of each weir, from which was calculated the bankfull threshold. The bankfull threshold was represented by the difference between the highest plane of the weir and the lowest level through which water flowed past the weir structure.

Bankfull levels were used to calculate the river level threshold, exceedance of which would indicate overland flow beyond that of the river channel (localised flood conditions). Weir designs were only available for Le Chasseur and Swellendam weirs. Bankfull estimates for Karoo, Wolvendrift and Wagenboomsheuvel weirs were obtained by using the known bankfull estimate of the nearest weir site for which there was an available design schematic. Table 3.7 illustrate the stations and their respective bankfull thresholds, as derived from the weir design schematics.

Table 3.7 River level stations and their data description, including location, bankfull level.

Station name	**Identifier code**	**Date range**	**Number of observations**	**Location coordinates (Latitude, longitude)**	**Bankfull level (m)**
Karroo	H4H014	1973-1996	6927	-33.766763, 19.544126	3.1(*D)
Le Chasseur	H4H017	1980-2012	11468	-33.818056, 19.693333	3.1(*C)
Wolvendrift	H5H004	1970-2012	14968	-33.897778, 20.011667	3.1(*D)
Wagenboomsheuvel	H5H005	1975-1991	5357	-34.002311, 20.212756	3.3(*D)
Swellendam	H7H006	1966-2011	18396	-34.067500, 20.405556	3.3(*C)

*C = calculated; D = derived from closest weir with known bankfull level.

Through the use of bankfull levels, only values which had guaranteed localised flooding (localised overland flow), in addition to the criteria of being contained within the 99.5th percentile of the data, were selected for analysis.

These values were then used to identify rainfall and river level of ‘flood magnitude’, to use as reference during evaluation of patch and island site change over time. Rainfall and river level values were first explored for normality using the Shapiro-Wilk and Kolmogorov-Smirnov tests, and illustrated visually through interquartile (Q-Q) plots and histograms. Descriptive statistics (mean, median, minimum, maximum, range, standard deviation, skewness, kurtosis, standard error, coefficient of variation, quartile 50 and 99.5) were also calculated for each station. Spearman rank correlation analysis was then conducted between rainfall and discharge in order to ascertain suitable lag periods, employing lagged correlation analysis. Flood events were then calculated for each weir and rainfall station pairing employing Spearman rank correlation analysis in combination with the identified lag periods.

Of note is the contribution of the Riviersonderend River (a tributary of the Breede River) water beyond the confluence point near Swellendam. As the Riviersonderend Mountains to the South of the Breede Valley divides rainfall falling on the Southern side of the Mountain and rainfall falling in the Breede Valley to the North, the Swellendam weir represents a unique instance. Rainfall to the South of the Riviersonderend Mountains is thus only measured after the confluence, and variation in river level is thus expected for the Swellendam weir. As the Riviersonderend was not included in the scope of this study, only Breede River data was used for analyses. This issue is further addressed in the results chapter.

3.5.5 Aerial photographs

Area calculated for each patch and island site was then used for analyses, by separating each time stamp’s values and comparing them across years. Area and perimeter were calculated using the ArcMap Field Calculator (ESRI 2013), and comparison made between area (m^2) for all sites across the time stamps. For both the island and patch sites, statistical software ‘R’ version 2.15 (R Core Team 2012) was used for descriptive statistics, normality testing, correlation analysis and for testing differences between means between years.

Normality testing was conducted using the Shapiro-Wilk and Kolmogorov-Smirnov tests. These tests were applied in order to ascertain whether non-parametric or parametric testing needed to be applied to the difference-in-mean testing conducted further on during analysis. The Wilcoxon rank sum test was applied to test for differences in median between two groups (time stamps), as well as a Friedman rank sum test for difference in median across all the groups.

3.6 EXPERIMENTAL DESIGN

The experimental design thus followed for this research incorporates 31 island sites, digitised across all years, in order to establish the change across time. Fourteen island sites were also identified in a similar manner and analysed for change across time (GIS calculated area compared across time). Five flood dates were obtained for each weir-rainfall station pairing, and used to assist in interpreting the patch and island site change over time. Fieldwork data furthermore contributed to a 'current' (2013) understanding of the real-world extent of invasion, which included specifically density and age class measurements.

Interview data were then further employed to provide context and greater insight into the hydrological regime specifically with regards to patch and island site area change across time. Figure 3.9 illustrates the various steps taken during data analysis, in order to better illustrate the relationships between the hydrological, spatial and interview data.

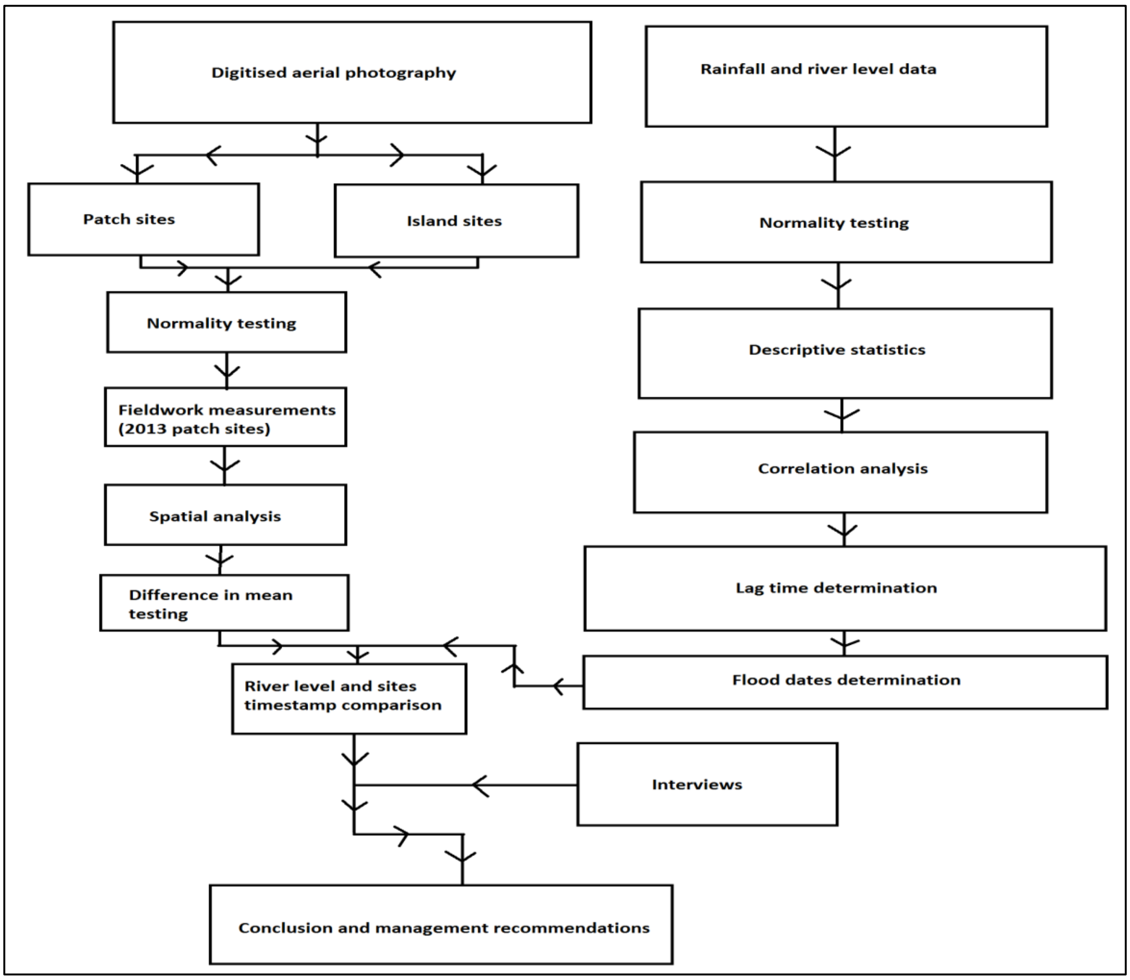


Figure 3.9 Research design displaying data relationships.

CHAPTER 4 RESULTS AND DISCUSSION

4.1 OVERVIEW OF RESULTS

Results obtained from the various data sources are reported and discussed in this chapter. Interview and field data were analysed, hydrological data explored, flood events and appropriate lag times established and the aerial photography analysed for change across time. The chapter concludes with a summary of the important findings.

4.2 INTERVIEWS

Interviews were conducted in order to indicate and emphasise any ancillary concerns surrounding the presence of *E. camaldulensis* within the study area. Fourteen interviews were conducted, therefore not allowing for statistical tests such as difference in mean testing. Proportionate (%) reporting was used as a means to indicate the viewpoints and concerns observed.

Interviewees were, barring one, all the owners of the properties (the exception being one general manager). The landowners obtained their chief income from farming (barring one, which was primarily managed as a vacation property). Table 4.1 details the various livestock proportions indicated to be on the various properties. Livestock not mentioned in any responses were pigs, chicken, goats and game, and subsequently only sheep and cattle were recorded. Livestock was included as a concern as they were believed to clear seedlings through trampling or grazing, which was validated by one respondent in an interview. The majority of the sites did not exhibit large quantities of livestock, making grazing impacts minimal within this study.

Table 4.1 Livestock summary on all properties of interviewees.

Type	Number respondents indicating livestock present (n)	Livestock sum	Mean of class total, from all responses (rounded down)
Sheep	4	2145	152
Cattle	7	353	25

A summary of the *E. camaldulensis* age class responses is provided in Table 4.2, indicating the number of times respondents indicated (cited) a specific class present, and the total proportion for each class for all classes mentioned. Respondents indicated that the largest age class present on their farms were predominantly that of the 'very mature' class, which represents the oldest growth stage of this species, suggesting the individuals present on respondents' property have been present between 30 and 50 years prior.

Table 4.2 Age class summary for all respondents.

Age class	**Citations per class (count)**	**Class proportion (%) of all citations**
Seedling	1	0.5
Young	2	3.2
Adult	5	23.6
Mature	8	21.8
Very mature	9	50.9

Results supported that *E. camaldulensis* trees had been present on the properties for long periods, with the mean time of their presence being 40 years (40.46 years; n = 13; SE = 5.57), which, if left undisturbed, would correspond to the presence of any 'very mature' individuals currently. Introductions into the Breede River in particular were noted going back as far as the 1950s by interview respondents, with recollections of purposeful planting of *E. camaldulensis* along riverbanks of the Breede River by respondents and their families going back three generations (roughly 75 years).

E. camaldulensis plantations were also noted along the Breede River as far back as the 1980s, where large plantations were initiated immediately north of Robertson for timber use (Bruwer 2013, Pers com). *E. camaldulensis* trees grown in forestry plantations would have to be cultivated for between five to 10 years before being commercially useful, as first harvest is commonly conducted at year six or seven (Bernardo et al. 1998), and one can thus reasonably assume the introduction of *E. camaldulensis* into the Roberson area, at least commercially, from at least the 1950s onward.

Non-commercial or private use of *E. camaldulensis*, however, can be regarded as having existed since at least the 1970s within the study area. Adult *E. camaldulensis* individuals would have added propagules into the river system, potentially contributing to the *E. camaldulensis* spread downstream. As *E. camaldulensis* seed bank is short lived (Tererai et al. 2013), the additional propagule pressure applied by the smaller populations of *E. camaldulensis* historically, would thus not necessarily have been the main driver of spread within the catchment. Nevertheless, the expansion of *E. camaldulensis* would be partially influenced by the propagule pressure applied through the existence of non-commercial and private-use forests on the river banks.

Larger populations such as those from commercial forestry, could conceivably have added larger amounts of propagules into the river system as more propagules would be produced, impacting to a high degree on the dispersal of *E. camaldulensis* downstream of their location. Gradients in propagule pressure such as this have long been recognised as important in determining invasion extent (Eschtruth & Battles 2009). Certainly any population increase of *E. camaldulensis* downstream of the Robertson area would have been partially influenced by the forestry populations present there. The occurrence of *E. camaldulensis* in the upper reach of the river (outside the study area), however, would suggest that Robertson was not the uppermost seeding source within the river.

Despite *E. camaldulensis* having been present along the Breede River since the 1970s, the majority (81.71%) of the respondents indicated never having planted *E. camaldulensis* purposefully, or having *E. camaldulensis* planted prior to their occupation of the property. The remainder (18.29%), represented by two instances, indicated purposeful planting, explained by the usefulness for shade trees and firewood of *E. camaldulensis*. Such a large proportion of respondents stating no involvement in propagating *E. camaldulensis*, could be an indicator of two factors; (i) stands of *E. camaldulensis* along the river are predominantly self-sown and self-maintained stands, and/or (ii) respondents were hesitant to state their involvement in what would be deemed an offense in terms of the law.

Most respondents (64.25%) indicated active clearing of the *E. camaldulensis* on their property, when compared to *not* clearing (35.75%). As such, clearing efforts by respondents were either not effective in the control of *E. camaldulensis*, or if any clearing occurred at all, were overestimated. The combination of few respondents involvement in propagation, coupled with statements of active clearing, suggest the first condition of self-sown and maintained stands to be the most likely, despite deliberate intervention and management. This conclusion concurs with Tererai et al. (2013), which make specific mention to the presence of self-sown stands along river systems within the Western Cape.

Reasons provided for respondents not clearing included the lack of appropriate permissions, a lack of time and finances as well as *E. camaldulensis* on the property not proving to be ‘problematic’ (in terms of ongoing farming activities) or even being too few to ‘be concerned about’. Contrastingly, reasons for clearing ranged from *E. camaldulensis* being regarded as undesirable (as a weed), to the need for firewood, to the species causing obstruction within the waterways on the property, as well as the stated relative high water usage of *E. camaldulensis*. It is clear, that regardless of the respondents’ personal involvement in clearing on their properties, that an appreciation for the ecological and financial threat posed by the presence of *E. camaldulensis* was well understood.

The majority (71.42%) of respondents indicated that other species in addition to *E. camaldulensis* had been cleared on their properties. The most frequently cited additional species cleared was *Acacia saligna* (80.00%), followed by *Acacia mearnsii* (10%) and other *Eucalyptus* spp. (10%). The presence of *Acacia mearnsii*, a dominant secondary invader, as indicated by the responses above, corresponds to that of the Berg River study conducted by Ruwanza et al. (2013). Reasons provided for such clearing included categories such as the obstructive potential and status as a weed (36.36%), agricultural land encroachment and their potential in terms of fire damage (9.09%). Study sites of the size used in this research, can be considered a fire risk due to their small size, as discussed by Van Wilgen, Higgins & Bellstedt (1990). This was confirmed through discussions with various farmers during field visits, as increased litter and fire risk was deemed one of the associated characteristics of *E. camaldulensis* from the farmers’ perspective.

Only 21.42% of respondents indicated that a clearing agency, or anyone other than the staff of the property, had cleared invasive species on the property in the past. The DWA was cited as the clearing agency involved in one instance, with clearing activities confined to the riparian fringe of the property, and having been conducted once only in this particular instance. The majority of the respondents thus indicated little involvement by state agencies in IAP management in their responses. Current planned DoA operations (2015 onwards), however, include the middle section of the river for clearing (Röscher 2012, Pers com), and as such the proportion of involvement is likely to be much higher once clearing operations have been concluded for this section.

When clearing was conducted by respondents, the methods were combined, mostly through felling and then spraying with a herbicidal agent (33.33%), or frilling. The single method cited most was felling (45%), followed by herbicidal agent (20%), bulldozing (15%), frilling (10%), hand-pulling and ring barking (5%). Despite most properties being invaded by more than one species of IAP, notably *Acacia saligna*, respondents most commonly still employed burning as a removal technique in order to 'gain control' of the IAP growth. Such methods may stimulate *A. saligna* occurrence.

In the case of *Eucalyptus* species, ring-barking and frilling are recommended (Martens, Waller & Delahunt 2003). The implication of these results are that the incorrect control method is currently being applied by the majority of the respondents, possibly due to a limited understanding of the species-specific response to treatment and the various methods available. Control methods chosen by respondents were predominantly more expensive in practice than that of the suggested methods (Martens, Waller & Delahunt 2003), eliminating cost as a possible explanation for the choice in control method.

Clearing actions conducted by the respondents are likely to remain less than optimal unless deliberate intervention and communication of the appropriate techniques are provided to the landowners. The implication thereof is a possible re-establishment of *E. camaldulensis* after clearing action, as well as ineffective control of current populations within the riparian fringe. The seeding source represented by the current populations of *E. camaldulensis* is thus likely to continue, despite the potential for greater proportions of landowner-instigated clearing along the river. It is the author's suggestion that the DoA employ a small-scale educational programme whilst liaising with landowners in the operational phase of their Breede River clearing programme to assist future clearing operations conducted by land owners. Improving the control techniques and exploring the various barriers to IAP clearing in and along the river could assist in control by landowners.

Further interview questions included background information on the flood history of the river. Respondents identified flood years as 2008 (31.83%), 1981 (18.19%), 2007 and 2009 (9.10%), as well as 1993, 2000, 2002, 2003, 2006, 2010 and 2011 (4.54% each). These flood dates correspond only to a 'memorable' flood events, and differs from one respondent to another, particularly as certain parts of the river may not have experienced memorable overland flow during any given event, whilst another may have. Generally, similar flood dates were observed in the hydrological results, indicating landowners were a good proxy for identification of flood events, although mostly restricted to recent events which could be remembered clearly. The flood results are discussed further in the hydrology section of this chapter.

The majority of respondents (53.84%) indicated they made no attempt to mitigate any flood damage on their properties, with the remainder doing so, by the placement of floodwalls (44.44%), and clearing of IAPs (55.56%). Respondents cited the impracticality of effectively limiting and mitigating large quantities of water (77.77%), along with the prohibitive costs involved, as the reasons for not attempting mitigation in all flooding instances. Importantly, most respondents (61.53%) believed the flooding to have influenced the occurrence of *E. camaldulensis* on their properties, with the minority (23.07%) stating they believed flooding had no influence, and the remainder indicating that they did not know. Opinion was varied on what exact influence flooding had on the *E. camaldulensis* presence, with 'improving condition' and 'allowing for new growth and seed dispersal after flood' being the two largest proportional responses (33.33% each).

Combined, the 'conditioning' and 'germination and seeding-dispersal' responses illustrate the majority of landowners believed *E. camaldulensis* to be related to the hydrological regime of the river, although to what extent remained unexplained from the interview data. Evaluated solely from the interview data, it is clear that respondents believed a relationship between flooding and *E.*

camaldulensis vigour and distribution does exist within the Breede River system, with annual flooding events, as is common, providing soil moisture to – and propagule transport from – the current populations. Germination of *E. camaldulensis* from layers of deposition or disturbed soil was also noted in 23.07% of responses, indicating active recruitment after overland flow was experienced and could potentially account for larger populations in the future.

Respondents, however, also validated the absence of a negative relationship, i.e. where water levels and discharge during floods events would be sufficient to remove and effectively reduce *E. camaldulensis* stands over time. Two respondents, a marked minority (15.38%), negated such an effect in their responses, stating *E. camaldulensis* seedlings germinated specifically from areas of disturbed and removed topsoil after a flood event. It is therefore likely the current population dynamics exhibited within the riparian zone, is influenced to a greater degree by the various competition interactions of the species present. Furthermore, establishment may be supplemented by a sufficient soil moisture supply from seasonal flooding and downstream dispersal of seed more so than by flood-induced vegetation removal. The current hydrological regime was not regarded as being generally prohibitive in terms of *E. camaldulensis* vigour, dispersal or spatial extent, but rather additive in that regard. Spatial analysis conducted later on in this report expounds the principle further.

Secondary growth (defined as new growth of any species from germination) after a flood event was not reported in 42.85% of responses, with the majority (57.15%) indicating secondary growth to have occurred in three distinctive ‘generations’, or stages of succession. Stage I growth was dominated by *Acacia saligna* (62.5%), followed by *E. camaldulensis* (25%). Stage II growth was predominantly *Acacia mearnsii* (50%), with *Eucalyptus* spp. following (33.33%). Stage III growth was composed of equal citation proportion of *Sesbania punicea*, *E. camaldulensis*, *Datura ferox* and *Acacia saligna*.

A small proportion (14.28%) of respondents, specifically made mention of *Acacia* species showing greater dominance, in terms of vegetation composition initially, which is then replaced by *E. camaldulensis* dominant vegetation within a few years. Ruwanza et al. (2012) found a high cover and richness of secondary invasion after *E. camaldulensis* clearing in the Berg River, however, the dominant alien life forms were herbaceous and graminoid, and not specifically *Acacia* species.

The interview responses, however, illustrate the predominantly alien re-establishment after flood events, suggesting that flooding events may promote alien vegetation such as *E. camaldulensis* and *Acacia* species above that of native vegetation. Restoration of similar sites in the Berg River required a combination of active and passive intervention (Ruwanza et al. 2013). Any clearing of *E. camaldulensis* and other alien vegetation after flood events should thus not be regarded as permanent gain, but rather a momentary disturbance of the vegetation structure. Such clearing is usually followed by alien herbaceous and graminoid secondary invasions (Tererai et al. 2013).

Additional questions focussed on the occurrence of island and bar sites within the river. Most respondents (71.42%) indicated that *E. camaldulensis* had an influence on the occurrence of islands. According to responses the influence was exerted primarily through the *E. camaldulensis* acting as obstruction to water flow, thereby trapping sediment and debris (35.71%). Respondents further indicated *E. camaldulensis* was not the only species implicated in the changing of island shape and occurrence, as *Acacia saligna* (31.81%), ‘reeds’ (27.27%) and *Eucalyptus* spp. were also cited, in addition to *E. camaldulensis* specifically (18.18%).

The minority of respondents regarded floods as impacting on the occurrence and size of islands within the river (42.85%), with the majority claiming no impact at all (57.15%). Those that indicated an influence, regarded the mechanism to be twofold; that of displacement of the islands (50%), and that of a size increase due to increased silt deposition after flood events (50%). The results indicate flood events *do* change the river morphology, shape and presence of islands. Vegetation implicated in exacerbating this impact was not restricted to *E. camaldulensis* specifically, but rather to alien vegetation in general. The presence of *E. camaldulensis* along the river can therefore not be regarded as impacting on the formation of islands to a greater extent than any other IAPs along the river.

Furthermore, from responses one can conclude that flood events would change the island formation and shape within the river, by transporting and depositing silt downstream. Depending on where in the river profile one was, respondents would state either a broad, shallow river which was becoming deeper and narrow due to IAP growth, or a narrow, deep river which was becoming broader with time. Such ‘discrepancies’ may be accounted for by considering the small-scale variation in the river profile, with much variation in terms of depth and breadth being common. The change in islands are spatially analysed further in this chapter.

Further results from the interviews included property considerations. Half of all respondents indicated that flooding does not influence their properties or business, with the other half responding in the affirmative. Those who stated no influence based their answer on the fact that no appreciable change was observed by them, due to flooding, and as such no impact was realised from the presence or absence of floods. Respondents who reported impacts from flooding, stated a loss of topsoil and arable land (66.66%) as the primary impact experienced, with silt deposition, and a legal liability, forming the remainder of the responses. Loss due to flooding and high rainfall events within the region was observed in the past and included municipal damages, such as roads (Holloway et al. 2010), and it is likely the respondents did not consider indirect influences of flooding, such as the loss of municipal services or road networks, on their businesses. These indirect and direct damages associated with flooding and high rainfall events are therefore likely to be higher than reported by the respondents.

Even so, 61.53% of the respondents made no attempt to mitigate flood impacts on their property. These results are contradictory in part, to the general observed responses of ‘no vegetation removal’ with flooding. The discrepancy can be explained by the question focussing on business impact experienced. Respondents may have been more aware of topsoil removal if it were sections of their property that was actively used or planned for agriculture, whereas ‘incidental’ removal along unutilised river banks may not have been noted.

Additionally, only half of the respondents indicated topsoil removal, and thus the representative proportion of respondents attributing topsoil removal to flooding was reduced even further. Flood damage was therefore not generally regarded as being due to vegetation, or large-scale topsoil removal, but rather through loss of crops after prolonged submersion. Many respondents stated that the clearance of *E. camaldulensis* on the river banks would allow for increased water flow, and thus lessen flood damage to their property. Still others stated the opposite, citing fast flowing flood waters as the main cause of flood damage during such an event.

4.3 FIELD DATA

In this section, the field measurements are analysed and discussed, to provide a broader perspective on the current (2013) characteristics of the patch sites. Across all 31 sites, exotic vegetation dominated the landscape, with only seven out of 31 sites (equivalent to 22.58%) having a majority (>50% canopy cover) native vegetation presence. Density of sites were also recorded, in accordance with WfW regulations, which required density to be expressed as the percentage of area covered, of the area of interest, by the stems of all target species present (Röscher 2012, Pers com).

Across all patch sites, *E. camaldulensis* covered large parts, however with stem densities never exceeding 45% coverage (mean = 30.22%). Stem densities of 45% however, represented almost complete canopy cover of the area, if the majority of the age class present was ‘very mature’ (>30m tall). Areas consisting predominantly of younger age classes were not expressed by proportions of such a large canopy cover, rather varying greatly, depending on the composition of the different classes within the site. This is to be expected, as seedling canopy size is much less than the canopy of a mature individual.

Using stem area coverage, across all the sites, seedlings were the least represented of the age classes, representing only 4.16% of the *E. camaldulensis* measured. The young age class represented 13.03%, with the adult class occupying 24.87% and the mature and very mature classes representing 30.75% and 27.19%, respectively. These results correspond to the study conducted by Tererai et al. (2013) on *E. camaldulensis* on the Berg River, finding stands to comprise mostly of mature individuals with little recruitment, which was suggested to be due to shade intolerance of seedlings and the short lived seed bank of this species.

The findings suggested an initial large-scale establishment of *E. camaldulensis* in some section along the upper stretches of the river, roughly 30 years ago. A possible explanation for such a sudden appearance could be the establishment and maintenance of an *E. camaldulensis* plantation adjacent to the river, immediately upstream of Robertson town around 1980 (Bruwer 2013, Pers com). Such a plantation would be a seeding source after maturation, causing downstream dispersal of seed and the subsequent downstream invasion.

The pattern of seedlings being the least represented age class, indicated either a low rate of establishment, or unsuccessful competition by seedlings for nutrients, water and sunlight. Factors such as flooding, anthropogenic soil disturbance (Brown & Gurevitch 2004) and clearing may have created suitable locations for the younger age classes to establish and develop, potentially resulting in seedlings forming a small proportion of the total composition, as was found in data.

Land use categories observed across the patch sites varied greatly, with agriculture representing the majority land use within the study area (Table 4.3). Land use categories for adjacent properties are also recorded, illustrating a similar composition.

Table 4.3 Land use categories for all patch sites and nearest neighbouring properties.

Category	**Representation of patch site land use (%)**	**Representation (%) of nearest neighbouring property**
Recreation	3.22	3.22
Wine grapes	61.29	54.83
Soft fruit	9.67	9.67
Citrus	6.46	6.45
Wheat	6.46	3.22
Undisturbed	12.90	3.22
Dairy	-	9.67
Cattle	-	3.22
Lucerne	-	3.22
Equestrian	-	3.22

As expected, viticulture dominates the land use of the area, followed by the growth of soft fruit, and thereafter citrus and wheat. As the Breede Valley is a large contributor to the wine industry in South Africa, this result is consistent with the overall land use pattern of the region.

Patch site size was generally small (mean = 14 484.23 m^2, SE = 2 768.07). Perimeter values for patch sites ranged from 63 to 2720 m (mean = 813.29 m), with an average of 18 minutes' walk required to cover the perimeter of each site (mean = 18.06 min). Slope of the sites ranged from 0 to 60° (mean = 17.25°), making certain sites almost inaccessible due to the steep gradient. Further difficulty observed during field visits include underfoot conditions, which ranged from stable grass covered areas, to knee-height mud or loose sand. Conditions experienced most across the sites were loose sand (45.16%) stable grass (35.48%), stable soil (12.90%), mud (3.22%) and loose debris/litter (3.22%).

Obstructive vegetation densities, measured as the percentage of area covered, were often also prohibitively dense in terms of area covered by the vegetation as a whole (mean = 46.58%, as canopy surface area cover), with taxa ranging from *Phragmites australis*, *Prionium serratum*, *Acacia saligna*, *Datura ferox* and common grasses. The obstructive vegetation was frequently found in the same region as that of the *E. camaldulensis* within these sites, and may be of concern for clearing agencies (due to lack of access). Exploratory site visits are recommended, as the height of such vegetation (mean = 2.45 m) can inhibit access to the target species, and may require additional planning by clearing organisations.

The most common vegetation associated with *E. camaldulensis*, was that of *Phragmites australis*, *Acacia saligna* and *Acacia mearnsii*, which was also found by Tererai et al. (2013) in the Berg River catchment. However, *Eucalyptus* species remained the dominant vegetation in terms of height, biomass and litter. The presence of less dominant *Acacia* species was only observed where *E. camaldulensis* was near a water source, suggesting competition between the two species was in favour of the *E. camaldulensis*, however, only in the presence of a sufficient supply of moisture. *Acacia* species could thus be more competitive under drought conditions. This specific interaction, requires further research to verify beyond anecdotal observation, and may provide useful insight into the second generation IAPs that resurface after initial clearing of *E. camaldulensis*.

A final result from field data was the confirmation of bare patches in the immediate local vicinity beneath the *E. camaldulensis* canopy. With the exception of one site, all sites visited displayed similar characteristics. These results are similar to what was found in the study on *E. camaldulensis* impacts by Tererai et al. (2013), in which reduced light by *Eucalyptus* canopy and the allelopathic potential of *E. camaldulensis* was singled out as a possible cause for the apparent decline in species richness, vigour and composition near or beneath *E. camaldulensis* stands. Not conforming to the abovementioned observation however, was one site in particular that had a large area covered with associated *Pennisetum clandestinum* (kikuyu). This site was however near a recreational facility, and the grass may have been purposefully planted prior.

4.4 HYDROLOGICAL DATA

Hydrological data were further analysed for the general hydrological regime of the study area. The results are discussed below.

4.4.1 Rainfall data

Normality testing of the daily rainfall data using the Shapiro-Wilk tests (Table 4.4) indicated that the data were non-normal in all cases (combined and for individual stations), with $p < 0.001$ against the H_0 of normally distributed data. The data were also strongly skewed, i.e. a skewness value greater than zero, as is standard for the normal distribution (Kim 2013), thus further suggesting nonparametric data (Doane & Seward 2011). The kurtosis value, which is a measure of the ‘peakedness’ of data (Kim 2013), also indicated a strong leptokurtic congregation about the mean. Kurtosis values greater than seven were regarded as a substantial departure from normality (Kim 2013), which can clearly be seen in Table 4.4. An inspection of the raw data revealed the majority of the daily rainfall values were either zero, or below 10mm. This was due to the presence of ‘dry days’, or days with very little precipitation, being the most frequent within the data.

Table 4.4 Daily rainfall (mm) descriptive statistics summary for all stations and years.

	Kwaggaskloof Dam	**Robertson**	**Rhebokskraal**	**Ashton**	**Marloth**	**Bontebokpark**	**Combined**
Date range	1980-2012	1980-1986; 2000-2012	1980-2012	1980-2012	1980-2012	1980-2012	1980-2012
Mean (mm)	8.24	5.32	7.39	6.37	10.6	6.47	7.41
Median (mm)	4.5	2.3	4	3	5.5	3.4	3.60
Min (mm)	0.1	0.1	0.3	0.1	0.1	0.1	0.1
Max (mm)	123	185	127	122	149.2	103.5	185
SD	10.98	9.06	10.88	9.18	14.17	9.25	10.95
Skewness	3.65	7.13	5.05	3.91	3.45	3.57	4.27
Kurtosis	**23.05**	**102.8**	**39.12**	**26.5**	**18.33**	**18.89**	**31.25**
SE	**0.3**	**0.21**	**0.3**	**0.24**	**0.27**	**0.18**	**0.1**
Q99.5	55.8	47.55	67.45	53.689	84.17	55.85	65.50
S-W: p (α = 0.05)	**W = 0.66; p < 0.001**	**W = 0.52; p < 0.001**	**W = 0.56; p < 0.001**	**W = 0.63; p < 0.001**	**W = 0.65; p < 0.001**	**W = 0.63; p < 0.001**	***D = 0.67; p < 0.001**

*Due to Shapiro-Wilk test sample-size limitation, Kolmogorov-Smirnov normality test was conducted for the combined station data.

Days without precipitation were excluded from the analysis, as they would have no impact on the hydrological dispersal of *E. camaldulensis* within the context of this study. The remaining days for which precipitation was recorded, consisted predominantly of low rainfall events, indicating that on days that rain occurred, very little rain actually fell (mean = 7.41 mm; n = 12 000). These results are not unexpected for rainfall data, as the oscillations of ‘wet’ and ‘dry’ days are a common understood reality of weather patterns. Of note in this data is not the large deviation from normality, but rather the incidence of low rainfall values experienced on ‘wet’ days, as flooding would logically necessitate large volumes of rainfall within a short period of time.

Figure 4.1 indicates the overall data shape, as a combined histogram of all values across all years. The red line depicted is the 99.5th percentile boundary, with outliers observed to the right of the line. Figure 4.1 further highlights the long ‘tails’ common to a leptokurtic distribution (Doane & Seward 2011). As zero values were omitted from the graph, the observed close proximity of values to zero is represent by the high number of values obtained close to zero (days on which small amounts of rainfall were recorded).

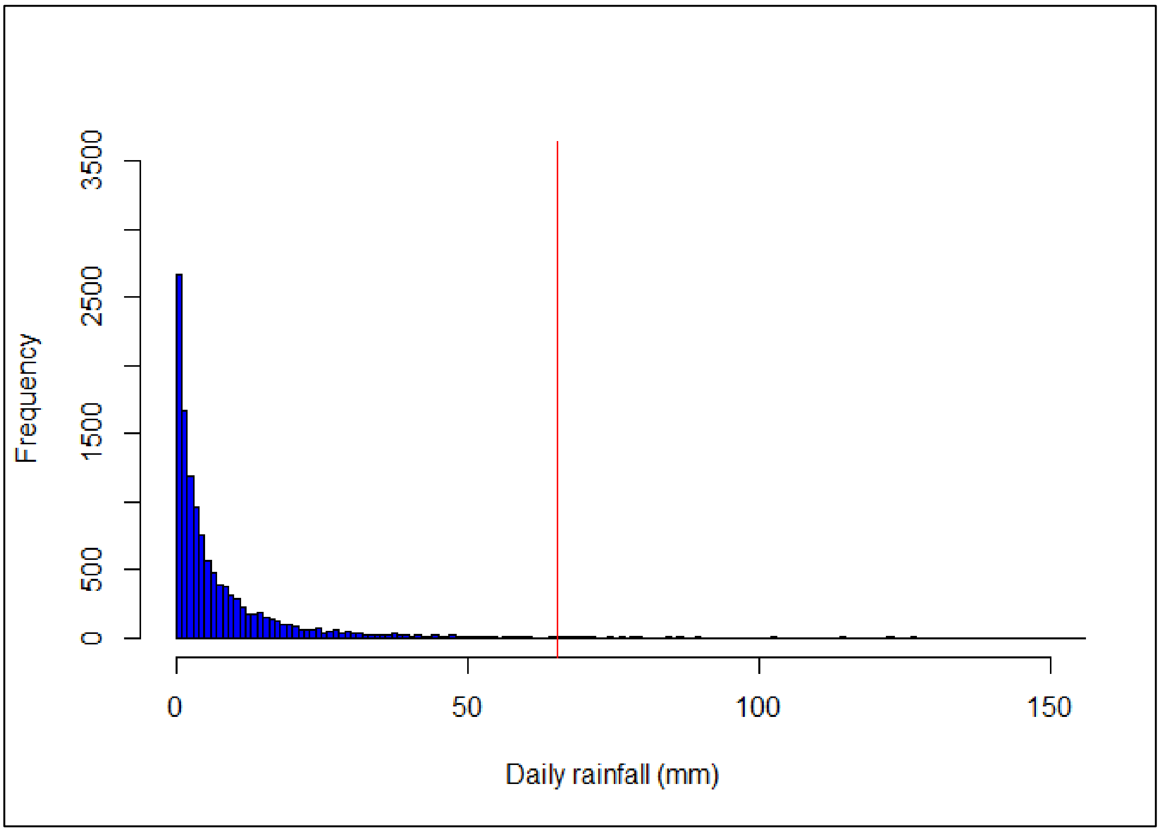


Figure 4.1 Histogram of daily rainfall values (mm) for all stations combined, 1980-2012 (n = 12 000). The red line indicates the 99.5th percentile boundary of the data. Zero values were omitted. Frequency here refers to the count of any given rainfall value within the data.

The distribution of the data were further visually compared to randomly produced datasets of the normal distribution, using identical parameters to the rainfall data (mean, standard deviation and skewness coefficient), through the Q-Q plot function of 'R' (R Core Team 2012) (Figure 4.2). Rainfall data deviated strongly from the linear representation of normality, supporting the normality testing results. Deviation is noted here by the non-conformity of the values to the trend line of a simulated Gaussian population, represented by the linear feature in Figure 4.2. If the data were more 'normal approximate', the values would adhere to the trend line, however, a clear curve is observed in the sample quantiles.

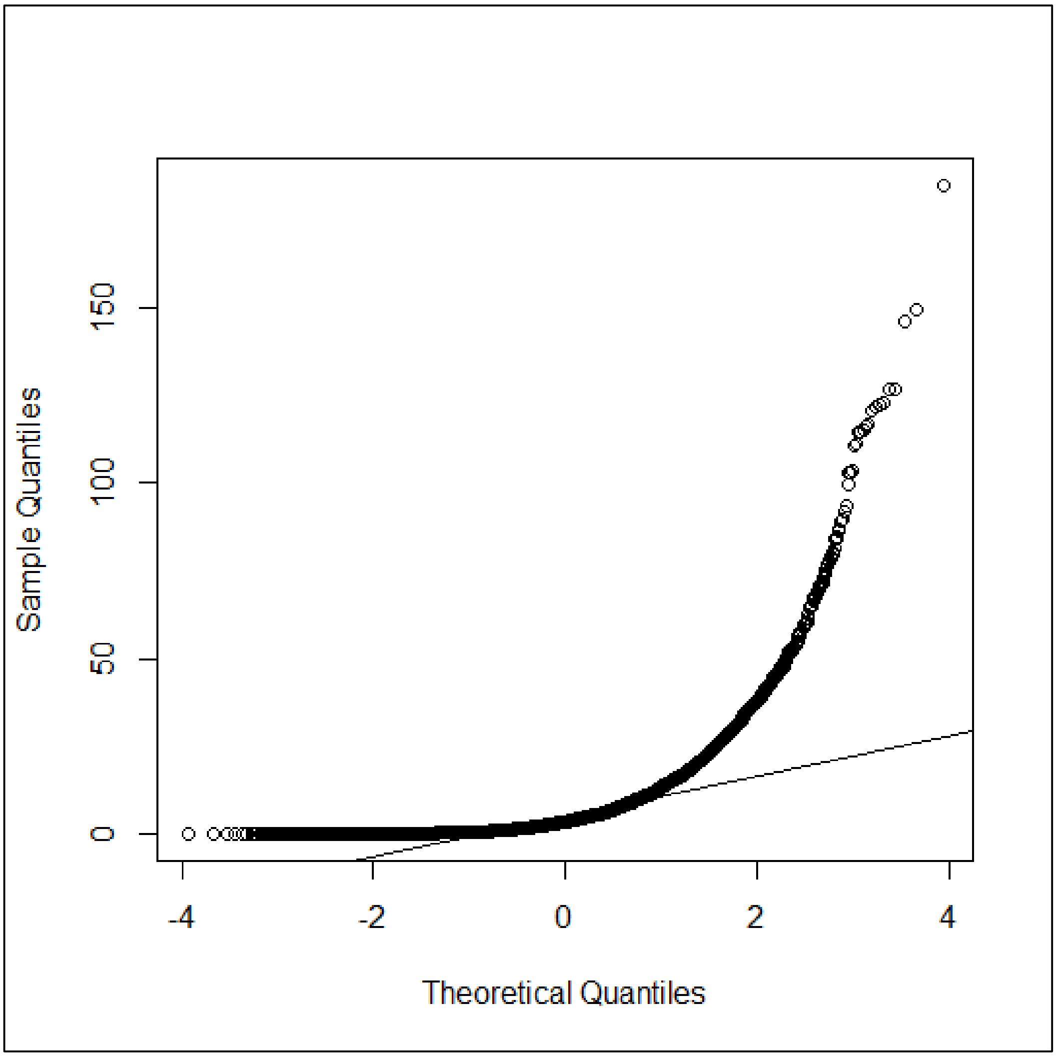


Figure 4.2 Daily rainfall (mm) Q-Q plot for combined stations, 1980-2012 (n = 12 000). The straight line represents the trend of simulated normally distributed data.

Outliers, above the 99.5th percentile horizontal red line in Figure 4.3, represent high rainfall and possibly extreme flood events (Alexander 2002). Sixty-one entries above the 99.5th percentile (>65mm per day) illustrate the many potential flood events. Values above the 99.5th percentile mark were considered 'high rainfall' events due to the rare frequency of such relatively large events within a 24 hour interval. These values were subsequently used to identify dates of flood events.

It was clear from the results that the data for rainfall were non-normally distributed, which was not unexpected for data such as this. Further results indicated low values for days when rain actually fell, and a high number of potential flood events over the study period.

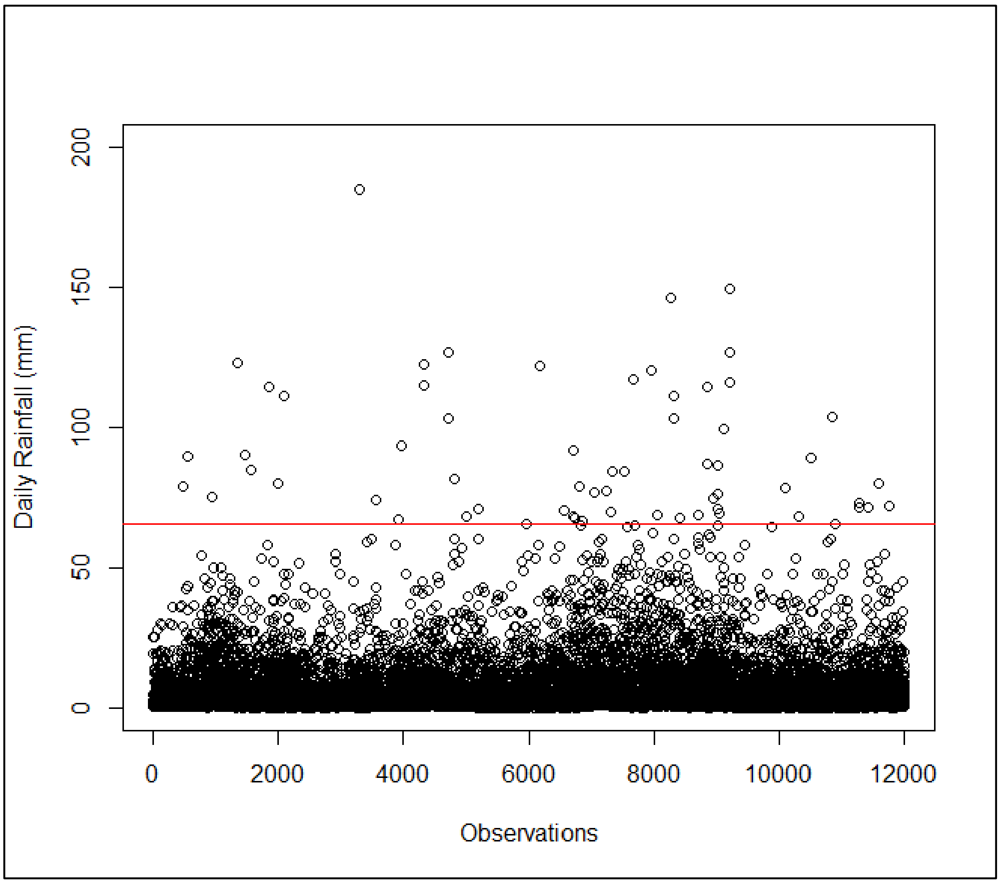


Figure 4.3 Plot of daily rainfall combined for all stations and years. The red line indicates the 99.5th percentile.

The large range of daily rainfall values (0.1-185 mm per data point) serve to illustrate the highly variable nature of rainfall magnitude within the study area. It was also clear from the results that intervals of high rainfall were not uncommon. As they were extreme events, the values were outliers in terms of the data, being less frequent and greater in magnitude than the remainder of the data. Establishing the presence of high rainfall events was an important observation in the context of this study, as the occurrence and magnitude of rainfall was logically an influencing factor on the presence and magnitude of flood events. Potential for high rainfall events were thus established from the data, illustrating that strong rainfall events were indeed a feature of the hydrological regime within the study area, albeit that they occurred at a lower frequency than events of smaller magnitude.

Even so, the specific dates of high rainfall events, and any associated high river levels, remain obscured. River levels are discussed further in a similar manner, in order to evaluate the general data shape and parameters, and to allow for the independent identification of flood events.

4.4.2 River data

Normality testing of the river level data (Table 4.5) using the Kolmogorov-Smirnov normality test indicated that the data were non-normal in all cases (combined and for individual weirs), with $p < 0.001$ for all weirs against the H_0 of normally distributed data. The data were also strongly skewed (Table 4.5), thus further suggesting non-normally distributed data. The kurtosis value furthermore indicated a strong leptokurtic congregation about the mean. All but one station (Karroo) reported Kurtosis values greater than seven, which was regarded as a substantial departure from normality (Kim 2013).

Figure 4.4 illustrates the combined weir data histogram, showing the positive skewness of the data. Outliers were isolated through the boundary of the red 99.5th percentile line (Figure 4.5), where all the outliers lay to the right of the line. The graph was also distinctly leptokurtic, further suggesting the non-normal distribution of the data.

The distribution of the data were further visually compared to randomly produced datasets of the normal distribution, as was done previously for rainfall data. Identical parameters to the river level data (mean, standard deviation and skewness coefficient) were used in the Q-Q plot function of ‘R’ (R Core Team 2012). Figure 4.5 shows that river level data deviated strongly from the linear representation of normality, again supporting the normality testing results. Deviation was noted by the non-conformity of the values to the trend line of a simulated Gaussian population, represented by the straight line (Figure 4.5). Greater adherence to the line would be expected for more ‘normally approximate’ data.

Table 4.5 Daily river level (m) descriptive statistics summary for all stations and years.

Zero included	**Karroo**	**Le Chasseur**	**Wolvendrift**	**Wagenboomsheuvel**	**Swellendam**	**Combined**
Date range	1973-1992	1980-2012	1970-2012	1975-1991	1966-2012	1966-2012
Mean	0.60	0.90	0.67	0.93	0.79	0.77
Median	0.34	0.75	0.49	0.7	0.72	0.67
Minimum	0.00	**0.22**	0.00	**0.20**	**0.01**	0.00
Maximum	4.58	4.82	7.13	6.27	7.03	7.13
SD	0.67	0.39	0.60	0.70	0.51	0.57
Skewness	**2.21**	**2.86**	**3.54**	**2.35**	**1.77**	**2.45**
Kurtosis	**5.57**	**13.84**	**18.33**	**7.72**	**7.98**	**11.04**
SE	0.01	0.0037	0.0049	0.01	0.0038	0.0024
Q99.5	3.61	2.94	4.05	4.22	3.00	3.60
K-S: p	**D = 0.50; p < 0.001**	**D = 0.66; p < 0.001**	**D = 0.55; p < 0.001**	**D = 0.59; p < 0.001**	**D = 0.53; p < 0.001**	**D = 00.52; p < 0.001**

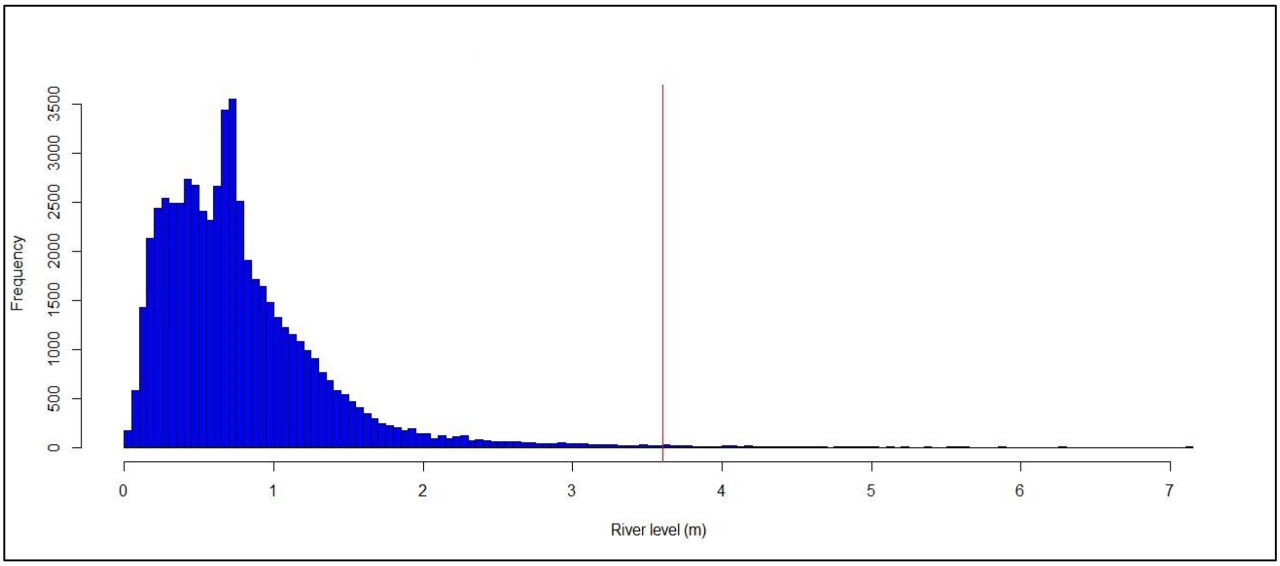


Figure 4.4 Histogram of daily river level values (m) for all stations combined, 1966-2012 (n = 57 116). The red line indicates the 99.5th percentile boundary of the data. Zero values were omitted. Frequency here refers to the count of any given river level value within the data.

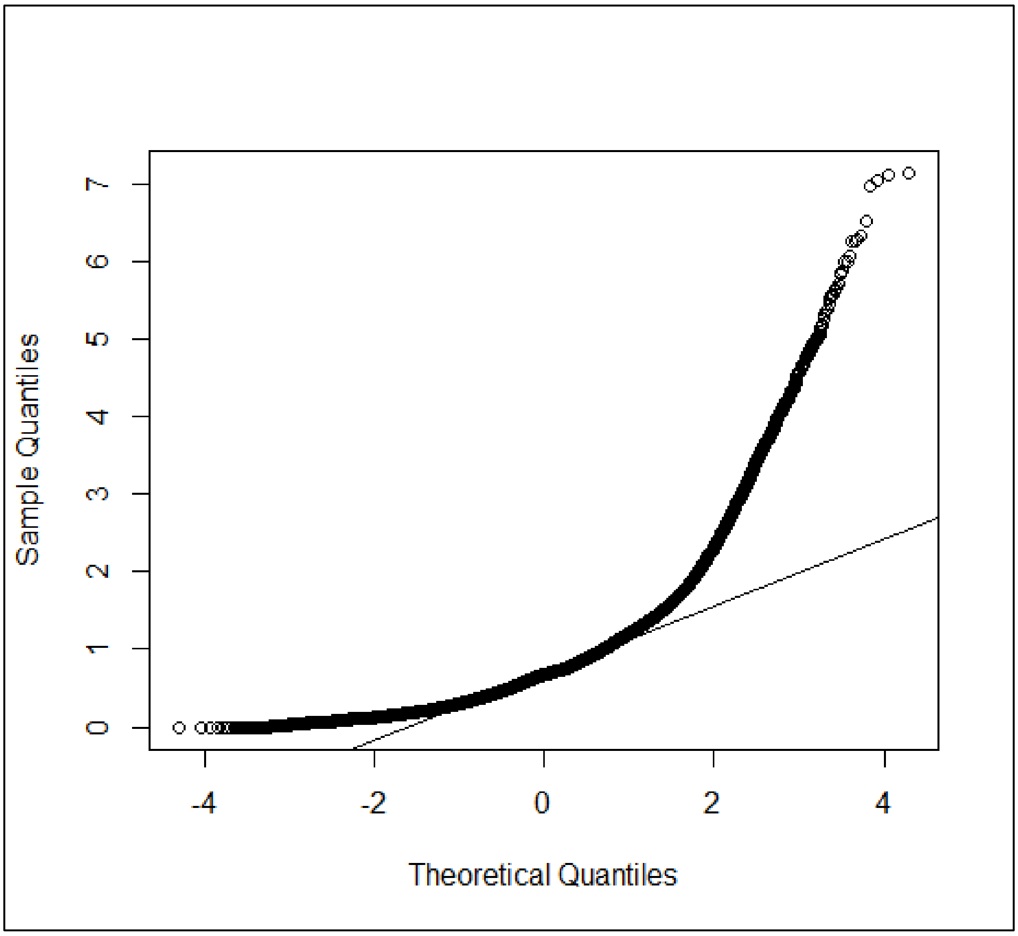


Figure 4.5 Daily river level (m) Q-Q plot for combined stations, 1966-2012 (n = 57 116). The straight line represents the trend of simulated normally distributed data.

The minimum values presented in Table 4.5 illustrate that three of the five weirs, namely Le Chasseur, Wagenboomsheuvel and Swellendam, never displayed flow values of zero. Flow thus never ceased at those locations. Even so, river level results suggested water level within the study area was generally low (mean = 0.77 m; n = 57 116). These results could potentially be explained by the large volumes of anthropogenic water abstraction from the river historically, as well as seasonal oscillation in the water levels of any given river channel. Furthermore, potential flood 'pulses', or periods where elevated levels of water was observed, could also have been influenced by water releases from the Brandvlei Dam. Steynor, Hewitson & Tadross (2009) also note the influence of abstraction and dam releases as impacts on the hydrological regime of the river.

Although such water releases have been confirmed from the literature, interview respondents suggested the release of dam water was not sufficient to allow for flooding within the river. In addition, the water released and subsequent elevated river level form part of the weir measurements, and as such form part of the hydrological regime of the river (regardless of the anthropogenic nature thereof). As such, these releases would also impact on the distribution of *E. camaldulensis* similarly to natural phenomena such as high rainfall and associated flood events. As both sources of water input (rainfall and released dam water) form part of the weir measurements, both are evaluated in the context of this study, and no distinction between the two sources were drawn.

The large range observed between minimum and maximum values (0-7.13 m) further served to illustrate the high degree of variation within the river levels. Additionally, the maximum values shown for river level exceeded the bankfull threshold, as identified in chapter three, for all weirs. The presence of overland flow, and thus flooding, was confirmed through the maximum values from the results. This is an important observation, as the aims and objective of this research could not be satisfied without the verified occurrence of floods within the study area.

Furthermore, as values greater than 3.6 m (99.5th percentile delineation for outliers used in Figure 4.6), numbered 96 entries (comprised of three day events), one could conclude that a high number of potential flood events were observed from the data. Values above the 99.5th percentile mark were considered outliers due to the proportionately few values of such magnitude present in the data. These values were subsequently used to relate to rainfall events and identify flood dates. Due to flood events not having upper limits in terms of severity (Alexander 2002), no defined upper boundaries were self-evident from the data. Further refinement of the extreme river level values was thus required, as an impractically large number of values above the 99.5th percentile threshold were found. The values of greatest extreme were isolated and only considered if they corresponded with elevated rainfall values (within an acceptable lag period discussed later in this report).

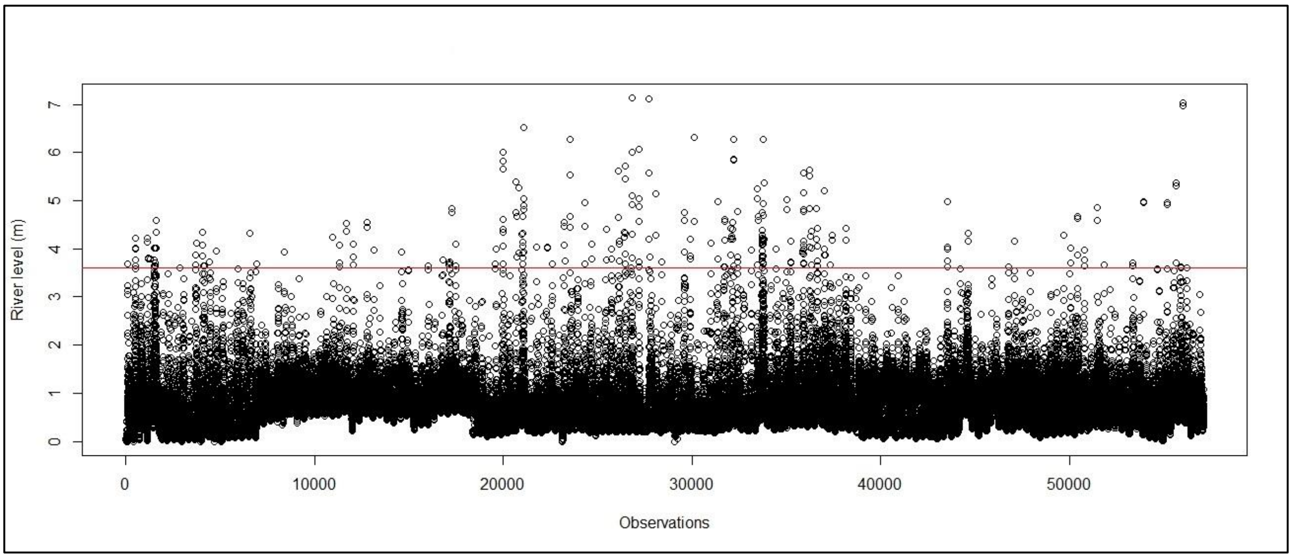


Figure 4.6 Plot of daily river level combined for all stations and years. The red line indicates the 99.5th percentile.

Analysis of the rainfall and river level allowed for certain conclusions to be drawn about the historical hydrological regime of the river. Data were found to be strongly skewed towards low rainfall events and low river levels. These findings were typical for such data, as found by Gericke (2010) in a study making use of rainfall, discharge and river level data, such as was used in this study. Even so, numerous flood magnitude events were recorded for every station across the data period, in addition to the occurrence of high rainfall events having been verified. Such findings served to illustrate that even though the region generally does not experience very severe (large amounts of precipitation within a small period of time) rainfall events frequently, localised flooding was an almost yearly event, which would conceivably influence *E. camaldulensis* populations. As a means of exploring the relationship between rainfall and river level, correlation and lag times were evaluated further.

4.4.3 Correlation and lag times

Lag time refers to the time between peak rainfall and peak runoff (Langbein & Iseri 1983), measured as m^3 per second. Steynor, Hewitson & Tadross (2009) found a lag period of one day was sufficient to relate rainfall input to increased discharge, specifically within the Breede River catchment. However, Lloyd (2010) indicated lag periods could be very large within the region. Lloyd (2010) noted lag periods significantly longer than one day were more feasible, as only the Karroo station was satisfactorily explained by a one-day lag period in his critique of Steynor, Hewitson & Tadross (2009). Additionally, lag periods of four days were observed in a study conducted by Alexander (2002). Employing data from corresponding rainfall stations and weirs within the closest proximity to each other (Table 4.6), a Spearman rank correlation was conducted to evaluate lag periods further.

Table 4.6 Weir and corresponding meteorological station pair, including the data period for which complete data was available and the distance between stations.

Weir	**Meteorological station**	**Corresponding data period**	**Euclidian distance between weir and meteorological station pair (km)**
Karroo	Kwaggaskloof	1980-1992	6.33
Le Chasseur	Kwaggaskloof	2000-2012	21.12
Wolvendrift	Robertson	2000-2010	15.22
Wagenboomsheuvel	Ashton	1980-1991	23.1
Swellendam	Marloth	1980-2000	6.43

Appropriate lag times were established through simple lagged-correlation analysis (Figure 4.7) (Steynor, Hewitson & Tadross 2009). The best suited lag time was regarded as displaying the greatest level of positive correlation between rainfall and river level. It is acknowledged, however, that rainfall is by no means the only factor involved in determining runoff and consequently the water level of any given river. Other factors may readily influence the relationship, for example groundwater seepage, evaporation and riverbank attenuation (Gericke 2010). Lloyd (2010) used discharge data from identical or closely situated weirs (as used in this study) and found that observed flow changes within the Breede River stemmed primarily from increased abstraction, creation of impoundments, and changes in land use along the river. In this study, high precision of lag times was unnecessary; and lagged cross correlation deemed sufficient to obtain useful estimates for the various stations.

Seven consecutive days were used for the meteorological station and weir pairs (results shown in Table 4.6), and lagged cross-correlations tested for each arrangement thereof (shifting pairs by seven days). The seven days were arbitrarily chosen to account for all reasonable expected lag periods such as the four days and one day found, respectively, by other authors (Alexander 2002; Steynor, Hewitson & Tadross 2009). The results are reported in Appendix C, and summarised in Figure 4.7, illustrating the peak correlation between rainfall and river level for the various station-pairs (weir and rainfall station pairings).

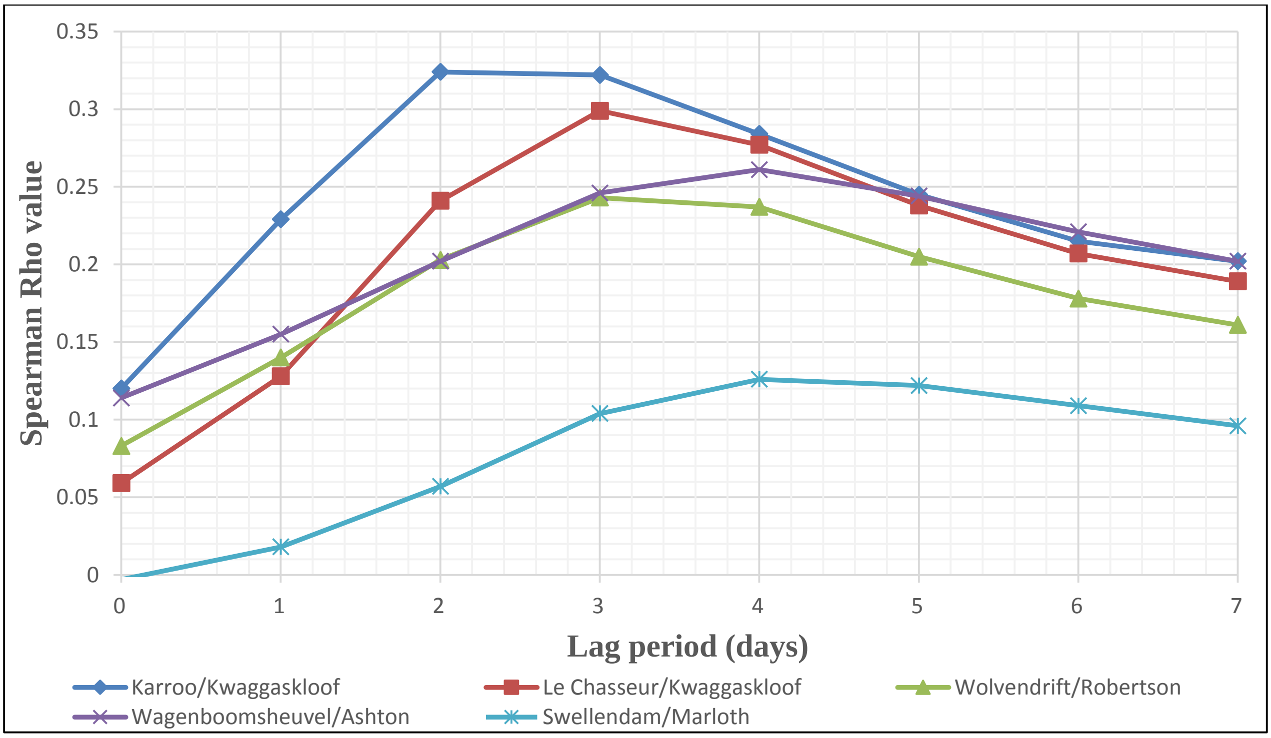


Figure 4.7 Spearman's correlation results for rainfall (mm) and discharge (m^3/s) for all five station pairs used in lag estimation.

It was clear from Table. 4.7, that the correlation between river level and rainfall was weak, yet positively correlated, which indicated there exists a relationship between rainfall input and river discharge, albeit not a strong relationship. The strongest relationship found between rainfall input and discharge was Karoo weir and Kwaggaskloof meteorological station (Spearman rho (4304) = 0.324, $p < 0.001$). As the Karoo weir is upstream from all the other weirs, less cumulative correlation, or 'smearing' of correlation between stations would be evident. This means weirs downstream of Karoo weir, by virtue of being within the same channel, and thus receiving increasingly more water (not differentiated in the analysis), would thus experience a high degree of long term cross-correlation between them and their upstream counterparts.

Stations could therefore not be regarded as independent from each other, and thus the correlation value obtained by the bottom-most weir would include water from the upper-most weir, whereas the uppermost weir would have no such interference. Comparison made between weirs thus implied an increasing measure of influence between the weirs, which would result in a reduced potential for comparative conclusions to be drawn. This cumulative correlation smearing was also noted by Lloyd (2010) in his critique of Steynor, Hewitson & Tadross (2009).

The relative strength of the relationship between Karoo weir and Kwaggaskloof meteorological station, when compared to the values obtained with the other stations downstream, could thus be due to less cumulative upstream influence between rainfall and discharge. This was observed in the optimal lag periods becoming longer as one evaluated weirs in a downstream direction (Table 4.7). The results of the lag correlation analysis were summarised in Table 4.7, and used during patch and island area analysis in the next section.

Table 4.7 Suggested lag periods between rainfall and discharge for all station pairings.

Weir Station	**Meteorological Station**	**Strongest Correlation (Spearman rank rho)**	**Suggested lag period (days)**
Karoo	Kwaggaskloof Dam	Rho (4304) = 0.324, $p < 0.001$	2
Le Chasseur	Kwaggaskloof Dam	Rho (4257) = 0.299, $p < 0.001$	3
Wolvendrift	Robertson	Rho (3914) = 0.243, $p < 0.001$	3
Wagenboomsheuvel	Ashton	Rho (4081) = 0.261, $p < 0.001$	4
Swellendam	Marloth	Rho (3628) = 0.126, $p < 0.001$	4

Lag periods (Table 4.7) were found to be consistent with those of Alexander (2002), in addition to showing similarities in downstream direction (with lag times increasing along a gradient in a downstream direction), as discussed by Lloyd (2010). The results did however allow for an approximation of the relationship between rainfall, discharge and river level, in particular the lag period between input and runoff. One could thus conclude that rainfall within the catchment, at distances of six to 23 km between source and runoff, ranged from two to four days depending on where in the river profile one evaluates this relationship. Despite the clear indication of a cumulative downstream effect, and the inter-relationship between weirs within the same channel, the lag periods obtained from the results were in accordance with other studies conducted, and could be regarded as sufficient for the purposes of this study.

These lag periods had more application in terms of further data analysis than assisting in the understanding of the *E. camaldulensis* distribution along the Breede River, as water input (rainfall) into the upper region of the study area inevitably flowed past the lower region, and thus the expected influence of rainfall and flooding on seed dispersal, soil moisture, vegetation and topsoil removal, was expected to be reasonably uniform between the weirs. No appreciable difference in impact on *E. camaldulensis* was thus expected from the variation in lag periods between the different subsections of the study area. These results allowed for the identification of five flood events of greatest magnitude, for each station pairing, across all the time stamps available and employing the appropriate lag time for each.

4.5 HIGH RAINFALL AND FLOODING EVENTS

Although other factors such as catchment area and shape, slope, drainage density and the antecedent soil moisture conditions within the basin (Gericke 2010) influence the onset of a flood, flooding was generally believed to follow large amounts of precipitation within a short period of time (Barredo 2007). Rainfall volume, and the duration of precipitation, should be sufficient to allow for saturation of the soil for subsequent runoff to occur (Barredo 2007). Once runoff is readily established, catchments may retain additional moisture via the river channels (Barredo 2007). Only when river channels are overtopped and overland flow occurs can an event be deemed a ‘river flood’ (Barredo 2007). Identifying flood events thus required the isolation of runoff instances greater than the river channel dimensions, which in combination with elevated rainfall events, were deemed a flood.

In order to evaluate the change over time of study site area, it was necessary to first scrutinise rainfall and river level data. Lag periods identified in the analyses, were used to elucidate the rainfall and flood events of the greatest magnitude across the study period, employing the appropriate lag period identified (Table 4.7) and the corresponding rainfall station and weir pairing.

Table 4.8 indicates the various flood events and dates that were obtained from the data. Only events after 1987, or 1989 (depending on river location due to the flight path for those years) were chosen, as 1987 was the earliest imagery data was available for, and thus the earliest for which analyses was possible. Due to insufficient data, area analyses for patch and island size was not possible for Wagenboomsheuvel weir.

Table 4.8 High rainfall and flooding events for all stations. Missing or incomplete data points were indicated with asterisks, to illustrate where data irregularities existed.

Weirs - lag time (days); bankfull threshold (m)	**Events**	**Combined Rainfall (mm)**	**Peak river level (m)**	**Spearman Correlation**
Karroo* **(**2; 3.13 m)	1989/09/04 – 1989/09/07	42.9	3.594	**Rho(7) = 0.74; p = 0.052**
	1990/07/10 - 1990/07/13	30.0	3.281	**Rho(7) = 0.77; p = 0.039**
	1991/06/24 – 1991/06/27	61.1	4.330	**Rho(7) = 0.70; p = 0.074**
	1991/08/01 - 1991/08/04	41.0	3.507	Rho(7) = 0.63; p = 0.12
	1992/06/03 - 1992/06/06	42.8	3.685	Rho(7) = 0.30; p = 0.50
Le Chasseur (3; 3.13 m)	1993/07/10-1993/07/13	128.4	4.535	Rho(7) = 0.12; p = 0.78**
	1996/06/16-1996/06/19	147.5	4.541	Rho(7) = -0.65; p = 0.11**
	2005/05/28-2005/05/30	30.9	3.647	Rho(7) = 0.29; P = 0.51
	2008/11/13-2008/11/16	93.5	4.826	**Rho(7) = 0.66; p = 0.09**
	2009/06/25-2009/06/28	63.9	4.091	**Rho(7) = 0.78; p = 0.03**
Wolvendrift (3; 3.13 m)	1992/06/24-1992/06/27	12.4	5.723	Rho(7) = -0.19; p = 0.67***
	1993/07/10-1993/07/13	42.5	7.132	**Rho(7) = 0.28; p = 0.53**
	1996/06/16-1996/06/19	24	7.114	**Rho(7) = 0.25; p = 0.57**
	2006/08/23-2006/08/26	112.4	4.988	**Rho(7) = 0.70; p = 0.07**
	2008/11/13-2008/11/16	144.8	6.268	**Rho(7) = 0.95; p < 0.001**
Wagenboomsheuvel ****(4; 3.27 m)	1977/08/20-1977/08/23	N/A	6.274	N/A
	1976/06/07-1976/06/10	N/A	5.250	N/A
	1990/08/12-1990/08/15	9.9	3.393	**Rho(7) = -0.089; p = 0.84**
	1983/07/16-1983/7/19	3.7	5.581	Rho(7) = 0.57; p = 0.17
	1984/05/17-1984/05/20	46	5.634	**Rho(7) = 0.85; p = 0.014**
Swellendam (4; 3.27 m)	1993/07/11-1993/07/14	53.3	4.673	**Rho(7) = 0.72; p = 0.063**
	1996/06/17-1996/06/20	9.9***	4.863	Rho(7) = 0.20; p = 0.65
	2003/03/24-2003/03/27	164.2	4.995	**Rho(7) = -0.18; p = 0.69**
	2007/11/22-2007/11/25	201.3	5.365	**Rho(7) = 0.90; p = 0.0056**
	2008/11/13-2008/11/16	397.7	7.031	**Rho(7) = 0.84; p = 0.016**

*Weir data available only for 1972-1992. **Insufficient data - nearest possible data arrangement used.
***No clear corresponding rainfall event exists. ****Weir data only available for 1975-1991.

Although five flood dates per station were provided in Table 4.8, flood events were in most cases highly variable in magnitude, with numerous cases recorded per year and per station. As such, five 'flood magnitude' events per station across a timeframe of 32 years were insufficient to account for all such events. Flood magnitude events were defined as high rainfall and river level events corresponding to an elevated river level above the weir threshold height. Flood threshold values were defined as the river level values for any given weir station which exceed the weir design schematic height (i.e. values greater than the weir measuring capacity). These values were identified in chapter three and reported in Table 3.7.

Flood threshold values (i.e. isolated data which represented flood events) for the two different flood threshold values (3.13 m and 3.27 m respectively) were relatively low, with mean = 3.64 m; n = 128 for Karroo and Le Chasseur weirs, as well as mean = 4.15 m; n = 284 for values measured at weirs Wolvendrift, Wagenboomsheuvel and Swellendam. It was clear from the means that flood events were generally only slightly above the exceedance level of the respective weirs, with the greatest mean for flood events being found for Swellendam weir; the lowermost weir in the study area. As flood events were outlier events, and thus less frequent in comparison to the rest of the hydrology data, this conclusion was unsurprising.

Two-sample Kolmogorov-Smirnov tests for the data associated with the two different weir threshold values (i.e. upper and lower sections of the river), displayed results of $D(128) = 0.31$, $p < 0.001$. This result suggests that the two flood threshold values obtained for the upper (mean = 3.64) and lower sections (mean = 4.15) of the river had significantly different distributions, and thus exhibited statistically different flow characteristics from each other.

From this could be concluded that upstream and downstream flow were different in terms of their hydrological regime, which could be a function of the data smearing effect mentioned above, and would thus confound the exact cause of such differences. Another possible influence on the observed differences between the sections, was the additional water input into the river downstream of the confluence.

The data compared above, was a grouping of Karoo, Le Chasseur and Wolvendrift weirs together, and was compared to the grouped data of the Wagenboomsheuvel and Swellendam weir. Even though the Riviersonderend tributary forms part of the Breede River catchment, the separation represented by the Riviersonderend Mountains to the south of the Breede River (Figure 4.8), results in the Riviersonderend water only entering the Breede River at the confluence. As such, the upper three weir stations in the Breede River did not contain the additional water supplied by the Riviersonderend, whereas Swellendam weir did. It could thus be concluded that the significant difference found between the upper three weirs and the lower two weirs was exacerbated by the addition of water from the Riviersonderend at the confluence point, which was not contained in the data for the upper three weirs.

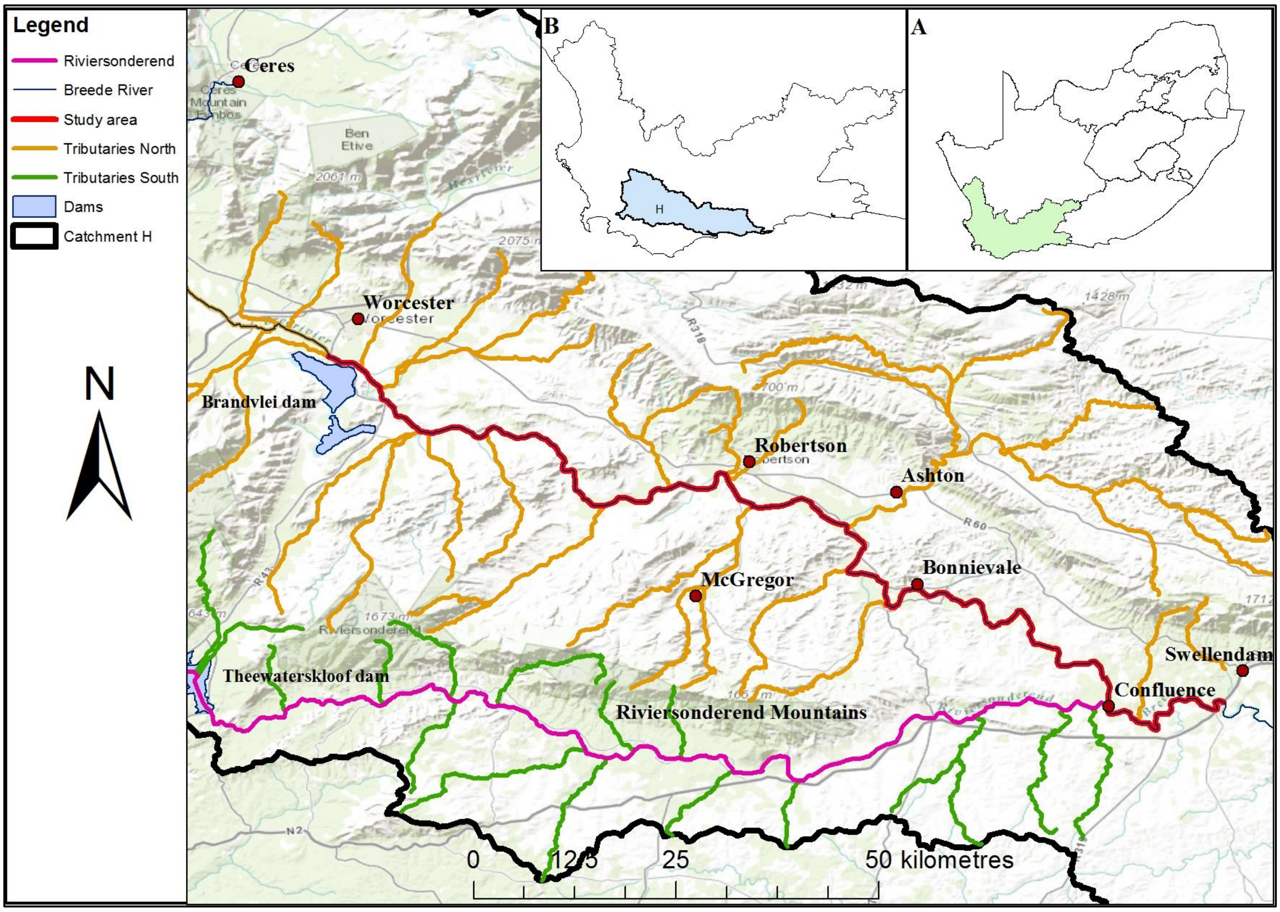


Figure 4.8 North-South tributary divisions of the Breede catchments, showing the confluence point near Swellendam and the Riviersonderend Mountain range as the separation between the two regions. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014).

A further conclusion from the difference found between the two groups was that the influence of the Riviersonderend water at the confluence was sufficient in magnitude to results in the statistical difference found between the two sections. However, other factors, such as the cumulative effect of flow downstream mentioned above, may also have contributed to the statistical results found. It is clear that the influence of the Riviersonderend tributary produced a marked difference in flow regime downstream of the confluence.

Such influence could contribute to further downstream differences (not included in this study), and warrants future research. It was sufficient in the context of this study, however, to note that there existed flow differences between the upper and lower sections of the study area. However, the contribution of water from the Riviersonderend into the lower section of the river was outside of the scope of this study, especially in the light of no significant *E. camaldulensis* sites being found further downstream of Swellendam.

The overarching conclusions that were drawn from this subsection were the identification of high rainfall and flood events (and dates) for each station pairing. Overland flow, and thus localised flooding was verified, and a statistical difference found between the three upstream and two downstream weirs. The difference was then explained through the addition of water from the Riviersonderend tributary, as well as cumulate downstream effects of measuring and comparing flow within the same channel.

4.6 AERIAL PHOTOGRAPHS

In this section, the patch and island sites are analysed in order to evaluate change over time.

4.6.1 Patch sites

A total of 31 patch sites were digitised for the year 2013, shown in Figure 4.9 along with the island site locations. Historical patch sites were digitised as discussed in chapter three, and the area (m^2) values analysed here. Patch site data for each times stamp were tested for normality using the Anderson-Darling normality test (Table 4.9). In all cases, non-normality of the data were found with $p < 0.001$ for all years. These results were visually verified through the use of Q-Q plots and histograms for each time stamp (not included in this report). Area values obtained for the different patch sites are shown in Table 4.10, indicating interval change in area.

Table 4.9 Patch site normality testing.

Year	Anderson-Darling Normality test	Conclusion
1987/89	A (31) = 2.66, $p < 0.001$	Not normal
2003	A (31) = 2.98, $p < 0.001$	Not normal
2007	A (31) = 2.83, $p < 0.001$	Not normal
2010	A (31) = 2.61, $p < 0.001$	Not normal
2013	A (31) = 2.49, $p < 0.001$	Not normal

Table 4.10 indicates the sites with maximum change (both increasing and decreasing), as well as the sites with the minimum amount of change (increasing and decreasing). Site 19 indicated the greatest amount of reduction between 1989 and 2013, reducing from 804 m^2 to 264 m^2. Site 23 decreased by the least amount proportionately (36 736 m^2 – 36 584 m^2). Sites that exhibited overall increase across the years were sites 28 (8 050 – 8 343 m^2) and 24 (665 – 10 669 m^2) respectively.

Of note was the near proximity between site 23 and 24, where one represented the least amount of decrease and the other the greatest amount of increase. These two sites were 22 m apart in 2013, which indicates the fine scale hydrological influence experienced by these sites across the years were similar due to their close proximity. This results suggested the local conditions experienced within the region were conducive to an increase in patch site area, despite the slight decrease in site 23, as the greatest increase was observed, combined with the least decrease, for this particular location.

Another conclusion drawn from the data was that despite the close proximity of the two sites, the fact that one increased and another decreased served to illustrate the fine scale at which hydrological influences could impact on the distribution of any given patch. Variation between sites, regardless of their proximity, could thus be expected to be highly variable in both magnitude and direction. Proximity of patch sites could thus not be regarded as an indicator of predicted magnitude and/or direction of change.

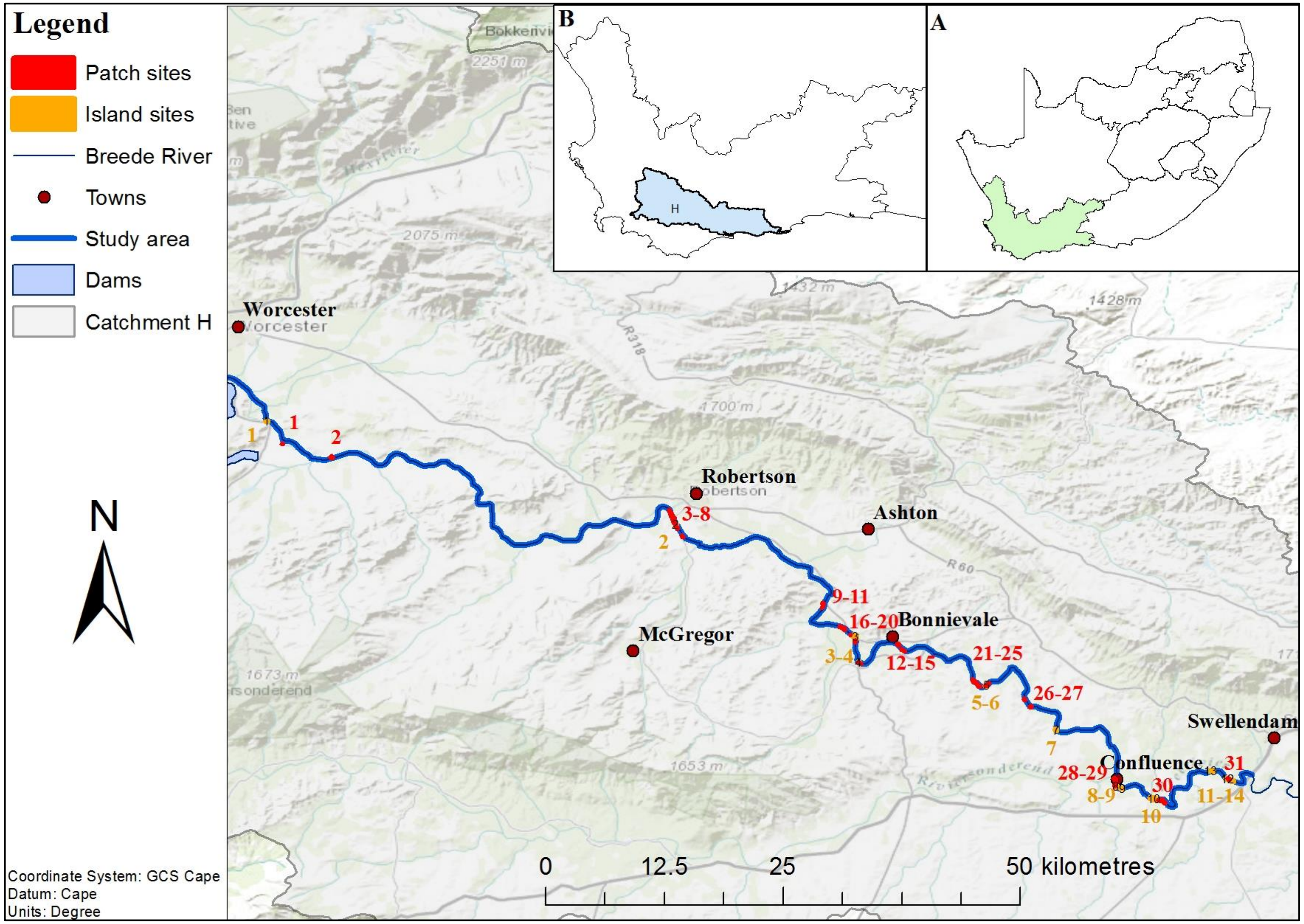


Figure 4.9 Location of patch and island sites within the study area. Inset map (A) depicts the province of interest, and inset map (B) the catchment location within the Western Cape. Topographic base map provided by ESRI (2014) (Red labels: patch sites; orange labels: island sites).

Table 4.10 Patch sites area (m^2) obtained from Field calculator, for all years. Percentage change between years are provided in grey shaded columns. Positive values indicate patch area increase, and negative values indicate decrease in area.

Site ID	1987/89	2003	% Change 1987/89-2003	2007	% Change 2003-2007	2010	% Change 2007-2010	2013	% Change 2010-2013	Overall % Change 1987/89-2013	Overall change (m^2) 1987/89-2013
1	2445.5	3864.6	58.03	4741.9	22.70	4724.9	-0.36	5308.3	12.3	117	2863
2	21158.4	31221.5	47.56	20699.1	-33.70	27422.8	32.48	27195.1	-0.8	29	6037
3	6166.6	11580.2	87.79	10539.3	-8.99	13239.7	25.62	8673.1	-34.5	41	2507
4	45471.2	61877.7	36.08	54582.3	-11.79	50198.1	-8.03	55167.7	9.9	21	9696
5	17320.0	26440.8	52.66	28997.6	9.67	26442.2	-8.81	25365.8	-4.1	46	8046
6	26199.2	19715.7	-24.75	18852.0	-4.38	18216.2	-3.37	18498.0	1.5	-29	-7701
7	6533.3	8335.6	27.59	7075.2	-15.12	6179.9	-12.65	10509.2	70.1	61	3976
8	8310.4	6202.0	-25.37	5147.7	-17.00	5162.1	0.28	5435.4	5.3	-35	-2875
9	2988.9	1624.9	-45.64	1618.7	-0.38	2352.1	45.31	1235.3	-47.5	-59	-1754
10	1761.9	3188.2	80.95	3455.7	8.39	3672.8	6.28	4858.0	32.3	176	3096
11	5665.8	2884.5	-49.09	4660.7	61.58	2753.5	-40.92	2880.6	4.6	-49	-2785
12	3383.4	2851.3	-15.73	2384.2	-16.38	3634.4	52.44	7746.1	113.1	129	4363
13	5723.7	5832.3	1.90	2043.4	-64.96	2590.7	26.78	4445.4	71.6	-22	-1278
14	1538.5	1518.1	-1.33	596.3	-60.72	1367.2	129.28	2950.8	115.8	92	1412

Continued overleaf.

Table 4.10 continued.

Site ID	1987/89	2003	% Change 1987/89-2003	2007	% Change 2003-2007	2010	% Change 2007-2010	2013	% Change 2010-2013	Overall % Change 1987/89-2013	Overall change (m^2) 1987/89-2013
15	4174.6	2447.3	-41.38	4824.9	97.15	8189.0	69.72	12929.9	57.9	210	8755
16	3291.0	2321.4	-29.46	3601.3	55.14	6866.4	90.66	7970.8	16.1	142	4680
17	835.0	2012.2	140.97	668.7	-66.77	1468.3	119.59	1023.3	-30.3	23	188
18	3704.1	4089.9	10.42	4134.6	1.09	4078.4	-1.36	8243.0	102.1	123	4539
19	**804.4**	**226.4**	**-71.86**	**378.4**	**67.19**	**280.3**	**-25.93**	**264.6**	**-5.6**	**-67**	**-540**
20	307.6	280.5	-8.82	108.3	-61.39	16100.9	14768.54	169.5	-98.9	-45	-138
21	16450.6	13002.5	-20.96	16888.6	29.89	530.1	-96.86	29376.2	5442.0	79	12926
22	8437.5	6528.9	-22.62	8282.2	26.85	29538.9	256.66	11804.7	-60.0	40	3367
23	**36736.9**	**38941.1**	**6.00**	**43600.1**	**11.96**	**523.7**	**-98.80**	**36584.5**	**6886.2**	**-0.4**	**-152**
24	**665.5**	**128.8**	**-80.65**	**148.0**	**14.90**	**975.4**	**559.16**	**10669.9**	**993.9**	**1503**	**10004**
25	439.9	335.6	-23.71	788.7	135.02	18950.2	2302.61	4944.0	-73.9	1024	4504
26	22398.5	14386.9	-35.77	18175.9	26.34	10314.1	-43.25	19265.8	86.8	-14	-3133
27	8964.2	10577.6	18.00	10773.7	1.85	8342.6	-22.56	10718.0	28.5	20	1754
28	**8050.4**	**5633.2**	**-30.03**	**3607.5**	**-35.96**	**43107.9**	**1094.96**	**8343.5**	**-80.6**	**4**	**293**
29	39489.2	39605.1	0.29	39843.4	0.60	39361.8	-1.21	49637.5	26.1	26	10148
30	33074.4	33276.3	0.61	33148.5	-0.38	2689.6	-91.89	51681.3	1821.5	56	18607
31	1373.9	2274.5	65.55	2415.9	6.22	2689.6	11.33	5115.9	90.2	272	3742

It was for this reason, and the lack of sufficiently fine-scale data on the large number of factors that influence the spatial extent of *E. camaldulensis*, that time stamps were grouped for comparison.

Results further indicated that not all sites exhibited growth, with nine out of the 31 sites, representing 29% of the total sites, displaying a reduction in invaded area between the years 1987/98 and 2013. Furthermore, sites showed a relatively small increase in invaded area (mean = 5704.6 m^2; n = 22) between year 1987/89 and 2013. For sites that displayed a decrease in size, changes were even smaller (mean = 2261.8 m^2; n = 9). A Wilcoxon rank sum test showed a significant difference across the total timeframe of study between sites that exhibited a decrease and sites displaying an increase (W (31) = 158, p = 0.008), i.e. sites that cumulatively grew across time or decreased in size across time.

Additionally, certain patch sites exhibited greater growth (in area) than other sites, for example, sites 24 & 25, in comparison to sites 21 & 22 (Table 4.10). This was potentially due to a number of factors, including a greater total seed production and thus greater propagule pressure from larger sites, to different variations in immediate surroundings experienced by the different sites (Lockwood, Cassey & Blackburn 2005). Despite the greater observed increase in area between these sites, it was the initially smaller sites (24 & 25) that displayed a greater percentage increase when compared to the larger two sites (21 & 22). These results suggested that larger sites had a greater impact in terms of overall area increase, but that smaller sites exhibited faster increase comparatively.

A possible explanation for the disparity in 'growth' speed of the sites could be that smaller sites, being characterised by a lesser density and lower mean stand height, could be subject to less intraspecific competition. It was possible that *E. camaldulensis* competition interactions with its neighbouring vegetation could have been such that the seedlings and young classes were able to compete better for resources with other species, than compared to other *E. camaldulensis* of greater height. As such, stands where the dominant age class was seedling or young, would be able to increase in area at a faster rate due to a greater competitive ability towards other species, whereas stands dominated by mature or very mature age classes would exert prohibitive competition pressure on newly established seedlings or young individuals of *E. camaldulensis*, thus reducing the potential for growth in terms of area of a given patch.

Figure 4.10 indicates the general change of combined patches for each time stamp, illustrating how all patches varied across time. The Wilcoxon signed rank test was used to elucidate differences of the grouped data (time stamps). Results indicated that year 1987/89 (mean = 11092.4 m^2; n = 31), differed significantly (V (31) = 85, $p < 0.001$) to year 2013 (mean = 14484.2 m^2; n = 31). Seen in Figure 4.10, a generally small change in patch site area was observed between 1987/89 and 2010 (range = 34 386 – 36 196 m^2), where after a sharp increase in area was observed.

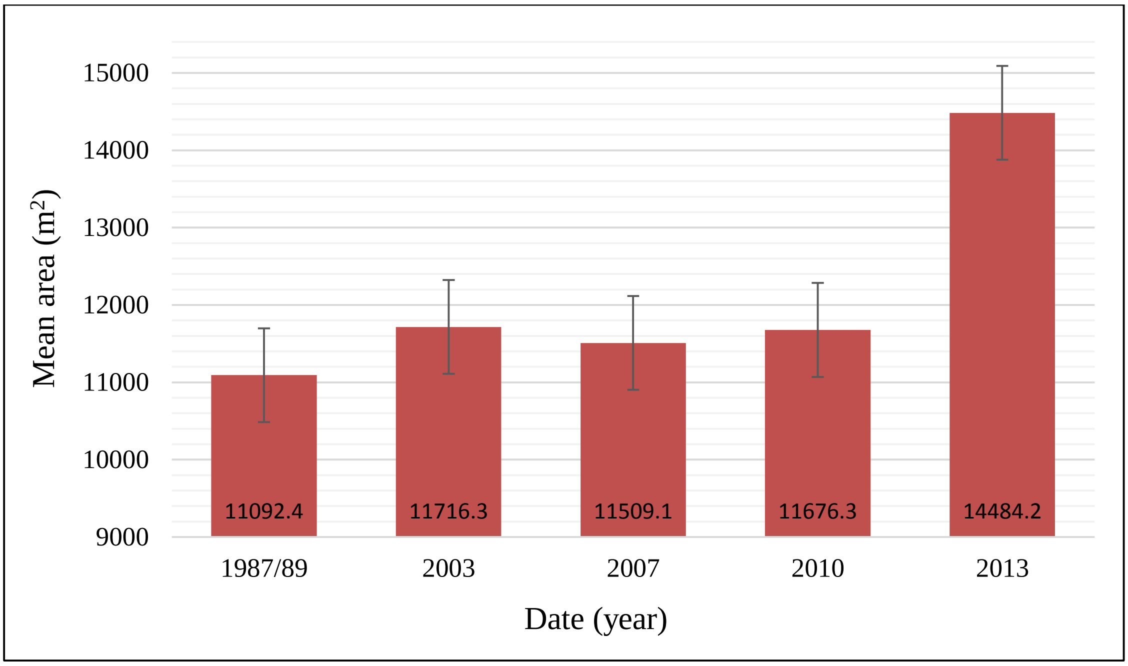


Figure 4.10 Patch site mean area (m^2) with standard error bars, indicating mean area change across time stamps.

The sharp increase observed could potentially have been influenced by changes in land use along the riparian zone, reduced clearing frequency along the river, or potentially by alteration of the mean river level due to increased or decreased water abstraction. Flooding may also have impacted on the change observed over that period, as a flood of considerable magnitude was observed in the river level data during November 2008.

During the 2008 flood event, which lasted three days, Le Chasseur weir experienced 93.5 mm rainfall and river levels of 4.82 m. Wolvendrift and Swellendam weirs also exhibited elevated rainfall and river level values of 144.8 mm, 6.26 m, 397.7 mm, and 7.03 m respectively. This flood event was named the 'Cape Winelands Flood' and caused R631.9 million in agricultural loss and damage to roads (Holloway et al. 2010). Water levels had risen to its highest level since 1806 in Swellendam, with water levels rising at a rate of 30 cm per 30 minutes during the flood (Holloway et al. 2010).

The severity of this event, and the resultant increase in river level and overland flow, could have potentially resulted in mass deposition and seeding over large areas of the riparian zone. *E. camaldulensis* stands would only be visible on aerial imagery to the naked eye once the mean stand size had increases sufficiently to be distinguished from neighbouring vegetation. As such, the 2010 patch sites may not have reflected the seedlings present for any given patch sites, and the site boundaries may thus have been under-reported. Contrastingly, field visits during 2013, would have provided accurate data on the precise extent of patch sites, as the effect of the previous flood would have been incorporated by the data for 2013.

The sharp increase could thus have been initiated during the Cape Winelands flood, with the precise extent of *E. camaldulensis* having been detrimentally influenced by the limited ability to distinguish seedlings from aerial imagery. An increase in 2013 could potentially have been the residual mass germination and establishment from the 2008 flood event, being observed in the data only at the time stamp of 2013. Despite the potential for this effect to have been the dominant factor in the observed patch site area of 2013, the exact cause remains unclear from the available data.

In addition to the statistical significance of the sharp increase of year 2013, differences in median between the others years were sought as well. A common non-parametric test applied to establish such differences is the Friedman rank test. The results indicated a significant difference across the groups with $\chi^2 = 114.67$, $p < 0.001$ (30, N = 155). The differences between years were compared for each pairing using the Wilcoxon signed rank test, with the results shown in Table 4.11.

Table 4.11 Wilcoxon signed rank pairwise comparisons for patch site area. Significant differences are marked with an asterisk (*) and bolding of the text.

Year	Comparison year	Test statistic (V)	Probability value (p)	Conclusion
1987/89	2003	245	p = 0.961	No significant difference
	2007	235	p = 0.809	No significant difference
	2010	210	p = 0.467	No significant difference
	2013	**85**	**p < 0.001***	**Significant difference**
2003	2007	233	p = 0.779	No significant difference
	2010	259	p = 0.839	No significant difference
	2013	**127**	**p = 0.016***	**Significant difference**
2007	2010	222	p = 0.621	No significant difference
	2013	**86**	**p < 0.001***	**Significant difference**
2010	2013	154	p = 0.066	No significant difference

The results obtained for the Wilcoxon signed rank test in Table 4.11 were required to be tested further using post-hoc tests, in order to account for the inflation of the alpha level when conducting multiple comparison tests (Abdi 2010). The Holm–Bonferroni post-hoc test was applied to the probability results of the pairwise comparisons. Table 4.12 illustrates the different ranked probability values for the different pairwise comparisons, and indicates which results were statistically significant.

Table 4.12 Holm–Bonferroni post-hoc test results for patch site area, indicating significant differences after correction was made for statistical multiplicity.

Year 1	Year 2	Probability	Holm-Bonferroni Critical p-value	Holm-Bonferroni Result
1987	**2013**	**0.0009***	**0.0050**	**Significant difference**
2007	**2013**	**0.0009***	**0.0056**	**Significant difference**
2003	2013	0.0166	0.0063	No significant difference
2010	2013	0.0664	0.0071	No significant difference
1987	2010	0.4678	0.0083	No significant difference
2007	2010	0.6219	0.0050	No significant difference
2003	2007	0.7793	0.0056	No significant difference
1987	2007	0.8092	0.0063	No significant difference
2003	2010	0.8393	0.0071	No significant difference
1987	2003	0.9615	0.0083	No significant difference

The Holm–Bonferroni post-hoc test requires cessation of significance evaluation after the first reported non-significant result in the stepwise process. Significance noted here after the 2003-2013 comparison is for illustrative purposes only.

Noticeable from Table 4.12 is the significantly different results obtained between years 1987/89 and 2013, and 2007 and 2013, respectively. Interestingly, the year 2010 and 2013 do not differ significantly, despite a large increase overall between the years.

The statistical difference obtained between these years could potentially have been due to the large time period between the data point 1987/89 and 2013, during which patch site area would have changed significantly. It was thus not unexpected to find differences in patch site area across such a long time frame, as river ecosystems are dynamic systems, and therefore constantly changing. The difference noted between years 2007 and 2013 could also potentially have been attributed to the occurrence of the Cape Wine lands flood, in addition to the November 2007 flood within the Breede River (Holloway et al. 2010). These events may have initiated large scale flooding, seeding, germination and disturbance, which would have contributed to *E. camaldulensis* patch size expansion within the region.

Patch size results served to illustrate a general increase in area across patches over time. The occurrence of *E. camaldulensis* has thus not been greater over the last 30 years, than it is currently. Even so, the increase was expected to be greater in magnitude. Even though mean stand area for the years 1987/98 was 0.011 km^2, an overall change of only 0.1 km^2 (net total) was observed cumulatively up to the year 2013. A potential reason for the less than expected increase in *E. camaldulensis* extent was the ongoing maintenance of agricultural land adjacent to the sites.

As the dominant land use within the study area is viticulture, for which gains can be made by increasing the area under cultivation, maximum proportions of available farmland would have been utilised for agriculture, with short-term maintenance of the cultivated lands ongoing in most cases, as was indicated during interviews. As such, established seedlings may have been removed if they were found within cultivated land, and thus area expansion could conceivably have been restricted to only fringe areas that were not maintained regularly. One respondent in particular indicated that after flood events, labourers were specifically requested to remove newly established seedlings (of specifically *E. camaldulensis*) from the vineyard regions of the property, thus validating maintenance as one possible explanation for the small change in area of patch sites.

4.6.2 Island sites

The motivation for the studying of island sites was that IAPS, specifically *E. camaldulensis* and *Acacia saligna* were according to interview respondents, contributing to island formation within the river channel, thus changing the river morphology and influencing the potential for future flooding. This was done primarily as a reaction to various interview responses, which indicated a general consensus that invasive non-native vegetation, was directly or indirectly influencing the river morphology. An evaluation was thus conducted to investigate the presence or absence of any specific change across time.

The locations of the island sites are indicated in Figure 4.9 (in the previous section). Table 4.13 indicates the results of the island site data distribution tests against that of the normal distribution, making use the Shapiro-Wilk normality test, which is appropriate when small sample sizes are used (Razali & Wah 2011). All data were not normal, with the Shapiro-Wilk normality test (S-W) results indicating probability values below the alpha threshold of 0.05 for all time stamps.

Table 4.13 Island site normality testing.

Year	**Shapiro-Wilk normality test**	**Conclusion**
1987/89	W (14) = 0.87, $p = 0.04$	Not normal
2003	W (14) = 0.60, $p < 0.001$	Not normal
2007	W (14) = 0.65, $p < 0.001$	Not normal
2010	W (14) = 0.58, $p < 0.001$	Not normal

Table 4.14 below provides the raw data for the island sites across all years, indicating interval change in percentage. The island sites with greatest and least increases, as well as greatest and least decreases, were noted by bolding of the text.

A very small combined change of 0.028 km^2 increase was found between time stamps 1987/89 and 2010, which, within the context of the mean island size of 0.014 km^2, was regarded as minimal. The results concur with responses received from interview data, as the island area increase was indicated as being small, since respondents stated little to no change in island formation following flooding (and due to the presence of *E. camaldulensis*).

Table 4.14 Island sites area (m^2) obtained from Field calculator, for all years. Percentage change between years are provided in grey shaded columns. Positive values indicate island site area increase, and negative values indicate decrease in area.

Site ID	1987/89	2003	% Change 1987/98-2003	2007	% Change 2003-2007	2010	% Change 2007-2010	Overall Change % 1987/89-2010	Overall change (m^2) 1987/98-2010
1	**6702.8**	**7089.8**	**6**	**4866.1**	**-31**	**4680.0**	**-4**	**-30**	**-2022.8**
2	7402.8	5500.7	-26	6456.3	17	2467.8	-62	-67	-4935.0
3	21660.3	67457.5	211	34002.2	-50	35335.6	4	63	13675.3
4	**0.3**	**0.5**	**80**	**1.6**	**226**	**1020.7**	**63296**	**371499**	**1020.5**
5	7.5	550.1	7252	481.3	-12	702.1	46	9283	694.6
6	5.0	1206.7	23838	94.6	-92	136.8	45	2614	131.8
7	3764.5	10252.5	172	9557.1	-7	8601.0	-10	128	4836.5
8	4.5	1659.5	36933	2838.8	71	3663.2	29	81647	3658.7
9	**9993.8**	**7284.2**	**-27**	**4381.9**	**-40**	**2834.3**	**-35**	**-72**	**-7159.5**
10	15755.6	59353.9	277	52959.4	-11	55000.0	4	249	39244.4
11	12209.0	12650.7	4	10869.6	-14	2703.1	-75	-78	-9505.9
12	19029.3	7254.9	-62	7272.5	0	6863.9	-6	-64	-12165.4
13	**2741.5**	**9277.9**	**238**	**4248.4**	**-54**	**4273.8**	**1**	**56**	**1532.3**
14	1507.6	2878.9	91	819.9	-72	719.5	-12	-52	-788.1

Furthermore, island site establishment and growth was indicated by respondents as having being initiated by not just *E. camaldulensis*, but rather any vegetation within the river course. As such, concluding that *E. camaldulensis* has a direct role in the formation of island sites may be erroneous, with the nearest accurate approximation being that *E. camaldulensis* may contribute to island formation, but not necessarily to a greater extent than any other vegetation found within the river channel.

Changes could possibly not have been as a direct result of IAP within the channel, but rather due to the presence of tributaries to the main Breede River channel. The input of sediment from the tributaries could have increased sediment load within the main Breede River channel, allowing for greater island formation and greater island area. However, the results indicated that islands did not increase by large amounts over the study period, suggesting that the historical and current trend, although being one of deposition and overall increase of island area, did not have a large effect on island formation and alien presence. Such a conclusion from the results were contrary to the expected effect, as the presence of numerous flooding events within the river was expected to produce large island area changes due to scouring of existing islands and increasing downstream sediment load. Sediment transported downstream would settle once flooding ceased and flow velocities lowered, conceivably resulting in an increase in island area.

Table 4.14 shows that not all sites exhibited growth, with six out of the potential 14 sites, representing 42.8% of the total sites, displaying a reduction in area between the years 1987 and 2010. Notable change across the 14 sites was that of numbers one, four, nine and 13. Sites four and 13 represented the greatest increase and least increase respectively, with sites one and nine representing least decrease and greatest decrease of area. In the case of the sharp increase in size of site four (0.3 m^2 in 1987/89 and 1020.7 m^2 in 2010), the increase could be attributed to the resulting deposition from a strong flood occurring in the Boesmans River (indicated from interview response). Site four was located at the confluence of the Boesmans and Breede River, and as such the deposition of sediment and change in river bank structure following the flood event may have deposited large amounts of sediment at the confluence point, resulting in the sharp increase in area.

Site 9, which represented the site with the greatest decrease over time, was located in the lower section of the river, in close proximity and downstream, of the confluence of the Riviersonderend and Breede Rivers. The large decrease in area could thus potentially be due to the addition of water from the Riviersonderend River, which would have possibly increased the flower velocity near the confluence. Faster flow velocity would thus potentially increase the amount of sediment uptake within the channel, and result in scouring of island sites present within that region, further resulting in a reduction in site area.

For all sites, the change across time was generally erratic, as a sharp increase in area between 1987/89 and 2003 was observed (Figure 4.11), with a seemingly steady decline thereafter. Even so, the overall change was that of increase between years 1987/89 and 2010. The difference between 1987/89 and 2010 was statistically tested using the paired Wilcoxon signed rank tests (V (14) = 51, p = 0.95) indicating no significant difference in median between year 1987/89 (mean = 7 198.89 m^2) and 2010 (mean = 9 214.41 m^2). Observed in Figure 4.11, however, was a sharp increase for year 2003, necessitating further analysis.

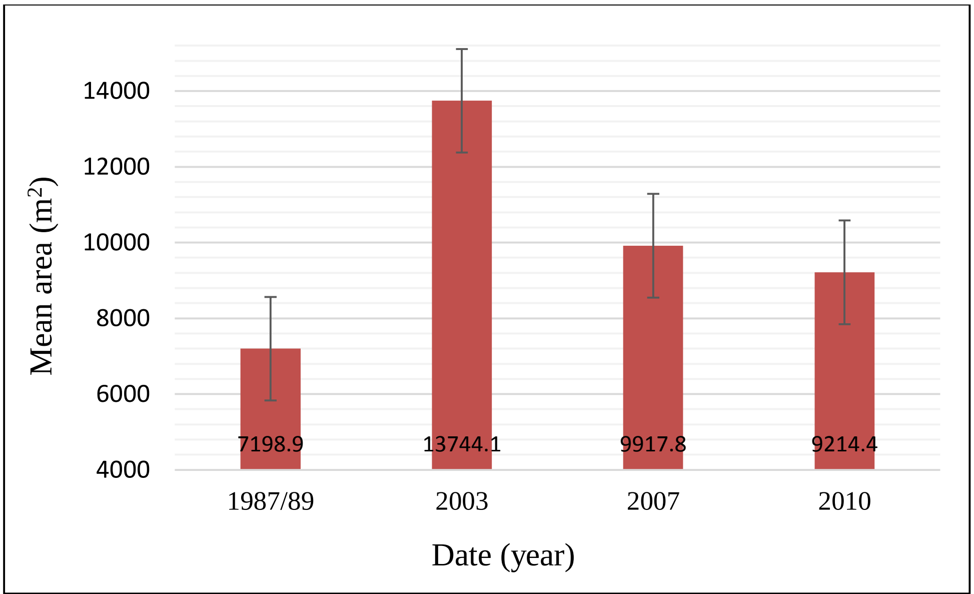


Figure4.11 Island site mean area (m^2) with standard error bars, indicating mean area change across time stamps.

The increase in island site area for the year 2003 could potentially have been a consequence of the relatively stable hydrological regime of the river between years 1997 and 2003, where few flood events of great magnitude were observed. The fewer occurrences of less severe flood events may have allowed for greater deposition of sediment during that time period, as less sediment could have been washed out of the system by low impact floods, allowing for greater island area increase. The 'Montagu Flood' which occurred in March 2003, however, may have washed large amounts of sediment downstream, which combined with more severe floods that occurred during 2007 and 2008 (Holloway et al. 2010), could have resulted in the decrease in island area observed for year 2010.

A Friedman rank sum test was used to elucidate whether there were any significant differences between islands across the years. The test results indicated a significant difference across the groups as $\chi^2 = 44.74$, $p < 0.001$ (13, N = 56).

Statistical testing applied to small sample sizes (such as this), and the conclusions one can draw therefrom, remains controversial within the scientific community (De Winter 2013). A significant difference observed for such tests, involving small samples sizes, may still be valid in certain cases however (De Winter 2013). However, since the significant results obtained indicates that the magnitude of the change observed (effect size), was sufficiently large, despite the small sample size, and resulting loss of power therefrom (Suresh & Chandrashekara 2012), the test was still able to find differences (Harvey 2014, Pers com). De Winter (2013) states that type II errors (false positive) are avoided only when the effect size is extremely large.

The small sample size used in this test is a result of unavoidable limitations inherent in the selection of the island sites, thus a large sample size was not possible. Additionally, due to the exploratory nature of this investigation (island site area change over time as a means only to evaluate whether any particular effect may be present), combined with the effect-size concern mentioned, the application of the difference between groups testing as conducted here were regarded as sufficient. Such results do, however, need to be viewed in light of the small sample size used, and inference beyond the limited capacity of the results of this study is not recommended.

Although the Friedman rank sum test indicated a significant difference between the years, a pairwise comparison (Table 4.15) was conducted to indicate where the difference were.

Table 4.15 Wilcoxon signed rank pairwise comparisons for island sites. Significant differences are marked with an asterisk (*) and bolding of the text. Results of $p < 0.05$ indicate rejection of the H_0 of similar medians between samples.

Year	Comparison year	Test statistic (V)	Probability value (p)	Conclusion
1987/89	2003	29	p = 0.15	No significant difference
	2007	45	p = 0.66	No significant difference
	2010	51	p = 0.95	No significant difference
2003	**2007**	**90**	**p = 0.016***	**Significant difference**
	2010	**95**	**p = 0.016***	**Significant difference**
2007	2010	59	p = 0.71	No significant difference

Results from Table 4.15 indicate significantly different medians between years 2003 and 2007, as well as 2003 and 2010, respectively. Post-hoc testing of these results using the Holm–Bonferroni post-hoc test was conducted to account for inflation of the alpha level when conducting multiple comparison tests (Abdi 2010). The results are shown in Table 4.16, indicating the significant differences found above, between years 2003 and 2007, as well as 2003 and 2010, were not statistically significant once the correction was applied.

Table 4.16 Holm–Bonferroni post-hoc test results for island sites, indicating significant differences after correction was made for statistical multiplicity.

Year 1	Year 2	Probability	Holm-Bonferroni Critical p-value	Holm-Bonferroni Result
2003	2007	0.016	0.0083	No significant difference
2003	2010	0.016	0.0100	No significant difference
1987/89	2003	0.15	0.0125	No significant difference
1987/89	2007	0.66	0.0167	No significant difference
2007	2010	0.71	0.0250	No significant difference
1987/89	2010	0.95	0.0500	No significant difference

Table 4.16 results indicate that there was no statistically significant differences observed between the years. These findings concur with the result of no significant difference observed between years

1987/89 and 2010 using paired Wilcoxon signed rank test. Despite the sharp increase in area for the year 2003, island site area stayed relatively stable over the years, with only a small increase observed cumulatively. The results of Table 4.16 thus support the observations of interview respondents, and the change over time of island area can thus be regarded as negligible. A conclusion of significant change in island area across the time period of this study was thus not warranted from the data.

Such results suggest that despite the observed peak in area for the year 2003, islands sites did not change significantly between 1987/89 and 2010. Therefore, one can only conclude that the long term outcome of island changes within the river, was not significant, and that if historical trends continue, a small but negligible increase in island area could be expected. Such a lack of apparent relationship was surprising, as the expected influence of flooding and elevated river levels would logically influence island area by scouring and deposition further downstream, resulting in a net decrease in area across the years. Scouring along tributaries (outside of the study area) may also have attributed to the apparent lack of relationship by deposition into the study area.

Furthermore, the results for the island sites suggest that *E. camaldulensis* influence on island site formation and increase, was not unique, in that other species may readily have contributed to the process. Island site formation and increase are thus not exclusively related to the presence of *E. camaldulensis* along the riparian zone.

4.7 RIVER LEVEL AND STUDY SITES COMPARISON

4.7.1 Patch sites

An interesting question, if not the overarching one of this study, is whether an influence can be observed between higher river levels and an increase or decrease in size of patch and island sites. One method of better understanding this relationship is the visual representation and interpretation of the data. To that end, graphs are provided (Figures 4.12-14) illustrating the three weir sites and the corresponding river level. Correlation between the rainfall and river level have been detailed earlier in this chapter and are not explained further.

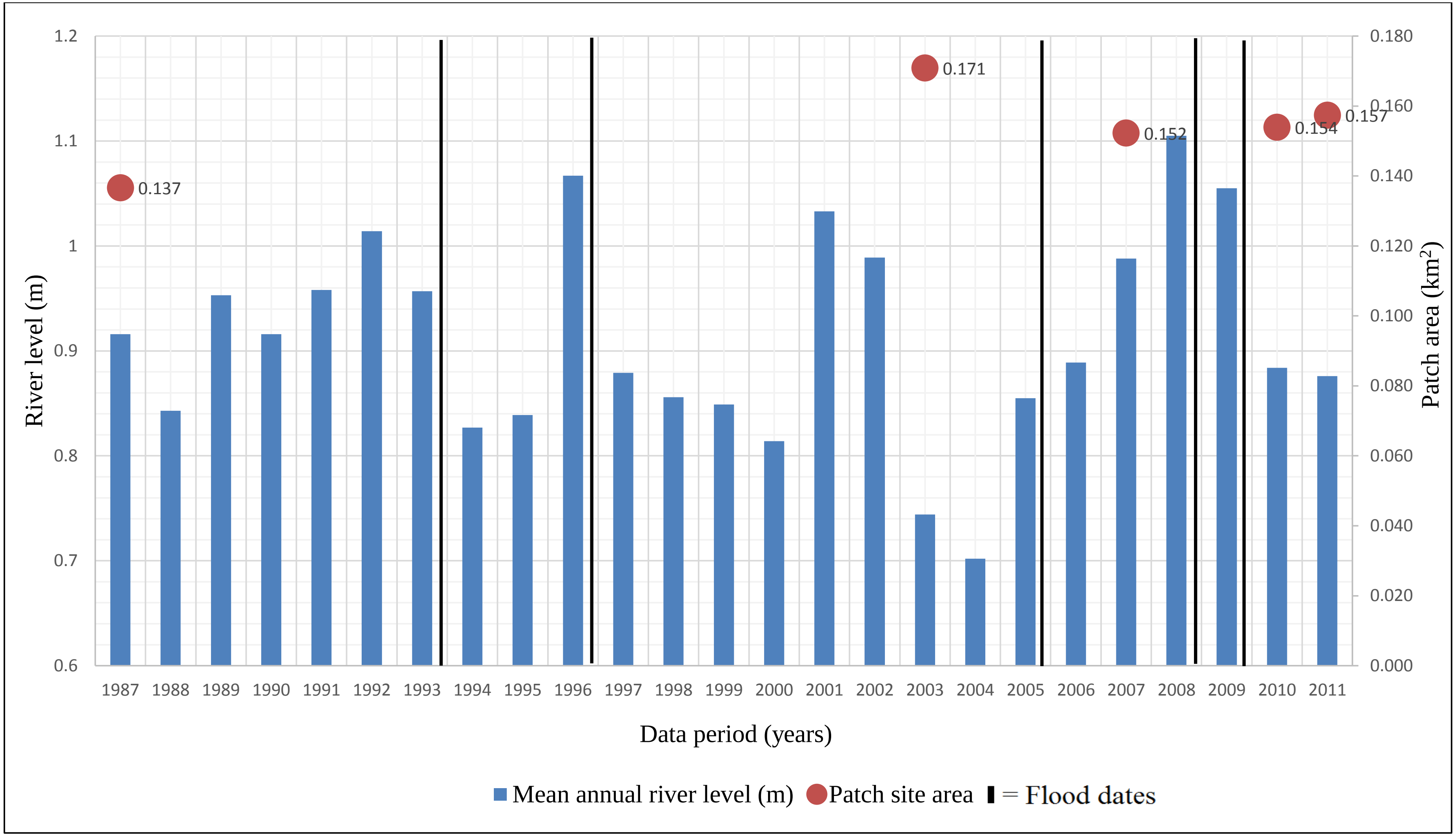


Figure 4.12 Le Chasseur weir flood, river level and imagery area comparison for patch sites.

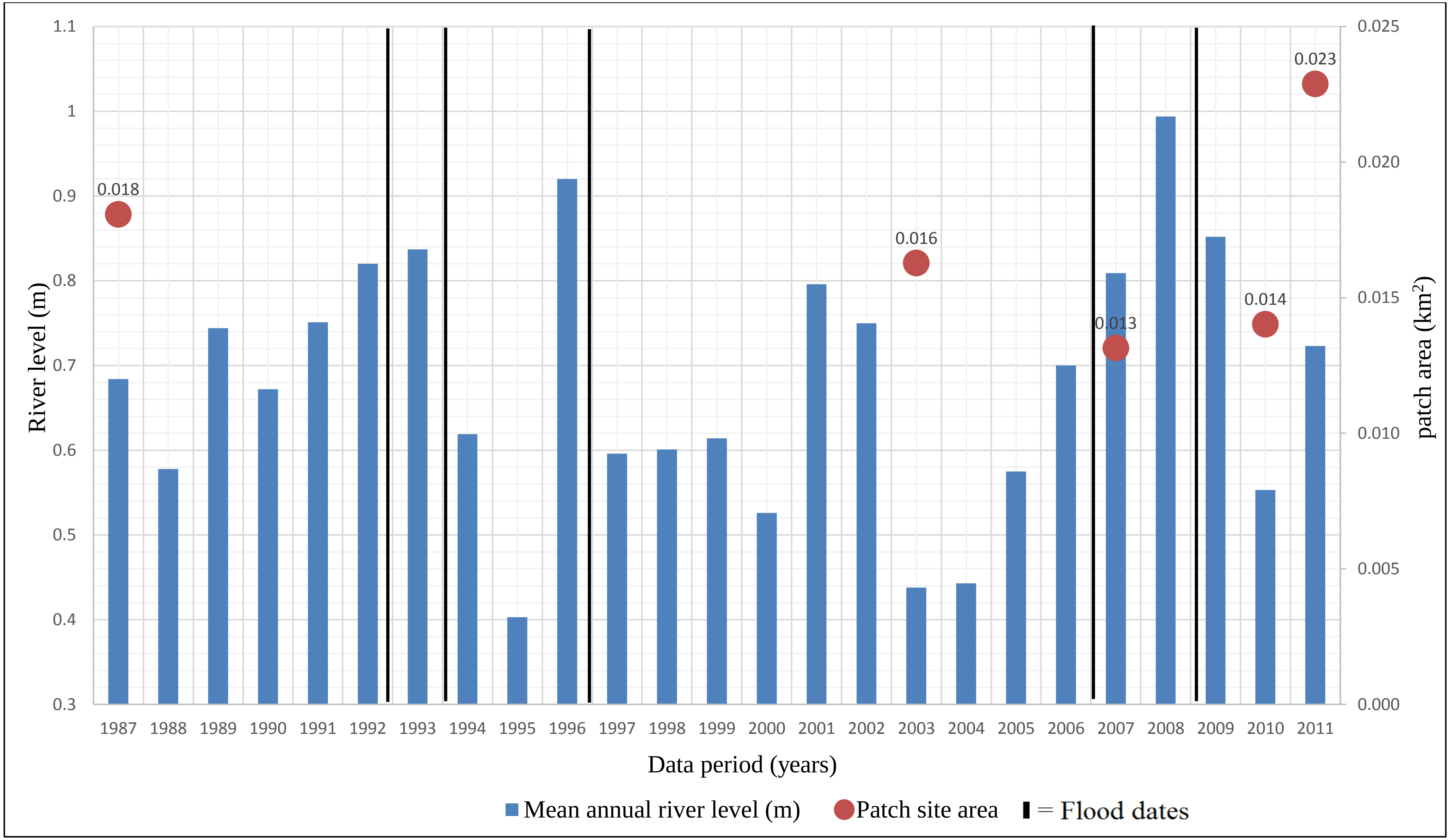


Figure 4.13 Wolvendrift weir flood, river level and imagery area comparison for patch sites.

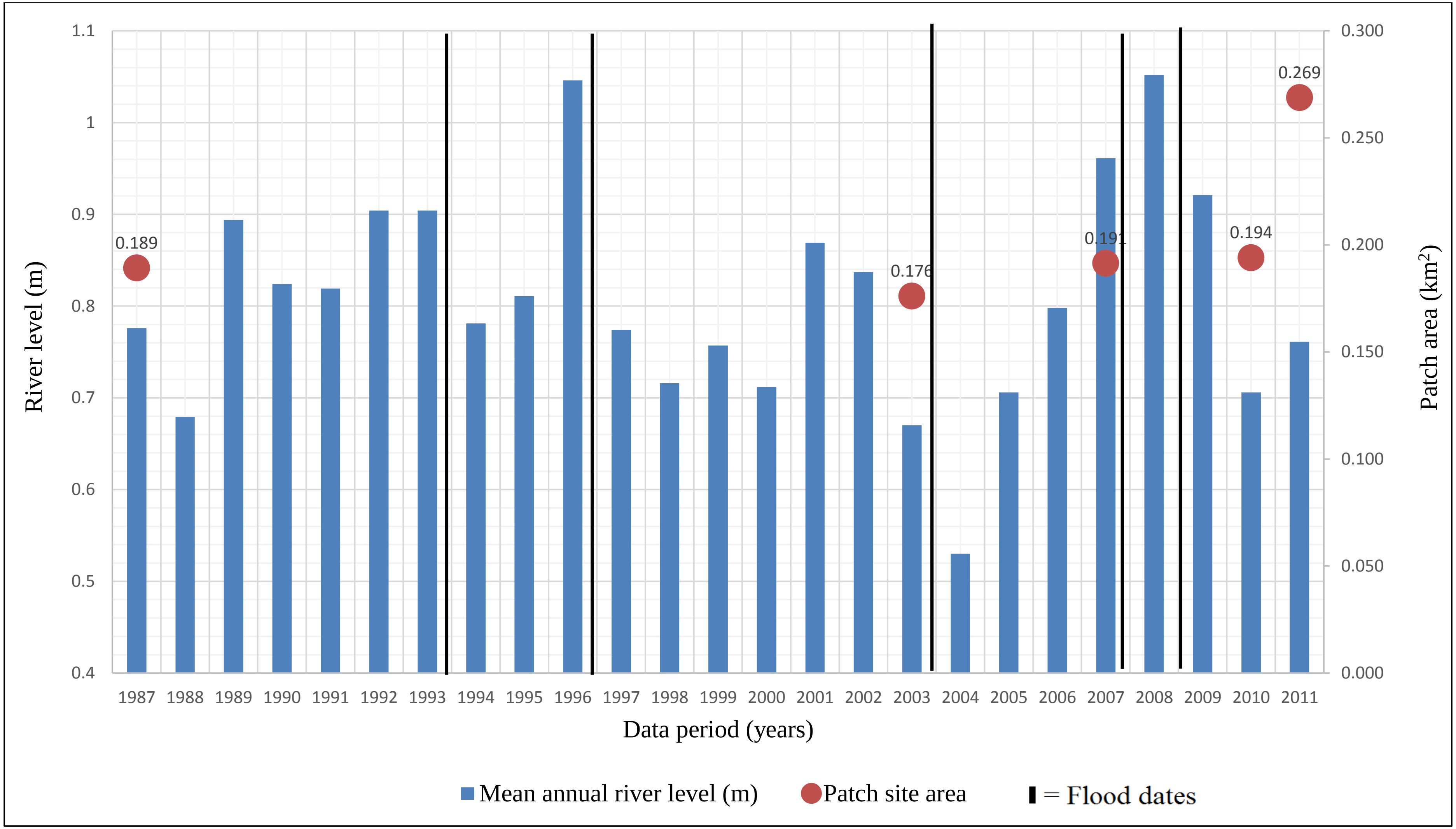


Figure 4.14 Swellendam weir flood, river level and imagery area comparison for patch sites.

Due to the non-availability of river level data for the Karoo weir after 1993, and a similar occurrence for Wagenboomsheuvel weir, no graphs could be meaningfully drawn, since there was only a single corresponding data point for the area of the patch and island sites.

Area was depicted as km^2 in all instances, in order to allow for better visualisation. Flood dates, as identified in in the hydrological analysis, are indicated on the graphs with black vertical lines spanning the full horizontal length of each graph. These are by no means the only floods, but the highest recorded peak river level events for that particular weir site and time span. Area data obtained from fieldwork in 2013 were also added to the last available river level data point (2011), and interpretation should be conducted with that in mind. No variation in interpretation was expected from such a data arrangement. Large orange dots depict the area data for each year. Comparison was made between mean annual river level to the patch area, as well as flood events to patch area.

From the figures above (Figures 4.12-14), the relationship between the patch site area, and higher river levels (which would be indicative of high rainfall or flood events), seemed weak at most. There was little definitive change in behaviour of river level and patch area that coincided meaningfully. Despite this, generalities could be observed from the graphs. In the upper reach of the study area at Le Chasseur (Figure 4.12), a long term increase in patch area was observed with oscillating but generally stable flow conditions between 1987/89 and 2003, with peaks and troughs either preceding or following on each other, and exhibiting relatively similar magnitudes.

The lowering river levels between the years 2002-2004 seemed to coincide with a small decrease in patch sites. Sustained, sharp increase in water levels during 2004-2008 period, and the concomitant patch size increase thereafter, suggested a lag period of two to three years was present between the mean river level and patch area values. Taking into account a lag period of two to three years, the decrease in patch area between 2003 and 2007 were observed to coincide with the reduction of river levels during a longer period between 2001 and 2004. Such a lag period could also explain the small but apparent increase of patch area after the year 2007, as coinciding with the increase in water level during 2004-2008.

A possible explanation for the decrease in patch area coinciding with an increase in river level of Figure 4.12, was the location of the weir within the upper region of the study area. Greater stream velocities are generally expected in the upper regions of rivers (Leopold 1953), which would allow for greater topsoil removal and vegetation disturbance during flood events (Steiger et al. 2005). The larger potential for disturbance by floodwater in the upper region could thus have impacted on the *E. camaldulensis* extent and distribution in an inhibitory manner, by removal of seedlings and young classes before they developed into mature adults (which would be better established and able to survive disturbance events). Despite this possible influence, the large presence of *E. camaldulensis* in the upper region of the river currently suggests that the influence exerted by floodwaters were not sufficient to eradicate *E. camaldulensis* patches, and the continued presence of *E. camaldulensis* within the region is therefore expected.

A possible explanation for the presence of a lag period was the use of aerial imagery for data acquisition. As the boundaries of *E. camaldulensis* stands were only clearly distinguishable from neighbouring vegetation once the *E. camaldulensis* individuals attain a certain height (of roughly 5m), it was plausible that the increase in patch area may only have been observed from aerial imagery a few years after the initial disturbance event, once the newly recruited and established seedling have developed sufficiently, to exhibit greater height than the surrounding vegetation. This would have suggested that the influence of any disturbance event such as a flood, could have coincide with an increase in patch area size, but would have only been observed within the data two or three years later (and only if data were obtained for that specific period).

In Figure 4.12, the presence of flood events, as identified previously in this chapter, and illustrated on the graphs with black vertical lines, seemed to suggest no specific interaction between patch size and flood frequency existed. One could conclude from such an observation that individual flood events, although potentially important in the seeding, topsoil removal, establishment and dispersal of *E. camaldulensis*, did not coincide with drastic patch area increase or decrease when observed across the time period. This assertion is verified in one study where propagule pressure was deemed the greater determinant of invasive extent, as opposed to an erratic flooding regime (Von Holle & Simberloff 2005). It was thus likely that flood events, if an influencing factor on *E. camaldulensis*, did not exhibit their influence at the chosen scale (yearly average of river level), and were thus obscured in their effect due to the coarse data resolution of patch area (five data points over 34 years).

In contrast to the change observed in Figure 4.12, patch area for Wolvendrift decreased with oscillating and fairly stable river levels between 1987/89-2003 (Figure 4.13). Taking into account the two to three year lag period mentioned above, a sharp increase in patch area between 2007 and 2011 coincided with the consistent and sharp increase in river level during the period of 2004-2008. A possible explanation for the observed change was that the channel velocity in the middle section of the study area, may have reached a 'tipping point', whereby vegetation disturbance and vegetation removal was no longer such a strong inhibitory force (as suggested for Figure 1.12), but rather a promoting force, through the addition of soil water and the transport of seed.

Patch areas could thus be seen to increase or decrease coinciding with an increase or decrease in river level respectively, taking into account a lag period of two to three years (Figure 4.13). As such, the hydrological regime of the middle study section coincided with an increase in the extent and distribution of *E. camaldulensis*. Additionally, flood events within the middle study section, were also not observed to coincide in any distinctive manner with either a decrease or increase in river level, possibly due to the coarse spatial data used in this study, as mentioned above.

The relationship between river level and patch area in the Swellendam region (Figure 4.14) exhibited a change similar to Figure 4.13, whereby a decrease in patch area coincided with oscillating but stable river levels between 1987/89 and 2003. Taking into account the two to three year lag period mentioned above, patch area could be seen to increase concomitantly to a period of decrease in river level between 2004 and 2004. It was possible however, due to the decrease and the increase of river levels between 2002 and 2007, that the observed increase in patch area between 2003 and 2007 could have been a cumulative effect (i.e. change in one direction due to river level decrease followed by a change in another direction as the river level increases thereafter). The small patch area increase noted between 2003 and 2007 would thus coincide with the cumulative change in river level of that period.

A sharp increase in river level thereafter (2004-2008) coincided with a strong increase in patch area (2007 and onwards), adjusted by the lag period. The hydrological regime of the Swellendam region thus coincided with a patch area increase within the region if account was taken of the lag period. As such, the relationship between river level and patch area was seemingly similar to that of the middle study section (Figure 4.13), possibly due to lower stream velocities expected for the lower reaches of rivers. Such lower velocities would thus influence *E. camaldulensis* distribution and extent in an additive manner, through the addition of ground water and the transport of seed, with little vegetation disturbance and topsoil removal.

This influence was noted during the interview responses as well, as respondents generally indicated water velocities during flood events to be insufficient to cause vegetation and topsoil removal. Respondents did, however, indicate that standing water following a flood may cause vegetation mortality (specifically in the context of vineyards), where certain vegetation do not tolerate submersion over long periods. The high submersion tolerance of *E. camaldulensis* (Akilan et al. 1997; Roberts & Marston 2011) may thus allow for greater survival of *E. camaldulensis* during extended periods of submersion (often up to a week in duration), and would thus increase patch area as the *E. camaldulensis* would therefore replace much of the vegetation killed during submersion. As such, the observed change in Figure 4.14, suggests an increase in river level coincided with an increase in patch area, and could potentially have been due to the soil moisture addition during flooding events, as well as the competitive advantage obtained from the high submersion tolerance of *E. camaldulensis*.

In contrast to Figure 4.12 and Figure 4.13, flood events appeared to coincide with an increase in patch area in Figure 4.14, specifically the three flood events occurring in short succession between 2003 and 2009. Although not visible from the graph, the increased mean river level values for each year accounted for the flood events as well (as all data for a year were used to compute the mean annual river level). The flood events thus contributed to the sharp increase in river level across those years.

These flood events could thus have accounted for the greater increase in patch area after 2010 observed in Figure 4.14, which was not observed so strongly for Figures 4.12 and Figure 4.13 (even though an increase across those years were observed). One could thus conclude that the presence of frequent flood events of a large magnitude could increase the mean river level of a region, which in turn would coincide with an increase in patch area concomitantly. Despite this assertion, it is unclear from the figures as to what extent flood events may have impacted on the patch site increase, beyond the general influence suggested in the discussion above.

Across all subsections of the study area (Figure 4.12-14), the change in river level and patch area did not appear to be constant between weir sites, however, as they displayed variation in relationship direction dependant on where in the river profile one measured, as well as which time frame one measured across. This would seem to suggest that the relationship between river level of patch size and area increase or decrease was dependent, and in this study actually changed, depending on where one's particular area of interest within the river profile lay.

Intuitively, one would expect the hydrological regime to coincide with similar directions of change for the upper (Figure 4.12) and middle (Figure 4.13) subsection of the study area, as they do not receive flow from the Riviersonderend River, whereas the lowermost weir (4.14) does. However, a relative similarity was observed for all three weirs in terms of mean river level, as the time periods corresponded with the different peaks and troughs noted during the study period. This would suggest that despite the addition of water from the Riviersonderend River, the hydrological regime of the lowermost weir reflects the general pattern from the upstream sections of the river than from the additional water input from the Riviersonderend tributary.

One could thus also conclude that the difference in relationship between river level and patch area observed between the weir sites, coincided less with specific flood events or water addition from tributaries, but more to the location within the river profile of any given weir. It was suggested that this may be due to the different stream velocities experienced by the different weirs within the river. A Spearman rank correlation could have been used to statistically test and quantify the strength or absence, as well as direction (positive or negative) of the relationships. Regrettably, such results could not be obtained, as only five data point pairs were available (for patch sites).

Additionally, any variation observed in river level at any given corresponding aerial image data point, may have be a function of season, daily variation, monthly variation, water abstraction or water input (from upstream dam reservoir) and as such any relationship that might have existed would have incorporated any one, or all of, the influential variables just mentioned. No meaningful conclusion could thus be drawn from such correlation results, and the use of correlation to quantify this relationship was therefore best avoided.

What was clear from the hydrology values was that for all stations river level oscillated in a fairly stable manner between 1987 and 2003, decreasing between 2002 and 2003, where after a steady increase was observed until the year 2008, following which river levels decreased consistently across the various weirs. The Breede Valley experienced a severe flood event during the year 2008 (Holloway et al. 2010), which could possibly account for the peak in values during that year. What remained unclear was whether the pattern observed for patch site area was due to any specific cause, or merely part of the long-term hydrological cycle, related to precipitation and climate within the region.

It was thus unreasonable to assume that individual flood events impacted directly upon the spatial extent of *E. camaldulensis*, but rather it is more probable, to assume that if any relationship existed at all, it would not be defined by any one flood event, but rather by a cluster of floods, or a lack of flooding, which would be reflected in the mean annual river values. A positive affirmation of a relationship between any given flood event, and a corresponding stand size, would thus be extrapolation beyond the statistical evidence. The data can thus be used to conclude the various generalities discussed above, but was not sufficient to elucidate any specific causal relationship between patch area and river level or flood events. Despite this, valuable insights were gained from the results above.

4.7.2 Island sites

Island site comparisons are also shown below for the three weirs (Figures 4.15-17). Data for the Le Chasseur weir (Figure 4.17) displayed a gradual decrease of island area in the presence of slightly oscillating but stable river levels between 1987 and 2003.

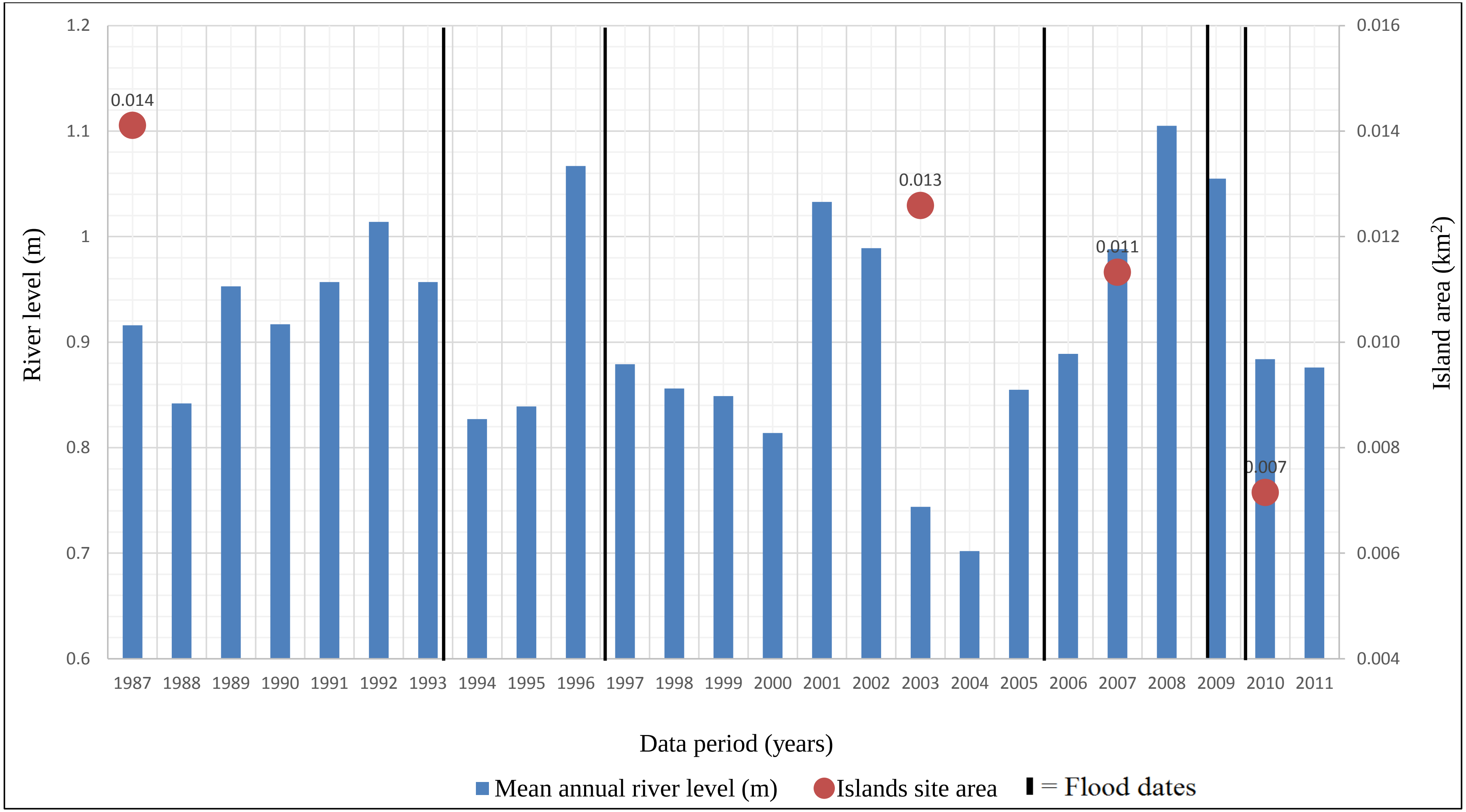


Figure 4.15 Le Chasseur weir flood, river level and imagery area comparison for island sites.

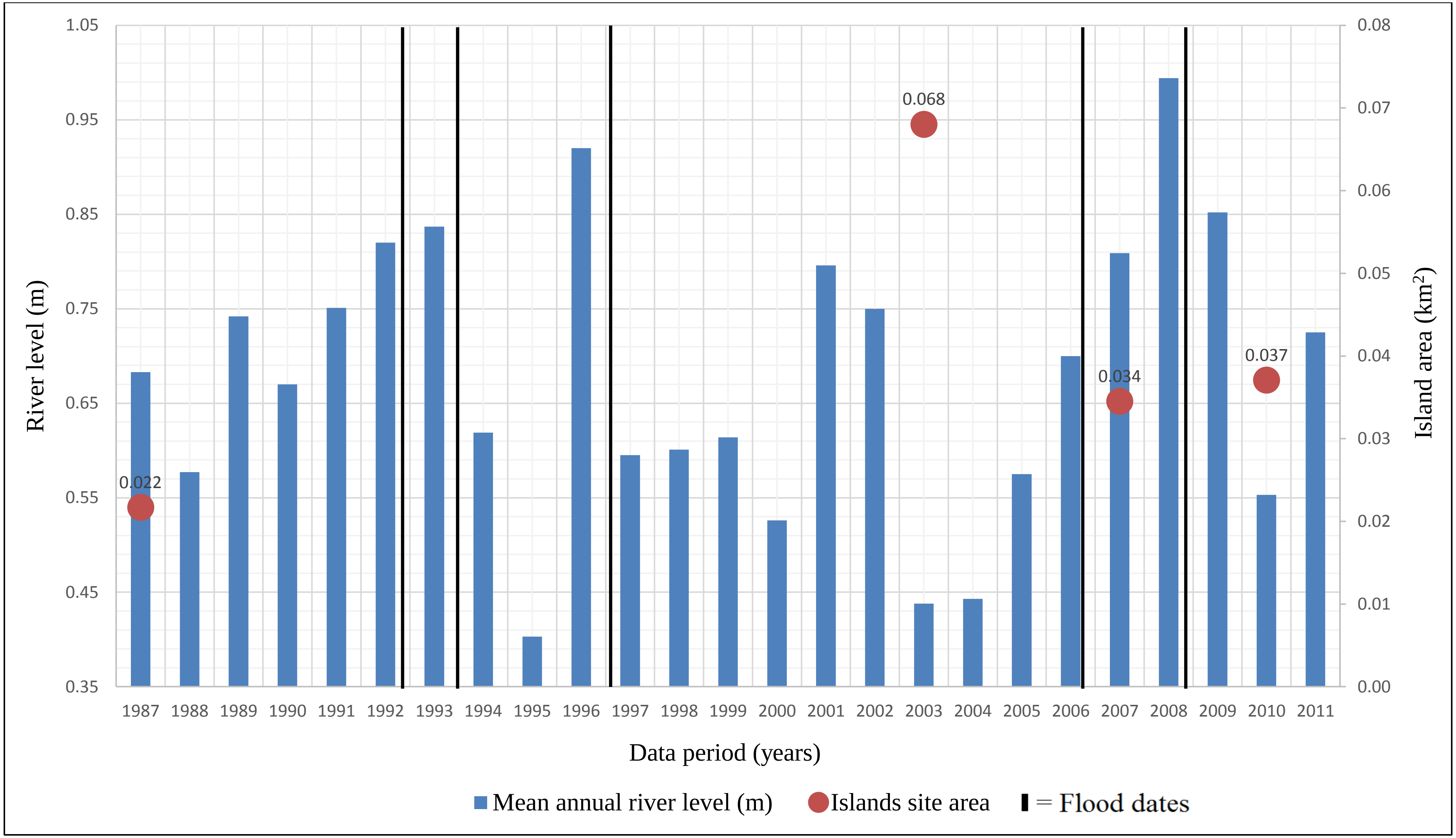


Figure 4.16 Wolvendrift weir flood, river level and imagery area comparison for island sites.

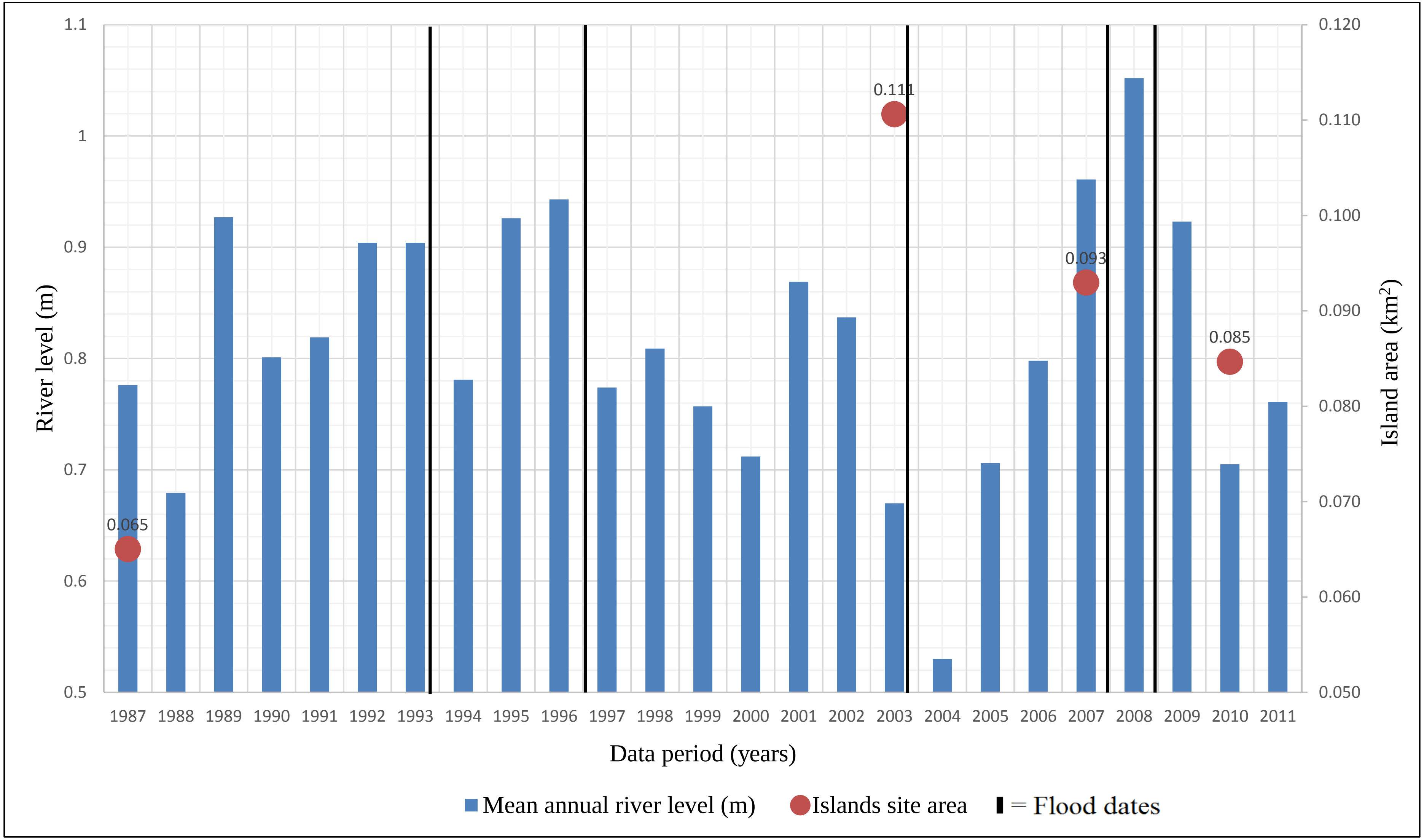


Figure 4.17 Swellendam weir flood, river level and imagery area comparison for island sites.

Island area decreased throughout the data period, indicating that the hydrological regime of the upper study section may have had a scouring effect on the island sites. As fast flow velocities from flooding were expected to dislodge and scour island sites within the channel, transporting sediment onto levee's and further downstream (Steiger et al. 2005), the general decline in island area in the uppermost reaches of the study area were not surprising, as channel velocity was regarded as potentially faster in the upper reaches of rivers (Leopold 1953) due to a reduction in slope in a downward direction. Indeed local slope and upstream drainage area have been shown to be important determinants of local stream velocity (Maidment et al. 1996).

As the Breede River exhibits a difference in elevation of roughly 75 m between Worcester and Bonnievale alone (eWisa 2012), the potential for an increase in stream velocity in a downwards direction exists. Additionally, due to the immediate scouring effect of high rainfall and flooding events within the channel, no lag period was expected as before with patch sites.

Furthermore, a sharp increase in annual water levels 2004-2008 coincided with a strong decline in island area. Between 2003 and 2007 the decline was gradual, whereas after 2007 the decline was markedly steeper. This may have been due to the immediate water levels being relatively high at the time (thus faster channel velocity and more scouring potential), in addition to the two large flood events seen between 2008 and 2010. A clear concomitant effect between river level and island area could thus be seen for Le Chasseur weir. One could conclude from this that the presence of high river levels, and especially the presence of flood events, may influence islands sites within the channel by scouring sediment, and sediment transporting downstream. This relationship is generally accepted (Steiger et al. 2005), and would result in the decline in island area.

Further downstream, at Wolvendrift weir (Figure 4.16), a contrasting relationship to that of Le Chasseur (Figure 4.15) was observed, in that island area increased across time with relatively stable and oscillating river levels between 1987 and 2002. Such increase suggested that the scouring was less of an effect within that section (as compared to further upstream), and rather deposition was the main influence with regards to island area increase. This could potentially have been possible due to the expected slower flow velocities within the channel by virtue of being further downstream. A possible 'tipping point' in flow velocity may have been reached between Le Chasseur weir and

Wolvendrift weir, where the influence of flooding on island area changed in direction from decreasing to an increasing effect.

The presence of three strong flood events during that period (1987-2003), however, seemed contradictory to the suggested influence mentioned above. The reduction in island area was intuitively related to the scouring influence represented by flooding events, however, the three flood events did not coincide visibly with the increase of island area between 1987 and 2003. It was thus reasonable to conclude, that if flooding events influence island area by scouring, that the immediate reduction in area was not observable from the figure, given the coarse temporal scale of data available during that period (2 data points spanning 1987-2003). Thus, whichever influence may have been exerted, it was not visible from the graph.

The two flooding events during 2006 and 2008 for Wolvendrift weir, did not however coincide with a decrease in island area. Specifically the 2006 flood, for which an island area data point was available in 2007, would have indicated that the expected scouring influence of flood events were present during that period. However, due to a decrease in island area, to a lesser extent, following the 2008 flood, it was suggested that if such an influence exists, that it would need to be viewed in combination with river levels. Flood events, thus had no observable relationship between island areas beyond what could be observed from the river level oscillation. Even so, a marked decrease in island area was noted during 2003 and 2007, with a concomitant sharp increase in water levels. This was followed by a slow increase in island area as river levels again lowered (2007-2010). It could therefore be concluded that river level were a more accurate indicator of the direction and magnitude of island area increase or decrease, with increased river levels occurring concomitantly to decreases in island area for Wolvendrift weir, with island area increasing where river levels remained relatively stable.

The Swellendam weir (Figure 4.17) exhibited similar change across time to that of Wolvendrift weir (Figure 4.16), with an increase in island area coinciding with a relatively stable and slightly oscillating river level between 1987 and 2003. Where river levels increased (2004-2008), island area decreased. This suggested that high river levels coincided with a decrease in island area, which provided further support for the possible scouring influence of islands by strong river velocities during periods of high flow.

The presence of three flood events between 2003 and 2008, where a concomitant and sharp decrease in island area was observed, additionally provides support for the scouring influence mentioned above. Unlike Wolvendrift weir, where island area increased slightly with the cessation of frequent flood events, island area continued to reduce within the Swellendam region. A possible explanation for this relationship is due to the additional water input from the Riviersonderend River. Such additional water input would potentially have resulted in greater flow velocity, and thus greater potential for scouring. As sediment from the upper regions would have been deposited as channel velocity slowed, the increased velocity after the confluence may have contributed to greater island area decrease within that region.

The general change across all weirs was thus one of increase in island site area with a relatively stable or slightly oscillating mean annual water level, with only a sharp decrease in island area being observed during times of consistent, and marked increase in river water level, or high frequency of flood events. As found for the patch sites, the hydrological regime between the weirs displayed similar change across years, due to being within the same channel.

4.8 SUMMARY OF RESULTS

In conclusion, this study has showed a general awareness of the problems posed by *E. camaldulensis* from interview responses, and indeed by other IAPS as well, despite clearing being conducted in a haphazard and ill-advised manner in many instances. Financial shortcoming and limited governmental intervention were singled out as possible reasons for this approach.

Furthermore, field data results indicated the high densities of *E. camaldulensis* along the riparian region of the river, serving to highlight the severity of the current invasion. Mature classes of *E. camaldulensis* dominated these regions, confirming the long standing invasion of the region, and providing support for the hypothesis that seeding sources were initially from commercial *E. camaldulensis* forestry practiced along the river banks since 1950.

Additionally, a steady increase of both patch and island sites across the study period, highlighting the increasing problem of IAPS, specifically *E. camaldulensis*, within the Breede River region. Hydrological results further indicated lag periods to vary with the river profile, and to be suited between 2-4 days generally. Flood dates were identified, and the resulting conclusion of a general increase of invaded area for patch sites after flooding was observed, however, the relationship was dependant on location within the river. Conversely, stable hydrological conditions were accompanied by a gradual decrease in the target species, although different locations within the river profile also showed different directions of change. An opposite relationship was found for islands sites, where a decrease in size after flooding events generally occurred, and an increase with stable flow over longer periods.

Although no causality can be inferred from these results, consideration of these findings may be prudent in resource management, specifically as it pertains to this region and target species. The presence of *E. camaldulensis* was thus likely to increase with the continuation of the current hydrological regime, in that no clear detrimental impact on the extent and distribution of the target species was observed. Caution is advised as no affirmation of positive causation was warranted from the results, in that a myriad of other factors may also readily influence the distribution of the target species within the study region.

E. camaldulensis was also further identified as one of many potential species contributing to island formation and change, although elevated river levels and frequent flood events coincided with a decrease in island area. The eradication of *E. camaldulensis* along the Breede River, should that ever be possible in practice, could thus not be expected to result in a marked difference in the formation of islands within the river.

CHAPTER 5 EVALUATION

5.1 SUMMARY

In this report, the severity and extent of IAPS invasions in South Africa was discussed, along with the pertinent literature regarding the presence and impact of *E. camaldulensis*, as well as the legal framework regulating various aspects of IAPS. The study area characteristics were described in terms of hydrology and composition of invasion. Management recommendations and insights into the stakeholders are further offered. Interview data, field measurements, rainfall and weir level data, in addition to spatial data were further used in analyses to address the aim of assessing the impact of high rainfall and high river level events on the distribution of *E. camaldulensis*.

Objective I, in which the current spatial extent of *E. camaldulensis* was required, was addressed through the various sites measured during fieldwork. The sites were invaded by a majority of mature to very mature *E. camaldulensis* populations, and expanded upon in the results and discussion sections. Identification of high rainfall and flood events, which represented objective II, were found through examination of weir threshold values and peak rainfall data. Lag times of between two to four days were identified and a weak positive correlation between the variables were found. Five flood events of the greatest magnitude were selected from the analyses, and used to evaluate the change over time of *E. camaldulensis*, together with historical spatial data which represented objective III and IV, respectively.

It was clear from the results that there was insufficient evidence to support a theory of a negative relationship between flooding and *E. camaldulensis* spatial extent. In fact, the relationship appeared positive in direction, whereby flooding events and high river levels corresponded to an increase in *E. camaldulensis* population size. The general trend between 1987/89 and 2013 was still one of an increase in stand size of *E. camaldulensis*. Results indicated that the overall hydrological regime was not sufficient to allow for a decrease in *E. camaldulensis* occurrence along the river over a longer time frame. Indeed, the physiological adaptations of *E. camaldulensis* ensure that the species has greater potential to adapt to water logging than drought.

As such the current hydrological regime of the Breede River may promote stand condition rather than promote water stress (in the form of drought), and subsequent die-back of the target species, even though short term reductions may be expected. Resource managers can thus safely assume flooding to have no strong, long term impact on *E. camaldulensis* distribution by diminishing stand size, but should rather expect a slow, yet gradual increase over the time span of decades.

Change in area across these study sites was found to be practically so low (or slow) across time, that clearing agencies may be better served by focussing their attention on other, faster spreading (and thus more urgent) species. The invasive potential of *E. camaldulensis* was not, however, in question, as has been shown by various other studies. Additionally it is evident from the widespread distribution of *E. camaldulensis* nationally, that to assume a slow rate of spread to be a characteristic of this species may be a costly error in natural resource management or clearing programme.

The overall increase in patch size of *E. camaldulensis* over time indicated past trends are likely to continue, thus an increase in patch size can be expected in future, albeit at a relatively slow pace. High rainfall and flooding events have not been observed to clearly coincide with the occurrence of *E. camaldulensis* along the river, and as such there is almost certainly other factors impacting significantly on the distribution of *E. camaldulensis*, apart from the hydrology of the region. Resource managers can be sure that unless intervention levels are increased, the area invaded by *E. camaldulensis* along the Breede River is likely to increase, and as such, noting this trend will be prudent and appropriate in future clearing planning. The final objective of providing management recommendations, was further addressed through the recommendations included further on in this chapter, as well as the discussion section.

Theoretical contributions made by this research is thus summarised as having shown that no strong long term relationship existed between river level and *E. camaldulensis* extent along the middle Breede River. The distinctive, highly dependent relationship observed in the Murray-Darling basin between flooding and *E. camaldulensis* distribution and vigour, does thus not correspond to the observed trend within the context of the middle Breede River. Current trends of an increase in area, however, necessitate the practical planning and execution of clearing efforts against the further spread of *E. camaldulensis*, which is of special importance to natural resource managers nationally. The theoretical, practical and data contributions gained from this study may thus serve as a basis for improved future management of *E. camaldulensis*, thereby assisting in the massive task of IAPS control faced by property owners, resource managers and residents of South Africa.

5.2 EVALUATION OF THESIS

Contribution made through this research project include an improved understanding of the interaction between *E. camaldulensis* and rainfall (and flooding); although not affirmed in the positive, was shown not to be restrictive. Furthermore, the general trend of change across time was established for both island and patch sites, in addition to establishing the current spatial extent, density and size composition of *E. camaldulensis* within the middle Breede area. This data will be compiled and added to the datasets of the DoA and WfW, assisting resource managers in their task of improved management of natural resources. All objectives were thus successfully achieved, and monitoring data obtained during the research will be communicated to the various interested parties.

The limitations of this study consist primarily of data quality. The quality of the conclusions obtained through research can only be proportionate to the quality of data used, and in this instance, a lack of useful aerial photography data limited the temporal scale at which conclusions could be drawn. The comparison between the much more comprehensive dataset for rainfall and river level, produced data restrictions as flood events did not necessary correspond with reliable aerial imagery data. Another limitation of the aerial imagery was the coarse resolution of the historical images, necessitating two time stamps of aerial imagery to be excluded from this work. This was due to poor visual detection of patches. It was simply not possible to reliably digitise patch and island sites with the available level of detail in two of the instances, which led to exclusion from this report.

Further limitations were the difficulties in estimating tree height. This limitation was addressed with the use of a vertex, by measuring tree heights at different patch sites across the study area, in order to ensure correct identification of tree height classes by the observer. Another limitation was the difficulty in establishing flood threshold values, as the river channel and river bank varied considerably in height depending on location within the river. This limitation was addressed by use of weir schematics, from which the exact height was calculated for weirs. Exact heights were then used to establish the weir threshold limit, allowing for certainty of overland flow (and thus flooding) when evaluating the river level data.

Visual interpretation also represented a concern in thus study. Only sites with *E. camaldulensis* were suitable for inclusion in data analysis, and as such only sites where the presence of the target species was confirmed (during fieldwork), could be included. Potential sites were excluded from the study on the basis that access was not granted and thus no confirmation of the target species could be made.

A final limitation was the sample size obtained for island sites and interviews. Although interview data were sufficient for the present inquiry (14 respondents), a sample of roughly 30 interviews would have been more useful, as reliable statistical inference beyond the proportional analysis used here would have been possible. Due to time constraints, and refusal of certain landowners to participate in the study, or simply their unavailability for the interview process, only a smaller sample size could be obtained. In the context of this report, where interview responses were used in a limited capacity to provide context only, the smaller sample size was sufficient.

Similarly, islands sites would have been better analyzed had there been a greater sample size. The island site sample size was constrained by suitability of potential sites (specifics provided in the methods section). As sites were required to be verified visually, and be visible on aerial imagery across all time stamps, many potential sites were excluded. The largest possible sample size was used after exclusion based on the criteria of suitability. Statistical analysis of the island sites was still valid and suitable for use in this study.

Strengths of this study included the use of reliable daily rainfall and river level data, which provided an accurate representation of the hydrological regime within the study area. Such fine scale measurement allowed the researcher to analyse the specific high rainfall and flood events to a high degree of accuracy. Another further strength of the hydrological data was the long time period for which data was available. This made possible the study over 26 years, which made the observations based thereon more reliable.

In addition, fieldwork data were also comprehensive in describing both the current extent (bounds) of invasion but also the density and age classes. A clear indication of the current spatial extent and vegetation structure was able to inform the conclusions. Specifically, the use of GPS waypoints for the study sites allowed for comparison of effects based on the location of a given site within the river, allowing for accurate understanding of the specific interactions and influences experienced at a given location.

A further strength of this study was the variety of data sources used. Interview responses and field data were invaluable in providing insight into the spatial changes observed across time. Indeed, without the qualitative data from interviews, and the quantitative data from fieldwork, conclusions drawn from spatial analysis would have been limited. The use of these varying data sources allowed for both breadth and depth into the research topic.

The use of field verification of sites was a further strength of this study. Many fine scale interactions were observed during fieldwork which allowed for greater understanding of the potential influences and interactions within the study area. An improved understanding further allowed for greater insight into the recommendations provided within this chapter. Verification of the target species additionally allowed for an estimation to be made regarding the historical presence of *E. camaldulensis*, which in combination with personal communications, interviews and literature allowed the researcher to establish valuable conclusions from the data.

5.3 RECOMMENDATIONS

5.3.1 Management

It is the author's recommendation, in order to specifically address objective V, that clearing programmes be conducted for *E. camaldulensis* along the Breede River only after clearing of tributaries are conducted, as the occurrence of *E. camaldulensis* is not confined to the Breede River alone, with large seed/seeding sources existing along the tributaries. Clearing the main river channel will thus be ineffective with the constant reseeding challenge represented by the tributaries. Clearing programmes should additionally be conducted with follow up programmes aimed at not only the juvenile clearing of *E. camaldulensis*, but that of competing IAPS such as *A. saligna* and *A. mearnsii,* as they are the major competitors to *E. camaldulensis* in this region. This is due to the secondary invasions observed in other studies (Ruwanza et al. 2012; 2013) which hamper the re-establishment of dominant native vegetation. Simply clearing a site and hoping for the return of natural vegetation is not enough, with even active restoration presenting significant challenges for improved ecosystem health of the future.

A further recommendation is to centralise land owners/users into one cohesive group represented by one person or small committee. The various water users associations currently represent the nearest approximation of this, however, fragmentation between these groups does occur. To illustrate this, the study area itself was represented by three distinct water users associations, which allowed for a separation of representation, and subsequently makes communication efforts between the DoA and land users more difficult. A further example of such fragmentation was the lack of a central contact person, or even a list of contact details for all members along the river. This was surprising given the personal, longstanding relationships shared between DoA staff, water users and land owners.

It was made clear during the project that interaction between the farmers was facilitated primarily through water users associations, with contact established mainly through personal relationships with individual land users along the river having been established over a long period of time. Certainly, there was no lack of skills or expertise from within the water users associations and DoA staff, and communication and interaction between the various stakeholders were maintained through these intimate contacts. However, the lack of a central authorised group or person does present an opportunity to improve on the current system, especially if the central contact groups are represented through those various water users associations, thereby incorporating their intimate knowledge of their respective sections and contacts into a cohesive, broad-scale collective. Through this collective, overarching IAP programmes could easier be implemented, managed, communicated and facilitated in the future.

Additionally, although farmers and land users interviewed indicated a great urgency for the removal of IAPS – and have a great appreciation for the risks posed, they still use incorrect clearing methods with little to no, or haphazard and unstructured follow up clearing programmes. Fire as a clearing method was commonly mentioned, in addition to the disposal of IAP remains directly into the river after removal (due to the ease thereof). These methods, although understandable practically, do not promote effective management of the IAPS along the river, and in certain instances promote the further spread and increase of IAPs.

Furthermore, individual land users could be better included in the broad IAP clearing strategy, through incentives and specialised training. Incentives could include subsidising herbicides or the hire of equipment used during clearing activities, which although not required of the DoA, would help fulfil their mandate of the management and control of IAP nationally. Reducing the financial burden of clearing IAPs may increase the efficacy and magnitude of removal conducted by the land users and farmers themselves. This is due to the fact that, in most instances, it is in the interests of land users and farmers to remove IAPs, and this understanding is prevalent among the interviewed respondents. The current invasion does not seem to be due to a lack of motivation or appreciation of risks involved with the presence of IAPs. Incentive schemes may thus promote self-management and clearing of IAPs from land user’s property, and when conducted in conjunction with education programmes to highlight the appropriate clearing methods, may reduce man-hours required for clearing by the DoA.

A final recommendation regards the reporting of IAP. Currently, the DoA manages IAP through mapping and field verification of each land parcel along the river prior to clearing operations. This allows for accurate monitoring of each parcel and effective communication between the DoA and the respective land user. If, however, the task of reporting IAP extent could be delegated effectively to the land user (who has it in their own interests to report), monitoring could be reduced to verification and scheduled survey only, which would further reduce man-hours and expenditure required by the DoA.

In light of the ecological sensitivity and importance of riparian zones, having a well informed and up to date, centrally organised, self-clearing and self-reporting riparian civil community can only bode well for current, and future clearing endeavours, especially as the tenacious nature of IAPs are becoming increasingly apparent and appreciated by the scientific community and civil society alike.

5.3.2 Further research opportunities

The research presented in this report, along with the conclusions drawn, would be most useful in conjunction with studies into the specific competition interactions between *Acacia* species and *E. camaldulensis* to allow for a greater understanding of the current distribution of *E. camaldulensis*, as compared to *Acacia saligna* or *Acacia mearnsii* along the river's edge. Further research opportunities presented by the conclusions of this study could be an evaluation of stakeholders' knowledge of effective clearing methods, species knowledge in terms of their biology and ecology, as well as an investigation into the various difficulties experienced by landowners and resource managers, in their attempts to manage and mitigate IAPS invasions in their respective spheres.

An additional research avenue for the future is the investigation into the effect of salinity on the distribution of *E. camaldulensis* within the lower section of the Breede River. Although the impact of salinity was outside of the scope of this research, the absence of large *E. camaldulensis* stands further downstream of the Swellendam region suggests that elevated salinity levels, which can reasonably be expected for sections of the river closer to the river mouth, may influence the establishment of *E. camaldulensis*. If a relationship exists, understanding the local impact of salinity levels may assist in prioritising clearing activities for the region.

Another avenue for research could be investigation into the influence of the Riviersonderend tributary to the Breede River downstream of the confluence. Specifically in attempting to quantify the magnitude of *E. camaldulensis* seeding and the water quantities added through the Riviersonderend.

Finally, research conducted into the classification and satellite identification of *E. camaldulensis* from sensor data would prove extremely useful to resource managers in particular, as immense gains into monitoring of the invasion could be made through remote sensing and GIS applications.

5.4 CONCLUDING REMARKS

Through this research report, a greater understanding of the relationship between the hydrological regime of the Breede River and *E. camaldulensis* was sought. Having successfully addressed the various research questions, objectives and aim, this study forms a small, yet important, contribution to our understanding of the interactions between the hydrological regime of the Breede River and the occurrence of *E. camaldulensis*, and indeed IAPS interactions within the study area in general.

The contribution of fine scale data of the real-world situation, combined with the results, provide greater insight into the myriad of factors influencing IAPs and *E. camaldulensis* in particular, thus assisting policy makers, natural resource managers, landowners and the scientific community in better understanding and managing our environment. This research also forms a basis for further inquiry, in that the relationship between rainfall and river level, as they relate to *E. camaldulensis* spread, has now been made clearer, and further inquiries can build upon the insights gleaned here.

Unless specific interventions are initiated and maintained, the urgent issue of IAPS and the related ecological harm they pose, will remain an ever present reality. Gains have been made in this regard, and the ingenuity of mankind may serve to address this specific problem within our society. Much can be hoped for, and indeed, striving for the ecologically sound management, greater understanding and conservation of our precious and only resource – the natural environment, can be a difficult but noble cause.

[END]

Words: 42 226.

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APPENDIX A: INTERVIEW SCHEDULE

Interviewer: **Date:**

SECTION A - HOME AND PROPERTY:

1. Which position do you hold on this property?

Owner		Worker	
Manager		Family member	
Foreman		Other (please indicate)	

2.1 Which active farming practices are conducted on this farm?

____

____

2.2 Which non-farming related activity does the property fulfil?

Residential		Business	
Vacation property		Recreation	

2.3 (If question 2.2 has been completed). What is the main economic activity of this farm?

____

____

3 Is there any livestock on the farm?

Type	Amount
Pig	
Sheep	
Cattle	
Chickens	
Goats	
Game	
Other (Please specify)	

SECTION B - RIVER RED GUM:

4. In your opinion, what is the percentage for each class of River Red Gum on the farm?

Age class	Percentage (%)
Seedling/sapling (0-30cm)	
Young (30cm-5m)	
Adult (5-15m)	
Mature (15-30m)	
Very Mature (30m+)	

TOTAL - 100%

5. Have you ever planted River Red Gum on the farm before? (Yes/No).

____

6. How many years has the River Red Gum been present on the farm?

____

7. Does anyone purposefully remove the River Red Gum on the farm, if so, how? If not, why?

____

____

8. Is there any other vegetation (Native/Alien/Weed) that is purposefully removed? (Yes/No). Which?

____

____

9. What is the reason for the abovementioned vegetation removal?

____

____

10. Has there been any vegetation clearing done on your property by anyone other than household members? (Ex. Working for Water, Dept. of Water Affairs)? (Yes/No). When and where?

____

____

11. How often was the abovementioned clearing conducted? (Weekly, monthly, yearly, 2-yearly, 5-yearly, 10-yearly, once-off).

____

12. Does your household employ any methods to reduce River Red Gum occurrence on the property? (Yes/No). Which?

____

____

SECTION C - FLOODS AND DAMS:

13. Do you remember any floods in the Breede River over the last 32 years? When and where do you remember them?

____

____

14. What actions did the household perform to reduce the flood effects *on your farm*?

____

____

15. Did the flooding affect the occurrence of River Red Gum in any way? (Yes/No). If so, how?

____

____

16. Which plants, would you say, established again after the floods, and in which order, if any?

____

____

17. Did the presence of the dam influence the water level and flow in the river? (Yes/No) If so, how?

____

____

18. Did the presence of dams affect the occurrence of River Red Gum in any way? (Yes/No). If so, how?

____

____

____

SECTION D – ISLAND FORMATION:

19. In your opinion did the construction of dams within the river influence the formation of island or bars within the river? If so, how? (example: occurrence, formation and/or shape).____

____

20. If, in your opinion, if the presence of River Red Gum influences the formation of islands within the river stream, how would you say that the River Red Gum changes the river channel, sandbanks and islands?

____

____

21. Which other species (native and/or exotic), if any, would you say are also involved in the formation of islands within the river stream?

____

____

22. In which way would you say does the presence or absence of the abovementioned species (question 24) influence island formation in the river?

____

____

23. Which impact would you say do floods have on the occurrence of islands and sand bars within the river?

____

____

24. In what way does the above change in the river influence your property and business?

____

____

25. In what way does your household attempt to mitigate the impact of these islands and sand bars within the river, if any?

SECTION E - CLOSING:

Thank you very much for participating in this survey. We appreciate your time and inputs. Is there anything else that you think would be helpful for me to know so that we understand the spread of River Red Gums better? Do you have any questions for me?

APPENDIX B: FIELDWORK DATA SHEET

Polygon Name	General GPS Position (GPS point code/name)	General GPS precision	Patch Land use category (1-8)	Nearest adjacent LU category (1-8)	Start GPS (GPS point code/name)	General Density % (Native/ Exotic)	Target Density (%) - trunks	Species; Age class; Density present (*positional code)	Underfoot conditions (stability)	Obstructive vegetation height	Obstructive vegetatino density (canopy cover)	Walk Time one way (min)	Slope	Notes
"1Worcester1"	"145"	"8m"	"3"	"2"	"112"	80/20	30	See "positional" code in 'Codes' tab for example	Large boulder	30cm	50% cover	12min	10 degree	Area was burned recently

APPENDIX C: RAINFALL AND RIVER DISCHARGE DATA LAGGED CROSS-CORRELATION RESULTS

Table C. 1 Karroo weir discharge (m^3/s) and Kwaggaskloof rainfall (mm) daily data lagged cross-correlation (1980-1992).

Days	**Rho value**	**P-value**	**Observations (N)**	**Result correlation**
0	0.120	$p < 0.001$	4306	Weak positive
1	0.229	$p < 0.001$	4305	Weak positive
2	0.324	$p < 0.001$	4304	Weak positive
3	0.322	$p < 0.001$	4303	Weak positive
4	0.284	$p < 0.001$	4302	Weak positive
5	0.245	$p < 0.001$	4301	Weak positive
6	0.215	$p < 0.001$	4300	Weak positive
7	0.202	$p < 0.001$	4299	Weak positive

Table C. 2 Le Chasseur weir discharge (m^3/s) and Kwaggasloof rainfall (mm) daily data lagged cross-correlation (2000-2012).

Days	**Rho value**	**P-value**	**Observations (N)**	**Result correlation**
0	0.059	$p < 0.001$	4260	Weak positive
1	0.128	$p < 0.001$	4259	Weak positive
2	0.241	$p < 0.001$	4258	Weak positive
3	0.299	$p < 0.001$	4257	Weak positive
4	0.277	$p < 0.001$	4256	Weak positive
5	0.238	$p < 0.001$	4255	Weak positive
6	0.207	$p < 0.001$	4254	Weak positive
7	0.189	$p < 0.001$	4253	Weak positive

Table C. 3 Wolvendrift weir discharge (m^3/s) and Robertson rainfall (mm) daily data lagged cross-correlation (2000-2010).

Days	**Rho value**	**P-value**	**Observations (N)**	**Result correlation**
0	0.083	$p < 0.001$	3917	Weak positive
1	0.140	$p < 0.001$	3916	Weak positive
2	0.203	$p < 0.001$	3915	Weak positive
3	0.243	$p < 0.001$	3914	Weak positive
4	0.237	$p < 0.001$	3913	Weak positive
5	0.205	$p < 0.001$	3912	Weak positive
6	0.178	$p < 0.001$	3911	Weak positive
7	0.161	$p < 0.001$	3910	Weak positive

Table C. 4 Wagenboomsheuvel weir discharge (m^3/s) and Ashton rainfall (mm) daily data cross-correlation (1980-1991).

Days	**Rho value**	**P-value**	**Observations (N)**	**Result correlation**
0	0.114	$p < 0.001$	4085	Weak positive
1	0.155	$p < 0.001$	4084	Weak positive
2	0.202	$p < 0.001$	4083	Weak positive
3	0.246	$p < 0.001$	4082	Weak positive
4	0.261	$p < 0.001$	4081	Weak positive
5	0.244	$p < 0.001$	4080	Weak positive
6	0.221	$p < 0.001$	4079	Weak positive
7	0.202	$p < 0.001$	4078	Weak positive

Table C. 5 Swellendam weir discharge (m^3/s) and Marloth rainfall (mm) daily data cross-correlation (1980-2009).

Days	**Rho value**	**P-value**	**Observations (N)**	**Result correlation**
0	-0.003	$p = 0.754$	7233	Weak negative
1	0.018	$p < 0.001$	7232	Weak positive
2	0.057	$p < 0.001$	7231	Weak positive
3	0.104	$p < 0.001$	7230	Weak positive
4	0.126	$p < 0.001$	7229	Weak positive
5	0.122	$p < 0.001$	7228	Weak positive
6	0.109	$p < 0.001$	7227	Weak positive
7	0.096	$p < 0.001$	7226	Weak positive