RESTORATION OF A DEGRADED SUBTROPICAL FOREST FOR CLIMATE CHANGE MITIGATION AND ADAPTATION IN THE CITY OF DURBAN, SOUTH AFRICA

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PREFACE

The research contained in this thesis was completed by the candidate while based in the Discipline of Land Use Planning and Management, School of Agricultural, Earth and Environmental Sciences, in the college of Agriculture, Engineering and Science, University of KwaZulu-Natal, Pietermaritzburg Campus, South Africa. The research was supported by eThekwini Municipality through the Durban Research Action Partnership: Community Reforestation Research Programme, and by the South African Research Chairs Initiative of the Department of Science and Technology and National Research Foundation of South Africa.

The contents of this work have not been submitted in any form to another university and, except where the work of others is acknowledged in the text, the results reported are due to investigations by the candidate.

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DECLARATION

I, Lutendo F. Mugwedi (Student number: 214584145), declare that:

(i) the research reported in this thesis, except where otherwise indicated or acknowledged, is my original work;

(ii) this thesis has not been submitted in full or in part for any degree or examination to any other university;

(iii) this thesis does not contain other persons' data, picture, graphs or other information, unless specifically acknowledged as being sourced from other persons;

(iv) this thesis does not contain other persons' writing, unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:

(a) their words have been re-written but the general information attributed to them has been referenced;

(b) where their exact words have been used, their writing has been placed inside quotation marks, and referenced;

(v) where I have used material for which publications followed, I have indicated in detail my role in the work;

(vi) this thesis is primarily the a collection of material, prepared by myself, published as journal articles or presented as a poster and oral presentations at conferences. In some cases, additional material has been included;

(vii) this thesis does not contain text, graphics or tables copied and pasted from internet, unless specifically acknowledged, and source being detailed in the dissertation and in Reference sections.

Signed: Lutendo F. Mugwedi

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GENERAL ABSRACT

With unprecedented changes in climate and land-use patterns, a decrease in global biodiversity and ecosystem services has been occurring at an alarming rate. This has resulted in a widespread damage to the life-support systems upon which every living organism depends on. Reforestation of degraded forest ecosystems is now globally recognized as one of the best natural capital investment options, owing its contribution to biodiversity conservation, climate change mitigation and adaptation, and ecosystem services provision. The aim of this study was (1) to unravel confusions caused by the inconsistent use of terminologies describing different reforestation initiatives; (2) to investigate motivations behind recent reforestation initiatives; (3) to demonstrate the use of a restoration decision-making tool, Robust offsetting (RobOff); (4) to investigate the influence of climatic and edaphic factors on reforestation initiative, (5) to assess reforestation initiative success, and (6) to assess the impact of drought on reforestation initiative. A comprehensive review was conducted to unravel the confusion caused by the inconsistent use terminologies describing different reforestation initiatives, and to gain insight into motivations behind reforestation initiatives in recent literature (2000 to 2016). The results showed that there are 10 most common terminologies used to describe different reforestation initiatives. These terminologies were categorized into five groups based on their motivations, namely, (1) *Creation* or *Fabrication*, *Reallocation* and *Replacement*, (2) *Ecological engineering*, (3) *Ecological restoration*, (4) *Reclamation*, *Reconstruction*, *Remediation*, *Renewal* or *Redemption*, and (5) *Rehabilitation*. The recent reforestation initiatives were motivated by the need to reinstate resilient and more functional forest ecosystems (through planting of a higher diversity of native tree species). This is because species diverse forests are more resilient and functional with significant contributions to biodiversity conservation (fauna and flora), climate change mitigation (carbon storage) and adaptation (e.g., flood control) and ecosystem services that sustain society (e.g., food) and economy (e.g., employment opportunities). Using the Buffelsdraai Landfill Site Community Reforestation Project (BLSCRP) as a case study, RobOff, was employed to plan a complex large-scale reforestation. The complexity was caused by a mosaic of habitats (‘extant forest’ and ‘former sugarcane fields’) with varying levels of degradation, diverse reforestation actions (*natural regeneration, current action, carbon action and biodiversity action*), a limited budget and multiple goals (biodiversity, carbon stock and employment). RobOff results showed that investing in the restoration of 'former sugarcane fields' through *biodiversity action* is preferable, because it achieved the highest biodiversity, carbon stock and employment opportunities. Field trials were conducted at the Buffelsdraai Landfill Site to assess the influence of microtopographic positions, and soil physical and chemical properties on the growth performance of the four most dominant planted native tree species (*Bridelia micrantha*, *Erythrina lysistemon*, *Millettia grandis* and *Vachellia natalitia*). Root-collar diameter, stem height and canopy width growth rates were assessed across the chronosequence of three habitats under restoration (0-, 3-, and 5-year-old), in the upland (dry) and lowland (wet) areas of each habitat. *Erythrina lysistemon* and *V. natalitia* were found to be good fast growing tree species suitable for restoration in both the upland and lowland areas, while *B. micrantha* was suitable for lowland area. Reforestation success of the BLSCRP was assessed using measures of plant richness, diversity, vegetation structure, invasive alien plants (IAPs) and ecological processes, contrasted across a chronosequence of habitats under restoration (0-year-old, 3-year-old and 5-year-old) and compared with a reference forest habitat (natural forest). The BLSCRP was largely successful, but low tree density and an increase in IAP cover with an increase in restoration age were identified as threats to the BLSCRP success. The 2015 El Niño event induced serendipitous drought occurrence in South Africa led to the assessment of its effect on planted tree sapling mortality and on the growth performance of the four most dominant planted tree species in the 0-year-old habitat. Drought effected mortality was highest in the lowland area (34.1%) and lower in the upland area (18.9%). Mortality rate of the nine most abundant species ranged from 10% to 52.5%. *Erythrina lysistemon* and *V. natalitia* had good growth rates in both the upland and lowland areas, and *B. micrantha* in the lowland area. The BLSCRP is highly likely to achieve its climate change mitigation and adaptation, biodiversity and ecosystem services restoration and employment creation in the city of Durban, provided the identified threats are addressed as soon as possible. The overall findings from this study showed that future large-scale reforestation initiatives around the globe should be designed to achieve biodiversity conservation, climate change mitigation and adaptation, and ecosystem services supply.

LIST OF PRESENTATIONS

2014

Symposium of Contemporary Conservation Practice, Howick - South Africa (November 2014) – Title: Restoration planning for Buffelsdraai Landfill Site in the eThekwini Municipality (Oral presentation).

2015

The sixth World Conference on Ecological Restoration, Manchester-UK (August 2015) – Title: Restoration planning for Buffelsdraai Landfill Site in the city of Durban, South Africa (Oral presentation)

Durban Action Research Partnership Year-end Symposium, Durban-South Africa (December 2015) – Title: Restoration planning for Buffelsdraai Landfill Site in the city of Durban, South Africa (Oral presentation). – **Prize won: 2nd Prize for the best student project presentation.**

2016

The third African Congress for Conservation Biology, El Jadida-Morocco (September 2016) – Title: Restoration planning for climate change mitigation and adaptation in the city of Durban, South Africa (Oral presentation).

2017

Durban Research Action Partnership launch, Durban-South Africa (January 2017) – Title: Restoration Trust Account Assessment (based on Chapter 5) (Oral presentation – Guest Speaker).

The seventh World Conference on Ecological Restoration, Foz Do Igua-Brazil (August – September 2017) – Title: Restoration planning for climate change mitigation and adaptation in the city of Durban, South Africa (Oral presentation).

Durban Action Research Partnership Year-end Symposium, Durban-South Africa (December 2017) – Title: Get your house in order first, before inviting new guests (Oral presentation based on chapter 2).

.

DEDICATION

This work is dedicated to my paternal grandmother Masindi Munari and my late uncle Nndavheleseni Mugwedi, their remarkable indigenous knowledge of forest ecosystem patterns and processes cultivated my interest in forest ecology.

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ONE DAY IS ONE DAY!

TABLE OF CONTENTS

GLOSSARY OF ACRONYMS

ACE – Abundance-based coverage estimator

ANOSIM – Analysis of similarity

BCEF – Biodiversity conservation and ecosystem functioning

BLSCRP – Buffelsdraai Landfill Site Community Reforestation Project

CW – Canopy width

EBA – Ecosystem-based adaptation

EMA – eThekwini Municipal Area

ENSO – El Niño Southern Oscilation

IAPs – Invasive alien plants

FIFA – Fédération Internationale de Football Association

FLR – Forest Landscape Restoration

GLMs – Generalized linear models

KZN – KwaZulu-Natal

MCPP – Municipal Climate Protection Programme

NEMBA – National Environmental Management: Biodiversity Act

NMDS – Non-metric multidimensional scaling

RobOff – Robust Offsetting

RCD – Root-collar diameter

SA – South Africa

SER – Society for Ecological Restoration

CHAPTER 1: GENERAL INTRODUCTION AND OVERVIEW

1.1. Study background motivation

Forests play a critical role in supporting and maintaining ecological systems and cycles (e.g., carbon and water recycling) (Bonan 2008; FAO 2016). Forests (especially closed-canopy tropical forests) have a major influence on local, regional and continental climate (Thompson et al. 2009; Swann et al. 2012). They regulate microclimatic conditions, reducing urban people's exposure to heat waves by providing shade (Oldfield et al. 2015; Zhang et al. 2016), and recycle and generate the atmospheric water vapour flows (Sheil and Murdiyarso 2009; Ellison et al. 2012) that can mitigate warming effects in arid areas (Locatelli et al. 2015).

Forests are major reservoirs of terrestrial biodiversity and contain about 50% of the total global terrestrial carbon stock (FAO 2000). Functional forests supply a wide range of ecosystem services to natural systems and humankind (Hassan et al. 2005). The ecosystem services include provisioning (e.g., food and timber), regulating (e.g., carbon sequestration and water purification), supporting (e.g., biodiversity refugee sites, and wildlife habitat and food) and cultural (e.g., recreation and spiritual fulfilment) (Thompson et al. 2009; FAO 2016). As a result, people have been depending on forests for their well-being and survival since time immemorial (von Maltitz et al. 2003; Mansourian et al. 2005; Thompson et al. 2009).

The management of forests and the use of their ecosystem services can affect their future critical role of maintaining biodiversity and ecosystem functions, and ecosystem services supply (FAO 2016). Most of the ecosystem services are often viewed as free societal benefits or 'public goods', and their critical contribution to humankind is often overlooked in individual, corporate and public decision-making (Roongtawanreongsri et al. 2015). When forests are undermined, they become increasingly susceptible to changes in land use practices (e.g., agriculture and development), thus resulting in their loss and degradation (Roongtawanreongsri et al. 2015). Forest loss and degradation as a result of changes in land use practices lead to habitat fragmentation, biodiversity and ecosystem services loss, soil degradation, increased risks of fire, and alien plant invasion, thus affecting societies and economies (von Maltitz et al. 2003; Mansourian et al. 2005; Thompson et al. 2009).

Forest loss and degradation is a problem that has been documented since the seventh century B.C., when it was mentioned in the *Epic of Gilgamesh* that was recorded on 12 cuneiform tablets found in Assyria (Dudley et al. 2005). Currently, forest loss and degradation is a global challenge (Mansourian et al. 2005; Thompson et al. 2009; FAO 2016), in 1990, the world's forested area was 4128 million ha, but this area has decreased to 3999 million ha by 2015 (FAO 2016). The remaining forests are under continuous pressure, and a considerable proportion has been severely degraded (Dale et al. 2001). The rapid and extreme impacts of global climate change are exacerbating forest loss and degradation (Dale et al. 2001; Bonan 2008). As a result of increased forest loss and degradation caused by current land use practices and climate change, there is a global concern about the long-term capacity of forests to maintain adequate resources (biodiversity and ecosystem services) to ensure ecological and socio-economic benefits for current and future generations (Dudley et al. 2005; Aronson et al. 2007; Thompson et al. 2009; FAO 2016). The need for reforestation has been promoted since the 1600s in England by English pamphleteer John Evelyn (Dudley et al. 2005). Reforestation is defined as the establishment of forests (using exotic or native tree species) on lands that historically had forests, but are now degraded or have been converted to other land use (after Lund 2014).

The motivations for reforestation have changed over time. Earlier reforestation initiatives (in the late nineteenth century to the twentieth century) were mainly driven by industrial demands (e.g., timber and pulp) and to a lesser extent by ecological and socio-economic demands (e.g., soil stabilization, timber and firewood) (Richardson 1998; Lamb and Gilmour 2003; Dudley et al. 2005). In this thesis, the focus is on reforestation to achieve ecological and socio-economic benefits. The use of fast growing exotic tree species was more dominant in the earlier reforestation initiatives (Richardson 1998; Dudley et al. 2005). Following the Kyoto Protocol in 1997, there has been a rapid increase in reforestation initiatives for climate change mitigation (Cannell 1999; Trotter et al. 2005; Zomer et al. 2006). This is because, reforestation is a low-cost approach to reduce greenhouse gas emission compared to other mitigation strategies (Turner et al. 2009). Developed countries are investing in reforestation initiatives in developing countries in order to offset their carbon footprint (Smith 2002). Although some projects use multi-species, most of them use monocultures of fast growing, especially exotic tree species (e.g., *Eucalyptus* spp., *Pinus* spp. and *Poplus* spp.) to sequester more carbon within a short period of time (Cannell 1999; Lamb and Gilmour 2003; Fernandes et al. 2016).

Reforestation initiatives can be aimed at achieving ecological benefits (e.g., biodiversity conservation), socio-economic benefits (e.g., timber, flood mitigation or employment) or both

(Mansourian 2005). Different approaches are taken to achieve specific reforestation goals. For example, planting of monocultures or a few tree species (less biodiversity benefits) has been an approach taken to achieve socio-economic benefits (Richardson 1998; Lamb and Gilmour 2003; Mansourian 2005). On the other hand, planting a higher diversity of native tree species is the approach taken to restore the pre-degraded biodiversity, ecosystem functions and services using a reference habitat (less disturbed forest) as a bench mark (Vallauri et al. 2005). As a result of different goals that reforestation initiatives aim to achieve, there has been a development of terminologies (e.g., rehabilitation and reclamation) describing different reforestation initiatives. These terminologies are often used indiscriminately, with no consensus among reforestation scientists and reforestation managers, thus leading to confusion amongst restoration scientists, restoration managers, policymakers, and other stakeholders (e.g., private land owners and local communities) (Mansourian 2005; Stanturf 2005; Hobbs et al. 2011). Therefore, there is a need to unravel this confusion by providing a comprehensive list of terminologies used to describe different reforestation initiatives’ motivations. This will allow the development of science-based strategic approaches, alignment of global reforestation activities, and the development of global learning networks around reforestation.

Alignment of global reforestation initiatives would allow reforestation managers and practitioners to learn different approaches used in the planning of large-scale reforestation projects around the globe. This is because, planning of a large-scale reforestation project can be a complex undertaking given multiple habitats with varying degradation conditions, diverse reforestation options (e.g., livestock exclusion, invasive alien plant management and tree planting), and multiple ecological (e.g., biodiversity) and socio-economic (e.g., water provision and employment) goals to be achieved simultaneously with a limited reforestation funding (Maron and Cockfield 2008; Turpie et al. 2008; Brancalion et al. 2013; Vogler et al. 2015). Without prioritization planning, resource allocation is likely to be made in an *ad hoc* manner and this might affect the reforestation success (Wilson et al. 2011). There is a need to incorporate systematic planning tools to offer guidance on planning and implementing reforestation projects (e.g., Thompson 2009; Wilson et al. 2011). This will ensure the efficient allocation of limited reforestation resources through prioritization of reforestation actions that provide greater ecological and socio-economic benefits and improve the long-term sustainability of reforestation projects (Maron and Cockfield 2008; Adame et al. 2015).

When planning a large-scale reforestation project, it is critical to know the intended reforestation tree species ecology and the reforestation site conditions. This knowledge is useful to ensure that species with higher survival rate and faster growth rate can be selected to reduce site maintenance costs (e.g., replanting to compensate for dead trees and weed control) and create microclimatic conditions conducive for natural succession, thus promoting rapid forest development (Walker and del Moral 2003; Montagnini et al. 2005; Omary 2011; Simmons et al. 2012; Goosem and Tucker 2013). The required knowledge include but not limited to species propagation, light requirements, growth under different climatic and edaphic conditions, tolerance to drought, resistance to disease and pest, ability to coppice after disturbance, and uses by local communities (e.g., food) or animals (Motangnini et al. 2005; Goosem and Tucker 2013). Once the project has been implemented, continuous monitoring and assessment using multiple indicators is crucial to guide the necessary management intervention to ensure reforestation success and to guide future reforestation projects (Ruiz-Jaén and Aide 2005; Wortley et al. 2013).

Most recent reforestation initiatives were motivated by climate change mitigation goal (carbon sequestration and storage) (Locatelli et al. 2015). However, climate change is now making reforestation more difficult (Biringer and Hansen 2005). Climate change induced stressors such as drought, diseases and pests outbreaks, forest fires and invasive alien plants (IAPs) can negatively affect forest growth and development (Biringer and Hansen 2005; Curiel-Esparza et al. 2015; Locatelli et al. 2015). Droughts associated with the El Niño Southern Oscilation (ENSO) events are becoming more frequent and more intense as a result of climate change (Edwards and Krockenberger 2006; Maza-Villalobos et al. 2013). Climatic modelling studies in southern Africa have shown that rainfall will be highly variable, accompanied by high temperatures, and more frequent and more intense droughts (Tadross et al. 2011; Naik and Abiodun 2016). Although drought events present a major challenge to reforestation, they also provide an opportunity to assess tree species that could promote rapid forest establishment in arid and drought prone areas (Edwards and Krockenberger 2006; Montwé et al. 2016) and create microclimatic conditions that are conducive to natural succession (Walker and del Moral 2003; Montagnini et al. 2005; Omary 2011; Simmons et al. 2012; Goosem and Tucker 2013).

In South Africa, forest is an important yet a rare biome covering approximately 4883.25 km^2 (Berliner 2009). The biome contains 568 woody plant species of which 64% is of economic importance (Geldenhuys 2007). The forest biome play a critical role in livelihood support (e.g., food, medicinal plants, timber, firewood, craft wood and fibre) (Shackleton and Shackleton 2004; Geldenhuys 2007; Fabricius and Cundill 2007), religious (spiritual inspiration) and cultural value (recreation and sense of place) (Fabricius and Cundill 2007). However, forest biome is highly fragmented as a result of changes in land use (e.g., agriculture, commercial plantations, development, mining, IAPs and fire), thus making it one of the most vulnerable biomes (von Maltitz et al. 2003). There are a considerable number of reforestation initiatives in South Africa. The motivations behind these initiatives include climate change mitigation and adaptation, ecological, hydrological and socio-economic goals (e.g., Untiedt 1992; van Aarde et al. 1996; Mills and Cowling 2006; Richardson and van Wilgen 2004; Scholtz 2008). There are synergies between reforestation initiatives motivations. For example, planting a higher diversity of native tree species achieve climate change mitigation, biodiversity restoration and ecosystem services provision (e.g., Cunningham et al. 2015; Oldfield et al. 2015).

Under the face of global environmental change, protection of functional and restoration of non-functional ecosystems is important to meet the health, social, cultural and economic needs of urban communities (Craig and Vesely 2007). EThekwini Municipality (city of Durban) is addressing the complex challenges of climate change through the development of city wide Municipal Climate Protection Programme (MCPP) initiated in 2004. The MCPP's strong and early focus on adaptation makes it unique compared to many other urban climate change initiatives around the world. One of the MCPP's adaptation work stream priorities is the prioritization and consolidation of ecosystem-based adaptation work (Roberts et al. 2012). The ecosystems within the municipal area provide a sustained supply of provisioning, regulating, supporting and cultural ecosystem services to the people of Durban. In 2003, these ecosystem services were valued at R3.1 billion per annum (US$227 million per annum), excluding their contribution to the tourism industry. The municipality is investing in the protection and restoration of ecosystems to enhance biodiversity and continued supply of these critical ecosystem services to meet the needs of the current and future generations (eThekwini Municipality 2015).

The city's hosting of 2010 FIFA™ World Cup and the United Nations Framework Convention on Climate Change COP17/CMP7 in 2011 led to the development and implementation of the greening programmes (Diederichs and Roberts 2015). One of the greening programmes was a large-scale community-based reforestation programme initiated in 2008 to offset a portion of CO^2 emissions associated with both events. Although climate change mitigation was the main motivation, the programme was designed to also enhance biodiversity, ecosystem services supply and socio-economic benefits, mainly employment opportunity to local impoverished communities. The reforestation programme was implemented in three sites, namely, Buffelsdraai Landfill Site, Inanda Mountain and Paradise Valley (Douwes et al. 2015). After the programme initiation, the municipality saw a need of using science-based approaches to (1) to access reforestation success of the current projects in order to inform the necessary management interventions needed to ensure project success, (2) to assess the likely impact of climate change on reforestation success, and (3) to plan for future reforestation projects. These issues were addressed in the context of the Buffelsdraai Landfill Site Community Reforestation Project (chapter 3 – 6).

1.2. Study aim and objectives

The aim of this study was to (1) to unravel confusions caused by the inconsistent use of terminologies describing different reforestation initiatives, (2) investigate motivations behind recent reforestation initiatives, (3) to demonstrate the use of a restoration decision-making tool, Robust Offsetting (RobOff), (4) to investigate the influence of climatic and edaphic factors on reforestation initiative, (5) to assess reforestation initiative success, and (6) to assess the impact of drought on reforestation initiative. To address this aim, the following objectives were developed:

1. To unravel the confusion caused by the inconsistent use of terminologies describing different reforestation initiatives.

2. To reflect on motivations behind recent reforestation initiatives (2000-2016).

3. To illustrate the use of RobOff to efficiently allocate resources in a large-scale restoration project with two habitats, multiple restoration alternatives and goals (biodiversity, carbon stock and employment) and a limited budget.

4. To assess the influence of microtopographic position, and soil physical and chemical properties on the growth of the four most dominant planted tree species at Buffelsdraai Landfill Site.

5. To assess early progress of the BLSCRP in terms of tree species diversity, vegetation structure and ecological processes (e.g., pollination) needed for creation of a self-sustaining, functional forest ecosystem that is resilient to disturbance, and the presence of IAPs.

6. To assess the effect of climate change induced drought on tree sapling survival and growth of the four most dominant planted tree species at Buffelsdraai Landfill Site.

1.3. Thesis outline

This thesis is based on five research chapters and written in paper format. Information for manuscripts that have been published and those in preparation for submission is provided on the title page of each chapter. Chapter 1 provide a short background, study motivation and a brief thesis outline. Chapter 2 addresses the first two objectives through literature review of issues pertaining terminologies describing different reforestation initiatives and motivations behind recent reforestation studies. Chapter 3 addresses the third objective through the use of a new publicly available restoration planning decision-making tool, RobOff. Chapter 4 addresses the fourth objective through the field study conducted at Buffelsdraai Landfill Site. This chapter assessed the effect of microtopographic position (upland and lowland areas) and soil physical and chemical properties (i.e., soil texture, soil moisture, pH, total nitrogen, phosphorus, potassium, calcium and magnesium) on the performance of the four most dominant planted tree species. Chapter 5 addresses the fifth objective through the field study that was also conducted at Buffelsdraai Landfill Site. The reforestation success was assessed in terms of vegetation structure, species richness and diversity indices, IAPs, and key ecological processes. Chapter 6 addresses the sixth objective through the field study conducted at Buffelsdraai Landfill Site. Drought effect on reforestation was assessed in terms of tree saplindsurvival and species growth rate. Chapter 7 provides synthesis of the entire study by integrating chapter 2 to 5 to address the overall aim of the study. It also provides conclusions and recommendations for the current and future reforestation initiatives elsewhere in the world.

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CHAPTER 2: RECENT ADVANCES IN REFORESTATION, AND A CALL FOR A CONSISTENT USE OF TERMINOLOGIES IN THE FIELD: A REVIEW

This chapter is based on

Lutendo F. Mugwedi, Mathieu Rouget, Benis Egoh, Rob Slotow (**In prep for submission to Restoration Ecology Journal**) Recent advances in reforestation, and a call for a consistent use of terminologies in the field: a review

Abstract

Human population increase, unsustainable land use practices, and extreme stress of climate change related impacts, have resulted in natural forest loss and degradation at an alarming rate of five million hectares per year from 1990 to 2015. Some of the forest loss and degradation consequences include a decline in biodiversity and ecosystem services provision, food insecurity, and social and political instability. In recent years, reforestation of degraded forest has emerged as a key component of biodiversity conservation, ecosystem services provision and climate change mitigation. This review is two-fold. First, the inconsistent use of terminologies describing different reforestation initiatives in the literature led to confusions amongst restoration scientists, restoration managers, policymakers, and other stakeholders (e.g., private land owners and local communities). Therefore, we reviewed terminologies describing different reforestation initiatives. Second, we assessed the motivations behind reforestation initiatives in recent literature (2000 to 2016). This is because the Forest Landscape Restoration approach formulated in 2000 called for reforestation initiatives to focus on achieving multiple reforestation goals. We found 14 studies that had 10 terminologies describing different reforestation initiatives. These studies provided either a definition or an explanation of the terminologies with no reference to any other literature. In this review, these terminologies were classified into five reforestation initiatives based on their motivations, namely, (1) *Creation* or *Fabrication*, *Reallocation* and *Replacement*; (2) *Ecological engineering*; (3) *Ecological restoration*; (4) *Reclamation*, *Reconstruction*, *Remediation*, *Renewal* or *Redemption*; (5) *Rehabilitation*. Results from the recent reforestation initiatives showed that, reforestation initiatives are taking place in six biogeographic realms of the world (i.e., Afrotropic, Australasia, Indo-Malay, Nearctic, Neotropic and Palearctic. Agriculture was found to be the main forest degrading factor. Most of the recent reforestation initiatives are planting a higher diversity of native tree species to achieve multiple ecological and socio-economic goals such as biodiversity conservation, climate change mitigation and adaptation, ecosystem services and economic benefits (e.g., employment opportunities). The consistent use of terminologies describing different reforestation initiatives would align global reforestation activities, and allow for effective development of a global learning network around reforestation. Reforestation initiatives that aim to achieve multiple reforestation goals would enhance the long-term sustainability of forest ecosystems.

Keywords: Biodiversity conservation; Ecosystem services; Ecological engineering; Forest degradation; Reallocation; Reclamation; Rehabilitation; Restoration.

2.1. Introduction

Globally, natural forest area is decreasing due to deforestation, while the planted forest area is increasing as a result of reforestation initiatives. Natural forest (hereafter referred to as 'native forest') is categorized into "primary forest" and "other naturally regenerated forest" (FRA 2015). Primary forest is defined as "naturally regenerated forest of native species, where there are no clearly visible indicators of human activities, and the ecological processes are not significantly disturbed". Other naturally regenerated forest is defined as "naturally regenerated forest where there are clearly visible indicators of human activities". Planted forest is defined as "forest predominantly composed of trees established through planting and/or deliberate seeding" (FRA 2015). Reforestation is defined as the establishment of forests (i.e., through active or passive actions) on lands that historically had forests, but are now degraded or have been converted to other land use (after Lund 2014). Despite all the conservation efforts in place, native forest loss and degradation is occurring on a continuous basis, especially in South America and Africa due to conversion to agricultural land. An annual net forest loss of 3.1 and 2.2 million hectares were recorded in Africa and South America, respectively, between 2010 and 2015 (FAO 2016). This loss was exacerbated by rapid and extreme stress of climate change related impacts such as drought, and disease and pest outbreaks (e.g., Ellison et al. 2005; Kirilenko and Sedjo 2007). However, in Europe, there has been a slight increase in native forest area (0.01 million ha/year between 2010 and 2015) (FAO 2016). Planted forest area is increasing as a result of commercial production (e.g., timber), climate change mitigation (carbon sequestration and storage) and adaptation (e.g., flood attenuation), and conservation (FAO 2016). In this review, reforestation initiatives for commercial production purpose were excluded.

Reforestation initiatives have different motivations, and this led to the development of terminologies describing different reforestation initiatives (Cooke and Johnson 2002; McDonald 2009). Some of the developed terminologies include creation, reallocation, rehabilitation, reclamation, renewal and replacement (Bradshaw and Chadwick 1980; Aronson et al. 1993; SER 2004; Clewell and Aronson 2013). These terminologies have been used inconsistently (Aronson et al. 1993; Cooke and Johnson 2002; Li 2006; Hobbs et al. 2011), thus causing confusion amongst restoration scientists, restoration managers, policymakers, and other stakeholders (e.g., private land owners and local communities). Examples of inconsistent use include the synonymous use of restoration, reclamation and reallocation (Aronson et al. 1993), and the use or *Rehanbilitation*, *Restoration* and *Reclamation* all implying land replacement (Gould 2011). The confusion leads to disconnection between restoration scientists, restoration managers (Hart et al. 2015), policymakers, and other stakeholders. This makes it difficult, (1) to develop science-based strategic approaches, (2) to align global reforestation activities, and, (3) to develop global learning networks around reforestation. Furthermore, it is also difficult to develop conceptual, institutional, legal and operational bases for the policies, and for the planning, implementation, monitoring and evaluation of reforestation initiatives (after Chazdon et al. 2016). For example, in order to account and credit carbon sinks under a cap-and-trade system of the United Nations Framework Convention on Climate Change and the Kyoto Protocol, hundreds of pages have been written trying to distinguish between afforestation and reforestation. Defining these terminologies has required people to think about the historical land cover type, but a critical question was "how far back in history can one go?" (Reilly et al. 2007). This is because humans and climate change have altered forest ecosystems for millennia (Hanberry et al. 2015).

Earlier reforestation initiatives focused on establishing a few tree species (mostly exotic tree species) to reinstate one, or more, lost or lacking forest service (Dudley et al. 2005), sometimes coupled with socio-political aims such as employment provision (Stanturf 2005). These initiatives were largely limited to a site level (Dudley et al. 2005). Examples of target forest services include provisioning services (e.g., food and timber), regulating services (e.g., water purification and urban climate regulation), supporting services (e.g., wildlife habitat and food, and nutrient cycling), and cultural services (e.g., aesthetic enjoyment and spiritual fulfilment) (e.g., Richardson 1998; Saxena 2001; Dudley and Stolton 2005; Groninger 2005; Regato and Berrahmouni 2005; Thompson et al. 2009; Negi et al. 2015; Oldfield 2015). These initiatives failed to address the drivers behind forest habitat loss and degradation (Dudley et al. 2005). On the other hand, the use of exotic tree species in reforestation initiatives has been a hot topic of debate over the years (Locatelli et al. 2015). This is because, although exotic tree species offer benefits such as timber, carbon sequestration and storage, and employment opportunities, there are many negative ecological and socio-economic impacts that they bring (Smith 2002). The impacts include, but are not limited to, biodiversity loss, reduced wildlife habitat and food (Lamb and Gilmour 2003), increased susceptibility of forests to climate change, and disease and pest outbreaks (Biringer and Hansen 2005; Pachauri and Reisinger 2007), reduced water production, and some exotic tree species becoming invasive (Richardson 1998; Lamb and Gilmour 2003) and changing fire regimes (Brooks et al. 2004), reduced livelihoods (e.g., food and medicinal plants) to local communities (Smith 2002; Zomer et al. 2006; Fernandes et al. 2016), and land tenure conflicts (Smith 2002).

Native forests have significant contributions to biodiversity conservation, human well-being and economic development through the supply of essential ecosystem services such as food, clean water and timber (MacDicken et al. 2015; FAO 2016). The loss of native forests has substantial and widespread consequences on climate, hydrological cycle, soil, biodiversity, and key ecosystem services that sustain societies, especially impoverished communities who rely on forests for livelihoods (e.g., Von Maltitz et al. 2003; Dudley et al. 2005; Ellison et al. 2005; Kirilenko and Sedjo 2007). Different measures such as protection, sustainable management and reforestation are needed to maintain biodiversity, and to ensure a continued supply of ecosystem services (Dudley et al. 2005; Thompson et al. 2009; Aronson and Alexander 2013). Among these measures, reforestation through planting of native tree species is now globally recognized as a key component in biodiversity conservation, and critical for the long-term sustainability of ecosystems (Aronson and Alexander 2013). The reforestation concepts discussed in this review are applicable to other terrestrial ecosystems (e.g., grassland), however, in this review, they are aligned to forest ecosystem.

There has been a call for recent reforestation initiatives to adopt a Forest Landscape Restoration (FLR) approach that was formulated in the year 2000. The FLR is defined as "a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscapes" (IUCN and WWF 2000). The FLR approach was followed by an influential publication that documented experiences from reforestation initiatives that tested the approach (Mansourian 2005). This approach focuses at a landscape-level, because at a landscape level, more comprehensive technical reforestation interventions can simultaneously contribute to both biodiversity conservation and human well-being (Dudley et al. 2005).

When carrying out reforestation, knowledge of the degrading factor/s, alternative reforestation actions, and understanding of the goals commonly achieved through alternative reforestation actions, is needed (Palma et al. 2015). The alternative reforestation actions differ in terms of intervention level needed, species type (e.g., exotic vs. native) and number of species used (single or multiple), and time and resources needed to achieve the desired outcomes (Rey Benayas 2005; Palma et al. 2015). Two main alternative reforestation actions include active and passive reforestation actions. Active reforestation action can be defined as activities that speed up natural processes. Examples of techniques used under active reforestation action include seeding and sapling transplanting (Rey Benayas 2005; Palma et al. 2015). Active reforestation action is often favoured to achieve the desired outcomes within a short time frame or in severely degraded sites that cannot recover naturally (Rey Benayas 2005). This action can also provide more employment opportunities (e.g., site preparation, seed collection and seedling growing, sapling transplanting and site maintenance) to local communities than the passive action (Rey Benayas 2005). Passive reforestation action can be defined as the reduction or removal of forest degrading activities to allow natural regeneration of forest (Morrison and Lindell 2011). Examples of techniques used under passive reforestation action include IAP management and grazing/browsing exclusion (Morrison et al. 2005; Rey Benayas et al. 2008). Passive reforestation action has been successful in reforesting more land, and at a lower cost, than active reforestation action (Rey Benayas 2005). However, this action is more successful in low or moderately degraded forest that still have propagules, i.e., live tree stumps and soil seed bank (Rey Benayas 2005).

The scope of this review is two-fold. First, given a number of publications on terminologies describing different reforestation initiatives, these terminologies were reviewed, and the reforestation initiatives under which these terminologies should be applied to, are discussed. This was done to encourage a consistent use of these terminologies amongst restoration scientists, restoration managers, policy makers and other stakeholders. Second, following the call for recent reforestation initiatives to adopt the FLR approach, the motivations behind recent reforestation initiatives were assessed in recent literature (2000 to 2016). To achieve this aim, the following key questions were developed; (1) What was the source or reason for forest loss and degradation in the first place? (2) What was the goal/s of reforestation intervention? (3) What were the anticipated benefits? (4) In which biogeographic realms were these initiatives implemented? (5) Which alternative reforestation actions and techniques were used? (6) Which species types (exotic, native or both), and how many species, were used?

2.2. Materials and methods

2.2.1. Scope of the study

As the focus of this study is on reforestation, only studies that reported on reforestation initiatives on sites where the historical land cover was forest were considered.

2.2.2. Literature survey

Thomson Reuters Web of Science database was used to search for peer-reviewed literature published in English language (https://apps.webofknowledge.com). Research papers, books, reviews, and proceedings papers in the fields of ecology, environmental science, biodiversity, and forestry, were considered.

To achieve the first aim, literature published from 1980 to 2016 was considered. The year 1980 was chosen as the starting point, because an influential work that listed and defined some of the terminologies describing different reforestation initiatives was published (Bradshaw and Chadwick 1980). The following keywords were used; “ecological restoration”, “ecological engineering”, “reconstruction”, “recreation”, “rehabilitation”, “reclamation”, “remediation”, “replacement”, “reallocation”, “afforestation”, “reforestation”, “revegetation”. Only articles that did not reference other sources in their terminology definition or explanation were selected, i.e., we were seeking the source article for a terminology. In cases where studies referenced other sources in their terminology definition or explanation, the original sources were traced. Literature from the Society for Ecological Restoration (SER) website (www.ser.org) was also used, because their ecological restoration definition is widely accepted and cited definition (e.g., Clewell and Aronson 2013, Hart et al. 2015; Aronson et al. 2016, Corlett 2016, Farag et al. 2016).

To achieve the second aim, literature published from 2000 to 2016 was considered. The year 2000 was chosen as the starting point, because a call for recent reforestation initiatives to adopt the Forest Landscape Restoration approach started in the year 2000 (IUCN and WWF 2000). The following keywords were used; “ecological restoration”, “ecological engineering”, “reconstruction”, “recreation”, “rehabilitation”, “reclamation”, “remediation”, “replacement”, “reallocation”, “afforestation”, “reforestation”, “revegetation”, “active/passive restoration”, “tree planting”, “seedling/sapling transplanting”, “seeding”, “invasive alien plants control/clearing/management”. It is acknowledged that this search may have missed some of the recent reforestation initiatives that never get published in scientific literature, but are documented on the implementing agencies' website or annual reports. The initial search yielded 1583 articles. The search results were screened to identify articles that reported on reforestation. One hundred and sixty nine studies were found and read in detail to identify relevant articles based on the following criteria; (1) studies that reported on reforestation initiatives initiated after the year 2000 or initiated before 2000, but the reforestation efforts still continued post-2000; and (2) studies that mentioned historical land cover, degrading factors, reforestation actions employed, and species type (i.e. native, exotic or both) and number used (single or multiple). Studies that matched the selection criteria were 33 (Table S2.1). Out of 33 studies, 10 studies were selected to showcase how the proposed terminologies could be used to reclassify terminologies used where they were not applicable. Data were then extracted from 33 studies based on six key questions (Table S2.2).

2.3. Results

2.3.1. Terminologies used to describe different reforestation initiatives

A total number of 14 studies that had 10 terminologies describing different reforestation initiatives were found. These studies provided either a definition or an explanation of the terminologies with no reference to any other literature. The 10 terminologies were classified into five reforestation initiatives based on similarities or differences of the reforestation initiative motivations, namely, (1) '*Creation* or *Fabrication*', '*Reallocation*' and '*Replacement*' which share a common motivation of establishing a forest ecosystem that has no resemblance to the historic one in terms of structure and function, in order to achieve ecological or socio-economic benefits; (2) '*Ecological engineering*' which includes the use of biological and engineering solutions (e.g., tree planting and the use of jute netting to stabilize soil on steep slopes) to establish a forest ecosystem that achieves both ecological and socio-economic benefits; (3) '*Ecological restoration*' which aims to reinstate the pre-degraded forest ecosystem conditions (species composition, structure and functions); (4) '*Reclamation*', '*Reconstruction*', '*Remediation*', '*Renewal* or *Redemption*' share a common motivation of returning land to useful state; and (5) '*Rehabilitation*' which aims to return forest to some of the desired structure and functions of the degraded ecosystem with less emphasis on pre-degraded forest ecosystem conditions (Table 2.1). The attributes of each reforestation initiative are summarized in Table 2.2. Out of the 33 reviewed studies, 19 were consistent with the proposed terminology and 14 were not (Table 2.3). Some of the reclassified studies used terminologies interchangeably (e.g., *Ecological Restoration / Rehabilitation / Reclamation*).

Table 2.1: The 10 commonly used terminologies describing different reforestation initiatives classified into five reforestation initiatives based on similarities and differences of their motivations. Definitions are within the quotation marks.

Terminologies	Definition or explanation	References
1. Ecological engineering	"Is the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both."	Mitsch and Jørgensen 2003
	"Involves manipulation of natural materials, living organisms and the physical chemical environment to achieve specific human goals and solve technical problems."	SER 2004
	"Manipulation and use of living organisms or other materials of biological origin to solve problems that affect people."	Clewell and Aronson 2013
2. Ecological restoration	Restoration may be used only where land is to be returned to its former use.	Bradshaw and Chadwick 1980
	The reinstatement of the original ecosystem structure and function.	Bradshaw 1996; Dyer 1990
	"The return of an ecosystem to a close approximation of its condition prior to disturbance."	National Research Council 1992
	"The manipulation of organisms and ecological processes to create self-organizing ecosystems that resemble pre-disturbance structure and functioning and promote conservation of biodiversity."	Allen et al. 2001
	"The process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed." A fundamental focus is on pre-degraded conditions as reference with the emphasis on the re-establishment of biotic integrity in terms of species composition, community structure, ecosystem processes, productivity and services.	SER 2004
	Repairing a disturbed site to a near natural state.	Rohr et al. 2016

3. Creation or Fabrication/ Reallocation / Replacement		
a. Creation or Fabrication	Establishment of forest ecosystem on the land that is devoid of vegetation for mitigation purpose.	SER 2004
b. Reallocation	Assignment of degraded land to a new use that does not necessarily contain the structure and function of the original ecosystem. Energy subsidies, e.g., nutrients and water may be required.	Aronson et al. 1993
c. Replacement	"To provide or procure a substitute or equivalent in place of."	Bradshaw 1996
	Substitution of species that are being displaced by climate by new species (or climate change adapted genotypes of the displaced species.	Stanturf et al. 2014
4. Reclamation / Reconstruction / Remediation / Renewal or Redemption		
a. Reclamation	Reclamation is often used where some new use of the land will be involved.	Bradshaw and Chadwick 1980
	The process of bring land to a useful state with no intention of returning to an original state.	Bradshaw 1996
	Land revegetation or management goal that has lower species diversity and may include introduction of new species to the area.	Allen et al. 2001
	Restoration of land for a useful purpose like soil stabilization, public safety and aesthetic improvement.	Mitsch and Jørgensen, 2003

	Reparation of degraded land for a useful purpose such as aesthetic improvement, stabilization of the terrain to ensure public safety. This could be achieved with one or a few species.	SER 2004
	Reparation of severely degraded land usually with no vegetation maybe as a result of mining, and the processes may include soil amelioration to improve physical, chemical and biological status of the land.	Stanturf et al. 2014
b. Reconstruction	Restoration of native plant species on land under other land use, e.g., agriculture.	Stanturf et al. 2014
c. Remediation	It is a process of rectifying or making good.	Bradshaw 1996
	Removing or reducing contamination.	Wagner et al. 2016
d. Renewal or Redemption	Used in the repair of degraded land in order to allow flexibility in planning a new purpose for the land.	Bradshaw and Chadwick 1980
5. Rehabilitation	Rehabilitation is sometimes confined to improvement of visual nature.	Bradshaw and Chadwick 1980
	An action that aims to repair damaged ecosystem functions and productivity to benefit the people.	Aronson et al. 1993
	Action of restoring an ecosystem to previous conditions with little or no expectations of returning the original conditions.	Bradshaw 1996
	All activities that are undertaken to make land useful again.	Burke 2003
	"Creation of an alternative ecosystem following a disturbance, different from the original and having utilitarian rather than conservation values."	Allen et al. 2001

	”Emphasizes the reparation of ecosystem processes, productivity and services.”	SER 2004
	Actions aimed at repairing structure and function of an ecosystem.	Mitsch 2012
	Restoration of desired species composition, structure or functions to an existing degraded ecosystem.	Stanturf et al. 2014

Table 2.2: Summary of reforestation initiatives attributes.

Reforestation initiative	Description	Degrading factor	Degradation level	Reforestation techniques	Species type	No. of species	Cost and time	Ecological and socio-economic value
1. *Creation* or *Fabrication* / *Reallocation* / *Replacement*	Establishment of forest on land that has no vegetation.	Agriculture, mining and other anthropogenic factors	High	Seeding and sapling transplanting	Exotic / native or both	One or a few	High	Lower due to single or fewer species used
2. *Ecological engineering*	Establishment of forest through the use of biological and engineering solutions.	Agriculture, mining and other anthropogenic factors	High	Seeding and sapling transplanting including engineering solutions	Exotic / native or both	One, a few or multiple	High cost and less time	Lower to medium depending on the number of species used
3. *Ecological restoration*	Establishment of pre-degraded forest conditions.	Agriculture, mining, other anthropogenic factors and IAPs	Low to High	Degrading factor exclusion, Seeding and sapling transplanting	Native	Multiple	Low to high depending on degradation level	High
4. *Reclamation* / *Reconstruction* / *Remediation* / *Renewal* or *Redemption*	Establishment of forest to make land useful.	Agriculture, mining and other anthropogenic factors	High	Seeding and sapling transplanting	Exotic / native or both	One or a few	High	Lower due to single or fewer species used
5. *Rehabilitation*	Establishment of forest to reinstate a few pre-degraded forest conditions.	Agriculture, mining, other anthropogenic factors and IAPs	Low to high	Degrading factor exclusion, Seeding and sapling transplanting	Exotic / native or both	Multiple	Low to high depending on degradation level	Medium

Table 2.3: Terminology reclassification of the recent reviewed reforestation initiatives based on our proposed list of terminologies. Only 14 studies that were wrongly classified were found.

Country	Terminology used	Proposed reclassification	Justification	Reference
1. China	*Afforestation / Reforestation*	*Rehabilitation*	Planting of multiple native and exotic tree species for climate change mitigation.	Ren et al. 2011
2. China	*Ecological restoration*	*Rehabilitation*	Planting of multiple native and exotic tree species to achieve climate change mitigation, provisioning and regulating ecosystem services.	Lin et al. 2012
3. Czech Republic	*Afforestation*	*Rehabilitation*	Planting of multiple native and exotic tree species for climate change mitigation through carbon storage.	Kotecky 2012
4. Hawaii	*Reforestation*	*Ecological Restoration*	Planting of multiple native tree species to achieve biodiversity and ecosystem functions, and regulating ecosystem services.	Li et al. 2010
5. India	*Ecological restoration / Reclamation / Rehabilitation*	*Rehabilitation*	Planting of multiple native and exotic tree species to conserve soil	Singh et al. 2012
6. India	*Ecological restoration / Rehabilitation*	*Ecological restoration*	Planting of multiple native tree species to achieve biodiversity and ecosystem functions, climate change mitigation, and provisioning, supporting and regulating ecosystem services.	Negi et al. 2015
7. Kenya	*Rehabilitation*	*Reclamation / Reconstruction / Remediation / Renewal or Redemption*	Planting of multiple exotic tree species to make land useful.	Kithiia and Lyth 2011
8. Kenya	*Ecological Restoration*	*Rehabilitation*	Planting of multiple native and exotic tree species to restore biodiversity and ecosystem functions.	Mullah et al. 2014

9. Namibia	*Rehabilitation*	*Reclamation / reconstruction / remediation / renewal or redemption*	Planting of multiple exotic tree species to make land useful.	Burke 2003
10. South Africa	*Ecological Restoration*	*Rehabilitation*	Planting of single native tree species to achieve biodiversity, climate change mitigation (carbon storage), provisioning and regulating ecosystem services and employment creation.	Mills and Cowling 2006
11. Spain	*Ecological Restoration / Rehabilitation / Reclamation*	*Ecological restoration*	Planting of multiple native tree species to extract heavy metal, restore biodiversity, provisioning and regulating ecosystem services.	Carreira et al. 2008
12. Spain	*Ecological restoration / Reclamation*	*Rehabilitation*	Planting of multiple native tree species to achieve regulating ecosystem services and to provide wildlife habitat.	De Torre et al. 2015
13. Uganda	*Ecological Restoration*	*Rehabilitation*	Planting of multiple native and exotic tree species to achieve biodiversity and ecosystem functions restoration, and provisioning ecosystem services.	Omeja et al. 2009
14. USA	*Ecological restoration / Reclamation*	*Rehabilitation*	Planting of multiple native tree species to re-establish plant community and hydrological processes.	Fields-Johnson et al. 2012

2.3.2. Motivations behind recent reforestation initiatives (2000 – 2016)

2.3.2.1. Geographic distribution of the recent reforestation initiatives

The reforestation initiatives were implemented in all the biogeographic realms of the world (except Oceania, hence excluded from the results) (Table 2.4). Overall, the highest number of reforestation initiatives were recorded in the Afrotropic biogeographic realm (10 studies), followed by the Neotropic, Nearctic, Palearctic, Australasia and Indo-Malay biogeographic realms. *Ecological restoration* and *Rehabilitation* initiatives were recorded across all the biogeographic realms. Under the *Ecological restoration* initiative, the highest number of studies was recorded in the Neotropic biogeographic realm (seven studies), followed by the Afrotropic, Nearctic, Australasia, Indo-Malay, and Palearctic biogeographic realms. Under the *Rehabilitation* initiative, the highest number of studies was recorded in the Afrotropic biogeographic realm (six studies), followed by the Australasia and Indo-Malay biogeographic realms, and then Nearctic and Neotropic biogeographic realms. Under the *Reclamation* / *Reconstruction / Remediation / Renewal* or *Redemption* initiative, studies were recorded only in the Nearctic and Palearctic biogeographic realms (two studies each). Under the *Creation or Fabrication / Reallocation* / *Replacement* initiative, only one study was recorded in the Palearctic biogeographic realm, and no study was recorded under the *Ecological engineering* initiative.

2.3.2.2. Drivers of forest loss and degradation

Overall, the main driver of forest loss and degradation was agriculture (19 studies), followed by other anthropogenic factors, mining, and then IAPs (Table 2.4). Forest loss and degradation drivers appear to have little influence on the choice of reforestation initiative, because *Ecological restoration* and *Rehabilitation* initiatives were implemented irrespective of the degrading factor type (e.g., mining). Under the *Ecological restoration* and *Rehabilitation* initiatives, the main driver of forest loss and degradation was agriculture (11 and six studies, respectively). The results showed that *Creation* or *Fabrication / Reallocation* / *Replacement* and *Reclamation* / *Reconstruction / Remediation / Renewal* or *Redemption* initiatives were implemented following substantial disturbance (e.g., mining and infrastructure development).

2.3.2.3. Reforestation actions and techniques, species type and number used

Overall, active reforestation through sapling transplanting technique was the most employed action in recent reforestation initiatives (27 studies). Passive reforestation through IAP management and exclusion of degrading factors, and a combination of active and passive reforestation actions (e.g., IAP management coupled with seeding/sapling transplanting) were also used (Table 2.4). Sapling transplanting was highest under the *Ecological restoration* initiative (12 studies), followed by the *Rehabilitation*, *Reclamation* / *Reconstruction / Remediation / Renewal* or *Redemption*, and then *Creation* or *Fabrication* / *Reallocation* / *Replacement* initiatives. The use of a higher diversity of native tree species is now widely adopted across different reforestation initiatives (28 studies), followed by the use of both exotic and native species, except in the *Ecological restoration* initiative. It was interesting to note that no study reported on the use of exotic tree species alone.

2.3.3.4. Goals and benefits of reforestation initiatives

The recent reforestation initiatives were designed to achieve multiple ecological and socio-economic goals simultaneously. Biodiversity conservation and ecosystem functions (BCEF) recovery was the main reforestation goal in most reforestation initiatives (30 studies), followed by supporting services, climate change mitigation, provisioning services, regulating services, economic benefits and then cultural services. The *Ecological restoration* initiative had the highest number of studies on the following goals; BVEF (15 studies), cultural services (three studies), economic benefits (five studies), provisioning services (seven studies), regulating services (six studies), and supporting services (eight studies). The *Rehabilitation* initiative had the highest number of studies on the climate change mitigation goal (10 studies) (Table 2.4). Reforestation goals linked benefits included, but were not limited to, fauna and flora conservation, carbon sequestration and storage, climate regulation, employment and carbon credits, food, medicinal plants, timber, firewood, soil stabilization, flood control and water purification, wildlife habitat and food, aesthetic, ecotourism, education, recreation, and spiritual fulfilment (Table 2.5).

Table 2.4: (a) Reforestation initiatives distribution across biogeographic realmsof the world and forest degrading factors; (b) Reforestation initiatives and their associated reforestation techniques, species type used and (d) target reforestation goals from recent reforestation studies, 2000 to 2016 (*n* = 33 studies). IAPs – Invasive alien plants; OAFs – Other anthropogenic factors; EDF – Exclusion of degrading factors; IAPM – Invasive alien plant management; IAPM+S – Invasive alien plant management plus seeding; IAPM+ST – Invasive alien plant management plus sapling transplant; ST – Sapling transplanting; E – Exotic tree species; N – Native tree species; BCEF. – Biodiversity conservation and ecosystem functions; CCM – Climate change mitigation; CS – Cultural services; EB – Economic benefits; PS – Provisioning services; RS – Regulating services; SS – Supporting services.

a. Reforestation initiatives	Study biogeographic realms						Forest degrading factors			
	Afrotropic	Australasia	Indo-Malay	Nearctic	Neotropic	Palearctic	Agriculture	Mining	OAF	IAPs
Creation or Fabrication / Reallocation / Replacement						1 (3%)	1 (3%)			
Ecological engineering										
Ecological restoration	4 (12.1%)	1 (3%)	1 (3%)	2 (6.1%)	7 (21.2%)	1 (3%)	11 (33.3%)	1 (3%)	3 (9.1%)	2 (6.1%)
Reclamation / Reconstruction / Remediation / Renewal or Redemption				2 (6.1%)		2 (6.1%)	1 (3%)	2 (6.1%)	1 (3%)	
Rehabilitation	6 (18.2%)	2 (6.1%)	2 (6.1%)	1 (3%)	1 (3%)		6 (18.2%)	1 (3%)	5 (15.2%)	1 (3%)

Total	**10 (30.3%)**	**3 (9.1%)**	**3 (9.1%)**	**5 (15.2%)**	**8 (24.2%)**	**4 (18.2%)**	**19 (57.6%)**	**4 (18.2%)**	**9 (27.3%)**	**3 (9.1%)**

b. Reforestation initiatives	Alternative reforestation techniques				Species type used				Reforestationn goals						
	EDF	IAPM	IAPM+S	IAPM+ST	ST	E	N	E/N	BCEF	CCM	CS	EB	PS	RS	SS
Creation or Fabrication / Reallocation / Replacement					1 (3%)			1 (3%)							1 (3%)
Ecological engineering															
Ecological restoration	1 (3%)	2 (6.1%)		1 (3%)	12 (36.4%)		17 (51.5%)		15 (45.5%)	5 (15.2%)	3 (9.1%)	5 (15.2%)	7 (21.2%)	6 (18.2%)	8 (24.2%)
Reclamation / Reconstruction / Remediation / Renewal or Redemption					4 (12.1%)		3 (9.1%)	1 (3%)	3 (9.1%)	1 (3%)			2 (6.1%)	1 (3%)	3 (9.1%)
Rehabilitation			1 (3%)	1 (3%)	10 (30.3)		8 (24.2%)	3 (9.1%)	12 (36.4%)	10 (30.3%)	1 (3%)	4 (12.1%)	4 (12.1%)	4 (12.1%)	7 (21.2%)
Total	**1 (3%)**	**2 (6.1%)**	**1 (3%)**	**2 (6.1%)**	**27 (81.8%)**	**0 (0%)**	**28 (84.8%)**	**5 (15.2%)**	**30 (90.9%)**	**16 (48.5%)**	**4 (12.1%)**	**9 (27.3%)**	**13 (39.4%)**	**11 (33.3%)**	**19 (57.6%)**

Table 2.5: Reforestation goals and their associated benefits from recent studies, 2000 – 2016 (*n* = 33 studies).

Reforestation goals	Reforestation benefits	Studies %
1. Biodiversity conservation and ecosystem functions	Fauna and flora diversity	90.9
	Soil formation, nutrient cycling and primary production	90.9
	Habitat connectivity	12.1
	Ecosystem resilience	6.1
2. Climate change mitigation	Carbon storage	24.2
3. Cultural services	Aesthetic	9.1
	Ecotourism	6.1
	Education	6.1
	Recreation	6.1
	Partnership	6.1
4. Economic benefits	Employment	9.1
	Carbon credits	6.1
5. Provisioning services	Food	6.1
	Firewood	15.2
	Fodder	6.1
	Compost	3.0
	Timber	3.0
	Water quantity	33.3
6. Regulating services	Air quality regulation	3.0
	Climate regulation	6.1
	Flood control	6.1
	Pollutants control (e.g., heavy metals in the soil)	6.1
	Soil conservation (e.g., soil erosion control)	21.2
	Water quality	33.3
7. Supporting service	Wildlife habitat and food	27.3

2.4. Discussion

2.4.1. Terminologies describing different reforestation initiatives and advances in recent reforestation initiatives

In this review, terminologies describing different reforestation initiatives were classified into five reforestation initiatives that are discussed below. This classification was based on similarities and differences between reforestation initiatives' motivations. In order to encourage the consistent use of terminologies, studies that used used wrong terminologies were reclassified following the description of five terminologies proposed in this study. For example, Burke (2003) described *Rehabilitation* as all activities that are implemented to make land useful again. This description was given with reference to the reforestation initiative at southern Namib mine land in Namibia (Burke 2003). Based on the results from this review, the reforestation initiative implemented at southern Namib mine land qualifies to be called *Reclamation / reconstruction / remediation / renewal or redemption* initiative.

2.4.1.1. Creation or *Fabrication / Reallocation / Replacement* initiative

Creation or *Fabrication* / *Reallocation* / *Replacement* share a common goal of establishing a forest ecosystem (that has no resemblance to the historic one in terms of structure and function) on land that does not have any vegetation, and the intended goals may include ecological and socio-economic benefits (e.g., Aronson et al. 1993; Stanturf et al. 2014). This is because the loss of forest as a result of activities such as mining and infrastructure development cause a substantial change to the environment, thus sometimes necessitating the establishment of a natural system, which could be a different ecosystem to what may have historically occurred there (Bradshaw 1996). According to SER (2004), the term *Creation* (sometimes referred to as '*Fabrication*') has been receiving increased attention in the last decade, especially in climate change mitigation projects. The *Creation* or *Fabrication* / *Reallocation* / *Replacement* initiative uses an active reforestation action through sapling transplanting technique. Any tree species (exotic, native or both) that can achieve the expected reforestation outcomes can be used. In some cases, there is a permanent on-going management interventions such as watering and fertilization to keep the forest productive (Aronson et al. 1993). The goals and benefits of this reforestation initiative, documented in this review, were supporting services (soil stabilization) and cultural services (recreation) (e.g., de Torre et al. 2015).

2.4.1.2. Ecological engineering initiative

The *Ecological engineering* initiative's motivation is to restore ecosystems that have been substantially altered by human activities (e.g., mining and infrastructure development) and to develop new ecosystems that contribute to both ecological and socio-economic benefits (Mitsch 2014). This reforestation initiative differs from the *Ecological restoration* initiative, because it incorporates natural materials and engineering solutions to achieve ecological and socio-economic goals (SER 2004; Mitsch 2012; Clewell and Aronson 2013). The *Ecological engineering* initiative is implemented on a forest ecosystem that has been substantially disturbed by anthropogenic factors such as intensive agriculture, mining and industrial pollution (Fanta 1994). For example, in agricultural landscapes that are affected by salinity, it can be very challenging to find tree species that grow in saline soils. However, this challenge can be solved by breeding of saline tolerant species (e.g., Dale and Dieters 2007). Another example of engineering solution can include either seeding, sapling transplanting, or the use of both reforestation techniques, coupled with engineering structures such as jute netting to stabilize soil in steep slopes (Papathoma-Koehle and Glade 2013).

Mitsch (2012) wrote that *Ecological restoration* initiative should be the heart and soul of *Ecological engineering* initiative. However, Aronson et al. (2016) argued that, although *Ecological engineering* and *Ecological restoration* initiatives draw some of the approaches and technologies from each other to restore a degraded ecosystem, and sometimes achieve similar benefits, they are not the same. They gave clear distinctions to show that these fields are different. For example, the end point in *Ecological engineering* initiative is predictable, because it is designed to provide desired services, while the end point in *Ecological restoration* initiative is unpredictable, because it is regulated by the ecosystem itself (Aronson et al. 2016). Furthermore, *Ecological engineering* initiative is usually implemented in areas where there is a higher risk of disasters such as flooding and landslides (e.g., Donat 1995; Renaud et al. 2013). It was unfortunate that none of the recent reforestation initiatives reviewed in this study documented *Ecological engineering* initiatives. This could be due to the fact that most of the *Ecological engineering* projects are mostly implemented in other ecosystems (e.g., rivers and wetlands) (Mitsch et al. 2012).

2.4.1.3. Ecological restoration initiative

Ecological restoration is an initiative that aims to reinstate the pre-degraded ecosystem conditions (Bradshaw and Chadwick 1980; Aronson et al. 1993; SER 2004). Restoration of forest ecosystem (and other ecosystems) to its pre-degraded state raises a lot of question about the date of definition of the original ecosystem that the restoration activity aims to achieve (Hobbs et al. 2011). In some cases, the pre-degradation baseline state might be known, and it may still be possible to restore those conditions, but, in some sites, this may be impossible as a result of unprecedented changes in land use patterns, climate, and in biodiversity (Choi 2007; Hobbs and Cramer 2008; Clewell and Aronson 2013; Aronson et al. 2016). For example, in the early years, it was probably possible to restore ecosystems that had low species diversity (e.g., herbaceous marshes) back to their original structure and functioning (Aronson et al. 2016). However, *Ecological restoration* initiative still seems possible in projects that are implemented where the degradation level has not passed the irreversibility threshold (Aronson et al. 1993).

Some restoration scientists now consider *Ecological restoration*'s focus (to return to pre-degraded state) as out-dated (Hart et al. 2015). Balaguer et al. (2014) stated that the focus of *Ecological restoration* on pre-degraded state as reference system remains the cornerstone concept, and conceptual tool that sets aside *Ecological restoration* from other similar reforestation initiatives. Furthermore, Hart et al. (2015) warned that *Ecological restoration* should focus on the recovery of pre-degraded state, because, once the objective is no longer about the pre-degraded state, it is no longer an *Ecological restoration* initiative. Hanberry et al. (2015) suggested that rather than giving new reforestation initiatives new terminologies, it may be more useful to emphasize that *Ecological restoration* is still about the maintenance of biodiversity, ecosystem function, structure and processes that prepare the forest ecosystem for an uncertain future. Hart et al. (2015) argued that broadening the definition of *Ecological restoration* to accommodate new reforestation initiatives will add to the existing confusion among restoration scientists, managers and practitioners who develop strategic decisions based on regulatory guidelines that focus on pre-degraded state. Rather than expanding *Ecological restoration* definition, restoration scientists should develop new terminologies that describe the new reforestation initiatives (Hart et al. 2015).

Although ecosystems change over time, the use of reference site or pre-degraded state as a reference system is still critical in reforestation. This is because reference site does not only provide a bench mark, it also guides the planning, implementation, monitoring, and evaluation of reforestation projects that aim to achieve long-term effectiveness, success and broad-scope impact (Balaguer et al. 2014). *Ecological restoration* initiative can take either a passive action, active action, or a combination of both actions. This initiative also differs from the other reforestation initiatives, because only native tree species are used (Cooke and Johnson 2004; Bradshaw et al. 2012). Most of the species present prior to degradation are included, because it is presumed that their co-occurrence led to a sustainable ecosystem that existed before the disturbance (Aronson et al. 2016). However, it is difficult to plant all the tree species that are available in the region due to factors such as higher tree species richness (e.g., there are over 11000 tree species in Brazil, Hubbell et al. 2008), suppliers of seeds and saplings tend to target species that are easy to find and/or grow (e.g., Roy 2015; Mugwedi et al. 2017), while some species can be difficult to find (Brancalion et al. 2012). Most projects that aim to restore the pre-degraded forest conditions in Brazil are planting more than 100 native tree species (Colmanetti et al. 2016).

Although only native tree species are used under *Ecological restoration*, there is a possibility that Tree-preneurs with little knowledge on exotic tree identification can grow seedlings of exotic tree species. If seedlings of exotic tree species manage to slip in under the careful watch of the nursery manager and get planted in the field, one can ask if this initiative would be still eligible to be classified under *Ecological restoration*. Under this scenario, this would be regarded as unitentional planting. Therefore, this intiative should be classified under *Ecological restoration*, but exotic tree speceis should be removed once noticed. This is because Ecological restoration aims to establish pre-degraded forest conditions free of exotic tree species, including other exotic plants (e.g., grass and forb species). The reforestation initiative that uses exotic tree species intentionally should be classified under rehabilitation, because it has violated the aim of establishing pre-deraded conditions.

In this review, a higher focus on *Ecological restoration* initiative across biogeographic realms showed that reforestation managers have now realized the significance of regaining ecological integrity to ensure long-term forest sustainability and human well-being enhancement. After reviewing two decades of research on biodiversity and ecosystem functioning, Cardinale et al. (2012) found that species diverse communities are multifunctional, more stable and more productive. This is because differences in species functional traits allow interacting species to fully utilize their limiting resources. Furthermore, species rich communities with multiple functions are more resistance and resilient to extreme climate change related disturbances, such as pest and disease outbreaks, and changing fire patterns (Biringer and Hansen 2005; Pachauri and Reisinger 2007). For example, pest and disease outbreaks are reduced due to low availability of their hosts in highly diverse forests (Chapin III et al. 2000). Most of the recent *Ecological restoration* initiative projects documented in this review were planned to achieve multiple goals simultaneously (e.g., biodiversity conservation, climate change mitigation and adaptation and ecosystem services provision) (e.g., Smith et al. 2016; Bare et al. 2016).

2.4.1.4. Reclamation / reconstruction / remediation / renewal or redemption initiative

Reclamation / *Reconstruction* / *Remediation* / *Renewal* or *Redemption* share a common motivation of returning land to useful state, either to achieve ecological (e.g., soil stabilization or biodiversity through habitat provision) or socio-economic benefits (e.g., agriculture, employment creation or aesthetic improvement) (Bradshaw 1996; SER 2004; Stanturf et al. 2014). *Reclamation* is the term used by practitioners in north-America and the UK, in the context of mined land (Bradshaw 1996) where there is a loss of topsoil (Stanturf et al. 2014). Stanturf (2005) used *Reclamation* and *Reconstruction* synonymously. However, these two terminologies were separated in Stanturf et al. (2014), but their separation was mainly based on the level of degradation, the degrading factor and the level of management techniques required. For example, *Reclamation* was applied to severely degraded land due to mining activities, thus requiring more intensive management techniques such as amelioration to soil physical, chemical and biological status, provision of regular irrigation and weed control to ensure early tree survival. *Reconstruction* was applied to moderately degraded land (e.g., recently abandoned agricultural land) that requires less management interventions (Stanturf et al. 2014). The *Reclamation* / *Reconstruction* / *Remediation* / *Renewal* or *Redemption* initiative employs active reforestation action through seeding and sapling transplanting reforestation technique due to substantial degradation of land. One or a few exotic or native tree species may be planted, but this is largely determined by the site conditions and reforestation initiative goals (SER 2004). However, exotic tree species are often used (especially in severely degraded sites where native tree species growth is hampered, e.g., mined land) and to extract heavy metals from the soil to create favourable growing conditions for native species (Sharma et al. 2004; van Rooyen et al. 2012; Stanturf et al. 2014). In this review, the recent reforestation studies showed that this reforestation initiative has moved beyond making land useful to achieving biodiversity and ecosystem functions, climate change mitigation, and ecosystem services.

2.4.1.5. Rehabilitation initiative

The *Rehabilitation* initiative resembles the *Ecological restoration* initiative, because of their fundamental focus on pre-degraded ecosystem species composition, structure and functioning as a reference (Aronson et al. 1993; SER 2004). However, the *Rehabilitation* initiative's motivation is to return degraded forest ecosystem species composition, structure and functions with less emphasis on near natural state (Bradshaw 1996; Allen et al. 2001; SER 2004), with an attempt to speed up the recovery process (Aronson et al. 1993). The *Rehabilitation* initiative can be implemented in moderately or severely degraded forest, and any of the active and passive reforestation actions (e.g., tree planting and invasive alien plant management) can be used to achieve reforestation goals (Stanturf et al. 2014). Furthermore, either native or a combination of exotic and native tree species can be planted. Although the motivation behind the *Rehabilitation* initiative is not to regain pre-degraded forest conditions, it can be designed to achieve all the goals and benefits that are achieved under the *Ecological restoration* initiative. For example, the *Rehabilitation* initiative studies documented in this review contributed to all the seven reforestation goals and their associated benefits.

2.4.2. Drivers of forest loss and degradation and potential solutions to reduce their impacts

2.4.2.1 Agriculture

Agriculture plays a fundamental role in human well-being (Meijaard et al. 2013), and a continued increase in human population results in a continued demand for more agricultural land (FAO 2016). In the tropics where forest is the dominant ecosystem, clearing of forest in search of more land for agricultural production is on a continuous basis (FAO 2016). However, some of the agricultural lands get abandoned as a result of ecological and socio-economic factors. For example, mismanagement of agricultural land leads to a decline in land productivity as a result of factors such as soil erosion (Lamb and Gilmour 2003; Rey Benayas 2005; FAO 2016). In other cases, land is abandoned as a result of reduced agricultural subsidies (Lamb and Gilmour 2003; Rey Benayas 2005), and the opening up of employment opportunities in industrial and other service sectors (Rey Benayas 2005; McGhee et al. 2007).

Agro-forestry practice in agricultural landscapes can be used to counteract land abandonment, because it has the potential to offer on-farm, landscape and global benefits (Gilmour 2005; Walker 2005; Jose 2009; Power et al. 2010; Adams et al. 2015). Agro-forestry benefits include increased agricultural productivity as a result of improved soil fertility (e.g., planting of leguminous tree species improve soil nitrogen, Walker 2005; Jose 2009). Other benefits include pollination and pest control, soil stabilization and erosion control, water purification, flood control, carbon sequestration, clean air, biodiversity conservation, forest habitat connectivity, improved species dispersal and gene-flow, aesthetic and recreation (e.g., Jose 2009; Gilbert-Norton et al. 2010; Power 2010). Approaches that could be used to encourage farmers to adopt agro-forestry practice include payment for ecosystem services (e.g., carbon credits and watershed protection) (Schuyt et al. 2007; Jose 2009; Winberg 2011), free or subsidised tree saplings, planting of trees that offer other income opportunities (e.g., fruit trees, medicinal trees, timber and firewood) (Saxena et al. 2001; Winberg 2011; Ruseva et al. 2015; Adams et al. 2016). A good example of encouraging the adoption of agro-forestry practice exists in the cloud forest of Chiapas, Mexico. The Starbucks Coffee Company and Conservation International have collaborated with farmers to conserve coffee farms while providing ecological benefits (Schuyt et al. 2007). The coffee farmers are encouraged to practice agro-forestry by planting native trees in degraded forest fragments, and to maintain the existing ones within the agricultural landscapes. The participating farmers have access to credit, to finance their crops, and those producing shade-grown coffee receive 44% more price premium over local prices (Schuyt et al. 2007).

2.4.2.2. Other anthropogenic factors, mining activities and IAPs

An increase in demand for livelihoods support, economic activities and residential areas lead to overharvesting of forest products and urbanization (Zhou et al. 2010; Adnan et al. 2014; Anton et al. 2015; Ferez et al. 2015; Knoke et al. 2014). Degradation of forest due to overharvesting of forest products such as timber and firewood is more common in rural and peri-urban areas (e.g., Von Maltitz et al. 2003). In urban areas, expanding cities need land to establish infrastructures such as buildings, roads and dams to support growing population (Chakravarty et al. 2012). Mining activities boost the economy of the country, but they leave the land unusable for any other purpose, because of the heavy metals that are highly toxic to life (e.g., Kithiia and Lyth 2011). The IAPs also cause forest degradation, but this usually occurs as a result of human-mediated disturbance and increased spread of propagules (e.g., Richardson and van Wilgen 2004). For example, the IAPs on forest margins and adjacent land spread into and establish in forest gaps created by overharvesting of forest products (von Maltitz et al. 2003).

A few reforestation initiatives in this review employed passive restoration action by excluding IAPs through clearing and allowing natural regeneration of native forest. Numerous studies have shown that IAP management alone can only achieve pre-invasion composition, structure and functions if there is a continuous management follow up to facilitate the establishment of recruiting native tree species (Beater et al. 2008; Reinecke et al. 2008, Simmons et al. 2016). However, in areas where IAP infestations are heavy, management should be coupled with seeding or tree planting (Holmes et al. 2008, Ruwanza et al. 2013; Kerr and Ruwanza 2016; Simmons et al. 2016). Although tree planting incurs more cost than seeding, it appeared to be highly favoured in reforestation initiatives, because it promotes a rapid recovery of forest (Palma and Laurance 2015). Seeding is usually not successful as a result of low seed quality, predation, dormancy and unfavourable site conditions to facilitate seed germination (e.g., Engel and Parrotta 2001).

2.4.2.3. Unavoidable forest loss or degradation

Unavoidable development activities in forest habitat could be compensated by biodiversity offset initiatives in degraded areas (Moilanen et al. 2009). Biodiversity offset can be defined as compensation for biodiversity loss caused by development activities, and is achieved by generating an ecologically equivalent biodiversity gains elsewhere (Maron et al. 2012). To make the mined lands useful for other purposes such as agriculture, development or recreation, the *Reclamation / Reconstruction / Remediation / Renewal or Redemption* reforestation initiative is usually recommended. Due to harsh environmental conditions created by mining operations, exotic tree species are mostly used to ameliorate soil and to create favourable conditions for native species establishment (Montagnini 2005; Kithiia and Lyth 2011). Some of the reasons that have promoted the use of exotic tree species include lack of native tree species seeds, lack of basic knowledge on native tree propagation techniques, some native tree species have slow growth rate and they are difficult to manage silviculturally than exotic tree species (Knowles and Parrotta 1995; Richardson 1998). However, the use of exotic tree species should be the last resort after screening a wide range of available native tree species (McNeely 2005). Various studies have shown that some native tree species can also be effective in ameliorating soil and create conditions suitable for native forest succession (e.g., van Aarde et al. 1996; Parrotta and Knowles 1999; Mulizane et al. 2005; MacDonald et al. 2015). In reforestation initiatives where exotic tree species are preferred, because of their valuable benefits such as soil amelioration and socio-economic benefits (e.g., fruits, firewood and timber), they should be screened for invasiveness potential through pre-planting tests (Richardson 1998), weed risk assessment model (Pheloung et al. 1999) and species distribution modelling (Thuiller et al. 2005).

2.5. Conclusion

Reforestation initiatives are being implemented in all the biogeographic realmsof the world, except in the Oceania biogeographic realm.This shows that governmental and non-governmental institutions around the world have now realized that reforestation plays a significant role in sustaining forest ecosystems, societies and economies. Most of these initiatives are planting a higher diversity of native tree species, to provide a wide range of ecosystem services that play a significant role in sustaining human well-being (Regato and Berrahmouni 2005). This is probably due to the fact that millions of households around the world depend on forest products to meet their nutritional (e.g., food) and health needs (medicinal plants), and for direct economic benefits (e.g., income from the sale of forest products) (Saxena et al. 2001; Shackleton and Shackleton 2004; Regato and Berrahmouni 2005). Furthermore, reforestation initiatives are also providing employment opportunities in form of reforestation site preparation, seed collection and seedling growth, tree planting and site maintenance, and carbon credits from the carbon offset trading schemes (Mills and Cowling 2006; Brancalion et al. 2012; Douwes et al. 2015) and carbon credits (e.g., Brown et al. 2011; Semwal et al. 2013).

All the reforestation initiatives have the potential to achieve the recovery of pre-degraded conditions, because most of them are now designed to achieve multiple ecological and socio-economic goals. Chazdon (2008) presented a restoration staircase model which showed that *Reclamation* and *Rehabilitation* initiatives can precede the *Ecological restoration* initiative. The number of steps is dependent on the level of degradation. For example, highly degraded land as a result of mining activities leaving heavy metals that are not conducive for biodiversity and ecosystem services restoration, would first require *Reclamation / Reconstruction / Remediation / Renewal or Redemption* initiative to make the conditions conducive. This is then followed by the *Rehabilitation* initiative to reinstate some species composition, structure and function, and this can ultimately lead to the *Ecological restoration* initiative. A similar approach to Chazdon (2008)'s restoration staircase model was also presented by van Rooyen et al. (2012) in the repair of mined sand dunes in Richards Bay, South Africa. SER (2004) noted that activities that aim to initiate ecosystem development along a preferred trajectory, that allows ecosystem establishment with little or no human influence can qualify as restoration.

Reforestation in a conservation perspective restores biodiversity that has been lost due to changes in land use patterns, in a socio-economic perspective, it restores ecosystem services from which people benefit, from a cultural perspective it strengthens communities, institutions and interpersonal relationships in pursuit of a common goal. From a personal perspective, it allows humans to reconnect with the nature and restore ourselves as we restore impaired ecosystem (Clewell and Aronson 2013). This review showed that recent reforestation initiatives were motivated by the need to achieve biodiversity conservation, more functional forest ecosystems, and to enhance ecosystem services and human well-being, and allow people to reconnect with nature. Reforestation initiatives should clarify drivers of forest loss and degradation, reforestation goals and their associated benefits and the employed alternative reforestation actions used. This information is critical to guide reforestation managers and practitioners in the planning and implementation of strategic reforestation approaches that address forest loss and to ensure the long-term sustainability of forests, and achieve ecological and socio-economic goals simultaneously.

The inconsistent use of terminologies describing different reforestation motivations lead to disconnection between restoration scientists, restoration managers (Hart et al. 2015), policymakers, and other stakeholders. This can result in the development of strategic approaches that are based on ecological, socio-economic and political value and/or agenda, instead of science-based approach. Therefore, the consistent use of terminologies based on reforestation initiatives' motivations, without broadening the terminology definition to accommodate new reforestation initiatives motivations, is encouraged. This will make it easier to for restoration scientists, restoration managers, policy makers and other stakeholders to develop science-based strategic approaches that will guide and improve reforestation practice, and provide conceptual, institutional, legal and operational basis for the policies, and for the monitoring and evaluation of the reforestation initiatives (after Chazdon et al. 2008). Furthermore, this would allow the global alignment of reforestation activities, and development of global learning network around reforestation.

2.6. References

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Zomer RJ; Trabucco A; Van Straaten O, Bossio DA (2006) Carbon, land and water: a global analysis of the hydrologic dimensions of climate change mitigation through afforestation/reforestation. Colombo, Sri Lanka: International Water Management Institute. IWMI Research Report 101.

2.7. Supplementary materials

Table S2.1: Complete list of recent studies on reforestation initiatives, 2000 – 2016.

Author	Date	Title	Reference	Country
Alexander HD, Moczygemba J, Dick K	2016	Growth and survival of thornscrub forest seedlings in response to restoration strategies aimed at alleviating abiotic and biotic stressors.	Journal of Arid Environments 124: 180-188	Mexico
Ancog RC, Florece LM, Nicopior OB	2016	Fire occurrence and fire mitigation strategies in a grassland reforestation area in the Philippines	Forest Policy and Economics, http://dx.doi.org/10.1016/j.forpol.2016.01.002	Philippines
Anderson PML, Avlonitis G, Ernstson H	2014	Ecological outcomes of civic and expert-led urban greening projects using native plant species in Cape Town, South Africa.	Landscape and Urban Planning 127: 104-113	South Africa
Anton V, Hartley S, Wittmer HU	2015	Survival and growth of planted seedlings of three native tree species in urban forest restoration in Wellington, New Zealand.	New Zealand Journal of Ecology 39: 170-178	New Zealand
Bare MC, Ashton MS	2015	Growth of native tree species planted in montane reforestation projects in the Colombian and Ecuadorian Andes differs	New Forests, doi:10.1007/s11056-015-9519-z	Columbia/Ecuador

		among site and species.		
Bonilla-Moheno M, Holl KD	2010	Direct Seeding to Restore Tropical Mature-Forest Species in Areas of Slash-and-Burn Agriculture	Restoration Ecology 18: 438–445	Mexico
Brown DR, Dettmann P, Rinaudo T, Tefera H, Tofu A	2011	Poverty Alleviation and Environmental Restoration Using the Clean Development Mechanism: A Case Study from Humbo, Ethiopia	Environmental Management 48:322–333	Ethiopia
Colmanetti MAA, Barbosa LM, Shirasuna RT, do Couto TZ	2016	Phytosociology and structural characterization of woody regeneration from a reforestation with native species in south eastern Brazil	Revista Árvore, Viçosa-MG 40: 209-218	Brazil
Costa AdS, Malhado ACM, Bragagnolo C, Correia RA, Ladle RJ	2016	Ecological outcomes of Atlantic Forest restoration initiatives by sugarcane producers	Land Use Policy 52: 345–352	Brazil
De Steven D, Faulkner SP, Keeland BD, Baldwin MJ, McCoy JW, Hughes SC	2015	Understory vegetation as an indicator for floodplain forest restoration in the Mississippi River Alluvial Valley, U.S.A.	Restoration Ecology 23: 402–412	USA
De Torre R,	2015	Use of restoration plantings to	Journal of Environmental Engineering and	Spain

Jiménez MD, Ramírez A, Mola I, Casado MA, Balaguer L		enhance bird seed dispersal at the roadside: failures and prospects	Landscape Management 23: 302-311	
Douwes E, Rouget M, Diederichs N, O'Donoghue S, Roy K, Roberts D	2015	Buffelsdraai Landfill Site Community Reforestation Project	XIV World Forestry Congress, Durban, South Africa	South Africa
Evans DM, Zipper CE, Burger JA, Strahm BD, Villamagna AM	2013	Reforestation practice for enhancement of ecosystem services on a compacted surface mine: Path toward ecosystem recovery	Ecological Engineering 51: 16-23	USA
Falcão JCF, Dáttilo W, Izzo TJ	2015	Efficiency of different planted forests in recovering biodiversity and ecological interactions in Brazilian Amazon	Forest Ecology and Management 339: 105-111	Brazil
Fields-Johnson CW, Zipper CE, Burger JA, Evans DM		Forest restoration on steep slopes after coal surface mining in Appalachian USA: Soil grading and seeding effects	Forest Ecology and Management 270: 126-134	USA
Kerr TF, Ruwanza S	2016	Does *Eucalyptus grandis* invasion and removal affect soils and vegetation in the Eastern Cape Province, South Africa?	Austral Ecology 41: 328-338	South Africa

Khamzina A, Lamers JPA, Martius C	2016	Above- and belowground litter stocks and decay at a multispecies afforestation site on arid, saline soil	Nutrient Cycling in Agroecosystems 104: 187-199	Uzbekistan
Kotecký V	2015	Contribution of afforestation subsidies policy to climate change adaptation in the Czech Republic	Land Use Policy 47: 112-120	Czech Republic
Matzek V, Warren S, Fisher C	2016	Incomplete recovery of ecosystem processes after two decades of riparian forest restoration	Restoration Ecology 24: 637-645	USA
Morris TL, Witkowski ETF, Coetzee JA	2008	Initial response of riparian plant community structure to clearing of invasive alien plants in Kruger National Park, South Africa	South African Journal of Botany 74: 485-494	South Africa
Morrison EB, Lindell CA	2011	Active or Passive Forest Restoration? Assessing Restoration Alternatives with Avian Foraging Behavior	Restoration Ecology 19: 170-177	Costa Rica
Mukul SA, Herbohn J, Firn J	2016	Co-benefits of biodiversity and carbon sequestration from regenerating secondary forests in the Philippine uplands: implications for forest landscape restoration	BIOTROPICA 48: 882-889	Philippines

Mullah CJA, Klanderud K, Totland O, Kigomo B	2014	Relationships between the density of two potential restoration tree species and plant species abundance and richness in a degraded Afromontane forest of Kenya	African Journal of Ecology 52: 77-87	Kenya
Negi VS, Bhatt ID, Phondani PC, Kothyari BP	2015	Rehabilitation of degraded community land in Western Himalaya: linking environmental conservation with livelihood	Current Science 109: 521-528	India
Ngugi MR, Neldner VJ, Kusy B	2015	Using forest growth trajectory modelling to complement BioCondition assessment of mine vegetation rehabilitation	Ecological Management and Restoration 16: 78-82	Australia
Oldfield EE, Felson AJ, Auyeung DS, Crowther TW, Sonti NF, Harada Y, Maynard DS, Sokol NW, Ashton MS, Warren II RJ, Hallet RA, Bradford MA	2015	Growing the urban forest: tree performance in response to biotic and abiotic land management	Restoration Ecology 23: 707-718	USA
Pohnan E, Ompusunggu H, Webb C	2015	Does tree planting change minds? Assessing the use of community participation in reforestation to address illegal logging in West Kalimantan	Tropical Conservation Science 8: 45-57	Indonesia

Santiago-García RJ, Colón SM, Sollins P, Van Bloem SJ	2008	The Role of Nurse Trees in Mitigating Fire Effects on Tropical Dry Forest Restoration: A Case Study	AMBIO: A Journal of the Human Environment 37: 604-608	Puerto Rico
Smith CMS, Bowie MH, Hahner JL, Boyer S, Kim Y-N, Zhong H-T, Abbott M, Rhodes S, Sharp D, Dickinson N	2016	Punakaiki Coastal Restoration Project: A case study for a consultative and multidisciplinary approach in selecting indicators of restoration success for a sand mining closure site, West Coast, New Zealand	Catena 136: 91-103	New Zealand
Stangeland T, Tabuti J, Lye KA	2011	The framework tree species approach to conserve medicinal trees in Uganda	Agroforestry Systems 82: 275-284	Uganda
Von Brandis RG	2012	Rehabilitation of abandoned coconut plantations at D'Arros Island, Republic of Seychelles	Ocean and Coastal Management 69, http://dx.doi.org/10.1016/j.ocecoaman.2012.09.003	Seychelles
Wilson SJ, Rhemtulla JM	2016	Acceleration and novelty: community restoration speeds recovery and transforms species composition in Andean cloud forest	Ecological Applications 26: 203-218	Ecuador
Wong JT-F, Chen X-W, Mo W-Y, Man Y-B, Ng C W-W, Wong M-H,	2016	Restoration of plant and animal communities in a sanitary landfill: a 10-year case study in Hong Kong	Land Degradation and Development 27: 490-499	China

Table S2.2: Assessed parameters in recent reforestation studies (2005 to 2016). Biogeographic realmswere adopted from Olson et al. (2001).

Parameters and categories	Features
1. Forest degrading factors	*Activities*
Agriculture	Crop and livestock production; forestry (timber and pulp)
Invasive alien plants (IAPs)	Unintentional and intentional introduction
Mine	Underground resources (e.g., coal and gold)
Other anthropogenic factors (OAF)	Infrastructure development (e.g., residential, industrial and roads), over-harvesting (e.g., medicine, firewood and timber).
Reforestation actions	*Techniques*
Passive reforestation	Exclusion of degrading factors (e.g., livestock fencing, fire breaks and IAPs management) to allow natural forest regeneration
Active reforestation	Seeding and transplanting of single or multiple native, exotic or both the nativeand exotic tree species
Reforestation goals	*Anticipated Benefits*
Biodiversity conservation and ecosystem functions	Fauna and flora diversity, ecosystem functions (e.g., nutrient cycling and pollination)
Climate change mitigation	Carbon storage
Cultural service	Aesthetic, ecotourism, education, recreation, spiritual
Economic benefits	Employment and carbon credits

Provisioning services	Food, firewood, fodder, medicinal plants, timber and water quantity
Regulating services	Air quality and climate regulation, flood control, pollutants control (e.g., heavy metals), soil conservation (e.g., soil erosion control), water quality
Supporting services	Habitat connectivity, wildlife food and habitat
Biogeographic realms	Biogeographic realm *location*
Afrotropic	Sub-Saharan Africa
Australasia	Australia and New Zealand
Indo-Malay	Southern Asia
Nearctic	South America
Neotropic	North America
Palearctic	Above sub-Saharan Africa, Europe and northern Asia

CHAPTER 3: RESTORATION PLANNING FOR CLIMATE CHANGE MITIGATION AND ADAPTATION IN THE CITY OF DURBAN, SOUTH AFRICA

This chapter is based on

Lutendo F. Mugwedi, Jayanti Ray-Mukherjee, Kathryn E. Roy, Benis Egoh, Federico M. Pouzols, Errol Douwes, Richard Boon, Sean O'Donoghue, Rob Slotow, Enrico Di Minin, Atte Moilanen, Mathieu Rouget. Restoration planning for climate change mitigation and adaptation in the city of Durban, South Africa. International Journal of Biodiversity Science, Ecosystem Services and Management (Accepted).

Abstract

Effective planning of a large-scale restoration project is challenging, because of the range of factors that need to be considered (e.g., restoration of multiple habitats with varying degradation levels, multiple restoration goals and limited conservation resources). Ecological restoration planning studies typically focus on biodiversity and ecosystem services, rather than employment and other co-benefits. Robust Offsetting (RobOff), a restoration planning tool, was used in a forest restoration project in Durban, South Africa, to plan forest restoration considering a mosaic of habitats with varying levels of degradation, diverse restoration actions, a limited budget and multiple (biodiversity, carbon stock and employment) goals. To achieve this, the restoration action currently being implemented (= *current action*) was compared to three restoration alternatives. The three restoration alternatives included: 1) *natural regeneration action*; 2) *carbon action*; and 3) *biodiversity action*. The results supported biodiversity action as most beneficial in terms of maximising biodiversity, carbon storage and job creation. Results showed that investing in *biodiversity action* is preferable to the status quo. The methodology presented proved to be useful in ensuring optimal biodiversity and ecosystem services conservation, enhance carbon storage and job creation.

Keywords: Biodiversity prioritization; Carbon stock; Conservation resource allocation; Decision-making tool; Creating employment; RobOff software; Tree planting.

3.1. Introduction

Many cities around the world are highly vulnerable to the adverse impacts of climate change (UN-Habitat 2011). The likely impacts include but not limited to species extinctions (Chapin III et al. 2000), a decrease in human health quality due to heat waves, poor air and water quality (Patz et al. 2005), an increase in frequency and intensity of floods, and an increase in erosion of coastal areas leading to infrastructure damage (Chapin III et al. 2000). These impacts will be exacerbated by poor governance, limited service delivery, and existing socio-economic challenges. This will result in a growing dependence upon ecosystem services, thus leading to degradation and fragmentation of functional ecosystems, and loss of ecosystem services critical for human well-being (United Nations-Habitat 2011; Oldfield et al. 2013; Elmqvist et al. 2015).

By 2030, the world rural and urban population is predicted to increase to 3.4 and 5.1 billion, respectively (United Nations 2015), with approximately 60% of urban land predicted to be under built infrastructure (Elmqvist et al. 2013). Environmental managers and city planners are faced with increasing pressure to protect ecosystems inside and outside of cities to ensure a continued supply of ecosystem services. This will ensure that the needs of the current and future generations are met (Schewenius et al. 2014), thus achieving more liveable and healthy cities that are prepared for the impacts of climate change (Elmqvist et al. 2015).

Ecosystem-based adaptation (EBA) is increasingly recognized as one of a toolbox of solutions for challenges faced by cities (Roberts and O'Donoghue 2013; Elmqvist et al. 2015). The EBA can be defined as 'the use of biodiversity and ecosystem services as part of an overall adaptation strategy.' (CBD 2009). It includes sustainable management, conservation and restoration of degraded ecosystems to provide services that help society adapt to the adverse impacts of climate change (CBD 2009; Colls et al. 2009; Munang et al. 2013; Doswald et al. 2014). For example, mangrove forest and coastal wetlands can help protect coastal communities from tropical storm surges (Alongi 2008).

Cities such as Auckland, Bangalore, Durban, London, Mombasa, New York, and São Paulo have embarked on large-scale tree planting (active restoration) projects in degraded forests, abandoned industrial mine and agricultural lands (Engel and Parrotta 2001; Kiithia and Lyth 2011; Rees and Everett 2012; Oldfield et al. 2013; Douwes et al. 2015). These projects seek to enhance biodiversity conservation (through planting of native flora) and contribute a range of critical ecosystem services. The critical ecosystem services include provisioning services (e.g., water, food and medicine), regulation services (e.g., carbon sequestration, water purification, storm-water regulation and microclimate regulation), cultural services (e.g., aesthetic, recreation, spiritual fulfilment and education) and supporting services (e.g., soil stabilisation and habitat provision) (Saxena et al. 2001; Dudley and Stolton 2005; Thompson et al. 2009; Negi et al. 2015; Elmqvist et al. 2015; Oldfield 2015). Tree planting can also provide employment opportunities to impoverished urban communities with limited access to other employment and government services (Douwes et al. 2015; Perring et al. 2015).

Although tree planting offers a wide range of benefits, funding for restoration is often limited, which constrains restoration actions and the extent of land that can be restored (Adame et al. 2015; Mazziotta et al. 2016). The limited restoration resources necessitate the identification of cost effective actions and selection of restoration areas which provide multiple restoration benefits (Crossman and Bryan 2009). Planning of a large-scale restoration project can be a challenging undertaking given multiple habitats with varying degrees of degradation, multiple restoration activity choices and (often competing) priorities for ecological and socio-economic goals to be achieved within limited restoration budgets (Maron and Cockfield 2008; Turpie et al. 2008; Brancalion et al. 2014; Vogler et al. 2015;). Socio-economic co-benefits are increasingly recognized in restoration projects, because they influence restoration success (Perring et al. 2015). For example, depriving local communities' access to forest services (e.g., food, medicine and fuelwood collection) that support their livelihoods without providing viable alternatives can lead to restoration project failure (Saxena et al. 2001; Le et al. 2011; Orsi et al. 2011; Barr and Sayer 2012).

Many restoration funding agencies, especially state agencies, are now funding restoration projects that aim to achieve stated ecological and socio-economic goals simultaneously (Maron and Cockfield 2008; Pendleton 2010). Deciding where to carry out restoration to achieve all target goals is a key challenge in large-scale restoration (Torrubia et al. 2014). Without prioritization planning, resource allocation is likely to be made in an *ad hoc* manner and this might affect the restoration success (Wilson et al. 2011). Managers of large-scale restoration projects are now incorporating systematic planning tools such as Marxan, Integer Linear-programming, and Zonation (Crossman and Bryan 2006; Egoh et al. 2014; Moilanen et al. 2014) in large-scale restoration planning (e.g., Thomson et al. 2009; Wilson et al. 2011; Perring et al. 2015). All these tools employ significant simplifications in objectives, biodiversity features and cost components, space, time, or problem dimensions, in order to manage analyses. Systematic planning tools also ensure the efficient allocation of available resources through prioritization of restoration actions. This can ensure improved ecological and socio-economic benefits, and long-term sustainability of restoration projects (Maron and Cockfield 2008; Adame et al. 2015; Rappaport et al. 2015).

A recently-released open-source software platform (see Table S3.1), Robust Offsetting (RobOff), differs from the other systematic conservation planning tools, in that it is primarily ‘action’ rather than spatially-based. It selects the best conservation action given a set of goals through modelling of uncertainty around alternative conservation actions (e.g., tree planting vs. invasive alien plants [IAPs] control) have on different biodiversity features (e.g., species richness) in different environments (Pouzols and Moilanen 2013a).

Systematic restoration planning studies done to guide restoration plans have accounted for either biodiversity features (e.g., species richness), ecosystem services provision (e.g., climate regulation or water supply) or both (e.g., Thomson et al. 2009; Orsi et al. 2011; Wilson et al. 2011; Budiharta et al. 2014; Egoh et al. 2014; Adame et al. 2015; Rappaport et al. 2015). This is because biodiversity and ecosystem services improve human well-being, for example, by providing humans with benefits such as clean water, climate regulation, recreation (Brancalion et al. 2014), and medicinal plants mostly for peri-urban and rural communities (Douwes et al. 2015). Although some restoration programs include financial benefits such as employment that enhances impoverished communities' adaptation capacity to adverse impacts of climate change (e.g., Roberts and O’Donoghue 2013; Brancalion et al. 2014; Wilson and Rhemtulla 2016), there is a lack of studies that prioritize employment creation in their restoration planning.

In South Africa, the city of Durban (managed by eThekwini Municipality) is a leader in climate change adaptation within developing countries (Diederichs and Roberts 2015). This is because the Municipal Climate Protection Programme is addressing the challenge of climate change vulnerability within the context of widespread poverty (Roberts and O’Donoghue 2013), intensifying urbanization and ecosystem degradation (Diederichs and Roberts 2015). The Buffelsdraai Landfill Site Community Reforestation Project (BLSCRP) was initiated to offset carbon emissions and increase climate change adaptation through biodiversity and ecosystem services restoration and employment creation (Douwes et al. 2015). This study is a post-hoc comparison of options that were not considered during the planning of the BLSCRP in Durban, to draw lessons for future similar planning efforts. Here, we show how RobOff could have been used to efficiently allocate resources in a large-scale restoration project with two habitats, multiple restoration alternatives and goals (biodiversity, carbon stock and employment), with a limited budget. To achieve this objective, a sequence of key research questions was addressed:

a) What is the recommended restoration action within each habitat?
b) How do different budget scenarios influence the optimal allocation of resources (hectares and budget) to alternative restoration actions?
c) Does prioritization of restoration benefits influence the division of resources to alternative restoration actions across the habitats?

This study differs from other restoration planning studies, because it prioritizes biodiversity, carbon storage and employment creation, as well as what restoration actions are appropriate to achieve these goals. Furthermore, it provides recommendations of where and how restoration should be carried out in order to maximise biodiversity, carbon storage and employment creation.

3.2. Materials and methods

3.2.1. Study area

The city of Durban, South Africa, harbours remnants of scarp forest, which is described as a refuge forest that survived the last glacial maximum (≈18000 BP) (Eeley et al. 1999). About 15–31 % of this forest type has been lost due to land transformation (e.g., because of logging and clearing for agriculture) and non-sustainable harvesting of forest products by rural communities (von Maltitz et al. 2003). The resulting fragmentation and landscape connectivity loss, between forest patches and the increased edge ecotone (Kotze and Lawes 2007), has led the eThekwini Municipality to engage in a range of land management and restoration related practices (Diederichs and Roberts 2010; Roberts and O'Donoghue 2013). One programme, namely, the Community Reforestation Programme, includes large-scale projects for carbon sequestration and climate change adaptation purposes. The Community Reforestation Programme has three projects, including the BLSCRP. In the BLSCRP, active restoration, i.e. planting of native trees, was employed within this project, to restore degraded scarp forest in a buffer area around the Buffelsdraai Landfill Site (29.62961S; 30.980392E). At least 51 of the 230 native tree species occurring in the region have been planted within a 580 ha (av. 1500 trees/ha) portion of the buffer zone. Aside from tree planting, control of IAPs and fire suppression (cutting fire breaks) are also implemented (Douwes et al. 2015).

Amongst the metropolitan areas in South Africa, Durban has the highest number of people living on less than US$2 per day (eThekwini Municipality, 2013). The BLSCRP directly supports two of the poorest communities, namely the Buffelsdraai and Osindisweni communities. This is in the form of direct job creation as well as through supporting local community members that grow trees for the project. In terms of tree growing, the project has trained community members (known as Tree-preneurs) to collect seeds, which they grow at their homesteads. Once the trees are big enough (greater than 30 cm in height) they are supplied to the project. Tree-preneurs source native tree seeds from the local forest and woodland patches. Seedlings are traded for credit notes, which can be exchanged for items such as groceries, clothes, building materials, bicycles, or to pay for school fees or for vehicle driving lessons. Land preparation, planting and maintenance are also done by community members, either permanently or temporarily employed by the project (Douwes et al. 2015).

3.2.2. Analytical framework

RobOff, can be used to determine optimal resource allocation solutions (Pouzols and Moilanen 2013a) using a resource allocation algorithm. For example, to identify a cost-effective and balanced set of alternative restoration actions across a range of restoration habitats (referred to as environments in the RobOff framework). The intention is to maximize the aggregate restoration benefits across several goals, limited by budget, and while accounting for different costs of actions in different restoration habitats and different temporal responses of biodiversity (and other) features (e.g., tree species richness) to alternative restoration actions in different restoration habitats (Pouzols and Moilanen 2013b). The restoration habitats used in this study included ‘former sugarcane fields’ and ‘extant forest’ with varying levels of degradation. RobOff inputs included condition of restoration habitats, external threats (e.g., IAPs), proposed alternative restoration actions (e.g., IAP control and tree planting) linked to each restoration habitat and their costs, available area to implement each action, features’ occurrence in the habitat (e.g., species richness), and the features response to each action (quantified on an annual basis). The response of features to alternative restoration actions is quantified under three uncertainty envelopes (lower, average and upper estimates) over a period of time (see Figure S3.1). Uncertainty envelopes are included to account for uncertainties in how well habitat restoration or environmental management operations really work (Pouzols et al. 2012). Overall, the extent to which an action is applied to a habitat is largely determined by budget availability. Here, these inputs are used by RobOff to determine the relationship between the restoration benefits and restoration actions (e.g., what actions can achieve which benefits?), restorations actions and restoration habitats (e.g., which restoration actions can be carried out in which restoration habitats?), restoration cost and actions (e.g., what costs are associated with what actions?) and what benefits could be obtained within the budget? (Figure 3.1) (See Pouzols et al. 2012).

3.2.2.1. Restoration habitats

There two restoration habitats at the Buffelsdraai Landfill Site: (a) 'former sugarcane fields' where the planting of at least 51 native tree species (current action) took place, and (b) 'extant forest' were used in this study. The 'former sugarcane fields' and 'extant forest' habitats covered 580 and 105.8 hectares, respectively. The 'extant forest' was slightly degraded by IAPs. Before restoration, the 'former sugarcane fields' habitat was dominated by sugarcane, IAPs, and weeds (graminoids and forbs).

3.2.2.2. Restoration actions and costs

The restoration action currently being implemented (= *current action* - discontinue sugarcane farming and reforestation using at least 51 native tree species to enhance biodiversity, carbon stock and provide project-linked employment for local communities) was compared to three restoration alternatives. The three restoration alternatives included: 1) *natural regeneration action*– discontinue sugarcane farming and allow natural recruitment of native tree species; 2) *carbon action* – discontinue sugarcane farming and undertake reforestation using 10 native tree species with a higher wood density to store a higher carbon stock; and 3) *biodiversity action* – discontinue sugarcane farming and implement reforestation using 80 native tree species to enhance tree species richness.

Each action was assessed in terms of tree species richness, carbon stock and employment creation and compared with the *current action* (planting of at least 51 native tree species). This was done through a series of workshops with 12 local restoration ecology and biodiversity experts (academics and practitioners). Experts were chosen based on their knowledge of local biodiversity, forest restoration management and cost, and the RobOff software. The *do nothing* (default in RobOff) and *natural regeneration* actions were proposed for the 'extant forest' while *do nothing*, *natural regeneration action*, *current action*, *carbon action* and *biodiversity action* were proposed for the 'former sugarcane fields' by the experts (Table 3.1).

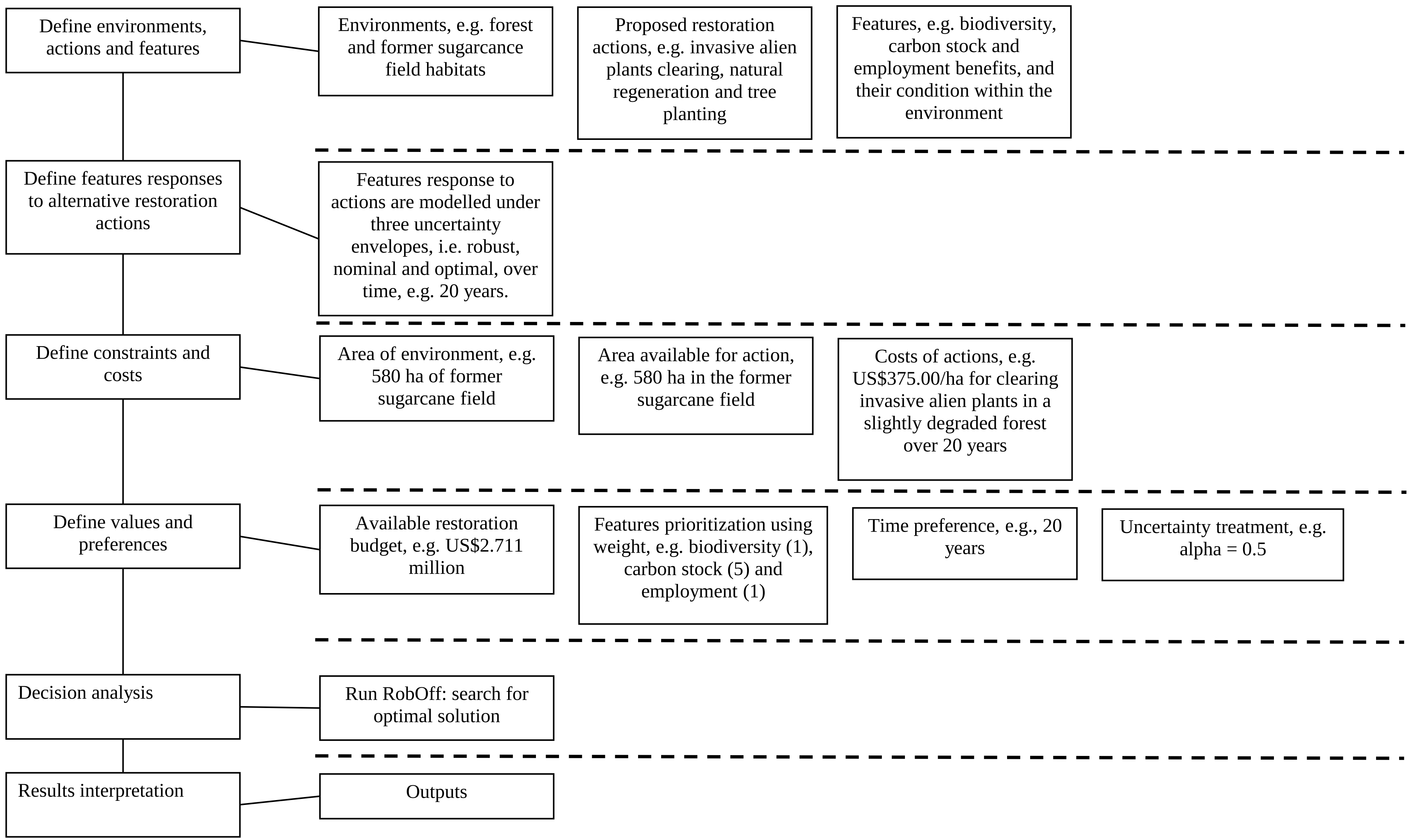

Figure 3.1: Flow chart of RobOff framework (modified from Pouzols et al. 2013b).

Table 3.1: Restoration actions and their associated costs per ha over a 20 year period in the ‘extant forest’ and ‘former sugarcane fields’ restoration habitats (see Table S3.1-3.4 for more details).

Habitats	Actions	Aims	Intervention	Cost (US$)/ha over 20 years
Extant forest (105.8 ha)	Do nothing (default in RobOff)	Control action; indicates areas where funds do not suffice for action	No intervention	0
	Natural regeneration action	To conserve the extant forest patches.	Allow natural regeneration of native plants supported by invasive alien plants (IAPs) clearing.	375
Former sugarcane fields (580 ha)	Do nothing	Same as in the extant forest	Same as in the extant forest.	0
	Natural regeneration action	To allow natural regeneration of native plants.	Allow natural regeneration of native plants and site maintenance (IAPS clearing and cutting fire breaks).	4260
	Current action	To restore biodiversity, sequester carbon, and to create project-related employment for the local communities.	Planting of 51 native tree species at 1000 trees per hectare and site maintenance. This is the status quo plan for the site.	4352
	Carbon action	To maximise carbon sequestration.	Planting of 10 native tree species with high wood density at 1000 trees per hectare and site maintenance.	4678
	Biodiversity action	To boost tree species richness and to increase ecosystem functioning.	Planting of 80 native trees species from the regional pool of scarp forest trees at 1000 trees per hectare and site maintenance.	4450

Restoration cost of each action within the habitats was estimated per ha, over a 20 year period (Table S3.1). Cost was divided into two categories: initial reforestation and subsequent site maintenance costs. Initial costs included seedling production (buying trees from Tree-preneurs), land preparation and planting, while site maintenance cost included removal of IAPs and fire suppression. Initial and fire suppression costs were estimated using expert knowledge and the financial report from the Municipality's reforestation program (EThekwini Municipality 2011) (Table S3.1). *Natural regeneration action* cost per ha in the 'extant forest' only included the cost of clearing IAPs, because forest fires are rare at this locality, hence it was lower than the cost of the similar action per ha in the 'former sugarcane fields', which included clearing IAPs and fire suppression. In contrast, *carbon action* cost per ha was higher than the *current* and *biodiversity* actions, because it included slow growing tree species with high wood density, thus requiring more intense site maintenance than the *current* and *biodiversity actions* (Table S3.1). The IAPs clearing cost was obtained from the South African National Parks' Working for Water Programme (IAPs clearing project, unpublished database) and from published scientific literature (Marais and Wanneburg 2008). The values were adjusted according using clearing costs from the SANPARKS database. The IAPs clearing method included uprooting of saplings and cutting of shrubs and trees.

3.2.2.3. Restoration features response (referred to as restoration benefits in this study) to alternative restoration actions across habitats

Three critical co-benefits from Buffelsdraai Landfill Site were chosen, namely, biodiversity, carbon stock and employment creation. Based on published literature of biodiversity, carbon stock and reforestation costs reports and local knowledge (Table S3.2-3.4), the experts estimated (annually over a period of 20 years) the response of benefits to alternative restoration actions across habitats under the lower, average and upper uncertainty envelopes (see Table S3.1-3.3 for a detailed description). Tree species richness was quantified in terms of no. of tree species/ha (Table S3.2), carbon stock benefits in terms of above-ground carbon storage in trees (tC/ha) (Table S3.3), and employment creation benefits in terms of no. of person days/ha (Table S3.4). Tree species richness was chosen as a simplified measure of biodiversity, because it correlates with the diversity of the ecosystem and is a measurement that can be easily monitored (Gotelli and Colwell 2001). Employment created was calculated as the sum of people involved in seedling production, land preparation, planting and site maintenance. One person day is equivalent to eight working hours per day with a daily remuneration rate of US$7 (as of November 2015, XE Currency Converter).

A 20 year restoration period was used, because it was estimated that, after this time, the trees would have grown into a forest, and the carbon stock in the restored scarp forest would be similar to woodland in the region (Glenday 2007).

3.2.2.4. Restoration budget

The cost of restoring all the 'former sugarcane fields' (580 ha) based on the most expensive action (i.e. *carbon action*) was calculated and used as the maximum budget (estimated at ZAR 38.5 million, or US$ 2.711 million as of November 2015). Optimal resource allocations were investigated for five budget scenarios: US$ 0.271 (10%), $ 0.677 (25%), $ 1.355 (50%), $ 2.033 (75%) and $ 2.711 (100%) million to explore the influence of budget availability on the allocation of resources to alternative restoration actions.

3.2.3. Data analyses

The cost of each alternative restoration action per hectare was quantified. To assess the effect of budget availability on resource allocation, analyses were replicated under alternative budget levels (previous section). Effect of biodiversity, carbon stock and employment prioritization on allocation of resources to alternative restoration actions was also assessed. Seven weighting schemes (e.g., relative weight of 1.0 for biodiversity, 1.0 for carbon stock and 2.0 for employment) were used to explore this effect. Prioritization of restoration benefits was done under seven permutations (i.e. seven prioritization scenarios) (Table S3.5). To assess the effects of prioritization of alternative restoration benefits, RobOff was run using the 50% and 100% budget scenarios over a 20 year period. Only the 50% and 100% budget scenarios were included, because similar restoration action to 50% and 100% was recommended under the 10% and 25% budget scenarios, although the benefits decreased with a decrease in available budget. To account for uncertainties of features' (e.g., biodiversity) responses to alternative restoration actions within the environment over a 20 year period, alpha was set at 0.5. The RobOff results take uncertainty level (alpha = 0.5) into account. A piecewise linear benefit function and a 2.5% time discounting rate were used (see Pouzols et al. 2012).

3.3. Results

When budget decreased, a focus on improving biodiversity (*biodiversity action*) through restoration of the 'former sugarcane fields' could maximise biodiversity, carbon and employment goals than the *current action*. Both biodiversity and employment goals prioritization achieved higher species richness, carbon stock and employment while carbon stock goal prioritization was at the expense of biodiversity.

3.3.1. How do different budget scenarios affect the allocation of resources to alternative restoration actions and benefits across different habitats?

Across all budget scenarios, degraded 'former sugarcane fields' should be prioritized for restoration rather than the 'extant forest'. Only under the 100% budget scenario (US$2.711 million), some provision was made also for the slightly degraded 'extant forest'. *Natural regeneration action* was selected for the 'extant forest', while *biodiversity action* was selected for the 'former sugarcane fields'. A reduction in budget eliminated the *natural regeneration action* in the 'extant forest', and prioritised the 'former sugarcane fields' with *biodiversity action* recommended. As the allocated budget decreased by an order of magnitude, fewer hectares were allocated to restoration, but *biodiversity action* remained the priority (Table 3.2); this is because *biodiversity action* offered the optimal solution.

Under the 100% budget scenario, allocation of resources to restoring slightly degraded 'extant forest' increased tree species richness from 70 to 75 over a 20 year period. Employment increased from 0 to 720 person days over a 20 year period. Carbon stock did not change (Figure 3.2a-c) over 20 years, because, if the IAPs are not cleared, they would also store carbon. When *biodiversity action* was the chosen restoration measure in the 'former sugarcane fields', the result was an increase in biodiversity (from 2 to 85 tree species per habitat over a 20 year period), carbon stock (from 0 to 13280 tC per habitat over 20 years) and employment (from 0 to 173451 person days per habitat over a 20 year period) gain (Figure 3.2a-c). A decrease in budget (75% to 10% budget scenarios) did not affect biodiversity (Figure 3.2a), because 80 tree species would be planted at an average density of 1500 trees per hectare, and five more species are expected to recruit within the planted habitat over a 20 year period; hence, 85 tree species over 20 years. Although, a decrease in budget did not reduce tree biodiversity, it reduced tree population density and the planted area. For example, under the 100% budget, 866100 trees would be planted in 577.4 hectares whereas under a 10 % budget, 91200 trees would be planted in 60.8 hectares. Therefore, it would take a long time for the city to achieve its restoration objectives under a 10% budget (e.g., high carbon stock).

Table 3.2: Allocation of resources by RobOff to alternative restoration actions under budget scenarios across habitats. *Carbon* and *current action*s were not selected under any budget scenario.

Budget scenarios (US$)	Restoration habitats	Recommended actions	Allocated area (ha)	Allocated budget (US$)
2.711 million (100%)	Extant forest	Do nothing	33.3	0
		Natural regeneration action	72.5	0.13 million
	Former sugarcane fields	Do nothing	2.6	0
		Biodiversity action	577.4	2.564 million
2.033 million (75%)	Extant forest	Do nothing	105.8	0
	Former Sugarcane fields	Do nothing	121.8	0
		Biodiversity action	458.2	2.035 million
1.355 million (50%)	Extant forest	Do nothing	105.8	0
	Former sugarcane fields	Do nothing	276	0
		Biodiversity action	304	1.35 million
0.677 million (25%)	Extant forest	Do nothing	105.8	0
	Former sugarcane fields	Do nothing	428	0
		Biodiversity action	152	0.675 million
0.271 million (10%)	Extant forest	Do nothing	105.8	0
	Former sugarcane fields	Do nothing	519.2	0
		Biodiversity action	60.8	0.27 million

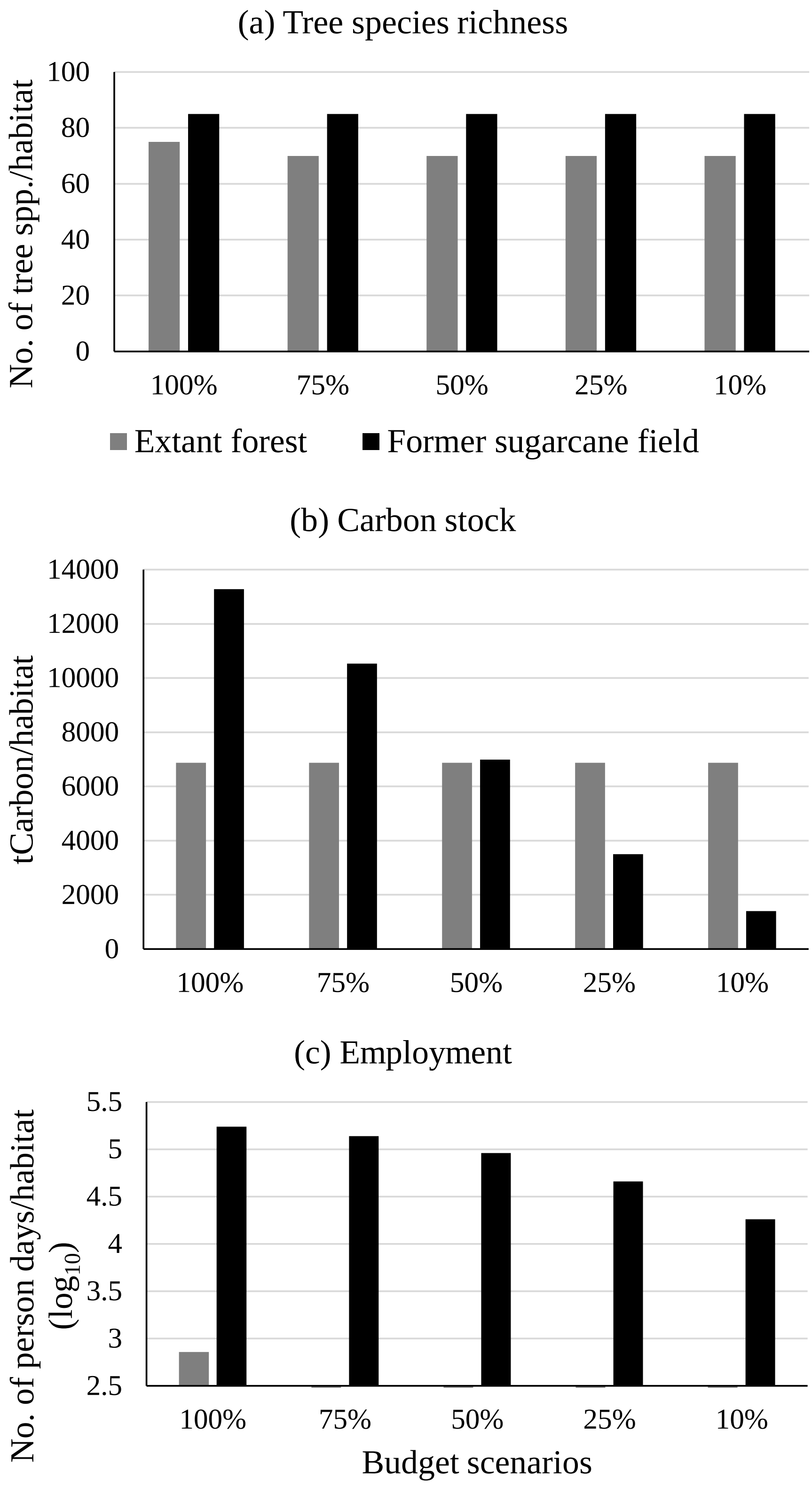

Figure 3.2: Restoration benefits, (a) biodiversity, (b) carbon stock and (c) employment under different budget scenarios, over a 20 year period. US$2.711 million = 100%, US$2.033 million = 75%, US$1.355 million = 50%, US$0.677 million = 25% and US$0.271 million = 10%.

3.2.2. Does prioritization of restoration benefits influence the allocation division of resources to alternative restoration actions and benefits across the habitats?

In the ‘extant forest’ under the 100% budget, giving more weight to one restoration goal (e.g., biodiversity) over the other did not affect the allocation of resources (Table 3.3), nor trade-offs in restoration benefits (Figure 3.3a-c). Under the 50% budget, a similar trend to the 100% budget was observed in terms of resource allocation (Table 3.3), although restoration benefits decreased (Figure 3.3a-c).

In the ‘former sugarcane fields’ under the 50% and 100% budget, restoration prioritization scenarios 1, 2, 3, 6 and 7 yielded similar resource allocation (Table 3.3). The *biodiversity action* was recommended under both the biodiversity and employment benefits prioritization scenarios, because it achieved more biodiversity and employment than the other alternative restoration actions. Carbon prioritization (scenarios 4 and 5) was at the expense of biodiversity, because this action only achieved 12 tree species over 20 years. However, when budget is not limited (US$2.711 million), carbon prioritization increased carbon stock by 8.2%, and employment decreased by 1.3% compared to the unweighted, biodiversity and employment scenarios (Figure 3.3a-c). The amount of benefits also decreased with a decrease in available budget (Figure S3.2a-c).

Table 3.3: The influence of budget scenarios on the allocation of resources to alternative restoration actions under restoration benefits prioritization scenarios (see Table S3.5 for scenarios description and the relative prioritization weight of their accociated benefits) across habitats. Scenarios that yielded similar results were lumped.

Budget scenarios	Prioritisation scenarios	Restoration habitats	Recommended actions	Allocated area (ha)	Allocated budget (million US$)
100%	1, 2 , 3, 6 and 7	Forest	Do nothing	33.3	0
			Natural regeneration action	72.5	0.013
	1, 2, 3, 6 and 7	Former sugarcane fields	Do nothing	2.6	0
			Biodiversity action	577.4	2.564
	4 and 5	Forest	Do nothing	33.3	0
			Natural regeneration action	72.5	0.013
	4 and 5	Former sugarcane fields	Do nothing	1	0
			Carbon action	579	2.7
50%	1, 2, 3, 6 and 7	Forest	Do nothing	105.8	0
	1, 2, 3, 6 and 7	Former sugarcane fields	Do nothing	276	0
			Biodiversity action	304	1.3
	4 and 5	Forest	Do nothing	105.8	0
	4 and 5	Former sugarcane fields	Do nothing	290.5	0
			Carbon action	289.5	1.3

Natural regeneration action cost per ha in the forest habitat only included clearing of fewer IAPs infestations, hence is lower than the cost of the similar action per ha in the 'former sugarcane fields', which included IAPs clearing and fire suppression. On the other hand, carbon action cost per ha was higher than the current and biodiversity actions, because it included slow growing tree species with high wood density, thus requiring more intense site maintenance than the current and biodiversity actions (Table S3.4).

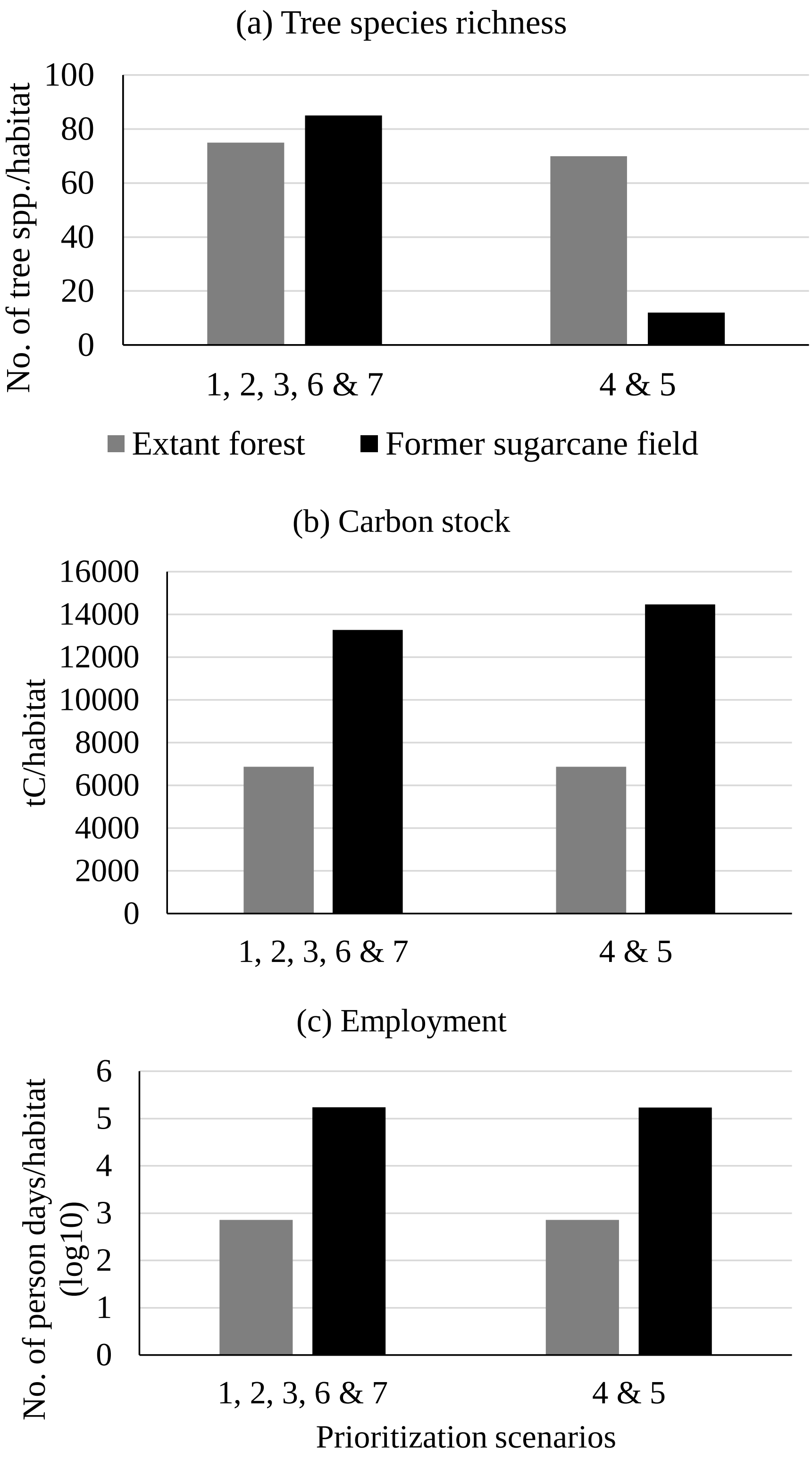

Figure 3.3: Restoration benefits, (a) biodiversity, (b) carbon stock and (c) employment under different restoration benefits prioritization scenarios (see Table S3.5) under 100% budget scenario, over a 20 year period. Lumped scenarios yielded similar results.

3.4. Discussion

Restoration ecologists and managers often find themselves with the challenge of how best to allocate limited resources to optimally achieve multiple restoration benefits (North et al. 2010). As a result, restoration planning is now moving away from traditional *ad hoc* approaches in the allocation of resources, to more coherent systematic approaches that offer higher rates of return on investment (e.g., Budiharta et al. 2014; Egoh et al. 2014). North et al. (2010), Orsi et al. (2011), Budiharta et al. (2014) and Egoh et al. (2014) have showed that decision-making support tools can assist in maximising outcome benefits of large-scale restoration projects. With multiple restoration goals to be achieved simultaneously, using tools such as RobOff can be helpful to systematically and effectively plan restoration interventions. RobOff compels the user (restoration planners) to identify objectives and intended outcomes of a restoration programme (e.g., climate change mitigation vs social advancement benefits), and to assess the suitability of multiple restoration actions to achieve the intended outcomes. RobOff identified better restoration interventions than current restoration approaches, that could have been employed to achieve greater biodiversity (= tree species richness), carbon stock and/or socio-economic advancement at Buffelsdraai Landfill Site. This is an important demonstration of how systematically planning for restoration can lead to greater benefits than *ad* hoc approaches.

In the restoration planning of moderately and highly degraded forests in Indonesia, Budiharta et al. (2014) found that, when budget is not limited, restoration of moderately degraded forest is additionally recommended to achieve restoration of threatened species habitat. However when the budget is limited, resources should always be allocated to the restoration of a highly degraded lowland forest to achieve carbon sequestration and to restore habitat of the threatened mammals (Budiharta et al. 2014). In our study, prioritization of the degraded ‘former sugarcane fields’ by RobOff shows that it is more beneficial to restore degraded land than restoring a partially degraded ‘extant forest’ which is still functional. This is especially so, if socio-economic benefit as job creation is targeted, as was the case at Buffelsdraai (Douwes et al. 2015). This is because restoring a partially degraded forest may typically achieve only a fraction of the potential employment benefits that could be realised when restoring ‘former sugarcane fields’. Furthermore, restoration of ‘former sugarcane fields’ simultaneously achieves the municipality’s other restoration benefits (biodiversity restoration, carbon sequestration and socio-economic advancement). Employment in the ‘extant forest’ only included IAPs clearing, whereas in the restoration of ‘former sugarcane fields’ employment opportunities included seedling production, land preparation, planting, and site maintenance, IAPs clearing and fire suppression.

A biodiversity-ecosystem functioning approach is increasingly being adopted in ecological restoration planning, because restoration of stable multiple ecosystem functions requires diverse species (Aerts and Honnay 2011; Cunningham et al. 2015). Species-rich systems are multifunctional, more stable and more productive, because different species functional traits allow species to fully utilize their limiting resources (Cardinale et al. 2012). They are more resistant and resilient to extreme climate change related disturbances such as pest and disease outbreaks, and changing fire patterns (Biringer and Hansen 2005; Aerts and Honnay 2011). Furthermore, numerous studies have shown that species-rich systems outperform species-poor systems in ecosystem functioning (e.g., carbon stock and nutrient cycling) (Aerts and Honnay 2011; Piotto et al. 2011; Hulvey et al. 2013; Cunningham et al. 2015), as well as in livelihood improvement in impoverished communities (e.g., food and medicinal plants provision) (Orsi and Geneletti 2010). In this study, *biodiversity action* was prioritized over alternative actions in the restoration of the ‘former sugarcane fields’. *Carbon action* was recommended when the main benefit sought was increased carbon stock, but the slight increase in carbon stock compared to biodiversity action does not warrant its implementation.

Our results have implication for the very popular tree planting initiatives around the world as a response to climate change mitigation (e.g., Piotto et al. 2011; Oldfield et al. 2015). Instead of focusing on reforestation to store carbon, restoration actions geared towards biodiversity as in the *biodiversity action* in this study, which achieved all three restoration benefits, might be a better alternative. Furthermore, an approach focused on improving diversity rather than tree density, is more likely to create a resilient socio-ecological ecosystem compared to the other actions (Biggs et al. 2015). While tree planting to store carbon contributes to climate change mitigation, if diversity is not considered, the system’s resilience is compromised, which could impact on climate change adaptation in the socio-ecological system. Species diversity is a key factor for the resilience of the socio-ecological system, particularly after disasters such as floods and fires (Adger et al. 2005; Leslie and McCabe 2014). Actions geared towards improving biodiversity are also important in the provision of other ecosystem services, since more and more studies are showing that biodiversity underpins most ecosystem services (Harrison et al, 2014). Interestingly, the biodiversity focused action also achieves employment opportunities, to alleviate poverty and improve lives of most people (Aronson et al. 2006).

Restoration practitioners are failing to show links between ecological restoration, society and policy (Aronson et al. 2010), and also underselling the full socio-economic benefits of restoration, which influence the society to invest in restoration as a primary tool of natural resource management (Aronson et al. 2010; Wortley et al. 2013). In developing countries, restoration projects are highly likely to receive more local support if they promote socio-economic development (Aronson et al. 2006; Le et al. 2011; Abram et al. 2014). The socio-economic benefits include income from forest products, employment opportunities, food and fibre provision and community capacity building (Chokkalingam et al. 2006; Le et al. 2011). Some of the past forest restoration projects have not been successful because they failed to account for the socio-economic needs of local communities in the planning phase (Saxena et al. 2001; Le et al. 2011; Barr and Sayer 2012). As a result, restoration projects are now designed to address both ecological and socio-economic needs (Mansourian and Dudley 2005). For example, communities are now involved in the selection of tree species that offer them benefits such as food, medicine, fuel and fodder (Saxena et al. 2001; Mekoya et al. 2008). Local community members can be recruited as labourers to carry out land preparation, planting, and site maintenance, to avoid undesirable social impacts. Furthermore, restoration projects that aim to achieve a balanced outcome (biodiversity and ecosystem services restoration, and socio-economic advancement) tend to be viewed more positively by funders, because of the broader benefits to the society (Sayer et al. 2004). Both biodiversity and carbon prioritization have more positive contributions to employment creation for local communities, compared to biodiversity benefit prioritization. Therefore, the perceptions of the local communities on BLSCRP are likely to be more positive. The likelihood of a negative perception is real, especially if local communities who were employed by sugarcane farmers in the area, suddenly find themselves without jobs as a result of the removal of land from sugarcane farming for conversion to ecological restoration sites.

3.5. Conclusion

This study has shown the value of adopting coherent and systematic approaches (decision-making support tools) to help improve the efficiency of large-scale restoration projects and their benefits. The consideration of different restoration options, and the inclusion of RobOff analyses in the original planning by the municipality of BLSCRP, would have shown how resources could have been more optimally allocated to alternative restoration actions that would achieve more biodiversity, carbon stock and employment benefits. Although our study assessed resource allocation within the forest ecosystem, RobOff is capable of solving very high dimensional problems, i.e. multiple ecosystems with multiple restoration actions and goals (Pouzols and Moilanen 2012). For example, ecosystem types such as forest, grassland and wetland require different restoration actions and achieve similar and different restoration benefits. RobOff could be used to efficiently and in a balanced manner allocate resources to these different ecosystem types.

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3.7. Supplementary materials

Table S3.1: Restoration actions and cost breakdown in the ‘extant forest’ and ‘former sugarcane fields’.

Habitats	Actions	Actions	Period (years)	Cost/ha (US$)	Total cost (US$)	Rationale
Extant forest (105.8 ha)	Do nothing (default in RobOff)	None	1-20	0	0	
	Natural regeneration action	Invasive alien plants (IAPs) clearing	1	38	38	It was assumed that the cost of IAPs clearing would decrease with follow up clearing and remains constant to cover maintenance (monitoring/clearing recruiting IAPs)
			2	31	31	
			3-20	17/year	306	
				Subtotal	**375**	
Former sugarcane fields (580 ha)	Do nothing	None	1-20	0	0	
	Natural regeneration action	IAPs clearing and fire suppression	1-20	213/year	**4260**	A constant cost over a 20 year period was used, because natural regeneration of trees is expected to be slow as a result of little or no soil seedbank depleted by intensive farming and recruitment will be dependent on seed dispersal
	Current action	Hole digging (1000 holes/ha)	1	97	97	Maintenance costs were decreased from years 8 to 20 by 40%. The reasoning provided was that an established forest cover would likely restrict the spread of IAPS. However, fire suppression would be still needed.
		Seedling price (1000 trees) Seedling transport (1000 trees)	1	131	131	

	Seedling transplant (1000 trees)	1	113	113	
	IAPs clearing and fire suppression	1	99	99	
		1-8	213/year	1704	
		9-16	191/year	1528	
		17-20	170/year	680	
			Subtotal	**4352**	
Carbon action	Hole digging (1000 holes/ha)	1	97	97	Based on the experts' knowledge, it was then estimated that the cost of producing tree seedlings with high wood density would be 50% greater than current action cost (experts' knowledge in the workshops), because most native trees with high wood density grow slow. This is supported by Shimamoto et. al. (2014) who found that fast growing trees can store more carbon over a shorter period of time (approximately 37 years), in the Brazilian Atlantic forest restoration site. However, over a longer period, slow growing trees can store twice as muchcarbon. Maintenance costs decreased from year 15 -20 by 20%, because most of the species in this action are characterized by slow growth.
	Seedling price (1000 trees)				
	Seedling transport (1000 trees)	1	196	196	
	Seedling transplant (1000 trees)				
	IAPs clearing and fire suppression	1	113	113	
		1	99	99	
		1-16	213/year	3408	
		17-20	191/year	765	
			Subtotal	**4678**	

	Biodiversity action	Hole digging (1000 holes/ha)	1	97	97	Tree species in this action are representative of the regional pool of scarp forest trees and some of these species might be rare and difficult to find and/or grow (seed dormancy that requires a long period or other treatment to break). As a result, it was estimated that the cost of producing their seedlings would be higher, hence a 75% increment to that of current action cost was employed. Maintenance costs were the same as for current action
		Seedling price (1000 trees)				
		Seedling transport (1000 trees)	1	229	229	
		Seedling transplant (1000 trees)				
		IAPs clearing and fire suppression	1	113	113	
			1	99	99	
			1-8	213/year	1704	
			9-16	191	1528	
			17-20	170/year	680	
				Subtotal	**4450**	

Table S3.2: Biodiversity (tree species richness) response to alternative restoration actions across restoration habitats over a 20 year period under three uncertainty envelopes. All carbon responses had a linear increase to the numbers reported in the table below.

Restoration habitats and actions	Uncertainty envelopes	Rationale	Period (years)		Source of information
			1	**20**	
Extant forest					
Do nothing (default in RobOff)	Average	Seventy tree species were recorded in the extant forest (Mugwedi et al. 2017). Species density (no. of species/habitat) is expected not to change, as a result of seed dispersal limitation from a distant forest patch and continuous habitat fragmentation caused by land-use change.	70	70	Bertoncini and Rodrigues (2008)
	Lower	A slight reduction in the species density is expected as a result of invasive alien plants (IAPs) infestation (e.g., shading, smothering native trees) and filling niche space. Over-harvesting of trees by the communities living adjacent to the forest could also reduce species number.	70	65	
	Upper	A slight increase in species density is expected as a results of seed dispersal from the nearby woodland fragments coupled with good growing conditions (e.g., more rain). For example, in Buffelsdraai there is a woodland fragment that has at least 10 species that were not found in the forest, but they could grow in the forest (Boon 2010).	70	75	
Natural regeneration action	Average	Species density is expected to increase slightly, as a result of seed dispersal from the nearby woodland fragment coupled with favourable	70	75	Bertoncini and Rodrigues (2008); Boon (2010); Chapman and Chapman (1999); eThekwini Municipality (2011); Expert

		climatic conditions (e.g., more rain).			knowledge; Hooper et al. (2005); Jogiste et al. (2003); Silverstrini et al. (2012).
	Lower	A slight reduction in species density is expected due to possible over-harvesting of trees with a few individuals.	70	68	
	Upper	An increase in species density is expected due to seeds being dispersed in from the surrounding woodland. Increased niche space created by IAPs removal is also expected to increase the chances of native tree recruitment.	70	80	
Former sugarcane fields					
	Average	Currently there are two tree species within the 'former sugarcane fields' habitat. Species density is expected to stay the same due to continuous farming activities.	2	2	Bertoncini and Rodrigues (2008);
Do nothing (default in RobOff)	Lower	A total loss of the existing tree species is expected due to possible removal by farmer.	2	0	
	Upper	A slight increase in species density is expected as a result of seed dispersed into the site from the surrounding woodland and forest fragments and recruitment on the edges of the sugarcane fields.	2	5	
Natural regeneration action	Average	Species richness is expected to increase as a result seed dispersal into the site from the surrounding forest and woodland. The richness is expected to recover slowly as a result of little or no seed bank depleted by intensive sugarcane farming (e.g., Botzat et al. 2015).	2	16	Bertoncini and Rodrigues (2008); Boon (2010);Botzat et al. (2015); Chapman and Chapman (1999); eThekwini Municipality (2011); Expert knowledge; Hooper et al. (2005); Jogiste et al. (2003); Silverstrini et al.

					(2012).
	Lower	Species density is expected to increase slowly as a result of grass colonisation and drought that may reduce tree recruitment and seedling establishment.	2	12	
	Upper	Species density is expected to be high as a result of good seed dispersal, high seedling recruitment and establishment coupled with favourable climatic conditions (e.g., more rain).	2	22	
Current action	Average	A slight increase in species density is expected as a result of seed dispersal from the surrounding forest and woodland fragments.	51	56	
	Lower	After 5 years, the species density is expected to decline as some species may not establish as a result of not enough shading, inappropriate soils or possible drought. After 10 years it is expected that some woodland species would also disappear due to too much shading.	51	41	Bertoncini and Rodrigues (2008); Boon (2010); Botzat et al. (2015); Chapman and Chapman (1999); eThekwini Municipality (2011); Expert's knowledge; Hooper et al. (2005); Jogiste et al. (2003); Silverstrini et al. (2012).
	Upper	Species density is expected to increase as a result of good seed dispersal, high seedling recruitment and establishment coupled with favourable climatic conditions (e.g., more rain).	51	61	
Carbon action	Average	Species density will remain the same due to the removal of any recruiting woody plants to facilitate faster growth of the carbon trees and to reduce competition for resources.	10	10	Bertoncini and Rodrigues (2008); Boon (2010); Botzat et al. (2015); Chapman and Chapman (1999); eThekwini Municipality (2011); Expert's knowledge; Hooper et al. (2005);

	Lower	Species density will remain the same due to the removal of any recruiting woody plants to facilitate faster growth of the carbon trees and to reduce competition for resources.	10	10	Jogiste et al. (2003); Silverstrini et al. (2012).
	Upper	In the first 10 years, naturally recruiting trees would be removed to facilitate faster growth of the carbon trees and to reduce competition. Thereafter species would be allowed to recruit.	10	15	
Biodiversity action	Average	A slight increase in species density is expected, as a result of seed dispersed into the site from the surrounding woodland and forest.	80	85	Bertoncini and Rodrigues (2008); Boon (2010); Botzat et al. (2015); Chapman and Chapman (1999); eThekwini Municipality (2011); Expert's knowledge; Hooper et al. (2005); Jogiste et al. (2003); Silverstrini et al. (2012).
	Lower	Ten species are expected to be lost due to inappropriate abiotic (e.g., drought) and biotic factors (e.g., herbivory and over-harvesting).	80	70	
	Upper	Species density is expected to increase as a result of good seed dispersal, high seedling recruitment and establishment coupled with favourable climatic conditions (e.g., more rain).	80	90	

Table S3.3: Carbon storage (tC/ha) response to alternative restoration actions across restoration habitats over a period of 20 years under three uncertainty envelopes.

Restoration habitats and actions	Uncertainty envelopes	Rationale	Period (years)		Source of information
			1	**20**	
Extant forest					
Do nothing (default in RobOff)	Average	The extantg forest patches are classified as Scarpforest. Glenday (2007) and National Terrestrial Carbon Sink Assessment (2015) estimated 66 tC/ha and 65tC/ha, respectively, for a Scarp forest. It is expected that the carbon stock in the forest will remain the same.	65	65	Expert's knowledge; van Rooyen et al. (2012); Glenday (2007), Department of Environmental Affairs, South Africa (2015).
	Lower	The carbon stock is expected to decrease as a result of severe drought and firestorm killing the trees.	65	0	
	Upper	The carbon stock is expected to increase as a result of favourable climatic condition (e.g., good rain) and an increase in atmospheric carbon.	65	75	
Natural regeneration action	Average	It is expected that the carbon stock will remain the same.	65	65	Expert's knowledge; van Rooyen et al. (2012); Glenday (2007), Department of Environmental Affairs, South Africa (2015).
	Lower	A slight reduction in carbon stock is expected as a result of severe drought and over-harvesting of trees by the communities living adjacent to the forest.	65	55	
	Upper	An increase in carbon stock is expected as a result of favourable climatic conditions and an increase in atmospheric carbon, there will be more tree recruitments.	65	75	

Former sugarcane fields					
Do nothing (default in RobOff)	Average	Above-ground carbon stock in sugarcane fields is zero, because when the sugarcane is harvested, all biomass is removed from the site.	0	0	Glenday (2007).
	Lower	Carbon stock in sugarcane fields is zero, same reason as above.	0	0	
	Upper	Currently there are two tree species within the Sugarcane environment. A slight increase in species density is expected is expected to increase carbon stock.	0	5	
Natural regeneration action	Average	Carbon stock is expected to be low as a result of slow recruitment of trees. Furthermore, it is suspected that the field would still be open woodland at the end of a 20 year period.	0	10	Expert's knowledge; van Rooyen et al. (2012); Glenday (2007), Department of Environmental Affairs, South Africa (2015).
	Lower	Carbon stock is expected to be low as a result of slow tree recruitment and drought.	0	6	
	Upper	Carbon stock is expected to increase slightly as a result of increased tree recruitment coupled with favourable climatic conditions.	0	14	
Current action	Average	The carbon stock is expected to reach 20tC/ha over a 20 year period.	0	20	Same as Natural regeneration action.
	Lower	Carbon stock is expected to increase slowly as a result of a decrease in tree density caused by drought.	0	15	
	Upper	Carbon stock is expected to increase rapidly owing to favourable climatic conditions (e.g., good rains).	0	30	

Carbon action	Average	This action is expected to result in more carbon stored, because only trees with high wood density will be planted.	0	25	Same as Natural regeneration action.
	Lower	Unfavourable climatic conditions (e.g., drought) are expected to slow down tree growth, hence low carbon stock.	0	15	
	Upper	The carbon stock is suspected to surpass that of the Dry Valley Thicket/Broadleaf Woodland (Glenday, 2007) owing to favourable climatic conditions (e.g., good rains).	0	35	
Biodiversity action	Average	The carbon stock is expected to be below that of the carbon action. However, in 50 years this action is expected to store more carbon than carbon action.	0	23	Same as Natural regeneration action.
	Lower	Unfavourable weather conditions (e.g., drought) are expected to slow down tree growth, hence low carbon stock.	0	15	
	Upper	Carbon stock is expected to be high owing to favourable climatic conditions (e.g., good rains).	0	30	

Table S3.4: Employment response to alternative restoration actions across restoration habitats over a period of 20 years under three uncertainty envelopes.

Restoration habitats and actions	Uncertainty envelopes	Rationale	Period (years)		Source of information
			1	**20**	
Extant forest					
Do nothing (default in RobOff)	Average	No activity.	0	0	
	Lower	No activity.	0	0	
	Upper	No activity.	0	0	
Natural regeneration action	Average	Two person days/ha/year are needed to control IAPs and do follow up in the first and second year. From year three to year 20, only one person day/ha/year will be needed to do monitoring and clearing of any recruiting IAPs.	2	1	Marais and Wanneburgh (2008); South African National Parks' Working for Water Programme - invasive alien plants clearing project.
	Lower	Two person days/ha/year are needed to control IAPs and do follow up in the first and second year. From year three to year 20, only one person day/ha/year will be needed to do monitoring and clearing of any recruiting IAPs.	2	1	
	Upper	Three person days/ha/year are needed to control IAPs in year one, followed by two person days to do follow up in year two, thereafter followed by one person day/ha/year from year three to year 20 to do monitoring and clearing of any recruiting IAPs.	3	1	

Former sugarcane fields					
Do nothing (default in RobOff)	Average	No activity.	0	0	
	Lower	No activity.	0	0	
	Upper	No activity.	0	0	
Natural regeneration action	Average	Twelve person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	12	12	Same as Natural regeneration action in the extant forest.
	Lower	Ten person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	10	10	
	Upper	Fourteen person days are needed to control IAPs. Person days/ha/year needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	14	14	
Current action	Average	Fifty nine person days/ha/year were needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, twelve person days/ha/year are needed to control IAPs.	59	12	EThekwini Municipality (2011); Marais and Wanneburgh (2008); South African National Parks' Working for Water Programme -invasive alien

		Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.			plants clearing project.
	Lower	Fifty seven person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, ten person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	57	10	
	Upper	Sixty one person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, fourteen person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	61	14	
Carbon action	Average	Sixty eight person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, twelve person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and	68	12	Same as in Current action

		clear recruiting IAPs.			
	Lower	Sixty five person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, ten person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	65	10	
	Upper	Seventy person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, fourteen person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	70	14	
Biodiversity action	Average	Seventy two person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, twelve person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	72	12	Same as in Current action
	Lower	Seventy person days/ha/year are needed for producing seedlings, land preparation and planting	70	10	

		in the first year. From year two to year 20, ten person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.		
	Upper	Seventy four person days/ha/year are needed for producing seedlings, land preparation and planting in the first year. From year two to year 20, fourteen person days/ha/year are needed to control IAPs. Person days needed to control IAPs are expected to remain the same over a 20 year period, because it is expected that the same number of person days will be needed to make fire breaks, to monitor and clear recruiting IAPs.	74	14

Table S3.5: Restoration benefits prioritization scenarios and relative weight for each benefit. The 100% and 50% budget scenarios were used to run these scenarios.

Prioritization Scenarios	Prioritised feature	Biodiversity weight	Carbon storage weight	Employment creation weight
1	Unweighted	1	1	1
2	Biodiversity	5	1	1
3	Biodiversity	10	1	1
4	Carbon	1	5	1
5	Carbon	1	10	1
6	Employment	1	1	5
7	Employment	1	1	10

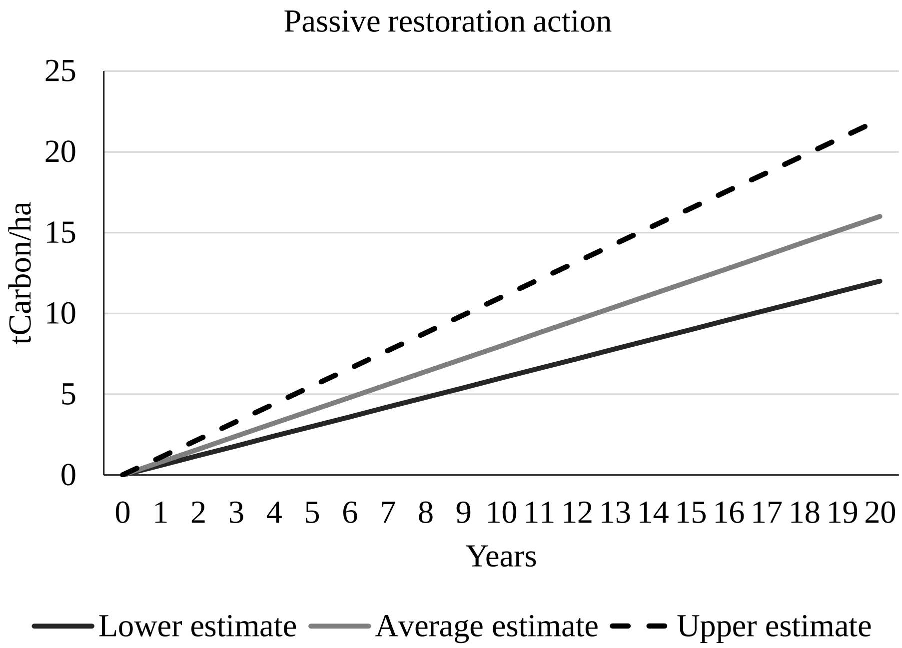

Figure S3.1: A typical example of carbon storage response to Natural regeneration action in the ‘former sugarcane fields’.

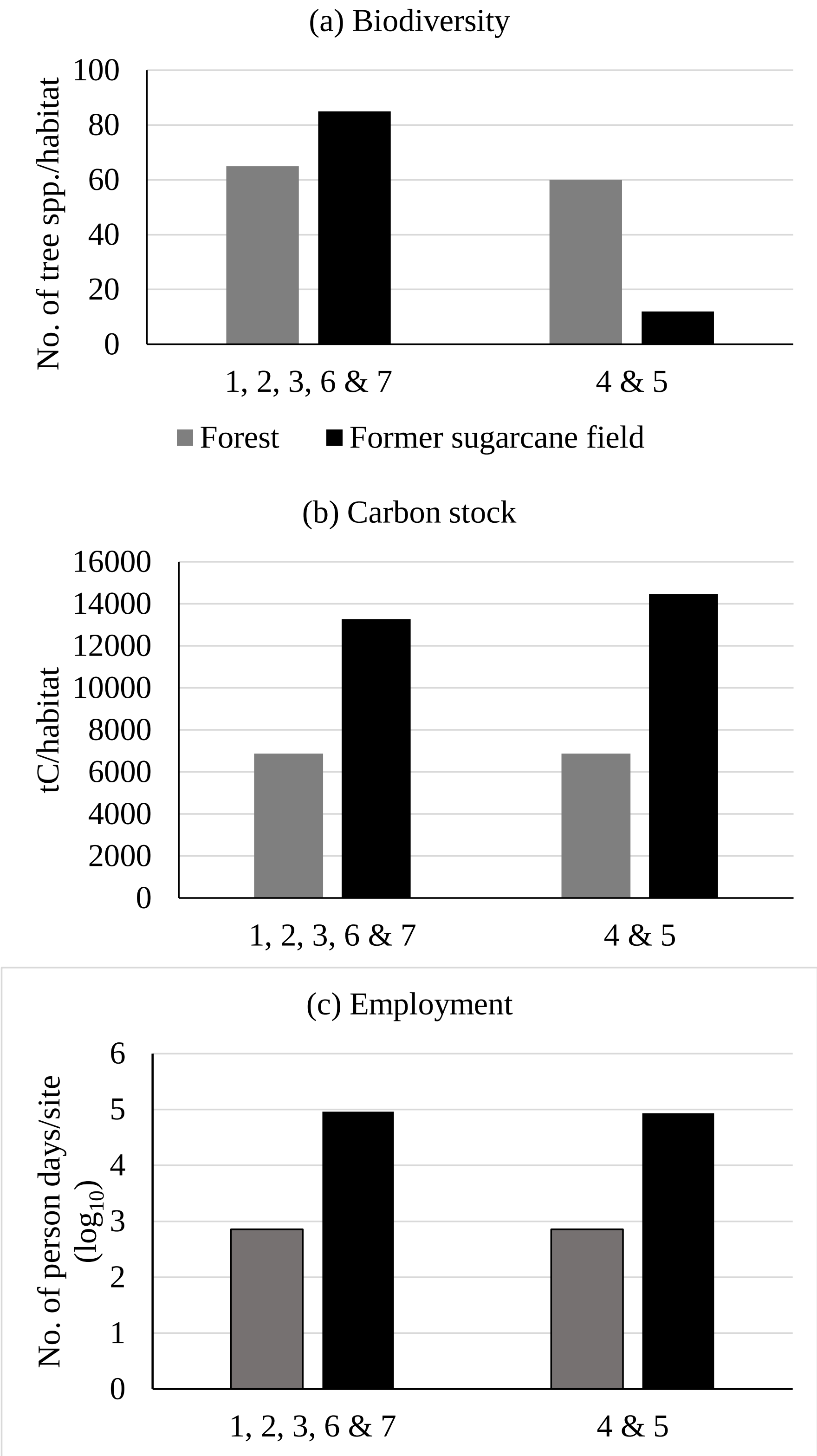

Figure S3.2: Restoration benefits, (a) biodiversity, (b) carbon stock and (c) employment under different restoration benefits prioritization scenarios (see Table S3.5) under 50% budget scenario. Lumped scenarios yielded similar results

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CHAPTER 4: GROWTH PERFORMANCE OF NATIVE TREE SPECIES IN RESTORATION OF A SUBTROPICAL BIODIVERSITY HOT SPOT FOREST IN DURBAN, SOUTH AFRICA

This chapter is based on

LF Mugwedi, M Rouget, B Egoh, Sershen, S Ramdhani, R Slotow (**to be submitted to the Durban Research Action Partnership Special Issue**) Growth performance of native tree species in restoration of a subtropical biodiversity hot spot forest in Durban, South Africa.

Abstract

Planting of native tree species is now a widely adopted approach to achieve biodiversity conservation, climate change mitigation and adaptation, and ecosystem services restoration, in degraded forests. However, knowledge on the growth performance of native species in relation to microtopographic conditions, and soil physical and chemical properties, is still lacking, especially in the subtropical forests. This knowledge is useful to inform the selection of tree species that promote rapid forest establishment, thus making the most of limited resources, and promoting rapid achievement of reforestation goals. Using a chronosequence of three habitats under restoration (0-,3-, and 5-year-old), and contrasting different microtopographic conditions (upland and lowland areas) within each habitat, root-collar diameter, height and canopy width of four most dominant planted tree species (*Bridelia micrantha*, *Erythrina lysistemon*, *Millettia grandis* and *Vachellia natalitia*) were assessed. Higher soil moisture coupled with higher nutrient levels in the lowland area promoted rapid establishment of *B. micrantha*, *E. lysistemon* and *V. natalitia*. The lowland area had the tallest species with wider canopies. *Erythrina lysistemon* was the best performing species in both the upland and lowland areas. *Erythrina lysistemon* and *V. natalitia* were found to be more suitable to both upland and lowland areas, and *B. micrantha* to lowland area. *Millettia grandis* did not perform well in both the upland and lowland areas, making it unsuitable to be used as a pioneer reforestation species. These results indicate that different species are better candidates for different microhabitats, and this can result in different reforestation trajectories. Therefore, to promote rapid forest establishment and restoration goals achievement, microtopographic conditions should form basis for selection of reforestation species selection.

Keywords: Growth rates; Microtopographic conditions; Severely degraded site; Soil nutrients and moisture

4.1. Introduction

With unprecedented changes in climate and land-use patterns, a decrease in global biodiversity has been proceeding at an alarming rate, causing widespread damage to the life-support systems upon which every living organism depends (Hobbs and Harris 2001; Sangermano et al. 2012). To counteract biodiversity and ecosystem services loss, conservation programmes are being established, of which restoration of degraded forest ecosystems is becoming an important component (Lamb and Gilmour 2003; Mansourian et al. 2005; Aronson and Alexander 2013). As a result, many countries around the world have embarked on large-scale reforestation initiatives, but planting monocultures of exotic tree species, especially for timber, is still a common phenomenon (Tolentino 2008; Davis et al. 2012). This approach has been widely criticized, because it offers few ecological and socio-economic benefits compared to diverse native forests (e.g., Smith 2002; Cao et al. 2009; Fernandes et al. 2016).

Planting a mixture of native species is increasingly adopted to safeguard biodiversity and ecosystem services in the face of global environmental change (Aronson and Alexander 2013; Cunninham et al. 2015). Species-rich systems are multifunctional, more stable and more productive, because differences in species functional traits allow interacting species to fully utilize their limiting resources (Cardinale et al. 2012). These systems are more resistance and resilient to extreme climate change related disturbances such as pest and disease outbreaks, and changing fire patterns (Biringer and Hansen 2005; Aerts and Honnay 2011). Furthermore, species-rich systems outperform species-poor systems in ecosystem functioning (e.g., carbon stock and nutrient cycling) (Aerts and Honnay 2011; Hulvey et al. 2013; Cunningham et al. 2015) and livelihoods improvement in impoverished communities (e.g., food and medicinal plants provision) (Orsi and Geneletti 2010).

Conversion of land area for agricultureal production is the main driver of biodiversity and ecosystem services losses in the world (Gibbs et al. 2010; Hosonuma et al. 2012). Extensive agricultural practices can lead to soil nutrients depletion, heavy metal build up, soil compaction and soil erosion, thus making it difficult for native tree species to establish (Eshetu 2002; Baumert et al. 2016; Khamzina et al. 2016). Planting a higher diversity of native tree species under such conditions in the initial planting phase can be technically challenging and costly (e.g., Lu et al. 2017). To counteract this challenge, inter-planting of selected fast growing native pioneer and native late-successional tree species has been found to complement and accelerate natural regeneration and enhance biodiversity recovery (Elliott et al. 2003; Goosem and Tucker 2013). Characteristics of a good reforestation pioneer species include: easy propagation, higher survival rate, and wide spread and dense canopies that suppress weed establishment, and creating microclimate that promotes late successional tree species establishment while reducing site maintenance costs (Elliott et al. 2003; Douterlungne et al. 2013; Goosem and Tucker 2013; Douterlungne et al. 2015). This is because their height and dense canopies attract seed dispersing wildlife (Reid et al. 2012; Goosem and Tucker 2013). This approach offers a win-win situation to developing countries and local communities with limited financial and technical assistance to carry out large-scale reforestation initiatives (Lu et al. 2017).

Microtopographic variation (upland and lowland areas) within a site effects hydrologic conditions, and soil physical and chemical properties (Simmons et al. 2012). As a result, within a species, individual tree growth rates may vary significantly (Omary 2011; Simmons et al. 2012; Li et al. 2014). Although reforestation has been implemented for more than three decades in South Africa (van Aarde et al. 1996), there are little available data on the field performance of many native species used in the reforestation initiatives. Knowledge on the influence of microtopographic position, and soil physical and chemical properties conducive to tree growth, can help to reduce site maintenance costs (e.g., weed control and fire suppression) needed to promote tree establishment (Henri 2001; Douterlungne et al. 2015), and help fast achievement of reforestation goals. In a desktop study, Roy (2015) identified 30 potential native tree species that could be used in the restoration of the severely degraded scarp forest in Durban, South Africa, but recommended that their field growth performance be assessed. Therefore, the aim of this study was to assess the influence of microtopographic position, and soil physical and chemical properties on the growth of the four most dominant planted tree species, by using a chronosequence of three habitats under restoration (0-year-old, 3-year-old, and 5-year-old). To achieve this aim, the following objectives were developed: 1) to identify tree species that could best be used to restore the upland area, lowland area, or both; 2) to understand the influence soil physical and chemical properties on tree performance. Findings from this study will be used to guide future reforestation planning in severely degraded sites within the eThekwini Municipal Area and reforestation projects elsewhere.

4.2. Methodology

4. 2.1. Study area

The study was conducted at Buffelsdraai Landfill Site (29.62961S; 30.980392E), the largest regional solid waste landfill in KZN, South Africa. The active landfill site is 116 ha and has a buffer zone of 757 ha, of which 580 ha (former low productive dryland sugarcane field) has been planted with over 51 native tree species. An average of 1000 (in the dry habitat) and 2000 (in the wet habitat) tree saplings were planted in the wet season (Douwes et al. 2015a). The remainder of the buffer zone is characterized by mosaic patches of native forest, woodlands and grasslands, with almost all the sites being invaded by alien plants (MacFarlane et al. 2011). The vegetation, broadly classified as KwaZulu-Natal Coastal Belt (grassland and subtropical forest), is highly transformed and fragmented with little formal protection. The vegetation type as a whole is classified as endangered (Mucina and Rutherford 2006). The remnants of native forest form part of Eastern Scarp Forest, usually located at an altitude of 100 to 1000 m (Von Maltitz et al. 2003), and is described as a refuge forest that survived the last glacial maximum (≈18000 BP) (Eeley et al. 1999). This region receives a high summer rainfall (440-1400 mm), with some rainfall in winter (Von Maltitz et al. 2003). The annual mean maximum temperature ranges from 22.2 °C in winter and 27.4 °C in summer (Roy 2015).

The South African Eastern Scarp Forest forms part of the Maputaland-Pondoland-Albany biodiversity hotspot (Mucina and Rutherford 2006). Approximately 15-31 % of Eastern Scarp Forest has been lost due to land use change (e.g., clearing for agriculture) and non-sustainable harvesting of forest products by rural communities, but it should be noted that this estimate is quite dated (Von Maltitz et al. 2003). This has resulted in fragmentation between patches and an increased edge ecotone (Kotze and Lawes 2007). The restoration of degraded forests is a conservation priority in South Africa (van Aarde et al. 1996; Scholtz 2007; Els 2010). The eThekwini Municipality (city of Durban) is addressing habitat transformation through the restoration of degraded forest ecosystems. This opportunity was presented by the city's hosting of 2010 FIFA™ World Cup and the United Nations Framework Convention on Climate Change COP17/CMP7 in 2011 (Diederichs and Roberts 2015). A major goal of the reforestation was to offset the carbon footprint associated with hosting of both mega events while enhancing the ecological capacity and potential to adapt to climate change, through socio-economic improvement (job creation), biodiversity protection and ecosystem services provision (e.g., water quality) (Douwes et al. 2015a).

The topography of the study area is characterized by the undulating and steep slopes (200 m to 325 m altitude) draining into the Black Mhlasini stream in the north and White Mhlasini stream in the south. A glacial conglomerate parent material that is base-rich, hard, and resistant to weathering, the Dwyka Tillite is the dominant geology within the site. The upland area is characterized by shallower Acrisol (20-40 cm). The soil is probably shallower as a result of higher soil erosion caused by runoff, exacerbated by cultivation. The lowland area is characterized by deeper lithol soil (60-110 cm). The deeper soil is probably as a result of the deposition of materials from the upper slopes (EThekwini Municipality 2014).

4.2.2. Annual rainfall pattern from the year 2010 to 2015

The year 2010 received a lower rainfall (564 mm), followed by a higher rainfall from the year 2011 to 2013 (1024 mm, 1008 mm and 886 mm, respectively), but the rainfall decreased in 2014 and 2015 (564 mm and 312 mm, respectively) resulting in one of the the worst drought in South Africa (Botai et al. 2016). Rainfall data were sourced from Phoenix weather station (S29.7043, E30.9761), approximately seven kilometres from the study area.

4.2.3. Experimental design and data collection

Tree planting started in 2009-2010 growing season (between November and February) and continued every year until 2014-2015 growing season. Planting coincided with the period in which more rainfall was received. Vegetation sampling was done at three habitats planted in 2009-2010 (hereafter referred to as the ‘5-year-old habitat’), 2011-2012 (hereafter referred to as the ‘3-year-old habitat’) and 2014-2015 (hereafter referred to as the ‘0-year-old habitat’).

In order to assess the impact of microtopographic position on tree growth performance, upland and lowland areas were selected within each restoration habitat (0, 3 and 5-year-old). Upland area is drier while lowland area is wetter (EThekwini Municipality 2014). A survey was carried out to identify the commonly planted native pioneer tree species that could be studied. This was achieved by establishing four walk-belt transects of 8 m by 200 m in both the upland and lowland areas within each habitat. Six dominant species were identified (Table S4.1), but only four species that had more than 30 individuals in both upland and lowland areas across the habitats were selected for the study (Table 4.1). These species are easy to propagate, an important factor to consider when selecting fast growing pioneer species (Elliot et al. 2003; Goosem and Tucker 2013). According to Boon (2014, pers. comm.), the selected species meet the criteria of scarp forest pioneer trees. Pioneer trees are species that require full sunlight for recruitment and growth, and, as a result, they cannot recruit under their own or other species closed canopy (Swaine and Whitmore 1988). They create more shady conditions under which more shady tolerant species (climax) can recruit and grow. *Bridelia micrantha* (Hochst.) Baill. (Phyllanthaceae) is associated with scarp forest, and it is usually found on forest margins. *Millettia grandis* (E. Mey.) Skeels (Fabaceae) is a fast growing forest pioneer that could be used to restore forest (Geldenhuys and Delvaux 2002). *Erythrina lysistemon* Hutch. (Fabaceae) occurs on scarp forest margins (Boon 2014, pers. comm.). *Vachellia* (former *Acacia) natalitia* E. Mey. (Fabaceae) is a potential bush encroachment species in savannah, which may ultimately lead to the establishment of forest, and it is occasionally found on forest margins (Mucina and Geldenhuys 2006). Tudor-Owen and Wyatt (1991) documented that *B. micrantha*, *V. natalitia* and *M. grandis* as suitable pioneer species to reforest areas degraded by sugarcane plantation. It is worth mentioning that *V. natalitia* is often confused with *V. karroo*, because they are very similar (Boon 2010; van Wyk and van Wyk 2013). As a result, *V. natalitia* was included under *V. karroo* in the older literature (e.g., Coates Palgrave 2002). *Vachellia natalitia* has leaves with 6-12 pairs of pinnae while *V. karroo* has 3-5 pairs of pinnae (Boon 2010; van Wyk and van Wyk 2013).

Table 4.1: Characteristics, distribution and economic uses of the four most dominant planted tree species in the reforestation of the former sugarcane field at Buffelsdraai Landfill Site.

Species	Family	Height	Leaf phenology	Distribution	Economic uses	References
Bridelia micrantha (Hochst.) Baill.	Phyllanthaceae	Up to 20 m	Deciduous	Angola, Burundi, Cameroon, Cote d'Ivoire, Democratic Republic of Congo, Eritrea, Ethiopia, Gambia, Ghana, Guinea, Kenya, Liberia, Malawi, Mali, Mozambique, Nigeria, Rwanda, Senegal, Sierra Leone, South Africa, Sudan, Tanzania, Togo, Uganda, Zambia, Zanzibar, Zimbabwe	Charcoal, Firewood, Fodder, medicinal, timber	Roothaert and Franzel (2001); Okullo and Waithum (2007); World Agroforestry centre (2009); van Wyk and van Wyk, 2013
Erythrina lysistemon Hutch.	Fabaceae	Up to 12 m	Deciduous	Botswana, Malawi, Mozambique, South Africa, Swaziland, Tanzania, Zambia and Zimbabwe	Ornamental, medicinal	Boon 2010; Nesom 2015; van Wyk and van Wyk, 2013
Millettia grandis (E.Mey.) Skeels	Fabaceae	Up to 25 m	Semi-deciduous	South Africa and Mozambique	Timber, medicinal,	van Wyk and van Wyk, 2013
Vachellia natalitia (E.Mey.) & Kyal. & Boatwr.	Fabaceae	Up to 12m	Semi-deciduous	South Africa, Mozambique and Zimbabwe	Medicinal, fodder, firewood and timber	van Wyk and van Wyk, 2013

In February 2015, 30 individuals of each species per microtopographic position per habitat were assessed (n = 720 individuals). The assessed individuals were chosen after establishing a centre line transect within the belt transect. The closest trees to transect were chosen. Tree root-collar diameter, height, and canopy width were measured. Root collar diameter was measured using callipers at 10 cm above ground. Although some of the trees in the 3-year-old habitat and 5-year-old habitat had a diameter at breast height (1.3 m) that could be measured, in order to ensure a consistent measure across species to test for significant difference in tree growth, root-collar diameter was measured (Oldfield et al. 2015). Height was measured from the soil level to the tip of the lead stem. Canopy width was measured along two directions (north-south and east-west) (Li et al. 2014).

To determine soil physical and chemical properties, soil was sampled from three depths, 0-10, 10-20 and 20-60 cm, in a pit (1m length x 1m width x 0.6m depth). Three pits per habitat were dug within each microtopographic position (upland and lowland areas). Soils were sieved at 2 mm to remove small stones, plant roots and other debris and stored at 4°C before analysis (Matías et al., 2011). Soil analyses were performed by the Soil Fertility Analytical Services Section, KwaZulu-Natal Department of Agriculture and Environmental Affairs. The following properties were analysed; soil texture, pH, total nitrogen, phosphorus, potassium, calcium and magnesium.

4.2.3. Data analyses

Tree canopy width was quantified as the mean of north-south and east-west lengths. Soil physical and chemical properties were statistically comparable across the soil depths (0-10 cm, 10-20 cm and 20-60 cm), hence the data were pooled. To fulfil homoscedasticity assumptions, values were either square-root or log-transformed. A three-way analysis of variance (ANOVA) was performed to test the effect of habitat, microtopographic position and species, on root-collar diameter, height, and crown width. A two-way ANOVA was used to test the effect of habitat and microtopographic position effect on soil physical and chemical properties. Tukey HSD post hoc test was used to separate means with significant differences at $P < 0.05$. All the analyses were performed in Statistica version 11 (StatSoft 2011).

4.3. Results

4.3.1. Species growth rate

Species growth rates among habitats, microtopographic position, and species were significantly ($P < 0.05$) different. Significant ($P < 0.05$) interaction effect among habitats, microtopographic position, and species, showed that species differed in their growth rates, and changed their ranking in relation to each other across habitats and microtopographic position. The overall results showed an increase in root-collar diameter (RCD), height and canopy width (CW) with an increase in restoration age in both upland and lowland areas. Overall, species growing in the lowland area performed better than species growing in the upland area (Table 4.2).

4.3.1.1. Species performance comparison within habitat

In the 0-year-old habitat, *E. lysistemon* showed a rapid establishment in RCD (1.57 cm) in the upland area and CW (0.47 m) in the lowland area. The RCD was significantly ($P < 0.05$) greater than the other species, except *E. lysistemon* (1.55 cm) in the lowland area and *V. natalitia* (1.09 cm) in the upland area while the CW was only significantly ($P < 0.05$) greater than *B. micrantha* (0.21 m) in the upland area. *Bridelia micrantha* in the lowland area showed a rapid growth in height (0.61 m), but only significantly ($P < 0.05$) taller than *M. grandis* (0.43) in the upland area (Table 4.2).

In the 3-year-old habitat, *E. lysistemon* in the lowland area had the highest RCD (9.17 cm) and CW (2.37 m). The RCD was significantly ($P < 0.05$) greater than the other species except *E. lysistemon* in the upland area (7.14 cm), while the CW was only statistically comparable to *B. micrantha* (2.25 m) and *V. natalitia* (1.86 m) in the lowland area. *Bridelia micrantha* in the lowland area was significantly ($P < 0.05$) taller (2.90 m) than the other species except *E. lysistemon* (2.84 m) and *V. natalitia* (2.26 m) in the lowland area (Table 4.2).

In the 5-year-old habitat, *Erythrina lysistemon* RCD (18.67 cm) and CW (3.23 m) showed a similar trend to the 3-year-old habitat. The RCD was only statistically comparable to *E. lysistemon* in the upland area (12.67 cm) while the CW was only statistically comparable to *B. micrantha* (2.77 m) in the lowland area and *V. natalitia* in both upland and lowland areas (2.73 m and 2.69 m, respectively). *Bridelia micrantha* height (3.09 m) showed a similar trend to the 3-year-old habitat, but statistically comparable to *E. lysistemon* (3.06 m) in the lowland area, and *V. natalitia* in both upland and lowland areas (2.79 m and 2.37 m, respectively (Table 4.2).

4.3.1.2. Within species comparison across habitats

Tree root-collar diameter (RCD)

The 0-year-old habitat in upland and lowland areas had the lowest RCD across all species, significantly ($P < 0.05$) lower than the other habitats (Table 4.2). In the upland area, *B. micrantha* RCD was statistically comparable between the 3-year-old and the 5-year-old habitats. In the lowland area, *B. micrantha* in the 5-year-old habitat had the highest RCD, significantly ($P < 0.05$) higher than the other habitats. *Erythrina lysistemon* had the highest RCD in the 5-year-old habitat in both upland and lowland areas, significantly ($P < 0.05$) higher than the other habitats. *Millettia grandis* RCD between the 3- and 5-year-old habitats in both upland and lowland areas were statistically comparable. *Vachellia natalitia* showed a similar trend to *E. lysistemon* (Table 4.2).

Tree height

Species height in the 0-year-old habitat in both upland and lowland areas showed a similar trend to the RCD. *Bridelia micrantha* was taller in the 5-year-old habitat, but statistically comparable to the 3-year-old habitat in both upland and lowland areas (Table 4.2). *Erythrina lysistemon* showed a similar trend to *B. micrantha*. In the lowland area, tallest *M. grandis* was found in the 3-year-old lowland area, but statistically comparable to the 5-year-old habitat. In the upland area, tallest *M. grandis* was found in the 5-year-old habitat, but statistically comparable to the 3-year-old. *Vachellia natalitia* was tallest in the 5-year-old habitat in both upland and lowland areas, significantly ($P < 0.05$) taller than the other habitats (Table 4.2).

Tree canopy width (CW)

Species CW in the 0-year-old habitat in both upland and lowland areas showed a similar trend to the RCD (Table 4.2). Highest *B. micrantha* CW was found in the 5-year-old habitat, lowland area. In the upland area, the 3- and 5-year-old habitats were significantly ($P < 0.05$) different, but statistically comparable in the lowland area. Highest *E. lysistemon* CW was found in the 5-year-old habitat in both upland and lowland areas, significantly higher than the other habitats. *Millettia grandis* CW showed a similar trend to *M. grandis* height. *Vachellia natalitia* showed a similar trend to *E. lysistemon* (Table 4.2).

Table 4.2: Species (mean±SE) root-collar diameter (RCD), stem height and crown width (CW) annual increments at the Bufflesdraai Landfill Site. Different letters in small caps depict significant difference between species, within each habitat at $P< 0.05$. Different letters in caps depict significant difference within each species across habitats at $P< 0.05$.

Microtopographic position	Species	RCD (cm)/habitat			Height (m)/habitat			Canopy width (m)/habitat		
		0-year-old	3-year-old	5-year-old	0-year-old	3-year-old	5-year-old	0-year-old	3-year-old	5-year-old
Upland area										
	Bridelia micrantha	0.79±0.04^{aA}	3.79±0.21^{deB}	4.58±0.34^{dBC}	0.54±0.03^{abA}	1.57±0.07^{cB}	1.94±0.09^{bB}	0.21±0.01^{bA}	1.23±0.08^{dB}	1.76±0.14^{cC}
	Erythrina lysistemon	1.57±0.08^{bA}	7.14±0.41^{abB}	12.67±1.73^{abC}	0.47±0.04^{abA}	1.83±0.08^{bcB}	2.19±0.12^{bB}	0.36±0.02^{abA}	1.79±0.12^{cB}	2.53±0.18^{bC}
	Millettia grandis	0.80±0.05^{aA}	2.96±0.12^{eB}	3.63±0.19^{deB}	0.43±0.03^{bA}	1.7±0.07^{bcB}	1.88±0.06^{bB}	0.30±0.02^{abA}	1.13±0.07^{dB}	1.35±0.09^{cdB}
	Vachellia natalitia	1.09±0.04^{abA}	3.44±0.16^{deB}	7.00±0.35^{cC}	0.52±0.04^{abA}	1.49±0.06^{cB}	2.37±0.08^{abC}	0.29±0.02^{abA}	1.24±0.07^{dB}	2.69±0.14^{abD}
Lowland area										
	Bridelia micrantha	0.79±0.03^{aA}	5.71±0.29^{bcC}	9.33±0.92^{bcD}	0.61±0.03^{aA}	2.90±0.15^{aC}	3.09±0.11^{aC}	0.26±0.01^{abA}	2.25±0.11^{abCD}	2.77±0.12^{abD}
	Erythrina lysistemon	1.55±0.09^{bA}	9.17±0.53^{aBC}	18.67±1.76^{aD}	0.54±0.03^{abA}	2.84±0.09^{aC}	3.06±0.11^{aC}	0.47±0.02^{aA}	2.37±0.13^{aC}	3.23±0.15^{aD}
	Millettia grandis	0.92±0.03^{aA}	3.64±0.17^{deB}	3.45±0.29^{eB}	0.48±0.02^{aA}	2.15±0.08^{bB}	2.02±0.09^{bB}	0.32±0.01^{abA}	1.52±0.09^{cdB}	1.16±0.10^{dB}
	Vachellia natalitia	0.86±0.06^{aA}	4.86±0.23^{cdB}	7.49±0.30^{bcC}	0.47±0.02^{abB}	2.26±0.11^{abB}	2.79±0.10^{aC}	0.38±0.04^{abA}	1.86±0.08^{abC}	2.73±0.15^{abD}

4.3.2. Soil physical and chemical properties

Significant interaction ($P < 0.05$) between habitat and microtopographic position occurred in total nitrogen, phosphorus, magnesium, calcim, pH and Clay. Total nitrogen, calcium, magnesium, pH and sand were consistently higher in the lowland area across habitats, while phosphorus and potassinm were consistently higher in the upland area (Table 4.3).

In the upland area, the 0-year-old habitat soil type was loam, clay loam in the 3-year-old habitat and clay in the 5-year-old habitat. The total nitrogen level was highest in the 5-year-old habitat, significantly ($P < 0.05$) higher than the 0-year-old habitat. Phosphorus level was highest in the 3-year-old habitat, significantly ($P < 0.05$) higher than the other habitats. Calcium level was highest in the 5-year-old habitat, significantly ($P < 0.05$) higher than the other habitats. Magnesium level was significantly ($P < 0.05$) different across habitats, but it was highest in the 5-year-old habitat and lowest in the 0-year-old habitat. Soil in the 3-year-old habitat, upland areas was more acidic, significantly ($P < 0.05$) different to the other habitats. Potassium levels were statistically comparable (Table 4.3).

In the lowland area, all habitats had clay loam soil. Soil nutrient levels were statistically comparable across habitats (Table 4.3).

4.5. Discussion

Knowledge on the reforestation species performance in a severely degraded site is important to guide the selection of species that will promote rapid forest establishment. Three out of the four examined tree species in this study are good candidates for restoring forest in severely degraded site. Therefore, native pioneer tree species can be used to restore severely degraded sites. Growth rates differed significantly within species, across species, microtopographic position, and habitats. Higher soil moisture coupled with higher nutrient levels in the lowland area promoted rapid tree establishment. Therefore, microtopographic position showed a greater influence on tree performance.

Table 4.3: Soil physical and chemical properties across three habitats (mean±SE). Different letters within columns indicate significant differences at $P < 0.05$.

Habitat	Microtopographic positions	Total N	P	K	Ca	Mg	pH	Sand	Silt	Clay	Soil type
		mg/kg						%			
0-year	Upland	1080 ± 380^{b}	7.12 ± 0.7^{b}	221.62 ± 34.46^{ab}	414.87 ± 30.00^{c}	344.62 ± 17.64^{c}	5.85 ± 0.08^{c}	44.66 ± 2.26^{a}	30.90 ± 0.80^{b}	27.46 ± 2.40^{b}	Loam
	Lowland	1830 ± 150^{a}	4.44 ± 0.62^{b}	155.88 ± 16.57^{abc}	1480.66 ± 101.87^{a}	901.11 ± 89.78^{a}	6.59 ± 0.01^{bc}	25.19 ± 2.57^{b}	36.13 ± 0.91^{ab}	40.31 ± 2.90^{a}	Clay loam
3-year	Upland	1180 ± 60^{ab}	25.00 ± 2.89^{a}	225.77 ± 17.86^{a}	371.33 ± 33.18^{c}	215.44 ± 18.21^{d}	5.39 ± 0.02^{d}	28.68 ± 1.36^{ab}	35.38 ± 0.95^{ab}	39.13 ± 0.95^{a}	Clay loam
	Lowland	1600 ± 210^{ab}	5.88 ± 1.19^{b}	118.00 ± 22.53^{c}	1517.66 ± 85.32^{a}	775.55 ± 81.09^{a}	6.82 ± 0.15^{a}	19.59 ± 0.97^{c}	40.15 ± 1.40^{a}	40.67 ± 1.69^{a}	Clay loam
5-year	Upland	1800 ± 220^{a}	6.87 ± 1.23^{b}	145.50 ± 25.32^{abc}	999.12 ± 66.44^{b}	538.87 ± 21.30^{b}	6.18 ± 0.12^{cd}	25.4 ± 1.44^{b}	29.58 ± 2.25^{b}	43.40 ± 2.67^{a}	Clay
	Lowland	1670 ± 180^{a}	5.33 ± 0.66^{b}	121.44 ± 16.89^{bc}	1430.33 ± 54.27^{a}	980.11 ± 26.87^{a}	6.70 ± 0.08^{a}	30.45 ± 6.42^{ab}	30.55 ± 2.37^{b}	39.30 ± 4.41^{a}	Clay loam

The use of native pioneer tree species, characterized by a rapid increase in height and canopy cover, is the most favourable strategy in reforestation initiatives (Blakesley et al. 2002; Elliot et al. 2003; Goosem and Tucker 2013; Roy 2015). This is because, fast growing, pioneer species are more adapted to grow in highly disturbed sites with harsh environmental conditions. Pioneer species are able to grow faster in the open harsh conditions because of their greater photosynthetic plasticity and a more improved growth response to irradiance (Strauss-Debenedetti and Bazzaz 1991; Baker et al. 2003). As a result, they create favourable understorey microclimatic conditions and provide attractive habitat for wildlife (Lugo 1997). Frugivorous birds and mammals that are attracted by the habitat provided by pioneer species disperse seeds and the favourable understorey microclimatic conditions promote seed germination and seedling establishment (Goosem and Tucker 2013). Furthermore, a rapid increase in tree height and canopy cover is critical to elevate tree canopy above weeds and invasive alien plants, shading them out, thus decreasing the likelihood of fire (Elliot et al. 2003). This trait is important in habitats that have been subjected to anthropogenic disturbance (e.g., agriculture) particularly in close proximity to cities are more vulnerable and highly likely to be invaded (e.g., Alston and Richardson 2006). In this study, *B. micrantha*, *E. lysistemon* and *V. natalitia* were found to be good pioneer reforestation species. However, *B. micrantha* performed well in the lowland area while *E. lysistemon* and *V. natalitia* performed well in both the upland and lowland areas. These species were tall with wide and dense canopies enabling rapid canopy closure with their neighbours. Tudor-Owen and Wyatt (1991), and Geldenhuys and Delvaux (2002) recommended the use of *M. grandis* in forest restoration. However, in this study, *M. grandis* performed well in terms of root-collar diameter and height, but performed poorly in terms of canopy width, which is one of the critical parameters of the reforestation species (e.g., Elliott et al. 2003). Therefore, *M. grandis* should be planted to enhance species diversity, canopy structure and niches for wildlife (see Elliott et al. 2003).

Species growth patterns may change in their life cycle as a result life history strategies, abiotic (e.g., microtopographic position), and biotic factors (e.g., stand competition) (Baker et al. 2003; Chi et al. 2015; Fernández-de-Una et al. 2015). Riedel et al. (2013) documented a change in species growth patterns and their associated life-history traits with stand development and resource use. In the reforestation of abandoned pasture in Central Panama, Riedel et al. (2013) found that *Tabebuia rosea* DC. (Bignoniaceae) was the tallest species, two years after planting; however, after three to five years, *Anacardium excelsum* L.

(Anacardiaceae) was the tallest species. They emphasized that reliable conclusions about species performance and their reforestation suitability can be drawn after allowing an adequate period of tree establishment to account for changes in growth patterns. In this study, tree growth rates recorded in the 0-year-old habitat showed that all species start-off with comparable RCD, height and CW irrespective of the microtopographic position. Furthermore, in the 3-year-old habitat, in the upland area, *B. micrantha* performed better than *V. natalitia* across all the growth parameters, but the opposite occurred in the 5-year-old habitat. A higher growth rate exhibited by *E. lysistemon* and *V. natalitia* could probably be explained by the following factors. A higher growth rate in deciduous trees compared to evergreens is possibly caused by their higher specific leaf area (e.g., Chi et al. 2015). Furthermore, species with a large canopy cover grow faster than species with a small canopy cover (Baker et al. 2003; Poorter et al. 2005).

An annual tree growth pattern is determined by the variation in soil moisture availability (Omary 2011; Reidel et al. 2013; Fernández-de-Una et al. 2015). In its natural habitat, *B. micrantha* occurs in wet habitats (e.g., swamps and riparian areas), and it is susceptible to drought (Schmidt and Mwaura 2010). In the 0-year-old habitat, it was the tallest species in both dry and lowland areas. However, in the 3- and 5-year-old habitats, it was the tallest species in the lowland area only. Furthermore, all the growth parameters in the lowland area were significantly different to the upland area in the 3- and 5-year-old habitats. Rainfall has a positive effect on tree growth, because of enhanced soil moisture availability (Omary 2011; Fernández-de-Una et al. 2015). A low rainfall in the year 2010 probably led to a poor establishment of *B. micrantha* after planting in the 5-year-old habitat. A good rainfall in the year 2011 to 2013 probably led to a rapid establishment of *B. micrantha* in the 3-year-old habitat lowland area, leading to a statistically comparable height and canopy width between the 3- and 5-year-old habitats. Therefore, it can be concluded that *B. micrantha*'s growth performance is mainly driven by water availability, because it only had a higher growth rate in the lowland area.

Some studies have documented a strong correlation between microtopography position and environmental variables such as soil texture, soil depth, soil moisture and nutrient levels (Grell et al. 2005; Simmons et al. 2012; Li et al. 2014). Soils in the upland areas are often shallower, limiting root space (Li et al. 2014), and coarse as a result of lower clay content (Simmons et al. 2012), while soils in the bottom lands have high silt and clay content (Grell et al. 2005). Similar findings were reported by Grell et al. (2005) and Simmons et al. (2012).

Clay loam soil in the lowland area has a positive influence on the tree growth (Henri 2001). After rains, lowland areas remain inundated for a longer period of time compared upland areas. Upland areas are often drier due to higher surface runoff, higher interflow rates and shallow soil. As a result, tree growth may be lower compared to lowland areas (Li et al. 2014). In our study, the soil in the upland area was shallow (20-40 cm) and dry while the soil in the lowland area was deeper and wet (60-110 cm) (EThekwini Municipality 2014) thus leading to higher tree growth rate. As a result, species planted in the lowland area exhibited a higher growth rate compared to species planted in the upland area. Similar findings were reported by Henri (2001) in a dry tropical region of western Venezuela.

During sugarcane farming at Bufflesdraai Landfill Site, there was fertilizer application to increase soil productivity (Winn R, pers. comm. 2014). Habitats under reforestation at Buffelsdraai Landfill Site had a lower pH (5.39 to 6.82) compared to the pH of native forest (7.5) in the region (van Aarde et al. 1998). This is largely attributed to continuous application of fertilizer during sugarcane farming. Other possible explanations include inundated conditions in the lowland area (Loeb et al. 2008) and low organic matter content in both the upland and low areas. Soils with lower organic matter content generally have a pH of less than 6.5 (Vepraskas and Faulkner 2016). In our study, lowland areas had a higher litter accumulation compared to the upland areas (Mugwedi et al. 2017).

On former agricultural land, soil nutrient levels are usually higher than in forest as a result of fertilizer addition to improve soil productivity (Cunningham et al. 2015). In our study, only magnesium and potassium levels in habitats under reforestation were higher than those reported in the native forest in the region (Van Aarde et al. 1998). Only the upland area of the 3-year-old habitat had phosphrus levels which were higher than those reported by Van Aarde et al. (1998). Calcium level in the soil can decrease (>20%) following reforestation due to translocation to plant biomass (Berthrong et al. 2009). However, this does not seem to be the case in our study, because the upland area of younger habitats (0- and 3-year-habitats) with smaller trees had lower calcium level compared to the 5-year-old habitat with bigger trees. There is also a strong correlation between soil organic matter content and soil nitrogen levels (Burke et al. 1999; Grell et al. 2005). Low levels of nitrogen and phosphorus in habitats under restoration could suggest that nitrogen and phosphorus mineralization rates may be low (van Aarde et al. 1998; Chen et al. 2009) probably as a result of low organic matter content in these habitats (Mugwedi et al. 2017). Van Aarde et al. (1998) reported an increase in soil nutrient levels with an increase in reforestation age, but our results did not show this trend.

However, it is expected that in increase in soil nutrient levels with an increase in reforestation age will be evident when the habitats are 16 years and older (e.g., van Aarde et al. 1998).

Soil nitrogen and phosphorus availability, both individually and in combination, affect plant productivity (Vitousek et al. 2010). In nitrogen deficient soils, nitrogen-fixing trees can fix atmospheric ntirgogen, thus leading to higher growth rate than non-nitogen-fixing trees (van Aarde et al. 1998; Wang et al. 2010; Hoogmoed et al. 2014a). In our study, nitrogen fixing species *E. lysistemon* and *V. natalitia* showed higher growth rates under nitrogen deficient soil, suggesting that these species fixed atmospheric nitrogen to enhance their growth. However, the growth rate of a nitrogen-fixing *M. grandis* was slower compared to *E. lysistemon* and *V. natalitia*. Nitrogen-fixing trees can benefit non-nitrogen-fixing trees through tight root connections or organic forms of nitrogen from the litter layer (Siddique et al. 2008; Hoogmoed et al. 2014b). Therefore, it is likely that higher domincance of nitrogen-fixing tree species at Buffelsdraai Landfill Site will benefit non-nitrogen-fixing tree sepcies such as *B. micrantha*. In our study, highest phosphorus levels were found in the upland area of the 3-year-old habitat, but this did not have a significacnt effect on tree growth, because highest growth rates occurred in the lowland area. In the tropical Andes of Ecuador and Colombia, Bare and Ashton (2015) found that magnesium had a positive influence on tree growth while calcium had a negative influence. However, in our study, higher levels of calcium and magnesium in the lowland area had positive influence on tree growth.

4.5. Conclusion

Microtopographic position had a significant influence on soil physical and chemical properties, which in turn influenced species growth rate. We emphasize the need to understand species growth preferences, the microtopographic position, and soil physical and chemical properties within the site, in order to enhance rapid forest establishment. This approach could promote rapid reforestation success, reduce reforestation costs, and enhance biodiversity, climate change mitigation and adaptation, and provision of ecosystem services. For example, the fast growing species such as *B. micrantha*, *E. lysistemon* and *V. natalitia* could play a huge role in climate change mitigation by storing more carbon within a short period of time, with the effect accentuated by selecting the correct species for the correct habitat. We recommend that sites under reforestation should be used to identify faster growing species that could be used in the future reforestation projects. Although the process of identifying more, faster growing species is labour intensive and time consuming, it is worth undertaking in order to save the limited conservation resources that are lost as a result of tree mortality and slow growth, necessitating replanting and long-term site maintenance (e.g., weed control and fire suppression).

4.6. References

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4.7. Supplementary materials

Table S4.1: The six most abundant tree species across three reforestation habitats at Buffledraai Landfill Site.

Restoration habitat	Microtopographic position	Tree species
0-year-old	Upland	*Vachellia natalitia* *Erythrina lysistemon* *Bridelia micrantha* *Syzygium cordatum* *Millettia grandis* *Trichilia dregeana*
	Lowland	*Tabernaemontana ventricosa* *Bridelia micrantha* *Syzygium cordatum* *Millettia grandis* *Erythrina lysistemon* *Vachellia natalitia*
3-year-old	Upland	*Erythrina lysistemon* *Millettia grandis* *Bridelia micrantha* *Protorhus longifolia* *Vachellia natalitia* *Syzygium cordatum*
	Lowland	*Senegalia caffra* *Vachellia natalitia* *Bridelia micrantha* *Erythrina lysistemon* *Millettia grandis* *Syzygium cordatum*
5-year-old	Upland	*Vachellia natalitia* *Erythrina lysistemon* *Bridelia micrantha* *Millettia grandis* *Syzygium cordatum* *Senegalia caffra*
	Lowland	*Vachellia natalitia* *Senegalia caffra* *Erythrina lysistemon* *Bridelia micrantha* *Millettia grandis* *Brachylaena discolor*

The species are arranged in the order of abundance, the first species being the most abundant and last species being the least abundant.

CHAPTER 5: AN ASSESSMENT OF A COMMUNITY-BASED, FOREST RESTORATION PROGRAMME IN DURBAN (ETHEKWINI), SOUTH AFRICA

This chapter is based on

LF Mugwedi, M Rouget, B Egoh, Sershen, S Ramdhani, R Slotow, JL Rentería (**2017**) An assessment of a community-based, forest restoration programme in Durban (eThekwini), South Africa. Forests 8: 255, doi:10.3390/f8080255

Abstract

The restoration of degraded forests to enhance biodiversity, ecosystem services, as well as climate change mitigation and adaptation is now a major priority in cities around the world. This study evaluated the success of the Buffelsdraai Landfill Site Community Reforestation Project in Durban, South Africa, by assessing ecological attributes. Measures of plant richness, diversity, vegetation structure, invasive alien plants (IAPs) and ecological processes were contrasted across a chronosequence of habitats under restoration (0-year-old, 3-year-old and 5-year-old), and compared with a reference forest habitat (remnant natural forest). Native tree species recruitment and vegetation structure increased with restoration age. Ecological processes, represented by the composition of pollination and seed dispersal traits in all the habitats under restoration, were similar to the reference habitat. However, low tree density and an increase in IAP cover with an increase in restoration age were identified as threats to reforestation success. We recommend enrichment planting and an effective IAP management strategy to promote more rapid habitat restoration while reducing site maintenance costs. Enrichment planting should not only focus on increasing tree species density and richness, but also on the inclusion of species with missing pollination and seed dispersal categories.

Keywords: Ecosystem processes; Invasive alien plants; Pollination; Restoration success; Seed dispersal; Species diversity; Vegetation structure

5.1. Introduction

Continuous environmental degradation, presently occurring at alarming rates around the world, has motivated restoration efforts that aim to enhance biodiversity and ecosystem functioning, and to ensure continued provision of ecosystem services (Aronson et al. 2006; Clewell and Aronson 2007). Furthermore, owing to a rapid expansion of urban populations and the threats of climate change, green infrastructure investment has become a necessity in large and developing cities across the globe (Oldfield et al. 2015; Simmons et al. 2016). In the quest of creating sustainable and resilient cities, governments are investing in the restoration of natural capital to improve human well-being (Oldfield et al. 2015).

An important objective of restoration initiatives is to create an ecosystem that is self-sustaining and resilient to disturbance (SER 2004). This can be achieved by creating an ecosystem that is closer or more similar to the former natural habitat (commonly labelled 'reference habitat') in terms of plant diversity, plant traits, and functional group diversity (Suding et al. 2004; van Andel and Aronson 2012). Self-sustaining and resilient forest ecosystems are characterized by species rich and multi-layered vegetation structure with key ecological processes such as litter accumulation, pollination and seed dispersal (Dorren et al. 2004; Norden et al. 2009). Assessment of vegetation structure provides information on habitat suitability for fauna such as insects, birds, reptiles and small mammals, and ecosystem productivity (Dorren et al. 2004; Suganuma and Durigan 2015). Multi-layered vegetation structure protects the forest from natural hazards (Dorren et al. 2004) such as IAP invasion (te Beeste et al. 2012).

Unfortunately, due to insufficient monitoring, the actual number of tree species and their density in most restoration projects is unknown (Woodford 2000; Kanowski and Catterall 2007). During the establishment phase, planted trees may also die, native tree recruitment may fail and sites can be invaded by alien plants and weeds, indicating the need for post-planting monitoring and management. Assessing reforestation success and challenges would provide beneficial insight which could guide the necessary management interventions and inform best practice in the future (Kanowski et al. 2010; Derhé et al. 2016). However, most restoration assessments often focus on species establishment (usually of indicator species), with at least one aspect of ecological processes necessary for long-term persistence of the ecosystem (Ruiz-Jaén and Aide 2006; Wortley et al. 2013). Furthermore, numerous studies have shown that the inclusion of a variety of indicators such as vegetation structure, species diversity and ecological process in assessing restoration success is imperative for comprehensive assessment of restoration outcomes (e.g., Ruiz-Jaén and Aide 2006; Monie et al. 2013).

An understanding of key ecological processes is essential to maximize the efficiency of restoration processes and managed restored systems (e.g., Menz et al. 2011; Abiyu et al. 2016). In this regard, plant litter is considered to be a valuable indicator to measure as it is a key factor in structuring many plant communities (e.g., seedling recruitment) due to its ability to modify micro-environmental conditions (Kostel-Hughes et al. 1998). Litter that accumulates on the forest floor forms an essential part of nutrient cycling (Barrientos 2012; Rubiano-Cardona et al. 2013) which is necessary for the long-term stability of an ecosystem (Herrick et al. 2006). Pollination, fruit production, seed dispersal and seedling establishment are also key ecological processes ensuring the long-term stability of the forest ecosystem (Dixon 2009; Ruxton and Schaefer 2012). Pollination has received little attention in restoration studies (Menz et al. 2011; Wortley et al. 2013). Pollination and seed dispersal are dependent on forest fauna such as insects, birds and mammals (Neuschulz et al. 2016). In most forest systems, recovery is limited by poor seed dispersal (Holl et al. 2000; White et al. 2004) and consequently, seed dispersal is often assessed by measuring seedling density and diversity (e.g., White et al. 2004 ; Aide et al. 2006).

Restoration success can be hampered by IAPs due to their competitive growth and reproductive strategies (Reed 2004; Padmanaba and Corlett 2014). Challenges posed by IAPs during habitat restoration are encountered particularly when species invade and establish large populations (Padmanaba and Corlett 2014). High IAP cover can compromise native seedling establishment (Simmons et al. 2016). Understanding IAPs distribution and expansion following forest restoration is the initial step towards the development of an effective control strategy and a determinant of when and where the control strategy should be implemented (Hernández et al. 2014).

The Buffelsdraai Landfill Site, situated within the eThekwini Municipal Area[1] (EMA), north of Durban in KwaZulu-Natal (KZN), South Africa was formerly under dryland sugarcane cultivation for over a century and is now under restoration (Douwes et al. 2015a). The restoration is a climate change driven project, which aims to offset the carbon footprint associated with the city's hosting of the 2010 FIFA World Cup, enhance biodiversity and ecosystem services, and provide socioe-conomic benefits to local communities (Douwes et al.

[1] The municipality manages the metropolitan region including the city of Durban.

2015a). The planting commenced in 2009 and since it is still in the establishment phase there is a need to assess its success in terms of ecosystem functioning, and to recommend the necessary management interventions to levels of functionality. Using a chronosequence (0-year-old, 3-year-old and 5-year-old) of land under reforestation, we assessed early progress of forest restoration in terms of tree species diversity, vegetation structure and ecological processes (e.g., pollination) needed for creation of a self-sustaining, functional forest ecosystem that is resilient to disturbance, and the presence of IAPs. The indicators assessed were compared to the reference (natural forest remnants) habitat in order to gauge restoration success. Additionally, various recommendations are made for future restoration planning within the EMA and other forest restoration projects elsewhere in the world.

5.2. Methods

5.2.1. Study area

The study was conducted at Buffelsdraai Landfill Site (29.62961S; 30.980392E), the largest regional solid waste landfill in KZN, South Africa. The active landfill site is 116 ha and has a buffer zone of 757 ha of which 580 ha (former low productive dryland sugarcane field) has been planted with over 51 native tree species. An average of 1000 (in the dry habitat) and 2000 (in the wet habitat) tree saplings per hectare were planted in the wet season, commencing from 2009-2010 (November to February) to 2014-2015 at a rate of about 100 ha per year (Douwes et al. 2015a). Species were planted randomly, but there is a lack of record on the number of tree species planted in the 3- and 5-year-old habitats, except for the 0-year-old habitat that we assessed one-month post planting. Furthermore, dead saplings replacement was done some months after planting in the 3- and 5-year-old habitat (Douwes et al. 2015a). Therefore, species richness and diversity results from these two habitat should be interpreted with caution. The remainder of the buffer zone is characterized by mosaic patches of native forest, woodlands and grasslands, with almost all the sites being invaded by alien plants (MacFarlane et al. 2011). The vegetation, broadly classified as KwaZulu-Natal Coastal Belt (grassland and subtropical forest), is highly transformed and fragmented with little formal protection. The vegetation type as a whole is classified as endangered (Mucina and Rutherford 2006). The remnants of native forest form part of Eastern Scarp Forest, usually located at an altitude of 100 to 1000 m (von Maltitz et al. 2003), and is described as a refuge forest that survived the last glacial maximum (≈18000 BP) (Eely et al. 1999). Approximately 15-31 % of Eastern Scarp Forest has been lost due to land use change (e.g., clearing for agriculture) and non-sustainable harvesting of forest products by rural communities, but it should be noted that this estimate is quite dated (von Maltitz et al. 2003). This has resulted in fragmentation between patches and an increased edge ecotone (Kotze and Lawes 2007). The remnants of natural native forest around the Buffelsdraai Landfill Site were used as a reference habitat, a requirement in reforestation assessment studies of this nature (Ruiz-Jaén and Aide 2006; Wortley et al. 2013).

Prior to the start of this reforestation project, tree species composition of the surrounding forest patches (reference habitat, 105.8 ha) was assessed to guide the selection of species to be planted in restoration site based on species importance value (after MacFarlane et al. 2011). This restoration initiative is a community-based forest restoration project, named the 'Buffelsdraai Landfill Site Community Reforestation Project'. Tree saplings for planting were supplied by Tree-preneurs (local community members who grow seedlin) within the Buffelsdraai, Osindisweni and KwaMashu local communities (peri-urban areas) who source the seeds from the reference habitat and forest patches within a 50 km radius of the site (after MacFarlane et al. 2011). These communities are plagued by poverty and unemployment (Statistics South Africa 2011).

The topography of the study area is characterized by undulating slopes (200 - 325 m altitude). A glacial conglomerate parent material that is base-rich, hard and resistant to weathering, the Dwyka Tillite, is the dominant geology within the site (EThekwini Municipality 2014). The upland area is characterized by shallower (20-40 cm) lithosol soil. The soil is probably shallower as a result of higher soil erosion caused by runoff, exacerbated by cultivation. The lowland area is characterized by deeper (60-110 cm) acrisol soil. The deeper soil is probably as a result of the deposition of materials from the upper slopes (EThekwini Municipality 2014). The upland area is drier while lowland area is wetter (EThekwini Municipality 2014). As a result, upland and lowland areas were subjected to different planting densities, 1000 and 2000 trees per ha, respectively (Douwes et al. 2015a).

5.2.2. Data collection

Vegetation sampling was done within three habitats under restoration, planted in 2009-2010 (hereafter referred to as the '5-year-old habitat'), 2011-2012 (hereafter referred to as the '3-year-old habitat') and 2014-2015 (hereafter referred to as the '0-year-old habitat'), and in the surrounding natural forest patches (reference habitat). The surrounding natural forest patches were approximately 40, 300 and 250 m from the 0-, 3- and 5-year-old habitats, respectively.

5.2.2.1. Vegetation composition and structure

Microtopographic positions (upland and lowland areas) within a site affect hydrologic conditions (Omary 2011). As a result, within a species, individual tree growth rates may vary significantly (Omary 2011; Simmons et al. 2012). To minimize the confounding effects of soil moisture and planting densities, upland and lowland areas were selected within each habitat under restoration (0-, 3- and 5-year-old) based on a soil survey done on the entire Buffelsdraai Landfill Site (EThekwini Municipality 2014) and in the reference habitat, in the upper slopes and riparian area. Twelve plots of 400 m^2 (20 × 20 m) (six plots in both upland and lowland areas) were randomly established within each restoration and reference habitats for vegetation sampling. All planted individuals within the habitats under restoration were counted. In the reference habitat, all trees excluding saplings (individuals with less than 4 cm stem diameter at breast height) were counted. Across all habitats, percentage tree canopy cover (visually estimated by one person to avoid estimation bias) and tree height were measured. Tree species within each plot were identified, and assigned to species successional types (pioneer vs. understorey vs. climax) using published accounts (Boon 2010; van Wyk and van Wyk 2013). Percentage herbaceous layer cover (graminoids, forbs and herbaceous climbers) was visually estimated in four 4 m^2 subplots that were established within each plot. Subplots were located at each cardinal point of the plot, 4 m from the plot boundary. In this study, climbers were defined as plants that use other plants or objects to support themselves in growing towards sunlight or into positions that permit attraction to pollinators or seed dispersers (Boon 2010).

5.2.2.2. Invasive alien plants

Only woody IAPs were considered, since their vigorous growth poses a huge threat to restoration success (Norton 2009). Percentage IAP cover was visually estimated within the subplots. The IAPs present in the subplots were identified and categorised using the National Environmental Management: Biodiversity Act (NEMBA), 2004 (Act No 10 of 2004) Alien and Invasive Species List, 2016 (South Africa 2016), which includes three invasive alien categories, 1–3.

5.2.2.3. Ecological processes

Forest litter accumulation was assessed by measuring litter cover and depth (Barrientos 2012; Rubiano-Cardona et al. 2013). Within each subplot, litter percentage cover was visually estimated. Litter depth was measured to the nearest millimetre using a ruler.

Pollination and seed dispersal traits for each tree species sampled were determined on the basis of field observations, herbarium specimens and published descriptions (Boon 2010; van Wyk and van Wyk 2013). Species were assigned to either one or a combination of six pollination categories, namely, bee, beetle, bird, butterfly, fly or 'other' pollination category. The 'other' pollination category included species with unknown pollinators and one wind-pollinated species that was represented by one individual in the reference habitat. Additionally, species were assigned to either one or a combination of five seed dispersal categories, namely, ballistic, bird, mammal, wind or 'other' dispersal category (species with unknown dispersal agents) (Mayfield et al. 2006; Seidler and plotkin 2006) based on field observations, herbarium specimens and published data (Boon 2010; van Wyk and van Wyk 2013).

To understand the influence of restoration age on native tree recruitment, all woody species (hereafter referred to as seedlings) greater than 0.05 m in height were counted and identified within each subplot (4 m^2).

5.2.3. Data analyses

Both the upland and lowland areas data were analysed separately to avoid the confounding effect of microtopographic position (upland and lowland area) and the consequent different tree planting densities. All statistical computations were performed in R statistics (R Development Core Team 2014).

5.2.3.1. Species richness and diversity

Estimate*S* package (Colwell 2013) was used to construct sample-based species-accumulation curves. To achieve this, we pooled all the species data per habitat. Two non-parametric estimators for abundance data, abundance-based coverage estimator (ACE) and Chao1 were used to estimate species richness (Gotelli and Colwell 2001). The level of sampling completeness was computed by dividing the number of species found by the projected number of species using ACE and Chao1 (Chacoff et al. 2012).

Simpson's evenness index ($E_{1/D}$) and Simpson's diversity index (*D)* were used as measures of tree diversity. Simpson diversity was computed in Estimate*S* package (Colwell 2013) using species abundance data (Magurran 2004). Simpson's evenness was computed as [$E_{1/D} = (1/D)/S$], where *D* represents Simpson's diversity index and *S* represents the number of species (Magurran 2004). Species richness (observed), evenness and diversity indices were compared across habitats using generalized linear models (GLMs), gaussian family (MASS package, Venables and Ripley 2002), followed by Tukey post-hoc test to separate habitats with significant differences at $P < 0.05$ (multcomp package, Hothorn et al. 2008).

5.2.3.2. Species composition and abundance

To assess whether restoration habitats are progressing towards or diverging from the reference habitat in terms of species composition and abundance, a non-metric multidimensional scaling (NMDS) ordination analysis based on Bray-Curtis index was used (Vegan: Community Ecology package, Oksanen et al. 2016). Two analyses of species composition were performed. The first analysis assessed species composition in the upland and lowland area of each habitat, separately. In the second analysis, both the upland and lowland areas within each habitat were combined in order to assess species composition per habitat. Patterns of species similarity shown in the NMDS were confirmed using analysis of similarity (ANOSIM) (Vegan: Community Ecology package, Oksanen et al. 2016).

5.2.3.3. Vegetation structure and litter accumulation

GLMs were applied using gaussian family (MASS package, Venables and Ripley 2002) to compare tree stem density, tree height, and litter depth across habitats. GLMs were also applied using logit function, quasibinomial family (MASS package, Venables and Ripley 2002) to compare percentage tree canopy cover, herbaceous layer cover, IAP cover and litter layer cover, and species successional type across habitats. Species successional type data were calculated in terms of proportional values per plot, i.e. the number of individuals within a successional type divided by the total number of individuals within the plot. Statistical analyses were followed by a Tukey post-hoc test to separate habitats with significant differences at $P < 0.05$ (multcomp package, Hothorn et al. 2008).

5.2.3.4. Pollination and seed dispersal

Community-weighted mean trait values for pollination and seed dispersal traits were computed (Cohen et al. 2014; Muscarella and Uriarte 2016). Traits were computed in terms of proportion per plot using species abundance. Species abundance data were used, as it shows the pollination/seed dispersal category state similarity of individuals, independent of species identity, within plots and habitats (Mayfield et al. 2006). We first compared each pollination and seed dispersal category across habitats using the GLMs, logit function, quasibinomial family (MASS package, Venables and Ripley 2002), followed by a Tukey post-hoc test (multcomp package, Hothorn et al. 2008). We then assessed the pollination and seed dispersal categories composition across habitats using the NMDS, followed by the

ANOSIM, to separate habitats with significant differences at $P < 0.05$ (Vegan: Community Ecology package, Oksanen et al. 2016).

5.2.3.5. Plant regeneration

Seedling recruitment levels were not compared across habitats statistically since only a few species with low densities recruited in the restored habitats. Only the mean density per subplot (4 m^2) of recruited species across the habitats was computed.

5.3. Results

5.3.1. Plant species richness and diversity

Species accumulation curves based on ACE and Chao1 showed that increased sampling would have revealed more species (Figure S5.1 in Supplementary Materials). Sampling completeness was adequate in all cases, with values ranging from 75.6% to 82.8% in the 0-year-old habitat, 93.7% to 95.3% in the 3-year-old habitat, 88.9% to 91.6% in the 5-year-old habitat, and 87.2% to 87.5% in the reference habitat. Tree species richness differed across habitats. In the upland area, 29, 22, 29 and 48 tree species were recorded in the 0-, 3-, 5-year-old and the reference habitats, respectively. In the lowland area, 18, 31, 36 and 40 tree species were recorded in the 0-, 3-, 5-year-old and the reference habitats, respectively (see Table S5.1). When data for all the species were pooled per habitat, 36, 40, 44 and 70 species were recorded in the 0-, 3-, 5-year-old and reference habitats, respectively. The shared species among restored habitats, and between restored and reference habitats ranged from 24 to 40 tree species. Overall, 59 tree species were recorded in habitats under restoration, of which 49 species were shared with the reference habitat (Table S5.1).

Species richness, Simpson's evenness, and Simpson's diversity varied significantly across habitats (Table 5.1). In the upland area, the reference habitat had the highest species richness, significantly ($\chi^2 = 9.36$; $P < 0.05$) higher than the 3-year-old habitat; however, richness among the habitats under restoration was statistically comparable. Species evenness was highest in the 0-year-old habitat, significantly ($\chi^2 = 8.97$; $P < 0.05$) higher than the 5-year-old habitat. Species were most diverse in the reference habitat, being significantly ($\chi^2 = 9.95$; $P < 0.05$) greater than the 3- and 5-year-old habitats.

Table 5.1: Measures of species richness, diversity and species successional type proportion across habitats (mean±SD). Different letters indicate significant difference across habitats at $P < 0.05$.

Upland area						
Habitat	Diversity indices			Tree successional type		
	Species richness	Simpson's evenness	Simpson's diversity	Pioneer	Understorey	Climax
0-year-old	11.3±1.86ab	0.87±0.03a	2.9±0.47ab	0.59±0.05a	0.17±0.05ab	0.27±0.03a
3-year-old	8.6±3.55b	0.83±0.04ab	1.9±0.85b	0.54±0.20a	0.05±0.06a	0.40±0.22ab
5-year-old	10.1±4.16ab	0.78±0.07b	2.2±0.85b	0.51±0.12a	0.15±0.09ab	0.32±0.10ab
Reference	14.1±2.85a	0.84±0.06ab	3.7±0.56a	0.26±0.11b	0.23±0.13b	0.50±0.09b

Lowland area						
Habitat	Species richness	Simpson's evenness	Simpson's diversity	Pioneer	Understorey	Climax
0-year-old	6.0±0.89a	0.90±0.02a	1.4±0.26a	0.53±0.08a	0.18±0.02a	0.27±0.10a
3-year-old	12.6±2.58b	0.80±0.06b	2.8±0.57b	0.63±0.09a	0.11±0.07a	0.25±0.08a
5-year-old	11.3±3.26b	0.89±0.04a	2.7±0.72b	0.57±0.10a	0.16±0.07a	0.25±0.04a
Reference	11.16±3.18b	0.87±0.05ab	2.8±0.60b	0.59±0.33a	0.07±0.04a	0.32±0.27a

In the lowland area, the 0-year-old habitat had the lowest species richness, significantly (χ^2 = 21.66; $P < 0.05$) lower than all the other habitats. The 3-year-old habitat had the lowest species evenness, significantly (χ^2 = 12.87; $P < 0.01$) lower than the 0- and 5-year-old habitats. Species diversity showed a similar trend to species richness (χ^2 = 25.55; $P < 0.01$).

5.3.2. Species abundance and composition

In the upland area, the NMDS ordination showed that habitats under restoration clustered together (Figure 5.1a), and this was confirmed by the analysis of similarity (ANOSIM) which showed that habitats under restoration were more similar in terms of species composition and abundance. The reference habitat was significantly (F = 6.241; $P < 0.01$) different from the habitats under restoration. In the lowland area, the NMDS ordination showed separation of habitats with low similarity (Figure 5.1b). The ANOSIM confirmed that habitats were significantly different (F = 21.080, $P < 0.01$) except the 5-year-old and reference habitats. For overall species composition per habitat, the NMDS showed variation across the habitats (see Figure S5.2 in Supplementary Materials), except between the 0- and 3-year-old habitats. The ANOSIM showed that the 0- and 3-year-old habitats were similar, and significantly different (F = 21.740; $P < 0.05$) from 5-year-old and the reference habitats which were themselves similar.

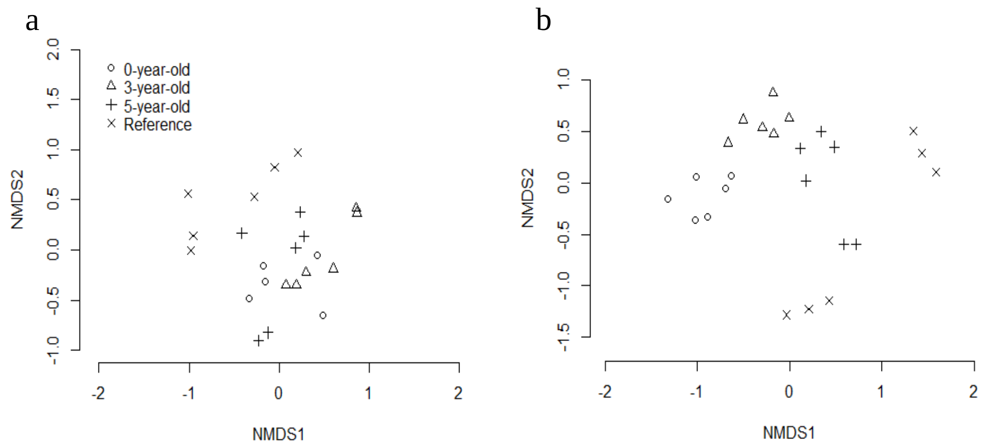

Figure 5.1: Non-metric Multidimensional Scaling (NMDS) ordination plot showing tree species composition in the (a) upland and (b) lowland areas. The NMDS was based on Bray-Curtis similarity index computed using species abundance data.

5.3.3. Vegetation structure

Species successional type proportion in the upland area varied significantly across habitats while no significant variation occurred in the lowland area (Table 5.1). In the upland area, pioneer species were most abundant in habitats under restoration, significantly (χ^2 = 20.44; P < 0.01) greater than the reference habitat. The reference habitat had the highest climax and understorey species richness significantly (χ^2 = 9.76; P < 0.05 and χ^2 = 11.62; P < 0.05, respectively) greater than the 0- and 3-year-old habitats, respectively.

Tree density, height and canopy cover, and herbaceous layer cover varied significantly across habitats (Table 5.2). In the upland area, tree density was statistically comparable across habitats. Unsurprisingly, the reference habitat had the tallest trees, significantly (χ^2 = 104.7; P < 0.01) taller than the other habitats. Tree height in the 3- and 5-year-old habitats was statistically comparable. Tree canopy cover varied significantly (χ^2 = 43.72; P < 0.01) across habitats. The reference habitat had the highest cover, followed by the 5-, 3- and then 0-year-old habitats. Herbaceous layer cover was statistically comparable across habitats. In the lowland area, the 3-year-old habitat had the highest tree density, significantly (χ^2 = 12.76; P < 0.01) higher than all the other habitats. Tree density per ha in the 0-year-old habitat was much lower (mean: 783 trees/ha) than the expected average density of 2000 trees per ha for lowland area. Tree height and canopy cover showed a similar trend to the upland area (χ^2 = 112.70; P < 0.01 and χ^2 = 206.36; P < 0.01, respectively). The reference habitat had the lowest herbaceous layer cover, significantly (χ^2 = 18.48; P < 0.01) lower than the 0- and 3-year-old habitats. Herbaceous layer cover was not statistically different across the habitats under restoration.

Table 5.2: Measures of vegetation structure and litter accumulation (mean±SD). TD – Tree density, TH – Tree height, TCC – Tree canopy cover, HLC – herbaceous layer cover, IAPsC – Invasive alien plants cover, LLC – Litter layer cover, LD – Litter depth. Different letters indicate significant difference across habitats at $P < 0.05$.

Upland area							
Habitat	Vegetation structure					Litter accumulation	
	TD (#)	TH (m)	TCC (%)	HLC (%)	IAPsC (%)	LLC (%)	LD (cm)
0-year-old	34.1±7.02a	0.52±0.06a	5.0±0.00a	60.8±8.7a	5.2±3.68a	19.5±13.82a	0.35±0.22b
3-year-old	51.8±8.79a	1.5±0.41b	20.8±9.17b	68.7±21.73a	14.4±3.29a	42.08±30.52b	0.49±0.36b
5-year-old	53.8±25.34a	2.0±0.47b	45.04.47c	70.8±25.8a	40.7±21.97b	40.83±32.97b	0.47±0.21b
Reference	34.3±12.59a	12.8±4.75c	79.1±11.58d	60.3±11.96a	8.1±5.74a	93.75±7.02c	1.14±0.32a
Lowland area							
Habitat	TD	TH	TCC (%)	HLC (%)	IAPsC (%)	LLC (%)	LD (cm)
0-year-old	31.3±7.08a	0.38±0.06a	5.0±0.00a	78.7±10.62a	3.8±0.95a	62.5±37.40a	0.72±0.56ab
3-year-old	58.1±17.73b	1.9±0.28b	27.5±5.24b	78.9±15.27a	20.0±18.6ab	68.9±27.00a	0.84±0.29ab
5-year-old	43.6±10.25a	2.1±0.23b	50.0±4.47c	63.7±19.03ab	33.7±25.46b	68.5±21.58a	0.52±0.19b
Reference	37.1±16.79a	9.8±3.37c	88.3±9.3d	45.0±11.58b	2.5±2.23a	98.3±2.17b	1.22±0.64a

5.3.4. Invasive alien plants

Five aggressive/widely spread woody IAPs in South Africa were recorded (See Table S5.3 in Supplementary Materials), with *Chromolaena odorata* (L.) King and Robinson (Asteraceae) being the most common invader. The IAP cover increased with restoration age in both the upland and lowland areas (Table 5.2). In the upland area, IAP cover in the 5-year-old habitat was significantly ($\chi^2 = 43.72$; $P < 0.01$) greater than the other habitats. In the lowland area, IAP cover in the 5-year-old habitat was significantly ($\chi^2 = 17.18$; $P < 0.01$) greater than in the 0-year-old and the reference habitat.

5.3.5. Ecological processes

Litter accumulation varied significantly across habitats (Table 5.2). In the upland area, the reference habitat had the highest litter layer cover, significantly ($\chi^2 = 31.77$; $P < 0.01$) higher than all the habitats under restoration. Among the habitats under restoration, the 0-year-old habitat had the lowest litter cover, significantly ($\chi^2 = 31.77$; $P < 0.01$) lower than the 3- and 5-year-old habitats. The reference habitat also had the deepest litter, significantly ($\chi^2 = 27.17$; $P < 0.01$) deeper than the habitats under restoration; however, litter depth was statistically comparable among the habitats under restoration. In the lowland area, the reference habitat had the highest litter layer cover, significantly ($\chi^2 = 11.32$; $P < 0.05$) higher than the habitats under restoration; however, litter layer cover among the habitats under restoration was statistically comparable. The reference habitat had the deepest litter, but was only significantly ($\chi^2 = 10.26$; $P < 0.05$) deeper than the 5-year-old habitat. Litter depth among the habitats under restoration was statistically comparable.

In the upland area, pollination categories were statistically comparable among the habitats under restoration; however, significant differences occurred between the habitats under restoration and the reference habitat, in some categories (Table 5.3). Bee-pollinated species were most abundant in the reference habitat, significantly ($\chi^2 = 13.35$; $P < 0.01$) higher than the 5-year-old habitat. Bird-pollinated species were most abundant in the 0- and 5-year-old habitats, significantly ($\chi^2 = 11.36$; $P < 0.01$) higher than the reference habitat. Fly-pollinated species were most abundant in the 5-year-old habitat, significantly ($\chi^2 = 10.13$; $P < 0.05$) higher than the reference habitat. The NMDS ordination for pollination trait composition showed that the 0- and 3-year-old habitats were more close to the reference habitat in the ordination space, but the 3-year-old habitat was also close to the 5-year-old habitat (Figure

5.2a). This was confirmed by the ANOSIM which showed that the 0-, 3-year-old and reference habitats were similar, and the 5-year-old habitat was only significantly ($F = 8.42$; $P < 0.01$) different from the 0-year-old and the reference habitats.

In the lowland area, some pollination categories varied significantly while some were statistically comparable across habitats (Table 5.3). Bee-pollinated species were most abundant in the 0-year-old habitat, but only significantly ($\chi^2 = 18.63$; $P < 0.01$) higher than the 3- and 5-year-old habitats. Bird-pollinated species were most abundant in the 5-year-old habitat, but only significantly ($\chi^2 = 44.64$, $P < 0.01$) higher than the 0-year-old and the reference habitats. Butterfly-pollinated species were most abundant in the 5-year-old and reference habitat, but only significantly ($\chi^2 = 20.78$; $P < 0.01$) higher than the 0-year-old habitat. Fly-pollinated species were most abundant in the 3- and 5-year-old habitats, but only significantly ($\chi^2 = 11.09$; $P < 0.05$) higher than the reference habitat. The 'other' pollination category was most abundant in the reference habitat, but only significantly ($\chi^2 = 9.24$; $P < 0.05$) higher than the 0-year-old habitat. The NMDS ordination showed that the 0-year-old habitat was close to the reference habitat, and the 3-year-old habitat close to the 5-year-old habitat (Figure 5.2b). The ANOSIM confirmed that the 0-year-old and the reference habitats were similar and significantly ($F = 9.34$; $P < 0.01$) different from the 3- and 5-year-old habitats, which were themselves similar.

Table 5.3: Species pollination and seed dispersal categories proportion across habitats (mean±SD). Different letters indicate significant difference across habitats at $P < 0.05$.

Upland area											
Habitat	Pollination categories						Seed dispersal categories				
	Bee	Beetle	Bird	Butterfly	Fly	‘Other’	Ballistic	Bird	Mammal	Wind	‘Other’
0-year-old	0.40±0.15ab	0.06±0.02a	0.29±0.06a	0.02±0.01a	0.35±0.08ab	0.30±0.11a	0.15±0.07a	0.40±0.20a	0.24±0.18ab	0.35±0.16a	0.13±0.02a
3-year-old	0.37±0.20ab	0.07±0.02a	0.20±0.15ab	0.02±0.01a	0.36±0.14ab	0.35±0.16a	0.12±0.10a	0.55±0.35a	0.35±0.07ab	0.28±0.06a	0.04±0.01a
5-year-old	0.26±0.02b	0.09±0.04a	0.29±0.10a	0.03±0.01a	0.54±0.14a	0.25±0.18a	0.20±0.04a	0.44±0.26a	0.17±0.13b	0.31±0.11a	0.08±0.03a
Reference	0.54±0.04a	0.07±0.02a	0.13±0.05b	0.06±0.01a	0.30±0.16b	0.27±0.12a	0.13±0.1a	0.55±0.18a	0.43±0.13a	0.25±0.14a	0.11±0.09a
Lowland area											
Habitat	Bee	Beetle	Bird	Butterfly	Fly	Other	Ballistic	Bird	Mammal	Wind	Other
0-year-old	0.58±0.14a	0.15±0.11a	0.04±0.01a	0.01±0.00a	0.33±0.14ab	0.13±0.05a	0.01±0.00a	0.66±0.17a	0.58±0.20a	0.17±0.13a	0.02±0.01a
3-year-old	0.39±0.11bc	0.08±0.01a	0.28±0.18b	0.04±0.03b	0.40±0.08b	0.24±0.11ab	0.16±0.11ab	0.59±0.07ab	0.24±0.11b	0.30±0.06a	0.08±0.05a
5-year-old	0.32±0.10c	0.15±0.03a	0.31±0.03b	0.06±0.04b	0.40±0.13b	0.27±0.08ab	0.24±0.15b	0.35±0.13b	0.24±0.18b	0.31±0.03a	0.09±0.06a
Reference	0.53±0.10ab	0.07±0.08a	0.04±0.01a	0.06±0.02b	0.22±0.04a	0.31±0.10b	0.28±0.10b	0.38±0.26b	0.28±0.18b	0.30±0.10a	0.03±0.01a

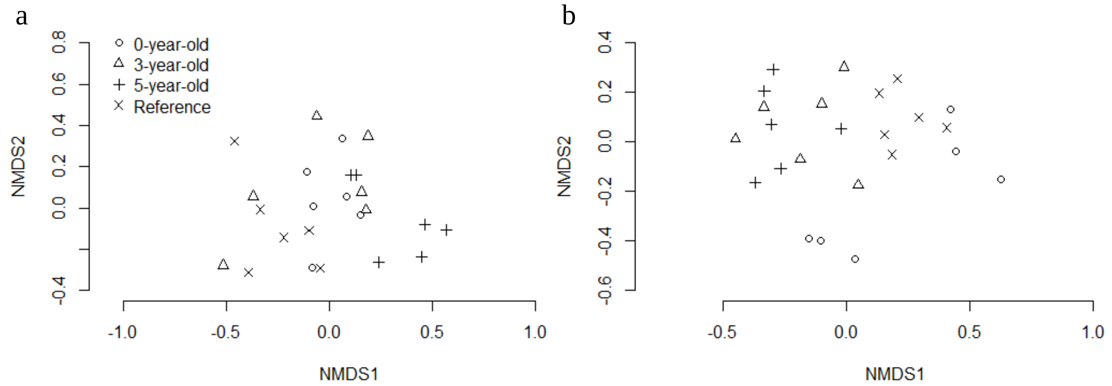

Figure 5.2: The NMDS ordination plot showing the species pollination trait in the (a) upland and (b) lowland areas. The NMDS was based on Bray-Curtis similarity index computed using species abundance data.

Seed dispersal categories were not statistically different across habitats in the upland area, except mammal-dispersed species. In the lowland area, only three categories differed significantly across habitats (Table 5.3). The 5-year-old habitat had the lowest abundance of mammal-dispersed species, but only significantly (χ^2 = 11.97; $P < 0.05$) different from the reference habitat. In the upland area, the NMDS ordination for seed dispersal trait composition showed that all the habitats were more close to each other (Figure 5.3a); the ANOSIM also showed that all the habitats were similar. In the lowland area, ballistic-dispersed species were most abundant in the reference habitat, but only significantly (χ^2 = 12.04, $P < 0.01$) higher than the 0-year-old habitat. Bird-dispersed species were most abundant in the 0-year-old habitat, but only significantly (χ^2 = 13.22; $P < 0.01$) higher than the 5-year-old and the reference habitats. Mammal-dispersed species were most abundant in the 0-year-old habitat, significantly (χ^2 = 14.22; $P < 0.01$) higher than all other habitats. The NMDS ordination showed no clear separation among the habitats (Figure 5.3b). However, the ANOSIM showed that all habitats were similar.

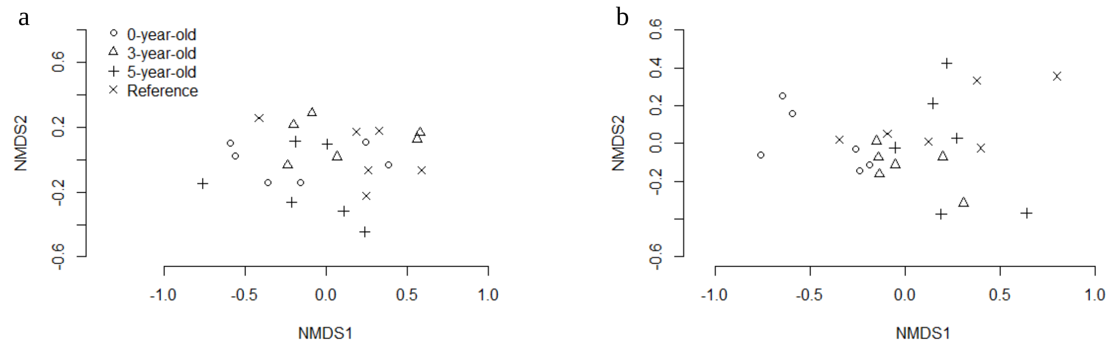

Figure 5.3: The NMDS ordination plot showing the species seed dispersal trait composition in the (a) upland and (b) lowland areas. The NMDS was based on Bray-Curtis similarity index computed using species abundance data.

Native tree seedling recruitment increased with restoration age (Table 5.4). In the upland area, two, one, eight and 21 species were recorded in the 0-, 3-, 5-year-old and the reference habitats, respectively. In the lowland area, three, 12 and 23 species were recorded in the 3-, 5-year-old and the reference habitats, respectively. The dominant seedling species were *Dalbergia obovata* E.Mey. (Fabaceae), *Combretum molle* R.Br. ex G.Don (Combretaceae), *Dalbergia armata* E.Mey. (Fabaceae) and *Kraussia floribunda* Harv. (Rubiaceae) in the reference habitat, *Diospyros lycioides* Desf. (Ebanaceae) in the 5-year-old habitat, and *Searsia rehmanniana* (Engl.) Moffet (Anacardiaceae) in both the 0- and 3-year-old habitats. Bird-dispersed species were the most dominant across all habitats, followed by mammal-dispersed species.

Table 5.4: Mean woody seedlings density (seedlings or saplings per 4 m^2) and their dispersal agents categories across habitats. For species authorship, see Table S5.2.

Plant species	Dispersal category	Seedling density							
		Upland				Lowland			
		0-year-old	3-year-old	5-year-old	Reference	0-year-old	3-year-old	5-year-old	Reference
Albizia adianthifolia	Gravity	0	0	0	0.83	0	0	0	2.66
Brachylaena discolour	Wind	0	0	0	0.67	0	0	0	0
Bridelia micrantha	Birds	0.17	0.17	0.33	0	0	0.17	0.17	0.33
Canthium inerme	Birds/Mammals	0	0	0	0	0	0	0	0.17
Celtis africana	Birds	0	0	0	0.17	0	0	0	0.67
Clausena anisata	Birds	0	0	0	0.50	0	0	0	0
Clerodendrum glabrum	Birds	0	0	0	0.17	0	0	0	0
Combretum molle	Wind	0	0	0	1.83	0	0	0	0
Croton sylvaticus	Birds/Mammals	0	0	0	0	0	0	0	0.33
Cryptocarya woodii	Birds	0	0	0	1.33	0	0	0	0
Dalbergia armata	Wind	0	0	0	1.83	0	0	0.17	5.67
Dalbergia obovata	Wind	0	0	0	8.67	0	0.17	0.33	1.16
Diospyros lycioides	Birds/Mammals	0	0	0.67	0	0	0	0	0
Diospyros scabrida	Birds	0	0	0	1.33	0	0	0	1.00
Erythrina sp.	Birds	0	0	0	0	0	0.17	0.17	0
Euclea natalensis subsp. *rotundifolia*	Birds/Mammals	0	0	0	0.17	0	0	0	0
Gymnosporia buxifolia	Birds	0	0	0	0	0	0	0	0.50
Harpephyllum caffrum	Birds/Mammals	0	0	0	0.17	0	0	0	0.17
Heteropyxis natalensis	Wind	0	0	0	0.50	0	0	0	0
Kraussia floribunda	Birds/Mammals	0	0	0.17	1.50	0	0	0.67	0.50
Macaranga capensis	Birds/Ballistic	0	0	0	0	0	0	0	0.17
Manilkara discolor	Birds/Mammals	0	0	0	0	0	0	0	3.67
Phoenix reclinata	Birds/Mammals	0	0	0	0	0	0	0	0.17
Protorhus longifolia	Birds/Mammals	0	0	0	1.00	0	0	0	2

Rauvolfia caffra	Birds/Mammals	0	0	0	0	0	0.17	0	0
Rhoicissus tomentosa	Birds/Mammals	0	0	0	0.17	0	0	0	0.83
Schrebera alata	Wind	0	0	0	0	0	0	0.5	0.33
Sclerocarya birrea subsp. *caffra*	Mammals	0	0	0	0	0	0.17	0	0
Sclerocroton integerrimus	Birds/Mammals	0	0	0.33	0.33	0	0	0	0
Searsia lucida	Birds	0	0	0	0	0	0	0.17	0
Searsia chirindensis	Birds/Mammals	0	0	0	0.33	0	0	0.5	0
Searsia dentata	Birds	0	0	0.17	0	0	0	0.83	0
Searsia rehmanniana	Birds	0.33	0	0.17	0	0	0.5	0	0
Seasia pentheri	Mammals	0	0	0.17	0	0	0	0	0
Senegalia caffra	Ballistic	0	0	0.17	0	0	0	0	0
Strychnos mitis	Birds/ Mammals	0	0	0	0	0	0	0	0.33
Tabernaemontana ventricosa	Birds/ Mammals	0	0	0	0	0	0	0	3.66
Tecomaria capensis	Wind	0	0	0	0	0	0	0.83	0
Trema orientalis	Birds/Mammals	0	0	0	0	0	0.33	0	0.17
Trichilia emetica	Birds/Mammals	0	0	0	0	0	0.17	0.17	0.83
Vachellia natalitia	Wind	0	0	0	0	0	0	0.17	0
Vangueria infausta	Mammals	0	0	0	0.33	0	0	0	0
Zanthoxylum capense	Birds	0	0	0	0.50	0	0	0	2.00

5.4. Discussion

Monitoring and assessment of restoration success are critical steps needed to provide beneficial insight into restoration challenges and successes which could guide the necessary management interventions and to inform best practice in the future (Kanowski et al. 2010; Florentine et al. 2013; Derhé et al. 2016). This study used measures of species richness, diversity, vegetation structure, IAP cover and ecological processes to assess the success of a climate change driven community-based reforestation project in the city of Durban, South Africa. Some of the assessed indicators, such as vegetation structure and native tree recruitment, are progressing towards the reference habitat. It was also interesting to note that some of the pollination and seed dispersal categories were more abundant in habitats under restoration. However, low tree species richness, and an increase in IAP cover with an increase in restoration age, were identified as the critical threats that could compromise the project success. Similar findings in the same area under restoration, but in completely different plots in the 3-, 4- and 5-year-old habitats were reported by Roy (2015). Other studies from Australia and Brazil also reported similar threats (Brancalion et al. 2012; Derhé et al. 2016). This suggests that possible management interventions are needed and recommendations on some potential interventions are made below.

Douwes et al. (2015a) assessed the nursery tree seedling stock ready for planting at Buffelsdraai Landfill Site and found that species that produce large fruits were the most dominant. For example, *Erythrina lysistemon* Hutch. (Fabaceae), *Millettia grandis* (E.Mey.) Skeels (Fabaceae), *Syzygium cordatum* Hochst. ex C.Krauss (Myrtaceae) and *Vachellia natalitia* E.Mey. (Fabaceae). Fruits of these species are easily noticeable, thus promoting their collection by Tree-preneurs. Furthermore, these species are easy to propagate and are also fast-growing (Douwes et al. 2015a). In the upland area, the 3-year-old habitat had lower species richness and diversity while the 5-year-old habitat had lower species evenness and diversity. In the lowland area, the 0-year-old habitat had lower species richness and diversity. Although the number of species planted into the 3- and 5-year-old habitats is unknown, a greater number of fast-growing species in the nursery probably contributed to lower species richness, evenness and diversity in the restored compared with the reference habitat. To overcome this challenge, Douwes et al. (2015a) recommended that Tree-preneurs should be incentivised to propagate less common species in order to increase species richness. In the Brazilian Atlantic Forest restoration study (Brancalion et al. 2012), this challenge was addressed by sourcing planting stock from both community-based seed collectors and professional seed collectors, to increase species richness.

Species richness, evenness and diversity indices (e.g., Simpson's evenness and Simpson's diversity) are widely used as indicators for assessing diversity between habitats under restoration and the reference communities, because they provide useful information on community state (e.g., Ruiz-Jaén and Aide 2006; Derhé et al. 2016). However, when assessing restoration, these indicators need to be carefully interpreted, because diversity indicators might be similar or higher in the habitats under restoration than the reference, but their species composition significantly different (Jaunatre et al. 2013). For example, in the lowland area, the 3-year-old habitat species richness, evenness and diversity showed no significant difference to the reference habitat, but these habitats differed significantly in terms of their species composition. Furthermore, tree density was not significantly different between the restored habitats and the reference habitat, but the tree height and canopy cover were visibly less established in the restored habitats in both the upland and lowland areas. However, the small size of the trees in the restored habitats (0.38-0.52 m) did not permit the measurement of parameters (e.g., diameter at breast height and canopy width) that could have allowed for proper structural comparisons between the reference and restored habitats.

The use of pioneer (fast-growing, but shade intolerant species) is the most recommended approach in forest restoration projects. These species create a canopy cover that shades out weeds, decreases fire risk and create conditions suitable for the colonization of understorey and climax species (e.g., Rodrigues et al. 2009; Goosem and Tucker 2013; Douterlungne et al. 2015). However, in the Brazilian Atlantic Forest, restoration of degraded lands using pioneer tree species resulted in the failure of many projects. When only pioneer trees were used, they matured and died before climax species could colonize the area (Rodrigues et al. 2009). In this study, all habitats under restoration in both the upland and lowland areas were dominated by pioneer tree species that have the ability to shade out weeds and create favourable conditions for understorey and climax species to colonize. Although climax and understorey species were present in the habitats under restoration, there were fewer climax and understorey species in the upland areas of the 0-and 3-year-old habitats, respectively. We recommend that their abundance be increased through enrichment planting (planting of more species in areas that have been planted before) under the established canopies of pioneer species to avoid the situation reported by Rodrigues et al. (2009).

Reforestation reports can sometimes contain information that is different from what was actually done in the field (Kanowski et al. 2010). For example, in the reforestation of the rainforest in North Queensland, Australia, audits found that the sites only had half of the plantings documented in the project reports. Furthermore, the forested area was unlikely to develop into rainforest as a result of poor tree establishment, probably caused by a lack of ongoing site maintenance (Kanowski et al. 2010). Therefore, it is critical to do a species assessment at an early stage so that necessary management interventions can be implemented to avoid poor tree establishment. For example, our study showed that an additional 14.8 and 60.8% of trees should be planted in the upland and lowland area at 0-year-old habitat to achieve an average density of 1000 and 2000 trees per ha, respectively. A higher tree density in the upland area of the 3 and 5-year-old habitats (29.5 and 34.5% more trees, respectively) probably occurred as a result of dead tree replacement intervention. We suspect that tree replacement was carried out based on dead stem observations, because after a careful assessment (removal of litter around the base of dead tree stems) of the planted habitats that were more than a year older (Mugwedi and Rouget, pers. obs. 2014) found that most of the dead tree stems were coppicing at the base of the stem. Furthermore, in both the upland and lowland areas at 0-year-old habitat, one month after planting (March 2015), some species lost their leaves accompanied by drying of stems, as a result of soil moisture stress, but re-sprouted eight months after. Therefore, the design of tree replanting interventions should be based on the outcomes of an assessment of tree mortality.

Mature native forests are characterized by a well-established structure that constitutes trees of different sizes, high canopy cover and understorey (Franklin et al. 2002). It takes decades to centuries for habitats under reforestation to develop the full structure of a mature forest (Cunningham et al. 2015). Tree canopy cover is an important developmental stage in reforestation habitats, because it creates suitable conditions for forest succession by reducing irradiance, soil temperature and shading out weeds (Wishnie et al. 2007; Coote et al. 2013; Suganuma and Durigan 2015). In reforestation habitats, tree canopy cover is a good indicator forest development within the first decade following reforestation (Suganuma and Durigan 2015). A full canopy cover can be achieved within two decades by planting more trees per hectare (e.g., 2500 trees/ha, Roy 2015), and fast-growing tree species, especially in the tropical and subtropical areas (Elliott et al. 2003; Wishnie et al. 2007). In this study, the reference habitat had tall trees with a higher canopy cover compared with the reforested habitats. However, tree height and canopy cover in the 3- and 5-year-old habitats in both the upland and lowland areas showed rapid development. This rapid tree height and canopy cover development is attributed to the dominance of fast growing species such as such as *Bridelia micrantha* (Hochst.) Baill. (Phyllanthaceae), *E. lysistemon*, *D. obovata*, *V. natalitia* (Mugwedi, pers. obs. 2014-2016). The 0-year-old habitat was also dominated with fast-growing tree species in both the upland and lowland areas. It is expected that tree height and canopy cover will also develop rapidly in this habitat.

The colonization of reforested habitats by weeds (graminoids and forbs) and IAPs is a serious concern, because this can lead to restoration failure if weed/IAP management is inadequate/ unsuitable (Kanowski et al. 2010; Simmons et al. 2016). It is therefore critical to continuously monitor the presence, expansion and distribution of IAPs. This is also because habitats that have been subjected to anthropogenic disturbance (e.g., agriculture) particularly in close proximity to cities are more vulnerable and highly likely to be invaded (e.g., Alston and Richardson 2006). Repeated removal of IAPs can increase native tree recruitment and tree species diversity (Simmons et al. 2016). Furthermore, past studies have found a decrease in IAP cover with an increase in natural forest tree canopy cover (e.g., Mandal and Joshi 2014). *Chromolaena odorata* is now becoming a serious problem in the upland area of the 5-year-old habitat, and in the lowland area of both the 3- and 5-year-old habitats. At the Buffelsdraai Landfill Site, *C. odorata* plants are cut (about 5 cm from the ground) once a year but there is no follow up to kill the trimmed stems via herbicide application. As a result, the stems coppiced, creating impenetrable *C. odorata* thickets (see Figure S5.3). Repeated cutting coupled with a higher abundance of fast growing pioneer species such as *B. micrantha*, *E. lysistemon*, *D. obovata*, *V. natalitia* (Mugwedi, pers. obs. 2014-2016) can lead to a decline in weeds and IAP cover. We therefore suggest that in recently (e.g., 0-year-old habitat) and future reforested habitats, IAPs should be uprooted and not cut, since these habitats are dominated by juvenile plants. Furthermore, all the common IAPs present in the restored habitats have established biological control agents (Baars and Nesser 1999; Zachariades et al. 2011). Therefore, biological control should also be prioritized as an alternative IAPs management option.

Additionally, the presence of weeds (graminoids and forbs) and IAPs (e.g., *C. odorata*) can alter fire regimes by increasing fuel load, fire frequency and intensity (Aronson et al. 2006; Brundu and Richardson 2016). Wildfire breakout is a major threat to forests and reforestation projects; therefore, fire prevention and suppression strategies should always be in place to avoid forest loss (e.g., Wang et al. 2007; Reyer et al. 2009; Brundu and Richardson 2016).

For example, in 2008, forest and thickets were lost as a result of fire in Hluhluwe Game Reserve, South Africa (Browne and Bond 2011). One of the possible contributing factors to the large impact of this fire was the heavy infestation of *C. odorata* that contains essential oils in the leaves, making it highly flammable (Goodall and Erasmus 1996; Witkowski and Wilson 2001) and capable of generating high flames that reach the tree canopy (te Beeste et al. 2012). Wildfire breakout, particularly in autumn and winter, is one of the biggest threats to the Buffelsdraai Landfill Site which can hamper reforestation success. If unmanaged, these fires can lead to seedling mortalities in forests (Wang et al. 2007). At present, graminoids and forbs surrounding the restored habitats are removed to create fire breaks and other fire management interventions include a fire fighting team that is always on standby to respond to any fire incident to curb fire-induced tree loss (Douwes et al. 2015b). In the event that such mortalities are incurred, a stock of insurance trees is stored in the on-site nursery to replace lost trees (Douwes et al. 2015a).

Litter that accumulates on the forest floor is a basic component of almost all forests and is an essential contributor towards nutrient cycling (Rubiano-Cardona et al. 2013). Litter accumulation is regulated by multiple biological (e.g., tree age, plant species and forest composition) and climatic (e.g., rainfall, temperature and humidity) factors (Rubiano-Cardona et al. 2013). Higher levels of litter accumulation were observed in both the upland and lowland areas in the reference habitat, than those under restoration. Higher litter accumulation levels in the reference habitat are attributed to its tall trees with wider canopies and a well-developed understorey. In the young, restored habitats litter accumulation presumably takes a relatively longer time than the reference habitat as trees do not have fully developed canopies and hence do not shed a lot of leaf litter (as reported in Rubiano-Cardona et al. 2013). At present, the herbaceous understory layer is contributing more to litter accumulation in both the upland and lowland areas of the restored habitats. However, it is expected that as tree height and canopy increases with restoration age, coupled with the development of understorey layer, this role will be fulfilled by the trees (as reported in Kanowski et al. 2003).

Numerous studies on reforestation have shown the importance of including tree species that are attractive to animals in order to promote key ecological process, such as pollination and seed dispersal (Kanowski et al. 2003; Goosem and Tucker 2013), that ensure the long-term stability of an ecosystem (McAlpine et al. 2016). Our study showed that though most of the pollination and seed dispersal categories were abundant in habitats under restoration, some habitats lacked certain pollination and seed dispersal categories entirely. For example, there were fewer bee-pollinated species in both the upland and lowland areas of the 5-year-old-habitat while ballistic-dispersed species were scarcer in the lowland area of the 0-year-old habitat. We therefore recommend that during enrichment planting, these habitats should be supplemented with species belonging to categories that are lacking in order to increase their ecosystem functioning. Some studies have shown that mobile pollination and seed dispersal agents actively move between established and non-established patches (Lundberg and Moberg 2003; Staddon et al. 2010), thus increasing chances of pollination and seed dispersal in non-established patches. Furthermore, attractiveness of areas under reforestation to pollinators and dispersal agents develops over time and can be assessed as the trees mature. In our study flowering, pollination and fruiting (Figure 5.4) observed in both upland and lowland areas of the 3- and 5-year-old habitats suggest that areas under restoration are becoming more attractive to pollinators and frugivores. An increase in bird species richness from 91 to 145 over a five-year period (Douwes et al. 2015a) supports this claim.

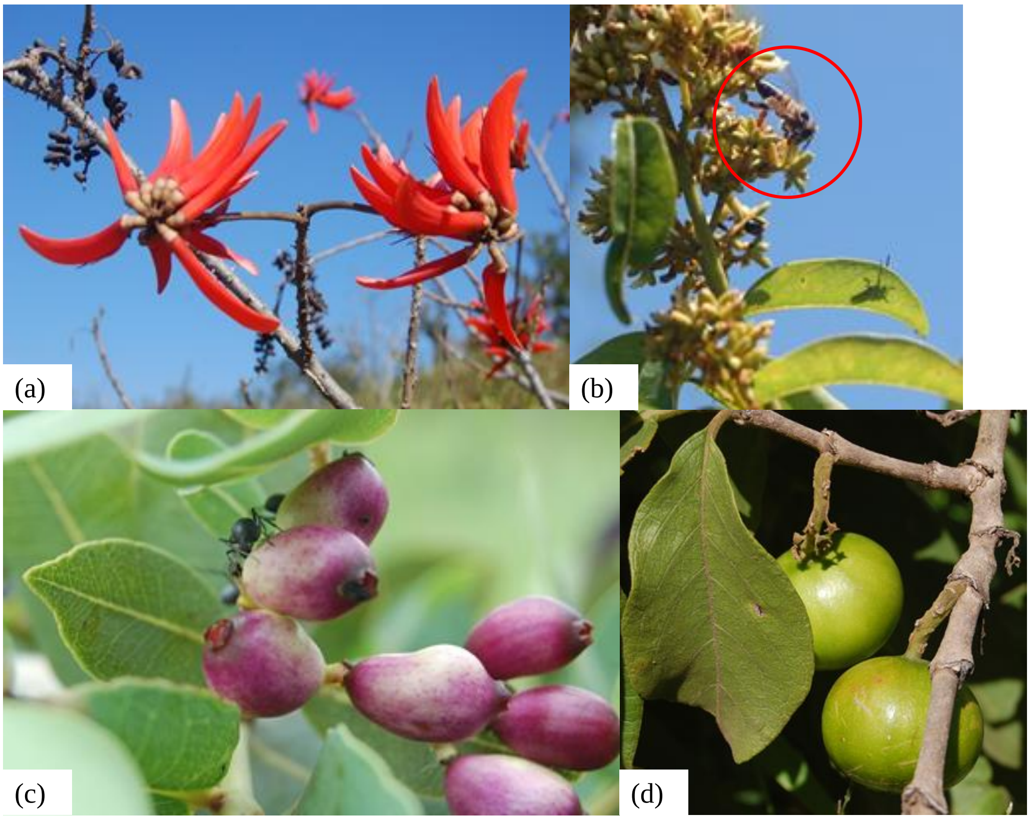

Figure 5.4: Flowering, pollination and fruiting in the 3- and 5-year-old habitats. (a) *Erythrina lysistemon* with flowers and pods, (b) bee pollination (within the red circle) on *Dalbergia obovata* flower, (c) *Syzygium cordatum* fruits, and (d) *Vangueria infausta* fruits.

Seed dispersal is the last phase in the plant reproductive cycle and the first phase in the process of population renewal (Harper 1977). As a result, seedling recruitment is considered to be one of the key factors that determine the success of the long-term vegetation restoration (McAlpine et al. 2016). Restoration habitats that are adjacent to the existing remnant forest are more likely to recover quickly as a result of colonization by animal seed dispersers (White et al. 2004; Monie et al. 2013; Abiyu et al. 2016). However, the recruitment of animal-dispersed tree seedlings is higher in the established habitats (Viani et al. 2015). In this study, habitats under restoration are adjacent to the reference habitat and animal-dispersed seedling richness and abundance increased with an increase in restoration age. The lowland areas in both the 3- and 5-year-old habitats had higher seedling richness compared with the upland area (12 spp. vs. eight spp., and eight spp. vs. one spp.). Higher seedling richness in the lowland areas could be attributed to microclimate conditions that are conducive for seedling recruitment, created by tall trees with wider canopy cover (as reported in Wishnie et al. 2007). The recruitment of animal dispersed species in these areas showed that frugivores are moving into areas under restoration and are most likely attracted by perches (established trees) and food (nectar and fruits). An increase in bird species richness in restoration habitats supports a higher recruitment of bird dispersed species. Most of the species that are mammal dispersed at Buffelsdraai Landfill Site are suspected to be dispersed by vervet monkeys (*Cercopithecus pygerythrus* F. Cuvier, 1821 (Cercopithecidae)) that move between the reference and restoration areas. In the restoration of mined coastal dune forest in Richards Bay, South Africa, Foord et al. (1994) found that vervet monkeys influence succession in habitats under restoration, because they disperse viable seeds of different tree species.

5.5. Conclusion

An important recommendation emanating from the present study is that reforestation success assessment should be carried out in the early stages of reforestation projects in order to understand the ecological development trajectories and to inform necessary management interventions, to maximise reforestation benefits (Monie et al. 2013; Roy 2015; Abiyu et al. 2016). The use of multiple indicators (e.g., vegetation structure, species diversity, ecological processes and IAP cover) also gave valuable insight into the ongoing ecological trajectories and enabled the identification of necessary management interventions. Flowering and fruiting of planted trees, seed dispersal and creation of favourable microclimatic conditions conducive for native tree recruitment signify restoration success (Monie et al. 2013) and the habitats under restoration showed some signs of success as early as three years after planting. However, enrichment planting is needed in certain habitats due to low tree density and richness. Enrichment planting should prioritize tree species in pollination and seed dispersal categories that are lacking. Pollination and seed dispersal processes should be monitored during the flowering and fruiting period, so that species that lack pollinators and dispersers can be identified and their agents introduced on site if possible (Menz et al. 2011; McAlpine et al. 2016). Most importantly, all these indicators should form part of a long-term monitoring and evaluation strategy.

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5.7. Supplementary materials

Table S5.1: The total number of recorded species per habitat (in brackets) and the number of shared species across habitats.

Habitats	0-year-old	3-year-old	5-year-old	Restored combined	Reference
0-year-old (36 spp.)	-	25 spp.	24 spp.	-	25 spp.
3-year-old (40 spp.)	25 spp.	-	28 spp.	-	30 spp.
5-year-old (44 spp.)	24 spp.	28 spp.	-	-	40 spp.
Restored combined (59 spp.)	-	-	-	-	49 spp.
Reference (70 spp.)	25 spp.	30 spp.	40 spp.	49 spp.	-

Table S5.2: Checklist of plant species found in the restored and reference habitats.

Species	Family	Upland area				Lowland area			
		0-year-old habitat	3-year-old habitat	5-year-old habitat	Reference habiat	0-year-old habitat	3-year-old habitat	5-year-old habitat	Reference habitat
Harpephyllum caffrum Bernh.	Anacardiaceae	X	X	X	X	X	X	X	-
Protorhus longifolia (Bernh.) Engl.	Anacardiaceae	-	X	X	X	-	-	-	X
Sclerocrya birrea (A.Rich.) Hochst. subsp. *caffra* (Sond.) Kokwaro	Anacardiaceae	-	-	-	-	-	-	-	-
Searsia chirindensis (Baker f.) Moffett	Anacardiaceae	-	-	X	X	-	X	X	X
Searsia dentata (Thunb.) F.A.Barkley	Anacardiaceae	-	-	-	-	-	-	-	-
Searsia lucida (L.) F.A.Barkley	Anacardiaceae	X	-	-	-	-	X	-	-
Searsia pentheri (Zahlbr.) Moffett	Anacardiaceae	-	-	X	-	-	-	-	X
Searsia rehmanniana (Engl.) Moffet	Anacardiaceae	X	-	-	X	-	X	X	-
Annona senegalensi Pers.	Annonaceae	-	-	-	-	-	-	-	-
Rauvolfia caffra Sond.	Apocynaceae	X	X	-	X	-	-	-	X
Tabernaemontana ventricosa Hochst. ex A.DC.	Apocynaceae	X	X	-	X	-	-	-	X
Cussonia spicata Thunb.	Araliaceae	-	-	-	-	-	-	-	-
Cussonia zuluensis Strey	Araliaceae	-	-	-	X	-	-	X	-
Phoenix reclinata Jacq.	Arecaceae	X	X	-	-	-	-	X	X

Aloe ferox Mill.	Asphodelaceae	-	-	X	-	-	-	X	-
Brachylaena discolor DC.	Asteraceae	X	X	X	-	-	X	-	-
Kigelia africana (Lam.) Benth.	Bignoniaceae	-	X	X	-	X	X	-	X
Tecomaria capensis (Thunb.) Lindl.	Bignoniaceae	-	-	X	-	X	X	-	X
Buddleja saligna Wild.	Buddlejaceae	-	-	-	-	-	-	-	-
Commiphora woodii Engl.	Burseraceae	-	-	-	-	-	-	-	-
Gymnosporia buxifolia (L.) Szyszyl.	Celastraceae	-	-	-	-	-	-	-	-
Chaetacme aristata Planch.	Celtidaceae	-	-	-	X	-	-	-	X
Trema orientalis (L.) Blume	Celtidaceae	X	X	-	-	X	X	X	-
Combretum kraussii Hochst.	Combretaceae	-	-	-	X	-	-	-	X
Combretum molle R.Br. ex G.Don	Combretaceae	X	-	-	X	-	-	-	X
Diospyros lycioides Desf.	Ebenaceae	-	-	-	X	-	-	X	-
Diospyros whyteana (Hiern) F.White	Ebenaceae	-	-	-	X	-	X	X	-
Euclea natalensis A.DC. subsp. *natalensis*	Ebenaceae	-	-	-	X	-	X	-	X
Nectaropetalum zuluense (Schönland) Corbishley	Eryrthroxylaceae	-	-	-	-	-	-	-	-
Antidesma venosum E.Mey. ex Tul.	Euphorbiaceae	-	-	-	-	-	-	-	-
Bridelia micrantha (Hochst.) Baill.	Euphorbiaceae	X	X	X	X	-	X	X	-
Croton sylvaticus Hochst.	Euphorbiaceae	X	-	-	-	X	-	-	-

Drypetes arguta (Müll.Arg.) Hutch.	Euphorbiaceae	-	-	-	X	-	-	-	-
Drypetes natalensis (Harv.) Hutch.	Euphorbiaceae	-	-	-	-	-	-	-	X
Euphorbia tirucalli L.	Euphorbiaceae	X	-	-	-	X	-	-	X
Macaranga capensis (Baill.) Benth. ex Sim	Euphorbiaceae	-	-	-	X	-	X	X	-
Sclerocroton integerrimus Hochst.	Euphorbiaceae	-	-	X	-	-	X	-	-
Shirakiopsis elliptica (Hochst.) Esser	Euphorbiaceae	-	-	-	-	-	-	-	-
Spirostachys africana Sond.	Euphorbiaceae	-	-	-	-	-	-	-	-
Acacia ataxacantha DC.	Fabaceae	-	-	-	X	-	-	-	X
Albizia adianthifolia (Schumach.) W.Wight	Fabaceae	X	-	X	X	-	-	X	-
Baphia racemosa (Hochst.) Baker	Fabaceae	-	-	X	X	-	-	X	-
Bauhinia tomentosa L.	Fabaceae	X	-	-	-	X	-	X	X
Dalbergia armata E.Mey.	Fabaceae	-	-	X	X	-	-	-	X
Dalbergia obovata E.Mey.	Fabaceae	X	X	X	X	X	-	-	X
Dichrostachys cinerea (L.) Wight & Arn.	Fabaceae	-	-	X	-	-	-	-	X
Erythrina caffra Thunb.	Fabaceae	X	X	X	X	X	X	X	X
Erythrina lysistemon Hutch.	Fabaceae	X	X	X	-	-	-	-	-
Erythrina latissima E.Mey.	Fabaceae	-	-	-	-	-	-	-	-
Millettia grandis (E.Mey.) Skeels	Fabaceae	X	X	X	X	-	X	X	-

Schotia brachypetala Sond.	Fabaceae	-	-	-	X	-	-	-	X
Senegalia caffra (Thunb.) Willd.	Fabaceae	-	X	X	X	X	X	X	-
Vachellia natalitia E. Mey.	Fabaceae	X	X	X	-	-	X	X	-
Vachellia nilotica (L.) Wild. ex Delile	Fabaceae	-	X	-	X	-	X	-	-
Vachellia sieberiana DC.	Fabaceae	-	X	-	X	-	-	-	X
Vachellia xanthophloea Benth. P.J.H. Hurter	Fabaceae	X	X	X	-	-	-	-	X
Xylotheca kraussiana Hochst.	Flacourtiaceae	-	-	-	-	-	-	-	-
Heteropyxis natalensis Harv.	Heteropyxidaceae	X	-	X	X	-	-	X	-
Apodytes dimidiata E.Mey. ex Arn.	Icacinaceae	-	-	X	X	-	-	-	X
Cryptocarya latifolia Sond.	Lauraceae	-	-	-	X	-	-	X	X
Cryptocarya woodii Engl.	Lauraceae	-	-	-	X	X	X	X	X
Ekebergia capensis Sparrm.	Meliaceae	X	-	-	-	-	X	-	-
Ekebergia pterophylla (C.DC.) Hofmeyr	Meliaceae	X	-	X	-	X	X	X	-
Trichilia emetica Vahl	Meliaceae	X	X	-	-	X	X	X	X
Ficus natalensis Hochst.	Moraceae	-	X	-	-	-	X	X	
Ficus sur Forssk.	Moraceae	-	-	X	-	-	-	-	-
Syzygium cordatum Hochst. ex C.Krauss	Myrtaceae	X	X	X	X	-	-	X	-
Syzygium guineense (Willd.) DC.	Myrtaceae	-	-	X	X	-	X	-	X
Ochna arborea Burch. ex DC.	Ochnaceae	-	-	-	-	-	-	-	-

Schrebera alata (Hochst.) Welw.	Oleaceae	-	-	-	-	-	-	-	-
Ziziphus mucronata Wild.	Rhamnaceae	-	-	-	X	X	X	X	X
Burchellia bubalina (L.f.) Sims	Rubiaceae	-	-	-	X	X	X	X	-
Canthium inerme (L.f.) Kuntze	Rubiaceae	-	-	-	X	-	-	X	X
Canthium spinosum (Klotzsch) Kuntze	Rubiaceae	-	-	-	X	X	-	-	-
Coddia rudis (E.Mey. ex Harv.) Verdc.	Rubiaceae	-	-	-	X	-	-	X	X
Kraussia floribunda Harv.	Rubiaceae	-	-	-	X	-	-	-	X
Psychotria capensis (Eckl.) Vatke subsp. *capensis*	Rubiaceae	-	-	-	X	-	-	-	X
Psydrax obovata(Eckl. & Zeyh.) Bridson	Rubiaceae	-	-	-	-	-	-	-	-
Rothmania globosa Thunb.	Rubiaceae	-	-	-	X	-	-	X	-
Tricalaysia lanceolata (Sond.) Burtt Davy	Rubiaceae	-	-	-	X	-	X	X	X
Vangueria infausta Burch.	Rubiaceae	X	-	X	-	-	X	-	X
Clausena anisata (Wild.) Hook.f. ex Benth	Rutaceae	-	-	-	-	-	-	-	-
Zanthoxylum capense (Thunb.) Harv.	Rutaceae	-	-	-	-	-	-	-	-
Hippobromus pauciflorus (L.f.) Radlk.	Sapindaceae	-	-	-	X	-	-	-	X
Manikara discolor (Sond.) J.H.Hemsl.	Sapotaceae	-	-	-	X	-	-	-	-
Dombeya cymosa Harv.	Sterculaiaceae	-	-	-	X	-	-	X	X

Dombeya rotundifolia Hochst.	Sterculaiaceae	-	-	-	-	-	-	-	-
Dombeya tiliacea (Endl.) Planch.	Sterculaiaceae	-	-	-	X	-	-	X	X
Strelitzia nicolai Regel & Körn.	Strelitziaceae	X	X	X	X	X	X	-	-
Grewia occidentalis L.	Tiliaceae	-	-	-	X	-	-	-	X
Clerodendrum glabrum E.Mey.	Verbenaceae	X			-	-	-	X	X

Table S5.3: Invasive alien plants recorded in the restoration (0, 3 and 5-year-old) and reference habitats.

Species	Family		Upland area				Lowland area			
		Invasive category	0-year-habitat	3-year-habitat	5-year-habitat	Reference habitat	0-year-habitat	3-year-habitat	5-year-habitat	Reference habitat
Chromolaena odorata (L.) R.M.King & H.Rob.	Asteraceae	1b	X	X	X	X	X	X	X	X
Lantana camara L.	Verbenaceae	1b	X	X	X	X	X	X	X	X
Melia azedarach L.	Meliaceae	1b, 3 in urban areas	X	X	X	X	X	X	X	X
Rubus cuneifolius Pursh	Rosaceae	1b	-	-	X	-	-	-	-	-
Solanum mauritianum Scop.	Solanaceae	1b	-	-	-	-	-	-	-	-

For a details on the National Environmental Management: Biodiversity Act (NEMBA) 2004 (Act no. 10 of 2004) invasive alien categories see South Africa (2016).

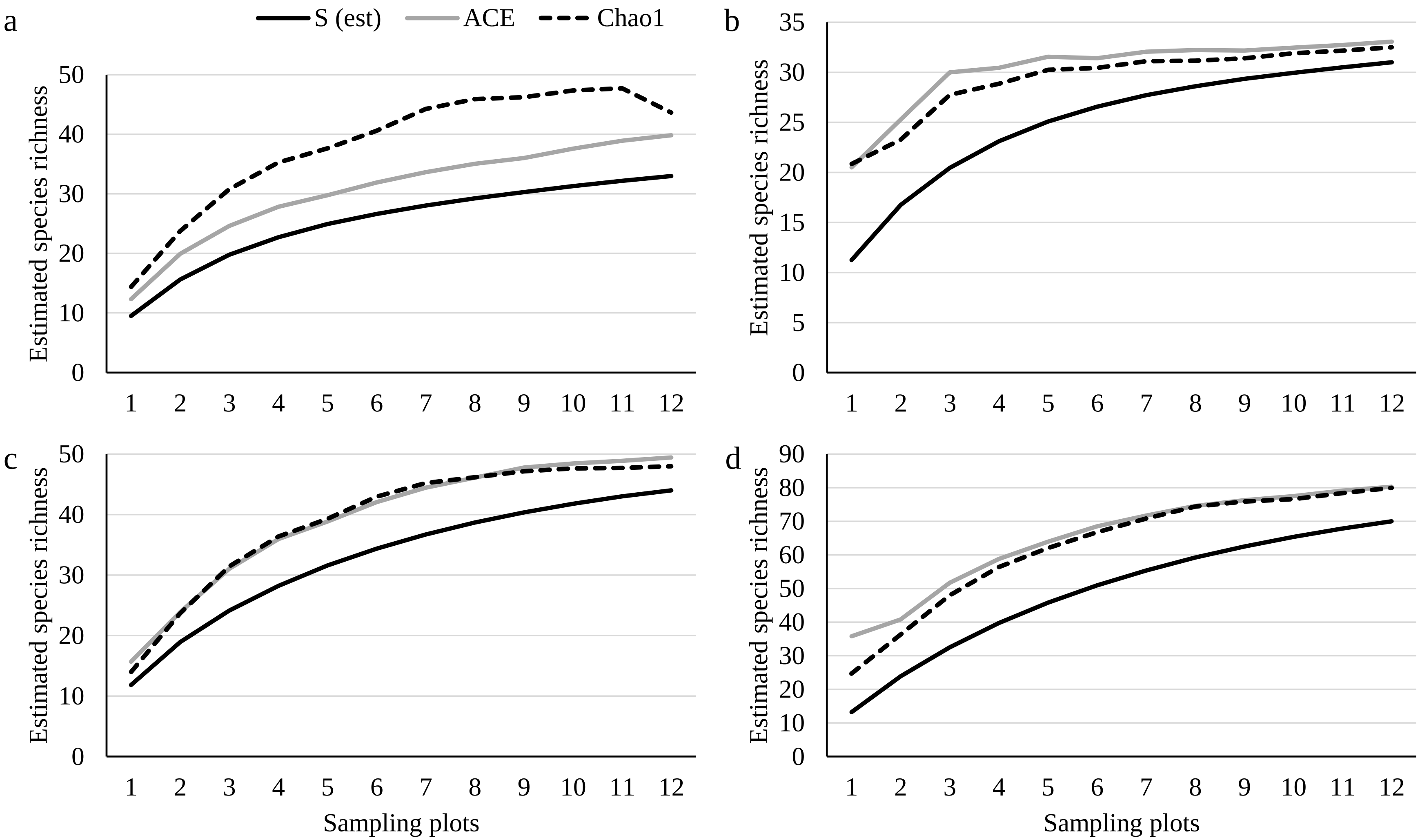

Figure S5.1: Species accumulation curves. Observed (S est.) and estimated (ACE and Chao1) species richness in the (a) 0-, (b) 3-, (c) 5-year-old and (d) reference habitats.

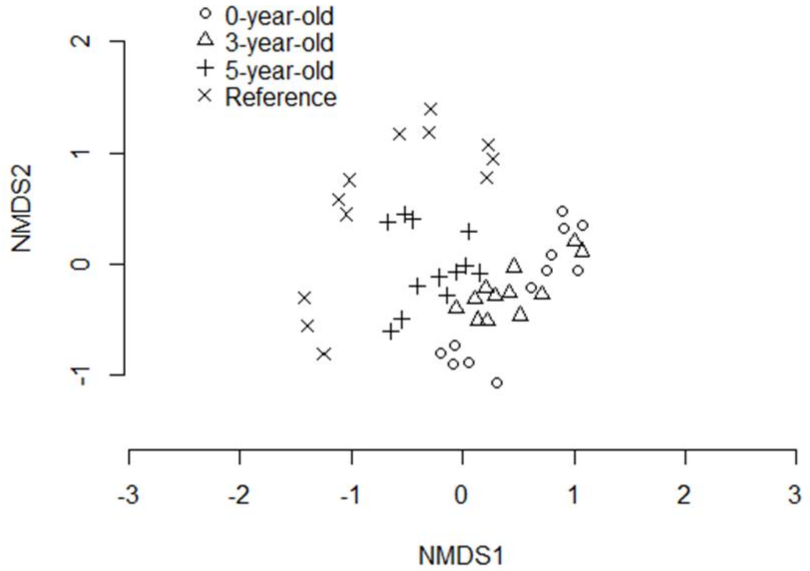

Figure S5.2: The NMDS ordination plot showing the overall tree species composition per habitat in both upland and lowland sections of the study site. The NMDS was based on Bray-Curtis similarity index computed using species abundance data.

Figure S5.3: Dense thickets of *Chromolaena odorata* infestation (foreground) in the 5-year-old habitat (Picture taken by Mugwedi L, 2015).

CHAPTER 6: MICROTOPOGRAPHY MODULATED SAPLING RESILIENCE TO AN EL NIŇO INDUCED DROUGHT WITHIN A RESTORED SUBTROPICAL FOREST IN DURBAN, SOUTH AFRICA

This chapter is based on

LF Mugwedi, M Rouget, B Egoh, Sershen, S Ramdhani, R Slotow (**Under review**) Microtopography modulated sapling resilience to an El Niño induced within a resotred subtropical forest in Durban, South Africa. New Forest Journal.

Abstract

The severe drought associated with the 2015 El Niño Southern Oscillation event presented an opportunity to assess its impact on the reforestation of a subtropical forest in Durban, South Africa. The aim of this study was to assess how microtopographic positions may have influenced species mortality at plot level, and the growth performance of the four most dominant planted tree species (*Bridelia micrantha*, *Erythrina lysistemon*, *Millettia grandis* and *Vachellia natalitia*) under a natural drought. Species growth rates were assessed in terms of root-collar diameter, stem height and canopy width. The assessments were carried out at planting (November 2014 and February 2015) and 13 months post-planting, in the upland and lowland areas of the habitat. Lowland area had deeper soils with a higher soil moisture level compared to the upland area. At plot level, overall mortality in the lowland area (34.1%) was higher than in the upland area (18.9%), at 13 months post-planting. However, the four dominant species had higher survival and growth rates in the lowland area. Superior growth performance of *E. lysistemon* and *V. natalitia* in both the upland and lowland areas, measured in terms of root-collar diameter, stem height and canopy width suggest that these species are good pioneer species for both dry and wet habitats, while *B. micrantha* may be more suitable in wet habitats. Apart from the drought adaptation strategies exhibited by some species, knowledge on species-specific habitat requirements and sapling hardening under nursery conditions can also increase species resilience to drought. The results highlight the importance of considering microtopographic positions and understanding species responses to water stress when selecting species when reforesting dry and drought prone areas.

Keywords: Climate change; Growth rate; Sapling hardening; Sapling mortality; Species selection

6.1. Introduction

Droughts associated with the El Niño Southern Oscilation (ENSO) events are becoming more frequent and more intense in the tropical, subtropical and temperate regions (Kocher et al. 2009; Allen et al. 2010; McDowell et al. 2011). These droughts have negative impacts on seedling growth and survival (Bunker and Carson 2005; Edwards and Krockenberger 2006; Maza-Villalobos et al. 2013), presenting a major challenge to reforestation initiatives in dry and drought prone areas. Climatic modelling studies in southern Africa have shown that rainfall is likely to be highly variable across the region over the next few decades, and accompanied by high temperatures, and more frequent and intense droughts (Tadross et al. 2011; Naik and Abiodun 2016). In 2015, South Africa (SA) experienced a drought associated with the El Niño event (Botai et al. 2016). District level rainfall analyses showed that 2015 was the driest year on record since 1921 (Botai et al. 2016). The drought was also accompanied by a 0.86°C increase in mean temperature, thus making it the hottest year on record since 1951 (Botai et al. 2016).

Comparing the responses of different tree species to severe drought is critical in the selection of species that are able to establish under limited soil moisture conditions (Edwards and Krockenberger 2006; Montwé et al. 2016), and create microclimatic conditions that promote natural succession (Walker and del Moral 2003). Studies have shown that among other abiotic (e.g., soil nutrients) and biotic (e.g., pests) factors, rapid forest establishment is also influenced by species selection (Li et al. 2014; Montwé et al. 2016). The use of fast growing indigenous pioneer tree species, inter-planted with late successional species to promote rapid forest establishment, is now a widely practiced approach in reforestation initiatives worldwide (Vallejo et al. 2012; Goosem and Tucker 2013). However, the selection of reforestation species that promote rapid forest establishment has now become more challenging as a result of global climate change (Poorter and Markesteijn 2008; Vallejo et al. 2012; Montwé et al. 2016). An increase in drought frequency and intensity, temperature, erratic rainfall, forest fires, and pest and disease outbreaks, due to global climate change will lead to reduced seedling growth and increased seedling mortality, compromising forest establishment (Zhao and Running 2010; Reichstein et al. 2013; Buhk et al. 2016). This is because the tree seedling establishment stage is considered to be the most important bottleneck for successful plant population regeneration (Poorter and Markesteijn 2008; James et al. 2011). Thus, ensuring successful seedling establishment represents a major challenge in reforestation, particularly in dry areas, and areas prone to drought.

Since droughts are episodic in many parts of the world event, few studies have assessed the response of tree saplings to natural drought under field conditions; this is especially true for subtropical forests (but see Luo et al. 2016; Kuang et al. 2017). Studies that have investigated seedling responses to drought have generally adopted one of the following approaches: (1) exposing potted seedlings to different watering regimes under the controlled environments such as greenhouse and growth chamber conditions (e.g., Palacios-Romero et al. 2017); (2) growing seedlings under rain shelter (e.g., Rosas et al. 2013; Kuang et al. 2017); (3) growing seedlings in the drier upper slopes and wetter lower slopes (Omary 2011; Li et al. 2014). Although these studies do provide some insights into species response to drought, the measured plant traits generally respond differently when subjected to natural drought in field conditions (Mokany and Ash 2008; Liu et al. 2011). For example, Lo Gullo et al. (2003) found that potted seedlings cannot demonstrate actual species responses to drought conditions, because plant roots in a pot tend to grow in the upwards direction where water potential is more negative, thus leading to false estimates of plant responses to water stress. Consequently, testing seedling responses under natural drought in field conditions is recommended to complement these studies (e.g., Liu et al. 2011; Vasques et al. 2016).

Most of the studies that have assessed how reforestation tree saplings respond to drought associated with ENSO events have been conducted in the tropical forests (e.g., Gilbert et al. 2001; Bunker and Carson 2005; Edwards and Krockenberger 2006; Craven et al. 2011; Maza-Villalobos et al. 2013; Maréchaux et al. 2016). When considering that drought events are likely to be more frequent in southern Africa (Tadross et al. 2011; Naik and Abiodun 2016), there is a need to assess the response of indigenous tree species to drought (Jooste 2015). This is because studies on the tree sapling responses to natural drought remain scant for subtropical forests, especially under field conditions (Luo et al. 2016; Kuang et al. 2017). Naidoo et al. (2013) recommended that reforestation initiatives in southern Africa should include drought tolerant tree species to enhance climate change mitigation and adaptation. For the reasons discussed above, the drought associated with the ENSO event that hit SA in 2015 to 2016 presented an ideal opportunity to assess the responses of reforestation tree species to a natural drought at Buffelsdraai Landfill Site Community Reforestation Project. The primary objectives were to, (1) to examine the influence of microtopographic conditions on tree species' responses to natural drought at plot level and, (2) to assess the effect microtopographic conditions on the growth performance of the four most dominant tree species under natural drought.

6.2. Materials and methods

6.2.1. Study site

In 2008, the eThekwini Municipality (city of Durban) embarked on a major reforestation programme to restore degraded forest ecosystems. This opportunity was presented by the city's hosting of 2010 FIFA World Cup™ and the United Nations Framework Convention on Climate Change COP17/CMP7 in 2011 (Diederichs and Roberts 2015). A major goal of the reforestation programme was to offset the carbon footprint associated with hosting of both mega events, while enhancing the city's ecological and adaptive capacity to climate change, through socio-economic improvement (job creation), biodiversity protection, and ecosystem services provision (e.g., water quality) (Douwes et al. 2015). This study was conducted at the largest reforestation site in Durban, the Buffelsdraai Landfill Site (29.62961S; 30.980392E). The active landfill site is 116 ha and has a buffer zone of 757 ha, of which 580 ha (formerly low productive dryland sugarcane field) has been restored to scarp forest by planting around 59 indigenous tree species (Mugwedi et al. 2017). Tree-preneurs (local community members who grow seedlings) source indigenous tree seeds from the local forest and woodland patches. Sapling hardening techniques are practiced by Tree-preneurs, and at the nursery to increase species resilience to harsh field conditions. Tree-preneurs grow seedlings using local soils without fertilizer or mulch. Once the saplings are received at the nursery, irrigation is progressively reduced for three to twelve months (Douwes et al. 2016). As of January 2015, 595 476 trees had been planted over 442 ha at Buffelsdraai Landfill Site (Douwes et al. 2016).

The topography of the study area is characterized by the undulating and steep slopes (200 m to 325 m altitude) draining into the Black Mhlasini stream in the north and White Mhlasini stream in the south. The dominant geology within the site is Dwyka Tillite, a glacial conglomerate parent material that is base-rich, hard and resistant to weathering. The upland area is characterized by shallower acrisol soil (20-40 cm) (EThekwini Municipality 2014). The soil is probably shallower as a result of higher soil erosion caused by runoff, exacerbated by cultivation (EThekwini Municipality 2014). The lowland area is characterized by deeper lithol soil (60-110 cm) (EThekwini Municipality 2014). The deeper soil is probably a result of the deposition of materials from the upper slopes (EThekwini Municipality 2014).

The vegetation, broadly classified as KwaZulu-Natal Coastal Belt (grassland and subtropical forest), is highly transformed and fragmented with little formal protection. The vegetation type as a whole is classified as endangered (Mucina and Rutherford 2006). The remnants of indigenous forest form part of Eastern Scarp Forest, usually located at an altitude of 100 to 1000 m (Von Maltitz et al. 2003), and is described as a refuge forest that survived the last glacial maximum (≈18000 BP) (Eeley et al. 1999).

The rainfall data from 2010 to 2015, sourced from Phoenix Weather Station (29.7043S, 30.9761E), approximately seven kilometres from the study area, showed that the rainfall from 2010 to 2014 ranged between 564 mm and 1024 mm, while 312 mm was recorded for 2015 and 525 mm in 2016. The annual rainfall received in 2015 was 29% lower than the average minimum summer rainfall in the region.

6.2.2. Experimental design and data collection

6.2.2.1. Soil moisture sampling

Gravimetric soil moisture sampling was done from March to December 2015, in the upland and lowland areas. Soil samples were collected from two soil depths of 0-10 cm and 10-20 cm within each plot (replicated five times) using an auger and placed in zip-lock plastic bags. Plant debris and gravel were removed before weighing the samples. A 30 g sample was weighed to obtain the wet mass, oven-dried at 105°C for 48 h and then reweighed to obtain dry mass (Matías et al. 2011).

6.2.2.2. Tree species sampling a plot level

The long-term monitoring plots were established in February 2015, on a site planted between November 2014 and February 2015 to study reforestation success. Microtopographic positions (upland and lowland areas) within a site affect hydrologic and soil conditions (Simmons et al. 2012). As a result, within a species, individual tree growth rates may vary significantly (Omary 2011; Simmons et al. 2012). To minimize the confounding effects of soil moisture and planting densities, upland and lowland areas were selected based on a soil survey done on the entire Buffelsdraai Landfill Site (eThekwini Municipality 2014). Twelve plots of $400m^2$ (20m × 20m) (six plots in both upland and lowland areas) were randomly established within the site. All planted tree species within each plot were identified and counted shortly after planting (February 2015) and 13 months post-planting (March 2016).

6.2.2.3. Sampling of the four most dominant plated tree species

The four most dominant planted indigenous tree species were chosen for the long-term growth monitoring study that commenced in February 2015. The selected species were

Bridelia micrantha (Hochst.) Baill. (Phyllanthaceae), *Erythrina lysistemon* Hutch. (Fabaceae), *Millettia grandis* (E. Mey.) Skeels (Fabaceae) and *Vachellia* (*Acacia) natalitia* E. Mey. (Fabaceae). The dominant species were chosen via 24 walk-belt transects of eight meters by 200 m in both the upland and lowland areas of habitats planted in the growing season (November to February) of 2009-2010, 2011-2012, and 2014-2015 (n = 8 transects per habitat). Six dominant species were identified (Table S6.1), but only four species that had more than 30 individuals in both upland and lowland areas across the habitats were selected for the long-term monitoring study. The selected species are easy to propagate, an important factor to consider when selecting fast growing pioneer species for reforestation (Elliott et al. 2003; Goosem and Tucker 2013). The effect of drought was only assessed in the habitat planted in 2014 – 2015 growing season, because sapling stage is the most susceptible and the most important bottleneck for successful plant population regeneration (Poorter and Markesteijn 2008; James et al. 2011).

Thirty individuals of each species per microtopographic position (upland and lowland areas) (n = 240 individuals) were measured for root-collar diameter, stem height and canopy width shortly after planting (February 2015) and 13 months post-planting (March 2016). The mortality of tagged individuals of the four species was also recorded 13 months post-planting (March 2016). The root-collar diameter was measured using callipers at 10 cm above ground. Stem height was measured from the soil level to the tip of the lead stem. Canopy width was measured along two directions (north-south and east-west) (Li et al. 2014). Since this was opportunistic study, lacking a control (i.e., irrigated saplings), annual growth rate (tree height) data of the study species under non-drought conditions were sourced from published descriptions and online databases (Johnson and Johnson 2002; Plant Resources of Southern Africa, www.prota4u.info; PlantZAfrica, www.pza.sanbi.org) for comparison with the annual tree height growth rate under drought conditions.

6.2.3. Data analyses

6.2.3.1. Rainfall and soil moisture

The total monthly rainfall (March to December 2017) was also plotted to help explain the soil moisture trend. Percentage soil moisture was then calculated as [(wet mass – dry mass)/dry mass × 100] (Yao et al. 2016). Percentage soil moisture between 0-10 and 10-20 cm showed no significant difference, hence the data were pooled for all subsequent analyses. Descriptive statistics (mean±SD) were used to interpret soil moisture data.

6.2.3.2. Drought effect on tree species mortality at plot level

Drought impact on tree species mortality at plot level was analysed using a paired t-test in Statistica version 11 (StatSoft 2012). No statistical test was employed to test the effect of drought within species, since the number of individuals per species across plots was highly variable (e.g., number of individuals per species within plots varied between one and eight). As a result, only percentage mortality rate per species was calculated [(number of individuals in a plot in 2015 – number of individuals in a plot in 2016)/number of individuals in plot in 2015) × 100].

6.2.3.3. Drought effect on the four most dominant planted tree species growth

The root-collar diameter, stem height and canopy width annual increments were computed. The annual increments were calculated as follows; root-collar diameter in 2016 – root-collar diameter in 2015. A paired t-test was used to test within species growth response (for root-collar diameter, stem height and canopy width) to natural drought under different microtopographic positions. To fulfil homoscedasticity assumptions, values were square-root-transformed. All statistical analyses were performed in Statistica version 11 (StatSoft 2012).

6.3. Results

6.3.1. Qualitative observations

Within two to six weeks after planting (March 2015), leaves started to wilt, followed by leaves shedding, and stem wilting, and sapling mortality in some cases (Figure 6.1). However, field observations in mid-January showed that most of the species which experienced leaf shedding and/or stem die-back, resprouted and/or coppiced, following rainfall events in December 2015 and January 2016 (Figure 6.2).

6.3.2. Rainfall and soil moisture

In 2015, the site received a considerable amount of rainfall from January to March (57.6 mm, 62.2 mm and 81.2 mm, respectively) and in December (61.4 mm). April to June, August and October were the driest months in 2015 (Figure 6.3a). The lowland area had higher soil moisture than the upland area across all the months in which measurements were conducted (Figure 6.3b). As expected, the higher soil moisture contents coincided with the months that received the high rainfall.

Figure 6.1: Dead *Erythrina lysistemon* sapling in the upland area.

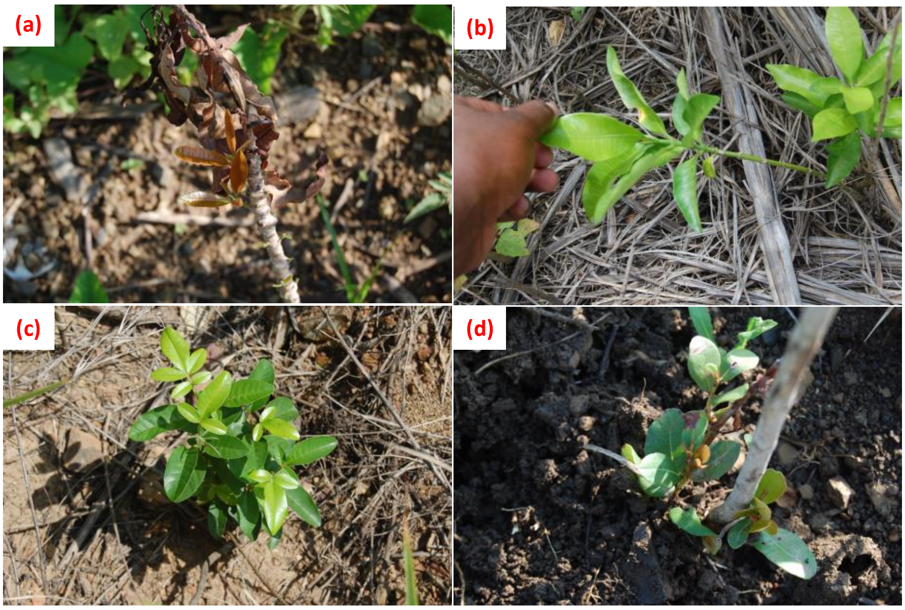

Figure 6.2: Resprouting (a) *Protorhus longifolia* and coppicing (b) *Tabernaemontana ventricosa*, (c) *Trichilia dregeana* and (d) *Syzygium cordatum*.

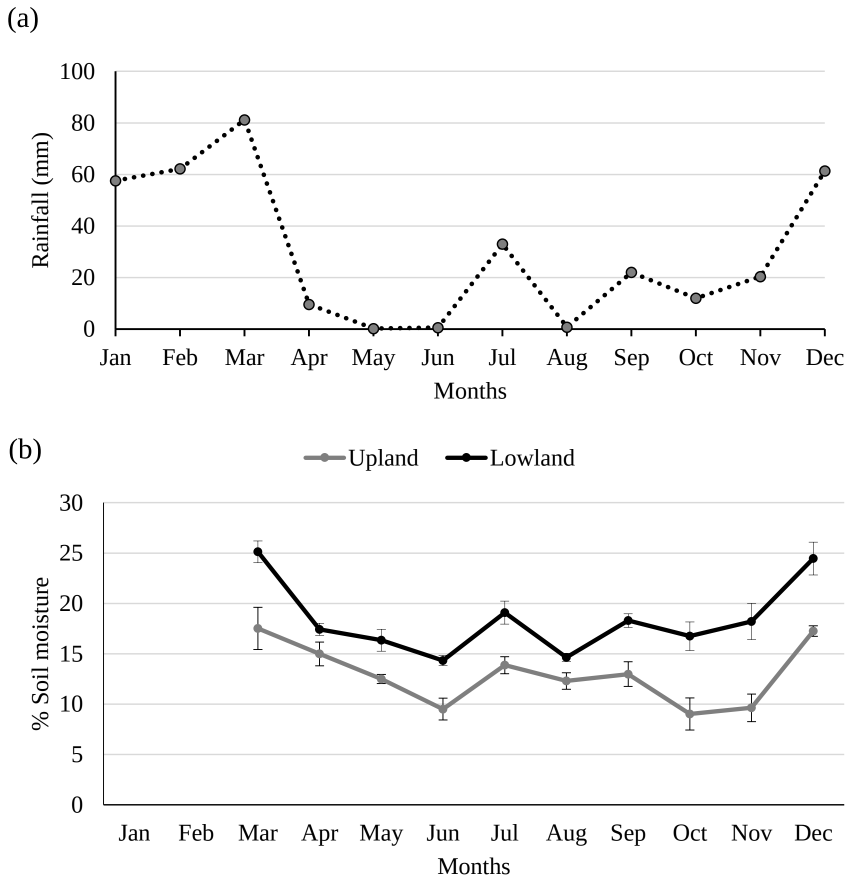

Figure 6.3: (a) Total monthly rainfall and (b) percentage soil moisture in the upland and lowland area at Buffelsdraai Landfill Site.

6.3.3. Effects of drought on sapling mortality at plot level

At plot level, there was significant higher mortality in both the upland and lowland areas between 2015 and 2016 ($P < 0.05$; Figure 6.4): 18.9% and 34.1% overall in the upland and lowland areas, respectively. Species survival rate varied slightlyfor species with more than 18 individuals in 2015. In the upland area, mortality rate ranged from 10% to 20%. In the lowland area, the highest mortality rate was recorded for *Protorhus longifolia* (Bernh.) Engl. (Anacrdiaceae) (42.9%) and *Syzygium cordatum* Hochst. ex C. Krauss (Myrtaceae) (52.5%) (Table 6.1).

6.3.4. Effects of drought on the survival and growth of the four dominant tree species planted

Some of the tagged individuals of the four dominant species suffered mortality in the 2016 sampling season. In the upland area, out of 30 individuals of each of the species tagged in 2015, 18 *B. micrantha*, 24 *M. grandis*, 26 *V. natalitia* and 27 *E. lysistemon* individuals survived the drought. In the lowland area, 23 *B. micrantha* survived the drought, and no mortality was recorded in the other three species. Growth rates of all four species (based on stem height) in both the upland and lowland areas was lower compared with the maximum growth rates reported in the literature for these species under non-drought conditions (Table 6.2). In the upland area, *B. micrantha*, *E. lysistemon*, *M. grandis* and *V. natalitia* achieved 5.6%, 7.3%, 9% and 10%, respectively, of the maximum annual growth rate under non-drought conditions. In the lowland area, *B. micrantha*, *E. lysistemon*, *M. grandis* and *V. natalitia* achieved 11.5%, 16%, 14% and 25%, respectively, of the maximum annual growth rate under non-drought conditions. Within species comparisons between the upland and lowland areas, showed that species in the lowland area had a significantly higher ($P < 0.05$) growth rate than the species in the upland area. All the four species had the highest root-collar diameter increment in the lowland area, significantly ($P < 0.05$) higher than the upland area, except for *M. grandis*. All the species had the highest stem height increment in the lowland area, significantly ($P < 0.05$) taller than the upland area. *Bridelia micrantha* and *E. lysistemon* had the highest canopy width increment in the lowland area, significantly ($P < 0.05$) greater than the upland area. *Millettia grandis* had the highest canopy width increment in the upland area, significantly ($P < 0.05$) higher than the lowland area. *Vachellia natalitia* canopy width increment between the upland and lowland areas was statistically comparable (Figure 6.5a-c).

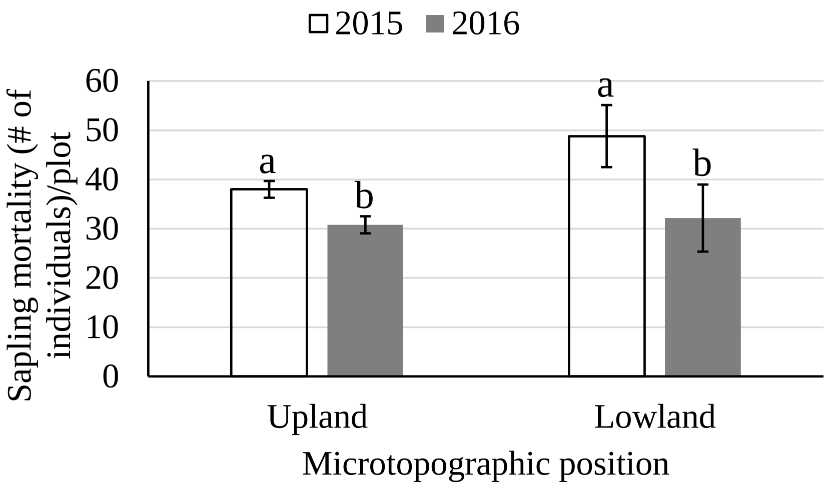

Figure 6.4: Tree mortality rate 13 months post-planting in the upland and lowland areas at Buffelsdraai Landfill Site. Different letters depict significant difference within each microtopographic position (upland and lowland areas) at $P < 0.05$.

Table 6.1: Species abundance and mortality (%) due to drought 13 months post-planting in the upland and lowland areas at Buffelsdraai Landfill Site.

Tree species	Upland			Lowland		
	Total 2015	Total 2016	% mortality	Total 2015	Total 2016	% mortality
Albizia adianthifolia (Schumach.) W.Wight	1	1	0	0	0	0
Baphia racemosa (Hochst.) Baker	0	0	0	1	0	100
Bauhinia tomentosa L.	28	23	17.9	0	0	0
Brachylaena discolor DC.	9	6	33.3	0	0	0
Bridelia micrantha (Hochst.) Baill.	8	6	25	53	44	17
Clausena anisata (Wild.) Hook.f. ex Benth	0	0	0	1	0	100
Clerodendrum glabrum E.Mey.	2	2	0	0	0	0
Combretum molle R.Br. ex G.Don	7	6	14.3	0	0	0
Croton sylvaticus Hochst.	1	1	0	0	0	0
Dalbergia obovata E.Mey.	8	6	25	12	12	0
Ekebergia capensis Sparrm.	1	0	100	0	0	0
Ekebergia pterophylla (C.DC.) Hofmeyr	3	2	33.3	0	0	0
Erythrina caffra Thunb.	3	1	33.3	0	0	0
Erythrina lysistemon Hutch.	16	13	18.8	0	0	0
Euphorbia tirucalli L.	6	6	0	0	0	0
Ficus natalensis Hochst.	0	0	0	1	1	0
Ficus sur Forssk.	0	0	0	53	44	17
Harpephyllum caffrum Bernh.	10	9	10	7	4	42.9
Heteropyxis natalensis Harv.	5	4	20	2	1	50
Millettia grandis (E.Mey.) Skeels	20	18	10	1	1	0
Phoenix reclinata Jacq.	1	1	0	2	2	0
Protorhus longifolia (Bernh.) Engl.	0	0	0	28	16	42.9
Rauvolfia caffra Sond.	1	0	100	14	8	57.1
Searsia lucida (L.) F.A.Barkley	4	3	25	0	0	0
Searsia rehmanniana (Engl.) Moffet	1	1	0	0	0	0

Strelitzia nicolai Regel & Körn.	1	1	0	2	2	0
Syzygium cordatum Hochst. ex C.Krauss	6	5	16.7	40	19	52.5
Tabernaemontana ventricosa Hochst. ex A.DC.	0	0	0	16	10	37.5
Trema orientalis (L.) Blume	1	1	0	0	0	0
Trichilia dregeana Vahl	20	16	20	18	3	16.7
Vachellia natalitia E.Mey.	49	40	18.4	1	1	0
Vachellia xanthophloea Benth. P.J.H. Hurter	1	1	0	2	2	0
Vangueria infausta Burch.	1	1	0	0	0	0

Table 6.2: Sapling height growth rate (m/year), 13 months post-planting, in the upland and lowland areas.

Species	Annual tree height growth rate (m/year)			References
	Upland area (mean±SD)	Lowland area (mean±SD)	Non-drought conditions	
Bridelia micrantha	0.11±0.05	0.23±0.10	Up to 2m	Johnson and Johnson (2002); online databases (Plant Resources of Tropical Africa; PlantZAfrica)
Erythrina lysistemon	0.11±0.06	0.24±0.09	Up to 1.5m	Johnson and Johnson (2002)
Millettia grandis	0.09±0.03	0.14±0.06	Up to 0.8-1m	Johnson and Johnson (2002); online databases (Plant Resources of Tropical Africa; PlantZAfrica)
Vachellia natalitia	0.10±0.03	0.25±0.08	Up to 1m	Johnson and Johnson (2002); online database (PlantZAfrica)

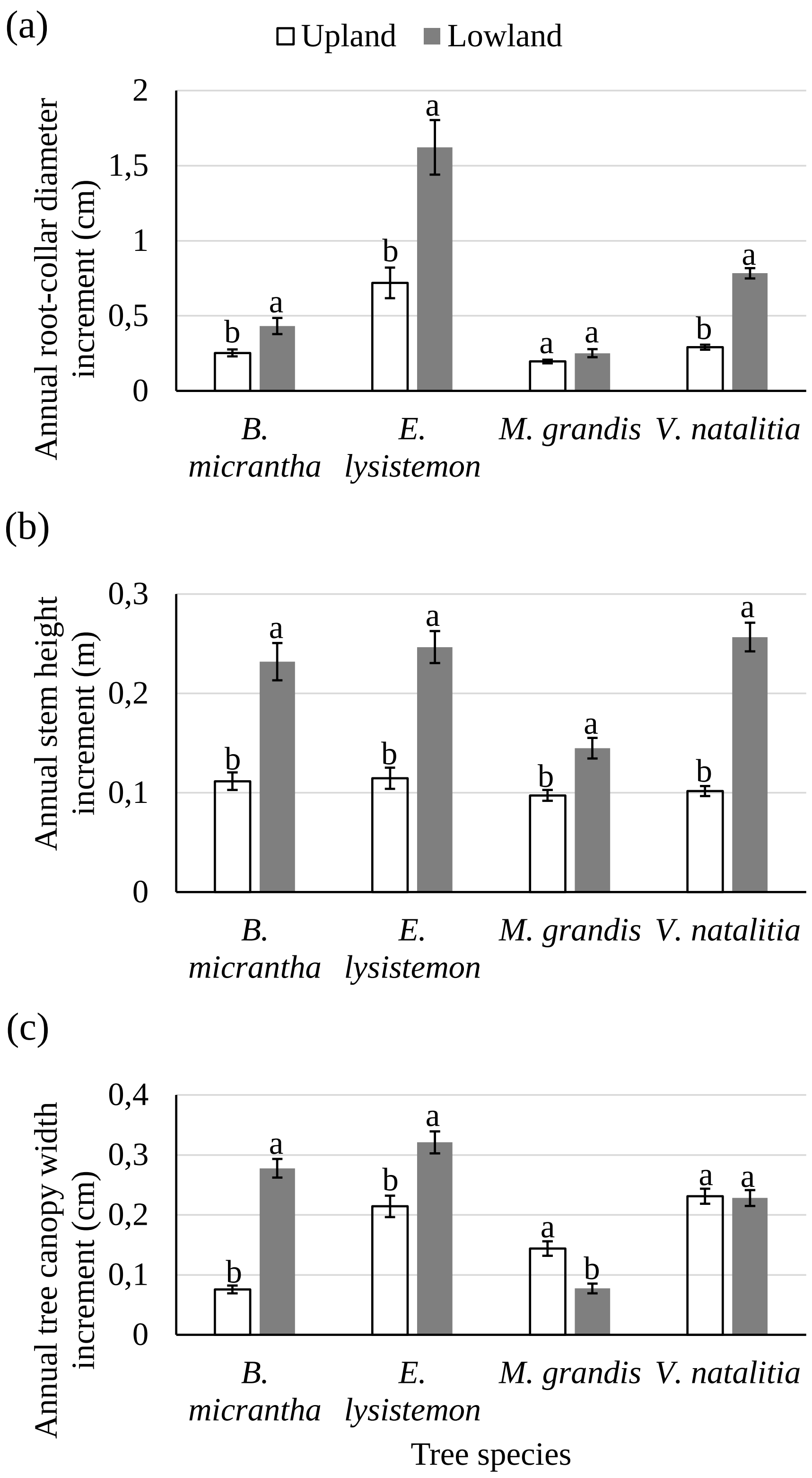

Figure 6.5: Drought effect on (a) annual root-collar diameter increment, (b) annual stem height increment and (c) annual canopy width increment (mean±SE) in the lowland and upland areas, 13 months post-planting. Different letters depict significant difference within each species, between the upland and lowland areas at $P < 0.05$.

6.4. Discussion

The 2015 El Niño induced drought was the most severe on record for Durban since 1921, and the year 2015 was the hottest since 1951 (Botai et al. 2016). However, the overall results showed that it did not hamper reforestation success at the Buffelsdraai Landfill Site entirely. In fact, sapling mortality was relatively low compared to other *in situ* studies, some of which reported up to 100% mortality, following the ENSO event (e.g., Gilbert et al. 2001; Engelbrecht et al. 2005; Edwards and Krockenberger 2006).

Nevertheless, the results suggest that microtopographic positions, in influencing soil moisture availability, modulated the survival and growth performance of the four dominant tree species significantly during the drought event. The four dominant tree species had a higher survival and growth rate in the lowland than in the upland area. However, at plot level, 34.1% and 19.5% mortality was recorded in the lowland and upland areas, respectively. A higher mortality rate in the lowland area despite a higher soil moisture than the upland area is attributed to the dominance of wet habitat adapted species in the lowland area. However, the available moisture was not sufficient to sustain the wet area adapted species. Similar to the current study's findings, Engelbrecht and Kusar (2003) and Poorter and Markesteijn (2008) also reported a higher survival rate in the upland area. They found that survival rate of species adapted to dry habitats was higher than that of species adapted to wet habitats, since the former are more drought adapted (i.e. are able to survive at relatively lower soil moisture contents).

Drought induced mortality differs considerably among species, and this is attributed to plant species-specific traits (Poorter and Markesteijn 2008; Lohbeck et al. 2015; Maréchaux et al. 2015). In our study, at plot level, species varied greatly in their response to drought. In both the upland and lowland areas, species exhibited different morphological responses to drought. The responses varied from leaf wilting and/or shedding, stem wilting a few weeks post-planting, followed by stem die-back two to six months post-planting. Leaf shedding under severe drought conditions is one of the adaptation strategies exhibited by deciduous species (Engelbrecht and Kursar 2005; Slot and Poorter 2007; Poorter and Markesteijn 2008). Leaf shedding exhibited by some species in our study may have helped the plants maintain non-lethal water potentials in meristems and roots (see Engelbrecht and Kursar 2005). In our study, species such as *P. longifolia* resprouted while *S. cordatum*, *Tabernaemontana ventricosa* Hochst. ex A.DC. (Apocynaceae) and *Trichilia dregeana* Vahl (Meliaceae) coppiced following rain events in December 2015 and January 2016. Species that were able to resprout and/or coppice after the drought event in our study may be also therefore be more favourable candidates for reforestation in areas prone to drought, since they contribute to forest ecosystem resilience (Vallejo and Alloza 1998; Vallejo et al. 2012).

A higher survival rate of dry habitat adapted *Bauhinia tomentosa* L. (Fabaceae), *M. grandis*, *T. dregeana*, and *V. natalitia* (van Wyk and van Wyk 2013), in the upland area, and wet habitat adapted *B. micrantha* (Tudor-Owen and Wyatt 1991) and *Ficus sur* Forssk. (Moraceae) (van Wyk and van Wyk 2013) in the lowland area, showed that species survival under non-drought conditions may also dependent on microtopographic positions. Although some species with less than 18 individuals had lower or no mortality rate in both the upland and lowland areas (e.g., *Dalbergia obovata* E.Mey. (Fabaceae), *Euphorbia tirucalli* L. (Euphorbiaceae), and *Harpephyllum caffrum* Bernh. (Ancardiaceae)), it is difficult to establish if this was random or true effects (e.g., Edwards and Krockenberger 2006), and further investigation of their performance under drought conditions is recommended.

Drought had a significant effect on the development of the four monitored species in both the upland and lowland areas. However, the influence of microtopographic position was evident in the lowland area where species survival and growth rates were greater than the upland area. Mortality rate among the four most dominant planted tree species differed greatlywithin microtoporaphic position; for example, the wet habitat adapted *B. micrantha* suffered 40% mortality in the upland area (vs. 23.3% in the lowland area. dry area adapted *E. lysistemon* and *V. natalitia* had a lower mortality rate in the upland area (10% and 13.3%, respectively) and no mortality in the lowland area. The lower survival rate of *B. micrantha* in the upland area suggest that it may be a more suitable pioneer species in wet habitats. Although *B. micrantha* is a wet habitat-adapted species, it was planted in the upland area because it was one of the most abundant species in supplied to the nursery by Tree-preneurs (local community members who grow seedlin), since it is relatively easy to propagate and has a rapid growth (Douwes et al. 2015). However, this species stem height growth did appear to be compromised under drought conditions when compared to data available for the species under non-drought conditions. Our findings are concurrent with other studies that reported reduced growth in saplings under drought conditions compared to those under non-drought conditions (Ito et al. 2000; Poorter and Hayashida-Oliver 2000; Engelbrecht and Kusar 2003). Higher survival and growth rates than *B. micrantha* exhibited by *E. lysistemon* and *V. natalitia* in both the upland and lowland areas make them ideal pioneer species for reforestation initiatives in drought prone areas. *Millettia grandis* demonstrated its drought tolerance, but it is not an ideal pioneer species for drought prone areas due to its slow growth.

In our study, it is evident that microtopographic positions and environmental variables played a critical role in tree species survival and growth performance. Some studies have showed a strong correlation between microtopographic position and environmental variables such as soil texture, soil depth and soil moisture (Grell et al. 2005; Simmons et al. 2012; Li et al. 2014). Soils in the upland areas are often shallower, limiting root space (Li et al. 2014), and coarse as a result of lower clay content (Simmons et al. 2012), while soils in the lowland areas have high silt and clay content (Grell et al. 2005). In our study, the soil in the upland area was shallower (20-40 cm) while the soil in the lowland area was deeper (60-110 cm) (EThekwini Municipality 2014). Therefore, it is likely that root growth was promoted saplings in the lowland area which enabled them to access moisture from deeper soil. This is supported by Vallejo et al. (2012) who reported that species growing in deeper soils (>40 cm) can survive drought through quicker development of roots to reach deeper moist soil layers. Padilla and Pugnaire (2007) also recorded a positive relationship between soil depth, soil moisture availability and sapling survival. Furthermore, after rains, lowland areas remain inundated for a longer period of time than upland areas. Additionally, upland areas are often drier due to higher surface runoff, higher interflow rates and shallower soil. As a result, tree growth may be lower compared to lowland areas (Li et al. 2014). The measurement of soil depth was beyond the scope of the present study but the results suggest that species found in lowland area displayed the potential to grow faster than the species growing in the upland area.

Once the saplings have been intoduced into the field, there is little or nothing that managers of reforestation initiatives can do to reduce the effects of drought. Irrigation of saplings is nearly impossible and non-existent at most reforestation sites (Lukac et al. 2011). Therefore, knowledge of species-specific habitat preference (Lukac et al. 2011; Omary 2011; Li et al. 2014) and/or preparing species for the predicted climatic conditions, are the only viable options to ensure optimal survival (Lukac et al. 2011). At the Buffelsdraai Landfill Site, silvicultural actions (sapling hardening and knowledge on species-specific habitat requirements) were employed to enhance species resilience to drought, and to guide the microtopographic placement of species. Species hardening techniques practiced in the production of saplings at the Buffelsdraai Landfill Site may have also contributed to higher species survival rate during the drought. Other studies also report the value of using hardening techniques to increase species resilience to harsh field conditions (e.g., drought) (Vallejo et al. 2012; Palma and Laurance 2015). Furthermore, Chirino et al. (2009) found that saplings that were exposed to drought stress prior to transplanting had a higher survival rate compared to the well-watered saplings. At the Buffelsdraai Landfill Site, knowledge on the species-specific habitat requirements was obtained from the species composition survey conducted in the remnant natural forest surrounding the site under reforestation (see Mugwedi et al. 2017). Furthermore, a soil survey of the reforestation site was also undertaken to identify dry and wet habitats (EThekwini Municipality 2014).

Some studies have reported an increase in saplings susceptibility to herbivory and pathogen attack during drought conditions, thus increasing sapling mortality (e.g., Engelbrecht and Kusar 2003; Zhao and Running 2010; Reichstein et al. 2013). Griscom et al. (2005) also reported a higher tree mortality caused by cattle trampling. These impacts can have critical implications on the success of reforestation initiatives (Palma and Laurance 2015). However, in our study, no signs of herbivory and pathogens were observed on the drought stressed saplings. Furthermore, at the study site, there are cattle from the local communities that graze on the site, but only one *E. lysistemon* sapling died due to cattle trampling, in the upland area. Thus herbivory, pathogens and cattle presence did not a major influence on sapling mortality.

6.5. Conclusion

This study showed that severe drought associated with the El Niño event caused moderate tree mortality and reduced tree species growth rates. However, the impact posed little threat to the Buffelsdraai Landfill Site Community Reforestation Project success. This is attributed to knowledge of species-specific habitat requirements and restoration site microtopographic positions and their conditions, and sapling hardening techniques employed at the nursery. With a predicted increase in frequency, duration and severity of droughts associated with the El Niño events, the above mentioned mechanisms should be adopted to increase reforestation species adaptation to severe droughts. Furthermore, there is a need to carry out long-term studies in order to understand how global climate change will affect secondary species succession in the reforested sites in subtropical areas. Future studies should assess the effect of drought focussing on all species used in the reforestation initiatives. This would provide insights into how climate change would affect species composition, diversity, and ecosystem functioning and resilience.

6.6. References

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6.7. Supplementary materials

Table S6.1: The six most abundant tree species across the three (time interval based) restoration habitat with both microtopographical areas at Buffelsdraai Landfill Site.

Restoration habitat (planting time)	Microtopographic position	Tree species
November 2014- February 2015	Upland	*Vachellia natalitia* *Erythrina lysistemon* *Bridelia micrantha* *Syzygium cordatum* *Millettia grandis* *Trichilia dregeana*
	Lowland	*Tabernaemontana ventricosa* *Bridelia micrantha* *Syzygium cordatum* *Millettia grandis* *Erythrina lysistemon* *Vachellia natalitia*
November 2011-February 2012	Upland	*Erythrina lysistemon* *Millettia grandis* *Bridelia micrantha* *Protorhus longifolia* *Vachellia natalitia* *Syzygium cordatum*
	Lowland	*Senegalia caffra* *Vachellia natalitia* *Bridelia micrantha* *Erythrina lysistemon* *Millettia grandis* *Syzygium cordatum*
November 2009-February 2010	Upland	*Vachellia natalitia* *Erythrina lysistemon* *Bridelia micrantha* *Millettia grandis* *Syzygium cordatum* *Senegalia caffra*
	Lowland	*Vachellia natalitia* *Senegalia caffra* *Erythrina lysistemon* *Bridelia micrantha* *Millettia grandis* *Brachylaena discolor*

The species are arranged in the order of abundance, the first species in each set being the most abundant and last species being the least abundant.

CHAPTER 7: GENERAL DISCUSSION

7.1. Introduction

Millions of hectares of world forests have been lost or degraded over centuries, as a result of increasing human population coupled with an increase in demand for food and land (FAO 2016), thus threatening biodiversity and life-essential ecosystem services for humankind and other organisms on the planet (Blignaut et al. 2007). Some species are already extinct, while the livelihoods of millions of people (especially the rural poor) have been severely affected (Dudley et al. 2005). Forest loss and degradation will be exacerbated by global climate change stressors such as drought, heat, pest and disease outbreaks, and invasive species, leading to more species extinctions, while the rural poor become more vulnerable (Biringer and Hansen 2005). Forest loss is now a global challenge, and the protection and management of remaining forests are no longer sufficient to address this challenge (Dudley et al. 2005), thus making reforestation a global priority (Aronson and Alexander 2013). The value of reforestation in climate change mitigation and adaptation is also starting to get full recognition (Biringer and Hansen 2005; Aronson et al. 2007; Locatelli et al. 2015). This is because reforestation can ameliorate climate change impacts through protection of coastal areas from storm surges (Locatelli et al. 2015), reduce the probability of species extinctions, by providing corridors between forest patches to facilitate the migration of species along the climatic gradients (Biringer and Hansen 2005; Young et al. 2007), and can act as a safety net for the rural poor through provision of different forest products (Locatelli et al. 2015). Governments and non-governmental institutions have now embarked on large-scale forest restoration initiatives (Aronson and Alexander 2013; Oldfield et al. 2013). However, "the key to any successful restoration programme lies in good project design that is based on sound science, a thorough understanding of threats and opportunities, and a strategic and pragmatic suite of interventions chosen to mitigate identified threats while capitalising on key opportunities" (Robinson 2005). This thesis aimed: (1) to unravel confusions caused by the inconsistent use of terminologies describing different reforestation initiatives; (2) to investigate motivations behind recent reforestation initiatives; (3) to demonstrate the use of a restoration decision-making tool, Robust offsetting (RobOff); (4) to investigate the influence of climatic, edaphic and biotic factors on forest restoration; (5) to assess reforestation initiative success; and (6) to assess the effect of drought on reforestation initiative.

7.2. Discussion

7.2.1. Confusions caused by the inconsistent use of terminologies describing different reforestation initiatives

Reforestation initiatives are taking place worldwide and this provides an opportunity for a global learning network around reforestation. Unfortunately, this cross learning can be difficult, because of the inconsistent use of terminologies describing different reforestation initiatives. The aim of this section was to unravel the confusion between restoration scientists, restoration managers, policy makers and other stakeholders caused by the inconsistent use of terminologies describing different reforestation motivations. This was achieved by providing a comprehensive list of terminologies used in literature and showing the reforestation motivations under which these terminologies should be applied to. The results showed that there are 10 most common terminologies used to describe reforestation in literature. We classified them into five groups based on their restoration motivations namely, (1) *Creation* or *Fabrication* / *Reallocation* / *Replacement*, (2) *Ecological engineering*, (3) *Ecological restoration*, (4) *Reclamation* / *Reconstruction* / *Remediation* / *Renewal* or *Redemption*, and (5) *Rehabilitation*. *Creation* or *Fabrication* / *Reallocation* / *Replacement* share a common motive of establishing an ecosystem that has no resemblance to the historic one in terms of structure and function mainly to achieve environmental and socioe-conomic benefits (e.g., Aronson et al. 1993; Stanturf et al. 2014). Ecological engineering's end point is predictable, because it is designed to provide desired services (Aronson et al. 2016). *Ecological restoration*'s motivation is to reinstate the pre-degradation forest conditions in term of biodiversity, ecosystem structure and functions, and ecosystem services (e.g., Bradshaw and Chadwick 1980; Allen et al. 2001). The end point in ecological restoration is unpredictable, because it is regulated by the ecosystem itself (Aronson et al. 2016). *Reclamation* / *Reconstruction* / *Remediation* / *Renewal* or *Redemption* share a common motive of returning land to useful state (e.g., Aronson et al. 1993; Stanturf et al. 2014). This review argued that a consistent use of these terminologies is important to avoid disconnection between restoration scientists, restoration managers (Hart et al. 2015), policymakers, and other stakeholders. Furthermore, the existing terminologies should not be expanded to accommodate new reforestation initiatives such as planting of trees to enhance forest resiliency in the face of uncertain climatic conditions (Hart et al. 2015). This will make it easier to for restoration scientists, restoration managers, policy makers and other stakeholders to develop science-based strategic approaches that will guide and improve reforestation, and provide conceptual, institutional, legal and operational basis for the policies, and for the monitoring and evaluation of the reforestation initiatives (after Chazdon et al. 2016). Furthermore, this would allow the global alignment of reforestation initiatives and the development of global learning network around reforestation.

7.2.2. Motivations behind reforestation initiatives in recent studies (2000-2016)

A comprehensive of literature on recent reforestation initiatives around the globe was undertaken. The first aim of the review was to determine the motivations behind recent reforestation initiatives (2000-2016). This aim was achieved by assessing reforestation goals and benefits, drivers behind forest loss and degradation, reforestation actions employed, species type (native or exotic) and number of species used (single or multiple), and the biogeographic realms (after Olson 2001) under which the studies were conducted. The results showed that over 90% of the recent reforestation initiatives were motivated by the need to achieve multiple goals simultaneously. The goals include regaining climate change resilient and more functional forest ecosystems coupled with provisioning, regulating, supporting, and cultural ecosystem services and economic benefits (e.g., employment and carbon credits).

Agriculture was the main driver behind forest loss and degradation, and this is largely attributed to an increase in human population coupled with an increase in food demand (FAO 2016). Active reforestation action through planting a higher diversity of native tree species was the widely adopted action (in over 80% of the reviewed studies). This is because this action has a significant contribution to biodiversity conservation, ecosystem services supply (e.g., Cunningham et al. 2015a; Locatelli et al. 2015), and a higher more carbon storage (climate change mitigation) than the highly favoured fast growing exotic tree species (Brown et al. 2011; Piotto et al. 2010; Hulvey et al. 2013; Cunningham et al. 2015b).

Restoration scientists and practitioners fully understand the value of restoration, and they will be increasingly called upon to address environmental crises and challenges through large-scale restoration programs (Aronson and Alexander 2013). The implementation of reforestation initiatives in all the biogeographic realms of the world showed that restoration scientists and practitioners are being called upon by government and non-governmental institutions to address environmental crises and challenges created by forest loss and degradation, through reforestation. A major outcome of this review was that, reforestation initiatives should be designed to achieve both ecological and socio-economic goals through planting a higher diversity of native tree species.

7.2.3. The use of restoration decision-making tool in the planning of a complex large-scale restoration programme

Biodiversity and ecosystem services loss has led to the adoption of systematic conservation planning tools to assist in the decision-making of conservation progamme planning (Sarkar et al. 2006). Tools such as Marxan and Zonation employ significant simplifications in objectives, biodiversity features and cost components, space, time, or problem dimension in order to manage the analysis (Schwartz et al. 2017). A recently released publicly available software, RobOff, differs from the other systematic conservation planning tools, because it focuses on 'what to do' rather than 'where' and on uncertain effects that alternative (conservation) actions (e.g., tree planting vs. invasive alien plants clearing) have on different (biodiversity) features (e.g., carbon storage vs. employment creation) in different environment (Pouzols and Moilanen 2013). Therefore, the aim of this study was to demonstrate how RobOff has been used to efficiently allocate resources in a large-scale restoration project with two habitats, multiple restoration alternatives and goals (biodiversity, carbon stock and employment), with a limited budget. Furthermore, most of the restoration planning studies done to guide restoration plans have accounted for either biodiversity features (e.g., species richness), ecosystem services provision (e.g., climate regulation or water purification) or both (e.g., Thomson et al. 2009; Orsi et al. 2011; Wilson et al. 2011; Budiharta et al. 2014; Egoh et al. 2014; Adame et al. 2015; Rappaport et al. 2015). This study is different, because it has also included employment creation, which has been shown to be a one of key factors that determine forest restoration success, especially in developing countries (Saxena et al. 2001; Le et al. 2011; Barr and Sayer 2012). Overall, the results showed when resources are limited or when biodiversity and employment are the main priority, it is more beneficial to restore a degraded land using *biodiversity action* (planting of 80 native tree species to boost tree species richness) than restoring a partially degraded forest which is still functional. Prioritization of carbon stock (through carbon action: planting of 10 native tree species with a higher wood density to achieve a higher carbon stock), instead, did not achieve the same return on investment compared to biodiversity and employment prioritization. These results showed that instead of focusing on planting trees (especially fast growing exotic tree species) to store more carbon, restoration actions geared towards biodiversity conservation which achieved all three restoration benefits might be a better alternative. These findings are supported by the findings from the review chapter which showed that most recent reforestation initiatives are planting a higher diversity of native species to achieve multiple ecological and socio-economic benefits. This study has also shown that consideration of different reforestation options and the inclusion of RobOff in the original planning by the municipality of the Buffelsdraai Landfill Site Community Reforestation Project (BLSCRP) would have shown how resources could have been allocated to alternative restoration actions that achieve a higher biodiversity, carbon stock and employment opportunities. A major outcome this study was that, RobOff is a coherent decision-making support tool that can be used for the optimal allocation of resources in large-scale reforestation projects to help maximise biodiversity and ecosystem services conservation, enhance carbon stock and provide employment benefits.

7.2.4. Growth performance of the four most dominant planted tree species at the Buffelsdraai Landfill Site

This study was motivated by a lack of knowledge on subtropical native tree species response to different microtopographic positions, and soil chemical and physical properties within a severely degraded habitat (Lu et al. 2017). Knowledge on pioneer native tree species performance is critical to inform the selection of fast growing species to promote rapid forest establishment in severely degraded habitats. Planting of fast growing native tree species is now a widely adopted approach in reforestation initiatives (Elliott et al. 2003; Wishnie et al. 2007; Lu et al. 2017). Using a chronosequence of three habitats under restoration (0-,3- and 5-year-old), and contrasting different microtopographic position within each habitat, this study assessed root-collar diameter, stem height and canopy width of four most dominant planted tree species (*Bridelia micrantha*, *Erythrina lysistemon*, *Millettia grandis* and *Vachellia natalitia*) at the Buffelsdraai Landfill Site. Results chapter showed that higher soil moisture coupled with higher nutrient levels in the lowland area (wet zone) promoted rapid establishment of *B. micrantha*, *E. lysistemon* and *V. natalitia*. *Erythrina lysistemon* and *V. natalitia* were found to be more suitable for forest restoration in both the upland (dry zone) and lowland areas, and *B. micrantha* in lowland area. *Millettia grandis* did not perform well in both upland and lowland areas, making it unsuitable to be selected as pioneer forest restoration species. These results indicate that different species are better candidates for different microtopographic positions within a site. Therefore, there is a need for more data on subtropical forest tree species performance under different microtopographic positions to inform the selection of species that promote rapid forest establishment and restoration goals achievement.

7.2.5. Assessment of the Buffelsdraai Landfill Site reforestation success

Assessment of restoration success is a key requirement in reforestation initiatives, because it helps to identify threats that can lead to restoration failure and guide the necessary management interventions needed to achieve the target goals (Ruiz-Jaén and Aide 2005; Wortley et al. 2013). Using a chronosequence (0-year-old, 3-year-old and 5-year-old) of land under reforestation (at the Buffelsdraai Landfill Site), this study assessed early progress of reforestation in terms of tree species diversity, vegetation structure and ecological processes (e.g., pollination) needed for creation of a self-sustaining, functional forest ecosystem that is resilient to disturbance, and the presence of IAPs. The overall results showed that the habitats under reforestation are progressing towards the reference habitat (remnant forest patches) in terms of vegetation structure (e.g., tree height and canopy cover) and native tree recruitment while the pollination and seed dispersal traits composition were similar to the reference habitat. The key identified reforestation success threats were low tree density and species richness, and an increase in IAP cover with an increase in restoration age. Enrichment planting to boost tree species density and richness should be implemented, but the focus should also be on the inclusion of species with missing pollination and seed dispersal categories in order to create a resilient and stable forest system in the long-term. An effective IAP management strategy should be investigated as a matter of priority, to promote more rapid habitat restoration while reducing site maintenance costs.

7.2.6. Impact of climate change induced drought on on reforestation trees at Buffelsdraai Landfill Site

Information on native tree sapling response to natural drought remains scant in the subtropical forests, especially under field conditions. The serendipitous drought occurrence associated with the El Niño event that occurred in South Africa in 2015 motivated this study. This chapter aimed to assess the effect of natural drought on tree sapling mortality at a plot level and to assess its effect on the growth performance of the four most dominant planted tree species, namely, *Bridelia micrantha*, *Erythrina lysistemon*, *Millettia grandis* and *Vachellia natalitia*. Species growth performance was assessed in terms of root-collar diameter (RCD), stem height and canopy width. The assessments were carried out just after planting (February 2015) and 13 months post-planting (March 2016) in the upland (dry) and lowland (wet) areas of the habitat planted in 2014-2015 (between November and February). Saplings in the lowland area suffered highest mortality (34.1%) than the saplings in the upland area (18.9%), and this was largely attributed to dry area adapted species survival under limited moisture (Engelbrecht and Kusar 2003; Poorter and Markesteijn 2008). At plot level, a higher survival rate of *Bauhinia tomentosa* L. (Fabaceae), *E. lysistemon M. grandis*, *T. dregeana,* and *V. natalitia* in the upland area and higher survival rate of *B. Micrantha* and *Ficus sur* Hochst. in the lowland area showed that these species are good candidates for restoring dry and drought prone areas. Results on species growth performance showed that *E. lysistemon* and *V. natalitia* performed well in both the upland and lowland areas making them good pioneer reforestation species for both dry and wet habitats, and *B. micrantha* for wet habitats only. However, drought significantly reduced the species' RCD, height and canopy width growth in the upland area compared to lowland area. Plant species traits, knowledge of species specific habitat requirements and species hardening techniques (the use of local soils and reduced irrigation in the nursery) had a positive influence on species survival and growth rate. The results of this chapter showed that more frequent and more intense droughts that have been predicted to occur in southern Africa will have negative impacts on the restoration of a subtropical forest. However, assessment of species performance under drought could inform the selection of fast growing drought adapted species that promote rapid forest establishment.

7.3. Conclusion

The findings from this study underscore the importance of understanding the drivers of forest loss and degradation, planning and implementation of large-scale reforestation initiatives in order to achieve biodiversity conservation, climate change mitigation and adaptation, and sustained human well-being. Although reforestation is no longer an emerging field, there are still challenges that can compromise the success of large-scale reforestation initiatives. This study enabled me to gain a broader understanding of the global native forest loss and degradation, drivers of forest loss and degradation, reforestation initiatives planning, implementation and assessment.

With all the conservation efforts in place, loss and degradation of native forest is still an ongoing challenge especially in developing countries as a result of anthropogenic disturbances (e.g., agriculture, mining and infrastructure development). Conservation efforts are largely perceived as obstacles to various anthropogenic activities that result in forest loss and degradation, because natural resources are viewed as public goods. At a landscape level where large-scale reforestation initiatives are implemented, there is competition for land use by different stakeholders who opt for land use alternatives that offer higher return on their investment (Droste et al. 2017). Some authors argue that conservation agencies could optimise land use by selling the full ecological and socio-economic benefits of sustainable use of natural resources (Aronson et al. 2010; Wortley et al. 2013). To optimise land use where there are competing interests, stakeholders need to work together (e.g., ecologists, architects, developers, land use planners, engineers, industrialists, local communities and state entities) (Bradshaw 1983; Standish et al. 2013). This will also enable the ecologists to sell the full benefits of sustainable development. The synergistic effect of the stakeholder partnership can lead to a co-design and co-production of a strategic investment of financial, human and social capital to achieve sustainable development that ensure the long-term persistence of the forest ecosystems.

A rapid increase in global population coupled with an increase in demand for food and infrastructure has resulted in competition for land use to achieve various interests (e.g., agriculture, development and mining) (Sayer et al. 2013; Haberl 2015). This is making the allocation and management of land to achieve ecological and socio-economic goals more challenging. The use of conservation planning tools is now a widely adopted approach to guide decisions about selection of conservation actions that achieve ecological and socio-economic goals in areas with competing interests (Sarkar et al. 2006). Decision-making tool suchs as RobOff are capable of balancing stakeholders' competing interests (e.g., conservation, job creation and development) that require careful considerations due to trade-offs. The use of decision support conservation tools should be a standard practice for conservation researchers and practitioners (Schwart et al. 2017). However, given the invaluable support that the conservation tools offer in decision-making, these tools require a commitment in financial and human capital investment. For example, the team that developed RobOff was invited to Durban, South Africa, to conduct one week training on the use of RobOff.

The drivers behind forest loss and degradation and the level of disturbance largely determine the intended reforestation initiative motivation. For example, mining activities cause intensive disturbance of forest ecosystem often leaving heavy metals on the soil surface. This level of disturbance often lead to *Reclamation* / *Reconstruction* / *Remediation* / *Renewal* or *Redemption* initiative which aims to return land to useful state, either to achieve either ecological or socio-economic benefits (Bradshaw 1996; SER 2004; Stanturf et al. 2014). Understanding of the degrading factor and site conditions (e.g., microtopographic positions, soil nutrients and moisture) informs the choice of species that can withstand the reforestation site conditions and promote rapid forest establishment. However, lack of knowledge on native tree species response to sites subjected to different degrading factors is still a challenge. As a result, exotic tree species (e.g., pines) that can withstand harsh conditions are often preferred in reforestation initiatives despite their ecological and socio-economic consequences (e.g., water capture, and reduced biodiversity and livelihoods). It was interesting to note that most recent reforestation initiatives around the world including developing countries are now planting a higher diversity of native tree species. Inexpensive techniques such as sapling hardening (e.g., limited irrigation in the nursery) and knowledge of species specific habitat requirements can be used to prepare native tree species to withstand climate change induced droughts. In nutrient limited soils, planting of a higher proportion on nitrogen-fixing tree species can benefit non-nitrogen fixing species through root connections or organic forms of nitrogen from the litter layer (Siddique et al. 2008; Hoogmoed et al. 2014b).

Reforestation initiatives around the world are now aimed at restoring high species diverse native forests. Diverse native forests are more functional, resilient to climate change, and offer multiple benefits such as climate change mitigation and adaptation, wildlife habitat and food, improved human well-being and economies. However, it can be difficult to plant a higher diversity of native tree species due to the following reasons; (1) in some areas, the remaining forest patches are heavily degraded with little tree species diversity; and (2) in community-based reforestation initiatives, community members tend to collect seeds of species that are easy to propagate, collect seeds of species that would benefit them (e.g., medicinal plants) and collect seeds from the nearby forests with low species richness. If funding is not limted, planting of a higher tree diversity can be achieved through contracting local communities and professional seed collectors. In highly degraded forests, tree species diversity could be enhanced by sourcing seeds from less disturbed forests (often found in protected areas). Incentives should be used to encourage local community Tree-preneurs to maximise tree species richness.

Planting of a higher diversity of native tree species (59 species) at Buffelsdraai Landfill Site has started to achieve the eThekwini Municipality's expected outcomes. The dominance of faster growing tree species such as *B. micrantha, E. lysistemon* and *V. natalitia* are contributing to rapid offset of the carbon footprint left by the 2010 FIFA World Cup event. The habitats under reforestation have now connected forest patches that were fragmented by sugarcane farming thus allowing species dispersal and gene-flow, and increasing their habitat. The trees in older habitats under reforestation (3- and 5-year-old habitats) are taller with denser canopies, flowering and fruiting thus attracting over 145 bird species. The trees have created microclimatic conditions that are promoting the establishment of native tree species dispersed by frugivorous animals. Other studies done in the same area under reforestation recorded an increase in invertebrate (Govender 2017) and small mammal (Lazarus 2017) species richness compared to the sites under continuous sugarcane farming. Higher tree species diversity at Buffelsdraai Landfill Site is also likely to increase forest resilience to climate change induced disturbances such as pest and disease outbreaks, fire and drought.

The BLSCRP has a significant contribution to local socio-economic benefits. The project has employed impoverished members of the local communities thus diversifying their livelihoods and enhancing their adaptive capacity to climate change. Apart from job creation, the habitats under reforestation will also contribute to human well-being through the supply provisioning ecosystem services such as wild fruits and medicinal plants. For example, bark harvesting for medicine was observed on *Sclerocarrya birea* sub spp. *caffra* (Sond.) Kolwaro (Anacardiaceae) in the fragmented forest. Furthermore, some of the planted trees such as *Harpephyllum caffrum* Bernh. (Anacardiaceae), *Schotia brachypetala* Sond. (Favaceae), *Trema orientalis* (L.) Blume (Celtidaceae) and *Apodytes dimidata* E. mey. (Icacinaceae) are highly valued medicinal plants in the Durban area (Coopoosamy and Naidoo 2012). The BLSCRP also has a significant contribution to cultural ecosystem service by serving as an outdoor classroom for primary and secondary schools in the area. The BLSCRP is also a centre of learning visited by a host on national and international government and non-governmental agencies to learn about forest restoration. For example, the site was visitied by the delegates of the XIV World Forestry Congress (7-11 September), Durban, South Africa. This cultural service is quite significant, because the young generation grow up knowing the significance of biodiversity conservation and reforestation of degraded forest ecosystems. Although other ecosystem services such as water purification have not been monitored yet, it is likely that the forest is fulfilling this role or will fulfil it once the forest is well established.

The significance of forests in biodiversity conservation, climate change mitigation and adaptation, ecosystem services provision and job creation is now evident around the globe. It is high time that state and private agencies channel more investment into forest conservation and reforestation initiatives. If we fail to act now, the rapid growing human population and economies will deplete the natural capital that we all depend upon for our survival and well-being thus resulting in the historic collapse of society and economy. Recent reforestation initiatives should aim to achieve ecological and socio-economic outcomes simultaneously, because this is the main focus of many state and private funding agencies (Maron and Cockfield 2008; Pendleton 2010). Care should be taken so that the reforestation initiatives are not implemented at the expense of local communities' livelihoods, but enhance their livelihoods through job creation and access to forest resources that do not compromise the initiative success. For example, most funding agencies are international institutions that do not understand the socio-ecological realities of a local community (Mitchell 2007). As a result, they may impose strict regulations that prohibit communities' access to certain forest resources (e.g., medicinal plants and firewood) that support their livelihoods. This might lead to conflicts between the project implementing agency (usually the local government) and the local communities, thus resulting in reforestation initiative failure (e.g., Saxena et al. 2001; Le et al. 2012; Orsi et al. 2011; Barr and Sayer 2012).

7.4. Recommendations

The ecological and socio-economic benefits of reforestation initiatives are now evident. As a result, many reforestation initiatives are being implemented around the globe. Based on the findings from this study, the following recommendations are given;

- Recent reforestation initiatives should be designed to achieve multiple ecological and socio-economic benefits.
- Active restoration through planting a higher diversity of native tree species should be practiced in moderately and severely degraded forest habitats.
- In agricultural landscapes with forest fragments, farmers should be encouraged (e.g., through incentives) to plant native tree species that provide both on-farm and off-farm benefits (e.g., biodiversity, carbon sequestration, corridors between fragmented forests, soil erosion control and soil nutrient cycling, fruits and medicine) to reduce or avoid land abandonment and clearing of more forest.
- Testing a wide range of native tree species in severely degraded sites with harsh conditions should be prioritized to avoid the use of exotic tree species that are associated with negative environmental and socio-economic impacts.
- With the rise in number of reforestation initiatives around the world, the consistent use of terminologies describing different reforestation motivations is needed to align

restoration initiatives and to aid the development of global learning network around reforestation.

- Robust Offsetting should be incorporated in the planning of a large-scale reforestation initiative planning to aid the efficient allocation of limited resources to actions and habitats that achieve a higher rate of return on investment.
- More trials on the subtropical native tree species performance under different microtopographic position and or natural drought conditions in severely degraded sites are needed to aid the selection of species that promote rapid forest establishment.
- Tree species that promote plant-animal interaction (e.g., pollination and seed dispersal) should be prioritized in reforestation initiatives in order to restore a more functional and resilient forest ecosystem in the long-term.
- Reforestation success assessment should be done in the early stages of the project using multiple indicators of vegetation structure, species diversity, ecological processes and invasive alien plants infestation to guide the necessary management interventions.
- The use of biological control agents should be implemented as an alternative IAP management option in habitats under reforestation, especially in areas where mechanical control is limited by financial constraints.

7.5. References

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