Spatiotemporal Dynamics of Afrotemperate Forests in KwaZulu-Natal, South Africa

MSc Dissertation by Research

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Declaration

I declare that this dissertation is my own, unaided work, and that all the sources that I have used or referenced have been indicated and acknowledged by means of complete references. It is being submitted for the Degree of Master of Science to the Faculty of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any other degree or examination in any other university.

L.Nyembezi

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Abstract

This study investigated Afrotemperate forests' extent and composition from 1940 to 2022 in KwaZulu-Natal, South Africa. Thirty patches were examined (10 small patches (~one hectare), 10 medium patches (~ between 1 and 1000 hectares) and 10 large patches (~1000 hectares)) over three time periods (1940s, 1980s and 2020s). Aerial photographs, with their high spatial resolution and long temporal extent, were used to estimate forest extent, composition (indigenous versus non-indigenous) and the surrounding land cover. Georeferencing and digitising of aerial photographs was done in QGIS. Fire extent and frequency from 1990-2024 was calculated seasonally using the difference in Normalised Burn Ratio (dNBR) on Landsat and Sentinel images in Google Earth Engine and ArcGIS Pro. Generalised Linear Mixed Models (GLMMs) were run to determine the relationship between forest extent, surrounding land cover and fire. The results of this study indicated that changes in forest extent were patch size specific, with smaller and larger patches increasing in extent and medium patches decreasing in extent. Smaller forests' expansion in extent was due to the invasion of non-indigenous vegetation. The invasion of non-indigenous vegetation was aided by the high fire extent and frequency that the small forests experienced. Medium sized patches had the highest amount of non-indigenous vegetation in their surrounding matrix which can contribute to the reduction of available water and the degradation of indigenous forests. Larger forest patches are able to buffer the effects of changes in fire and land cover patterns which can aid in indigenous forest expansion. The surrounding matrix of the larger forests had the highest amount of abandoned agricultural land which can promote forest expansion and bush encroachment. Fire extent was highest for the smaller forests possibly due to the high amounts of non-indigenous vegetation found within them. Fire frequency was highest in medium forests' matrices possibly due to the medium forests having the highest levels of non-indigenous vegetation in the surrounding matrix and the lowest reduction in herbaceous vegetation in the surrounding matrix. Statistically fire extent and land cover were found to impact forest extent independently. The results of this study indicate that small and medium sized forests are at a higher risk of degradation in comparison to larger forests. Therefore the results of this study can be used to inform decisions around where to focus prioritisation efforts for forest conservation.

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Rationale

Forests found in southern African can be defined as ecosystems with wood formations with one or more clearly developed tree layers and overlapping of crowns of trees occurring in at least one of the tree layers (Mucina et al., 2022). The main forests found in southern Africa are Subtropical, Tropical Dry and Afrotemperate forests (Mucina et al., 2022). Forests in this study were considered as any continuous tree cover (indigenous and/ or non-indigenous) and the composition of the forests were later determined. Forests provide various cultural, regulating, supporting and provisioning ecosystem services such as terrestrial biodiversity, soil and water protection, fibre, fuel, carbon sequestration, tourism and traditional medicine (Gonzalez et al., 2005). The loss of forests (deforestation) is a global issue with it being estimated that global forest area has decreased from 1990 to 2020 by 7.1% (Farrokhi et al., 2024).

The causes of forest decline in extent are mostly linked to overgrazing, logging for firewood or timber, repeated fires, diseases or parasites, land cover and land use change and natural disasters (such as cyclones) and increase in human population (Contreras-Hermosilla, 2000; Farrokhi et al., 2024). South African forests, similarly, to the rest of the world, have undergone forest decline in extent for centuries. South African forests have a history of anthropogenic drivers of change such as logging for personal use, food sources, traditional medicine, and firewood (Mucina and Geldenhuys, 2006a). There is also a history of logging for commercial use from 1652 when European settlers arrived in South Africa. Timber from trees were used to make furniture or railway sleepers which were sold by settlers (Mucina and Geldenhuys, 2006). The unsustainable logging and prevention of tree regeneration that occurred from 1652 has caused the conversion of primary forests into secondary forests. This conversion led to the degradation and reduction in the productivity of the forests present today (Adie et al., 2013; Contreras-Hermosilla, 2000). While degradation is a broad term, in this context it refers to a modification or permanent change in the forest's structure, function, species composition or productivity as a result of human actions (Vásquez-Grandón et al., 2018). Selective logging has primarily stopped since the 1940s, however secondary forests are still under further threat from indirect drivers such as reduced water availability from the planting of exotic plantations and the increase in the frequency of uncontrolled fires and direct drivers such as over-grazing and unsustainable harvesting of forest products (Lawes et al., 2004). Both of these factors cause a further reduction in the already small Afrotemperate forest patches (Lawes et al., 2004). While secondary Afrotemperate forests are not pristine and contain less aboveground carbon storage compared to old growth, they still provide an important role for carbon sequestration (Adie et al., 2013; Fisher, 2024). Therefore, the prevention of the further degradation of these forests is important. Investigating the extent and species composition of secondary forests over time will assist to examine whether they are being lost or are regenerating and examining different forest sizes will help to establish whether forest vulnerability varies across different forest sizes. The composition of forests will also help to indicate whether forests are being invaded by non-indigenous vegetation.

Aim and Objectives

Aim:

To establish how small, medium and large Afrotemperate forests in KwaZulu-Natal, South Africa, have changed in extent and composition (indigenous vs non-indigenous tree cover) from 1940 to 2022, and whether fire and land cover influenced these changes.

Objectives

1) Track the extent of 30 Afrotemperate forests over an 80-year period.
   - How did selected forests change in extent between 1940, 1980 and 2022?
   - If the forests had expanded or reduced in extent, what is their current composition (indigenous vs non-indigenous species)?
2) Identify the impact of the surrounding land cover and fire had on forest dynamics and vulnerability.
   - How did the surrounding land cover of the forests of interest change between the 1940s, 1980s and 2020s?
   - What was the fire history around the selected forests annually from 1990 – 2024?
   - Was there a relationship between land cover, fire, and forest extent over time?

Literature Review

Global Forest Trends

Forests play an important role globally in providing cultural, social, economic and environmental benefits and services (Maini, 2007). In 2015, it was estimated that forests cover approximately 31% of the Earth's land cover with 44% of them being tropical forests, 8% subtropical forests, 26% temperate forests and 22% boreal forests (Keenan et al., 2015; Khairoun et al., 2024). Forests are considered resilient ecosystems as they are able to renew themselves after a disturbance (Maini, 2007).

The causes of deforestation are primarily linked to economic development, demographic trends and technology factors (Geist and Lambin, 2001; P. Smith et al., 2010). Globally, forest area decline from 1990 to 2015 was primarily driven by commodity-driven deforestation, forestry, agriculture, wildfires and urban expansion (Curtis et al., 2018; Keenan et al., 2015). The rate of decrease was not consistent throughout the 25 years as the annual rate of net forest loss halved in 2015 compared to 1990. This could indicate that action was taken to reduce forest loss during the 25 years (Keenan et al., 2015). Global forest loss is predicted to further decrease by 0.06% per year until 2030 due to forest conservation efforts globally (d'Annunzio et al., 2015). Forest gains can be due to natural forest regrowth on abandoned agricultural land (Baumann et al., 2011) and afforestation and/ or reforestation (d'Annunzio et al., 2015).

South African Afromontemperate Forests

Forests are the smallest biome found in southern Africa (Eeley et al., 2001). In South Africa specifically, forests are found along the eastern and southern parts of the country in the Great Escarpment, mountain ranges and the coastal lowlands (Mucina and Geldenhuys, 2006a). One type of forest found in South Africa is the Afrotemperate forests which are usually made up of multistrata, evergreen, indigenous trees with a closed canopy (Eeley et al., 1999a; Rutherford et al., 2006). Afrotemperate forests can be characterized as either montane *Podocarpus* forests or mist-belt mixed *Podocarpus* forests (Eeley et al., 1999b; von Maltitz, 2003). These forests are found at high altitudes (ranging between 1000 and 1500m) (Mucina and Geldenhuys, 2006a; Wirminghaus et al., 2001a), have an archipelago distribution and occur as a range of patch sizes from one to 3200 hectares (Berliner, 2009). Their characteristics are due to them having experienced at least one major climatic extinction filtering event in their existence, for example the Last Glacial Maximum event which occurred roughly 18 000 years before present (BP) (Lawes et al., 2000; Lawes, 1990; Eeley et al., 1999a; Mensah et al., 2020). The expansion and contraction of Afrotemperate forests since the Quaternary period has been primarily due to the varying wetness levels in the different climatic extinction filtering events that occurred during this time period (Lawes, 1990). This indicates that forest change is natural for Afrotemperate forests.

The combination of the small size of these forests and the major climatic extinction filtering events that have occurred, has resulted in Afrotemperate forests being species poor (Didham et al., 1998; Laurance et al., 2002; Murcia, 1995; Yahner, 1988). The fauna and flora that are able to survive in these forests are more resilient to change, edge effects and isolation of these forests (Balmford, 1996; Bodin et al., 2006; Danielsen, 1997; Lawes, 1990; Stratford and Robinson, 2005). Some of the fauna that exist in these forest include land hoppers (*Talitriator africana)*, dung beetles (family: Scarabaeidae), crested guineafowls (*Guttera pucherani*), Cape robins (*Cossypha caffra)* and one of South Africa's endemic birds; the Cape parrot *(Poicephalus robustus)* (Kotze and Lawes, 2007; Wirminghaus et al., 2001a). While in general, Afrotemperate forests tend to have low species richness and diversity relative to other types of forests globally, there is variation in the number of species found in different Afrotemperate forests across the continent. For example a study that compared the South African northern mist-belt Afrotemperate forests to the Ethiopian Afrotemperate forests (Kebede et al., 2013; Tadele et al., 2014) found that even though the South African forests were smaller and had limited spatial coverage in comparison to the Ethiopian forests, the South African Afrotemperate forests had a higher floristic diversity (Mensah et al., 2020). This is likely due to Ethiopian forests having taller trees with larger diameters which reduces the number of species able to occupy each forest plot, decreasing the forests' floristic species richness (Mensah et al., 2020). Therefore, while South African Afrotemperate forests may be species poor, they have a higher floristic diversity in comparison to other Afrotemperate forests across the continent.

The dynamics of these small forest patches are impacted by both the habitat interior and the surrounding matrix (Kotze and Lawes, 2007). Historically, Afrotemperate forests were predominately surrounded by grassland or savanna matrices with very sharp forest-grassland boundaries (Lawes et al., 2004). These edges provide protection for the forest's interior against changes in microclimate conditions that usually occurs at the edge of the forest (Kotze and Lawes, 2007). The sharp forest-grassland boundary is due to the fires that take place in the matrix surrounding the forest (Beckett and Bond, 2019). While at a global scale, climate limits the distribution of forests (Eeley et al., 1999a; Taylor and Hamilton, 1994), on a local and regional scale, fire plays an important role in Afrotemperate forest extent (Eeley et al., 1999a; Geldenhuys, 1994; Meadows and Linder, 1993). Similar to other forests, South African forests have been exposed to severe and uncontrolled fires and unregulated harvesting which has contributed to these forest degrading and being considered secondary forests (Curtis et al., 2018; Geldenhuys, 2002). Both of these factors reinforce the small, fragmented and isolated nature of the Afrotemperate forests (Mensah et al., 2020). This also means that with reduced fire and harvesting, Afrotemperate forests should be able to expand in extent (Mensah et al., 2020). This was seen in a study done in Hluhluwe-iMfolozi Park in KwaZulu-Natal where forest extent increased by 77% from 1937 to 2013 (Beckett and Bond, 2019). Most increase in extent occurred in the 1990s and it is likely due to the reduction in extreme fires post 1990 (Beckett and Bond, 2019).

Fire

Forests and fires interact with each other regularly particularly in Africa where forests are found in grassland or savanna mosaics (Roy, 2003). Due to two-thirds of the world's savannahs being found in Africa, it is considered the 'burnt center" of the world (Roy, 2003). These fires have occurred for centuries and are responsible for shaping the landscape's structure, plant community development, land cover, and the distribution and biological diversity of grasslands, savannas and forests in the continent (Archibald et al., 2009; Bond et al., 2005; Higgins et al., 2000; Roy, 2003; Williamson et al., 2024). Afrotemperate forests have developed and persisted in environments that have regular fire exposure (Mensah et al., 2020). Fire predominantly occurs in savanna and grassland ecosystems (both of which are fire-adapted ecosystems) as grass acts as the main fuel in these ecosystems (Beckett and Bond, 2019). Fire then burns around Afrotemperate forests (fire-sensitive ecosystems), rarely penetrating within the forest canopy (Biddulph and Kellman, 1998; Hennenberg et al., 2008; Hoffmann et al., 2012a; Kellman and Meave, 1997). The interaction of the fire with grasslands, savannas and forest ecosystems allows for a forest-grassland-savanna mosaic to occur in the landscape (Brando et al., 2014, 2012; Cochrane and Laurance, 2002). Analysis from soil carbon isotope ratios indicates that some Afrotemperate forests were historically grasslands (Gillson, 2015; West et al., 2000) which is evidence that there has been expansions and contractions of both forests and grassland ecosystems driven by fire patterns (Beckett and Bond, 2019). This is further emphasized when the exclusion of fire in these mosaic landscapes leads to the increase in woody biomass from forested ecosystems

(Hoffmann et al., 2012a; Murphy and Bowman, 2012; Warman and Moles, 2009). This cycle is a positive feedback loop as the less fire that occurs in the ecosystem, the more forest expansion occurs (and vice versa) (Beckage and Ellingwood, 2008; Geiger et al., 2011; Staver et al., 2011). This provides evidence that forests and savannas and grasslands are alternative stable states for each other and the state that occurs is primarily driven by fire (Hirota et al., 2011; Scheffer and Carpenter, 2003).

Fire and its spread is influenced by various factors such as weather conditions; the presence of ignition sources; and the amount, type and arrangement of the available fuel (Archibald et al., 2009; Stott, 2000). Fire can be caused by natural (such as lightning) or anthropogenic forces. Lightning is the main cause of fires in remote areas where there is limited contact with human beings (Archibald et al., 2009; Roy, 2003). However, in Africa the main cause of fire ignitions is due to anthropogenic factors such as deforestation activities; land cover conversions; agricultural land clearings; grazing land management; improvement of yields of plants, fruits and other forest products; or cooking, torches and camp fires from human settlements (Archibald et al., 2009; Goldammer, 1988; Sheuyange et al., 2005). The intensity and severity of fires can be linked to weather conditions with very large fires being the result of several hot days with dry winds after a rainy season (Archibald et al., 2009; Bessie and Johnson, 1995; Moritz, 2003). Long term changes in weather conditions caused by climate change can also increase the frequency, severity and intensity of fires (Jones et al., 2024). Understanding the influence of different environmental factors on fire can be complicated as the influence may not be linear. For example, if there is an increase in rainfall, there may be an increase in the amount of available fuel which would increase the spread of fire but there is also an increase in fuel moisture which decreases the spread of fire (Scholes et al., 1996; Spessa et al., 2005). Therefore, it is important to understand the behaviour of fire in each context that it occurs. Using tools that can measure fire patterns in different contexts can assist with this. A study done in Australia in 2002 looked at the deforestation rates and the expansion and contraction of forests and woodland and found that aerial photographs are an important tool to assess vegetation change (changes in vegetation boundaries, tree density, community composition and crown dieback) particularly due to top-down drivers such as fire and grazing and can be more accurate method than using satellite-based data (Fensham and Fairfax, 2002). This indicates that the use of aerial photographs can be useful to determine changes in forest extent.

As in other forest-grassland mosaics, the dynamic between Afrotemperate forests and fire is affected by wind speed, fuel moisture, and the topography within and around Afrotemperate forests (Biddulph and Kellman, 1998; Clarke, 2002; Geldenhuys, 1994; Hoffmann et al., 2012b; Little et al., 2012). However Afrotemperate forest fire patterns are particularly influenced by the topography of the landscapes (Beckett and Bond, 2019). Afrotemperate forests are found on south-facing slopes which receives less solar radiation compared to north-facing slopes (Beckett and Bond, 2019; Bradstock et al., 2010). The less solar radiation received, the less evapotranspiration can occur which increases soil and fuel moisture (Beckett and Bond, 2019). Therefore, Afrotemperate forests tend to be fire refugia due to their position on the slopes in the landscape (Clarke, 2002; Geldenhuys, 1994).

Land cover

Forest fires impact climate change, land cover, land use, ecosystems and biodiversity (Roy, 2003). However, fires are also impacted by these factors (Archibald et al., 2009). Land use and land cover (LULC) has an impact on all disturbance regimes, particularly fire, as they impact the landscapes' structure, patterns and processes that occur (Viedma et al., 2006; Vitousek et al., 1997). How much LULC impacts fire and its rate of spread is dependent on the type and rate of change of LULC; the amount and type of vegetation in the landscape; the dynamics of the vegetation after fire; and the topography and the climate in the landscape (Bucini and Lambin, 2002; Fernandes, 2009; Moloney and Levin, 1996; Moreira et al., 2009; Turner et al., 1997; Viedma et al., 2006). Anthropogenic factors associated with LULC can also impact fire regimes such as the creation of grazing land, changes in population density and the density of and distance to roads (Archibald et al., 2009; Sparks et al., 2024). The impact of these factors on fire patterns are context specific for example areas with high population density have shown to increase the ignitions of fires (Keeley et al., 1999); however, increased population densities can also lead to decreased fuel loads and fragmentation in the landscape which would reduce the rate at which fire can spread (Archibald et al., 2009). Different land covers impact fire patterns differently due to differences in the microclimate and the biophysical conditions of the vegetation (fuel) in the landscape (Gutiérrez-Vélez et al., 2014a). The number of different land cover classes also impacts fire patterns, as landscape homogeneity promotes fire spread whereas heterogenous landscapes act as natural firebreaks in the landscape (Moreira et al., 2011; Vega-García and Chuvieco, 2006; Viedma et al., 2009). The more irregularly shaped, dispersed and scattered different land cover patches are, the less likely fire is to spread (Gutiérrez-Vélez et al., 2014b). Land use can also impact the fire regimes in the area as different land management practices by farmers or rural residents may be focused on either preventing, allowing or controlling fires in the area (Gutiérrez-Vélez et al., 2014a). Distance from roads can also be a factor that impacts the number of ignition of fires (Silvestrini et al., 2011; Uriarte et al., 2012). Land cover and land use also impacts the timing of fires. This is seen in a study done in the southwestern part of the Central African Republic in 2002 using Landsat TM images which found that in savannas that were surrounded by human settlements, fire tended to burn later compared to fires in areas where there were no human settlements (Bucini and Lambin, 2002).

How LULC impacts fire patterns will be context specific and dependent on the microclimate and the conditions of the vegetation in the area. For example in the Brazilian Amazon, fire activity was highly linked to deforestation, with increasing deforestation rates leading to increased fire frequency (Aragão et al., 2008; Aragão and Shimabukuro, 2010; Gutiérrez-Vélez et al., 2014b; Lima et al., 2012; Uriarte et al., 2012; Vasconcelos et al., 2013). Older forests were found to be less flammable compared to logged forests and secondary forests (Uhl and Kauffman, 1990) and secondary forests were found to be less flammable compared to logged forests due to secondary forests having a higher relative humidity compared to logged forests (Gutiérrez-Vélez et al., 2014b; Uhl and Kauffman, 1990). In the Peruvian Amazon, fire occurrence was mostly linked to distance to croplands and pastures of fire-prone shrubs (Uriarte et al., 2012). In a study done in the forests of Kapuas District, Central Kalimantan Province, Indonesia, it was found that fires mostly occurred in peat soil sites with marshland bush cover (closer to river sources) in comparison to plantation and secondary mangrove forests (Thoha et al., 2019). This was due to the marshland bush covers having more abundant and flammable fuel loads in comparison to the other land cover types (Thoha et al., 2019). This indicates that fires in these three regimes are influenced by different LULC. There may be differences in the flammability of the same land cover class. For example, while pastures are fairly flammable, degraded pastures are found to be less flammable compared to non-degraded pastures (Gutiérrez-Vélez et al., 2014b; Uhl and Kauffman, 1990). This is likely due to differences in their microclimates. Therefore, understanding the surrounding LULC in area and how it affects fire is an important component to understand the impact fire has on Afrotemperate forests.

Methods

Study Site

KwaZulu-Natal (KZN) is a province located on the eastern coast of South Africa. It is a naturally biodiverse region consisting of mesic grasslands, savannas, forests and wetlands (Jewitt et al., 2015; Rutherford et al., 2006; Wessels et al., 2016). It is also anthropogenically diverse consisting of exotic plantations, sugarcane fields, rural and urban settlements (Wessels et al., 2016). KwaZulu-Natal receives on average around 837 mm of precipitation per annum (Jewitt et al., 2015). Most rain occurs during summer between November and March and the lowest amount of rain during winter from June and July (Kotze and Samways, 2001). The mean monthly temperature is between 19- 20 °C in January (the hottest month of the year) and 8 °C in July (the coldest month of the year) (Kotze and Samways, 2001). The province's forests are usually small (between 10 and 100 ha) and scattered on south and south-east-facing slopes (Mucina and Geldenhuys, 2006). My study area was the montane *Podocarpus* Afrotemperate forests of KZN. The Afrotemperate forests are classified as being temperate forests and are found at altitudes ranging between 1 000 and 1 500 m (Mucina and Geldenhuys, 2006a). Afrotemperate forests are multi-species forests made up of *Podocarpus* and *Afrocarpus* species and angiosperms such as *Apodytes dimidiata, Ilex mitis, Nuxia congesta, Prunus africana, Nuxia floribunda, Celtis africana, Xymalos monospora,* and *Calodendrum capense* (Mucina and Geldenhuys, 2006b; Wirminghaus et al., 2001b).

Site Selection

I investigated 10 large (~1000 ha), 10 medium (~ between 1 and 1000 ha) and 10 small (~1 ha) randomly selected Afrotemperate forests to establish how their extent changes over time. Size categories were chosen due to most sizes being found between one hectare and 400 hectares and a significant number above 3200 hectares (Berliner, 2009). Most forests were selected from a similar region (central or lower part of KwaZulu-Natal) and subtype to ensure that the analysed forests had similar climatic patterns (Figure 1).

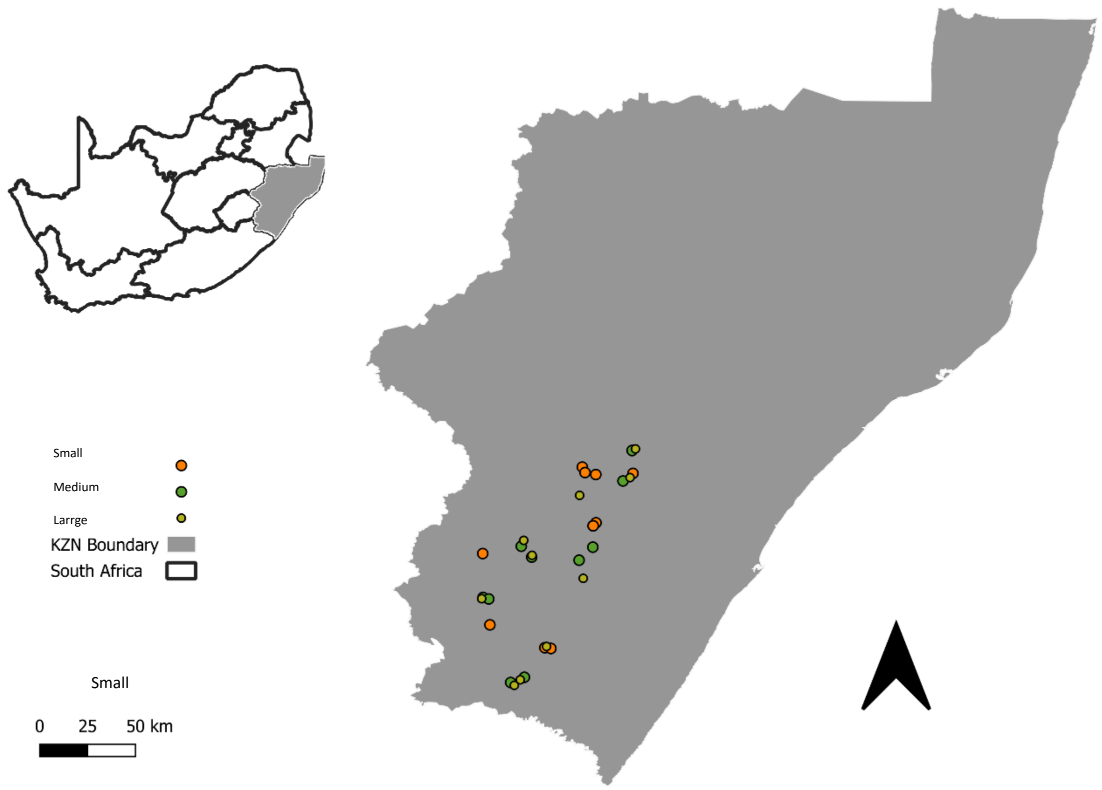

Figure 1: Locations of 30 Afrotemperate forests in KwaZulu-Natal, South Africa. Orange dots represent small forests (~1 ha), green dots represent medium forests (~ between 1 and 1000 ha) and yellow dots represent large forests (~1000 ha).

Data Collection

My primary source of data for forest extent and composition were high resolution aerial photographs. Aerial photographs are advantageous as they have a high spatial resolution and tonal detail which allows for an accurate assessment of landscape features (Fensham and Fairfax, 2002). This in turn helps with determining changes in vegetation boundaries, tree densities and vegetation composition (Fensham and Fairfax, 2002). Because aerial photos have been captured since the 1930s, they provide a long temporal extent which will allow for a long-term analysis (Fensham and Fairfax, 2002).

All aerial photos were downloaded from the National Geo-Spatial Information (NGI) site (National Geo-Spatial Information (NGI), 2024). Aerial photographs were taken by NGI on average every 10 years. Three sets of photos were downloaded (1937- 1978; 1975- 2000 and 2009-2016) for each forest (Appendix A). It was not possible to get images from the exact same years due to a lack of available data, but a minimum of 14 years was chosen between each period to ensure there would be enough of a time difference for a change to be observed. Images between 2009 and 2016 were georeferenced and orthorectified by NGI when downloaded.

Data Processing

Georeferencing

Pre-2009 images were georeferenced. In general, most images had between 30 and 40 ground control points (GCPs); however, some needed > 300 for accuracy purposes. I used a first-order polynomial transformation and ensured the Root Mean Square Error (RMSE) remained below 20 for all images (Ćalasan et al., 2020) which is done to ensure that images are georeferenced as accurately as possible. Images were projected into WGS84 UTM35S or 36S depending on their location. Georeferencing was done in QGIS version 3.34.3.

Orthorectification

Orthorectification is the process of geometrically correcting images to remove distortions that can be cause by camera tilts, topographic relief or optical properties of the camera lens (Hackeloeer et al., 2014). Orthorectification was not feasible for pre-2009 aerial photographs due to the absence of rational polynomial coefficients (RPCs) and a 3D transformation model. Additionally, the lack of fiducials prevented any metadata extraction, complicating the orthorectification process. Alternatively, rubber sheeting—a form of pseudoorthorectification—can be used when orthorectification is impossible, which closely resembles accurate georeferencing. This method aligns corresponding points between images, yielding results similar to the already georeferenced images in this project (T. J. Smith et al., 2010). Therefore I did not orthorectify the images but just accurately georeferenced them.

Objective 1 Methods: *Track the extent of selected Afrotemperate forests over an 80-year period.*

Forest Extent

Each of my 30 forests were digitised in QGIS using the shape, pattern, texture, and context of each forest. This was done for each of the three periods (see Appendix A). The absolute area change (in ha) and the perimeter-area ratio values (in km/ha) for each period were calculated. Perimeter primarily represents the shape and extent of the patch (Helzer and Jelinski, 1999). Looking at perimeter-area ratios is often used to calculate 2D shape compactness which allows for an analysis of the impact of edge effects which has implications on biodiversity (biotic and abiotic) and fire (Armenteras et al., 2013; Bogaert et al., 2000; Helzer and Jelinski, 1999; Laurance and Yensen, 1991; Murcia, 1995). Results were reported and analysed using mean values (± standard error value) in the results section.

Forest Composition

The forest species composition was determined visually for each forest patch using expert knowledge applied to the aerial photographs. The composition of forests was determined on a scale from one to three with one indicating indigenous vegetation (Figure 2, image A), three indicating exotic plantations (Figure 2, image C) and two a less distinctive composition (it was be referred to as 'mixed' in the investigation) (Figure 2, image C). Indigenous forests were classified as being less uniform in texture and structure in comparison to plantations but darker in colour in comparison to the mixed vegetation. Results were reported and analysed using mean values (± standard error value) in the results section.

Figure 2: Differences between types of vegetation used for forest composition analysis. **A-** indigenous forests, **B-** mixed vegetation and **C-** exotic plantation.

Objective 2 Methods: *Identify the impact of the surrounding land cover and fire has on forest dynamics and vulnerability.*

Land cover

I mapped the land cover of the surrounding matrix of the 30 forests for three time periods (Appendix A) using aerial photographs due to their high temporal extent and spatial resolution. I classified land cover within a 1 km buffer region of each of the digitised forests as it is a way to look at landscape change at an intermediate scale between local scale changes (100 m buffers) and large landscape changes (10 km) (Määttänen et al., 2022). All classifications were done in ArcGIS Pro version 3.3. I mosaiced the aerial photographs together and then clipped them to the 1 km buffer region of the digitised forests. The collection of training data was done using object-based classification which uses individual objects to classify the image instead of individual pixels (Liu and Xia, 2010). Advantages of using this method include that it decreases issues linked to the within- class spectral variation that may cause the 'salt-and-pepper' effects that are common in pixel-based classifications (Liu and Xia, 2010). Object-based classification is also able to incorporate changes in shapes and topography which pixel-based classifications do not usually include (Guo et al., 2007; Kelly et al., 2004) and this is particularly important for the mountainous landscape that the study sites are found on. However, there are issues with using object-based classifications which can include under (getting too few segments) and over segmentation (getting too many segments) (Delves et al., 1992; Möller et al., 2007). Both of these issues can cause misclassification of classes which can result in a reduction in accuracy (Liu and Xia, 2010; Song et al., 2005). Both of these issues arose in some of the classifications and when this occurred, I manually reclassified the misclassified areas. The training data was used to train the Support Vector Machine (SVM) classifier (Liu, 2005; Richards and Jia, 2006). Support Vector Machine classifiers use optimization algorithms which locate the optimal boundaries between different classes (Huang et al., 2002) which makes this classifier ideal for land cover classifications. Support Vector Machine classifiers have been found to outperform more commonly used classifiers such as maximum likelihood and multi-layer backpropagation neural network classifiers in classification accuracy (overall but also for individual classes) and have a faster computational time (Kavzoglu and Colkesen, 2009; Pal and Mather, 2003) which is why I selected it as the classifier for this land cover classification. The classifier was then used to assign each of the pixels of the raster to a land cover class according to the training data (Liu, 2005). I classified my land cover classes based on Thompson's (1996) classification (Table 1). Level two classification from Thompson (1996) was used for forests classification and level one for all other land cover classes. However due to visual issues with the older images, I classified grasslands and barren land as a single class as it was difficult to differentiate between these two land cover types. I grouped exotic plantations or any non-indigenous trees as non-indigenous forests as some of the exotic vegetation were not plantations but other types of alien trees, possibly *Solanum mauritianum* (also referred to as Bugweed) which is frequently found in the eastern part of South Africa (Witkowski and Garner, 2008) or *Acacia mearnsii* (also referred to as Black Wattle) which is frequently found in the rural parts of the Drakensberg regions of South Africa (De Neergaard et al., 2005). I did an accuracy assessment of the classified rasters manually by using approximately 100 stratified random accuracy assessment points per image (Appendix B) and visually determining the ground truth points to compare to the classified points. A confusion matrix was run for the ground truth points against the classified points and the user and producer accuracy, and Kappa Coefficient values were captured (Appendix C). The user accuracy is used to determine the false positives or errors of commission while the producer accuracy is used to determine the false negatives or the errors of omission (ESRI, n.d.). The Kappa Coefficient measures the overall agreement of classified results with the reference data using random chance (Congalton and Mead, 1983; ESRI, n.d.). I ensured that the Kappa and producer and user accuracy were higher than 70% for all classified images. The area of each land cover class was extracted in hectares. All values were converted to percentage in order to accommodate for the different sizes of the matrices (buffer zones) studied. The mean percentage of each land cover class for each size category (small, medium and large) was calculated for each time period to determine the change in land cover over time.

Table 1: Standard land cover classification for remote sensing applications in South Africa: class summary (Thompson, 1996)

	Level I	**Level II**
1	Forest and Woodland	Forest Woodland Wooded grassland
2	Thicket, bushland, scrub forest and high fynbos	Thicket Scrub forest Bushland Bush clumps High heathland (high fynbos)
3	Shrubland and low fynbos	Shrubland Low fynbos (heathland)
4	Herbland	
5	Grassland	Unimproved grassland Improved grassland
6	Forest Plantations	Pine species Eucalypt species Wattle/ other species Indigenous species
7	Waterbodies	
8	Wetlands	
9	Barren lands	Bare rock/ soil Degraded land
10	Cultivated land	Permanent crops Temporary crops
11	Urban/ built-up land	Residential Commercial Industrial/ transport
12	Mines and quarries	

Fire

I studied fire extent and fire frequency surrounding the selected forests annually from 1990 to 2024. Landsat images and Sentinel-2 images were used, accessed on Google Earth Engine. Landsat was launched in 1972 and has a spatial resolution of 30 m, with a 16 day repeat cycle (USGS, n.d.). Sentinel-2 images are also free and open access and were launched in 2015. Sentinel-2 satellites have a 10 day repeat coverage and between 10 and 20 m spatial resolution (Li and Roy, 2017). While Landsat provides a better temporal extent than Sentinel, Sentinel provides a better spatial resolution therefore I used both image types to increase accuracy of my analysis but also increase the possibility of finding images with the least amount of cloud coverage (Li and Roy, 2017). These two satellites are comparable due to them having similar sensors and resolutions compared to other open source satellite images commonly used for fire analysis such as the Moderate Resolution Imagining Spectroradiometer (MODIS) which has a minimum resolution of 250 m (Justice et al., 1998; Showstack, 2014). Landsat five, (1990-1998) seven (1999 -2013), and eight (2014-2016), and Sentinel-2 (2017-2024) images were used as these are the only satellites that contain both the near and shortwave infrared bands, which were needed to identify burnt area.

Using indices that measure burnt area is more accurate to measure fire than active fires as satellites may not be passing over that area when an active fire is present (Eva and Lambin, 2000). Therefore using a tool like Normalised Burn Ratio (NBR) is useful as it is based on burnt areas. The Normalised Burn Ratio (NBR) is used to determine the difference in infrared reflectance between burnt and unburnt areas with areas that have a high Near Infrared reflectance (NIR) and low Shortwave Infrared Reflectance (SWIR) indicating areas with lots of vegetation and areas with low NIR and high SWIR indicating areas that are burnt (United Nations, n.d.). The NBR is calculated using the following formula:

$$NBR = \frac{NIR - SWIR}{NIR + SWIR}$$

To estimate fire extent, it is necessary to determine the difference in NBR (dNBR) before and after the fire. That can be done by using this formula:

$$\Delta NBR(dNBR) = PreFireNBR - PostFireNBR$$

Positive dNBR values indicates that there has been reduction in vegetation and areas with a negative dNBR indicate areas with dense vegetation regrowth post-fire (Roy et al., 2006).

Fire extent is determined by combining the four fire severity categories (burnt) that is provided by NBR. The range of values of different fire severity levels can be seen below:

	Severity Level	dNBR Range (scaled by 10^3)	dNBR Range (not scaled)
Unburnt	Enhanced Regrowth, high (post-fire)	-500 to -251	-0.500 to -0.251
	Enhanced Regrowth, low (post-fire)	-250 to -101	-0.250 to -0.101
	Unburned	-100 to +99	-0.100 to +0.99
	Low Severity	+100 to +269	+0.100 to +0.269
Burnt	Moderate-low Severity	+270 to +439	+0.270 to +0.439
	Miderate-high Severity	+440 to +659	+0.440 to +0.659
	High Severity	+660 to +1300	+0.660 to +1.300

Table 2: Values of severity levels of Normalised Burn Ratio (NBR) and the groupings of fire extent (burnt and unburnt) excluding the low severity class (United Nations, n.d.)

I calculated dNBR on the 1 km buffer area surrounding each forest, seasonally. Seasonal analysis was done as changes in vegetation's flammability is highly impacted by vegetation moisture which is seasonally dependent (Veraverbeke et al., 2012). Seasonal divisions were determined by the hydrological year as fire is highly influenced by moisture (Scholes et al., 1996; Spessa et al., 2005). Summer was stated as November to March, autumn being April, winter from May to August and spring September to October (Nel, 2009; Smart, 2017). Pre-fire was the month before the season and post-fire was the month after the seasons. During the data collection of the fire results, I found that there was an overestimation of the low severity class (it made up to 59% of the classified fire extent) in the autumn and winter seasons. The initial results indicated that all forests patches had completely burnt for multiple years which is unlikely as the microclimate below Afrotemperate canopies is dark and moist which usually does not promote fire (Abiem et al., 2020). Difference in Normalised Burn Ratio relies on spectral response changes which can cause misclassifications of burnt and unburnt areas in areas with high heterogeneity (Eva and Lambin, 2000; Fraser et al., 2017; Ramjeawon et al., 2020; Santos et al., 2020). My study area was not only heterogenous but was also found at varying altitudes and with a steep topography which can also create changes in spectral signatures due to differences in sun angles. Three different studies in Brazil, Canada and the United States of America (USA) that used dNBR for fire detection found either over or underestimation depending on the land cover type (Fraser et al., 2017; Picotte and Robertson, 2010; Santos et al., 2020). Changes in environmental conditions due to seasonal change can also impact the accuracy of detecting fires as low sun angles and vegetation senescence can affect the spectral signatures the satellite picks up (Key, 2006; Robson, 2005). This is likely the reason for why there was a high overestimation of the low severity class in autumn and winter. Therefore I decided to exclude the low severity class from the study to prevent overestimation of my fire results. Mean percentage of fire extent was calculated in order to compare across the different sizes of the different forest patches. Annual fire frequency was determined by converting rasters into binary images, with one representing burnt area (dNBR values greater 270) and zero representing unburnt area ((dNBR values less than 270) (Table 2)) and adding all the burnt area together using band math, in QGIS.

Forest Change

To determine how fire and land cover change influence forest extent, I ran four Generalized Linear Mixed Models (GLMMs) in R version 4.3.0. My first GLMM had forest extent as the response variable and land cover as the predictor variable. This GLMM looked at all three time periods- 1940, 1980 and the 2020s. My second GLMM had forest extent as the response variable and fire as the predictor variable. This model included information from 1990 until 2024. My third GLMM had fire as the response variable and land cover as the predictor variables. This included all fires present in the buffer region and looked at the periods from 1990 to 2024. My last GLMM had forest extent as the response variable and land cover and fire as the predictor variables from 1990 to 2024. For all models, I had year and patch as random effects. For relationships that were not statistically significant, I evaluated results qualitatively to see if there were any relationships that might be ecologically relevant. This was referred to as an ecological analysis.

Results

Objective 1 Results: *Track the extent of selected Afrotemperate forests over an 80-year period.*

Small forests

Forest extent

The small forest patches (approximately one hectare) in this study had a general increase in extent over the three-time periods (Table 3). There was a general increase in forest extent between time one and two of 183.31% (± 394.39), a general decrease between time two and three of 1.92% (±38.74) and a general increase between time one and three of 131.97% (±304.33) (Table 3).

Forest Composition

Overall, 60% of the patches were or became mixed. Most mixed patches increased more in extent than natural forests (Table 3). Forests that became mixed in time two increased by 535.16% (± 642.24) from time one whereas forests that remained natural increased by 57.73% (± 11.30) (Table 3). However the forests that became mixed in time three increased by 48.23% (± 19.81) from time one, less than both natural and forests that became mixed in time two.

After the initial invasion of the non-indigenous species, forests tended to decrease in size with forests that had become mixed in time two decreased by 36.72% (± 18.65) in time three (Table 3). This is also reinforced by the fact that the only patch that was mixed throughout the study decreased in size by 54.39% in time three from time one (Table 3).

Table 3: Forest extent in hectares and composition of 10 one-hectare Afrotemperate Forest patches in KwaZulu-Natal, South Africa. Natural patches represent patches that are made up of indigenous trees and mixed patches are in bold and represent forest patches that are composed of indigenous trees and non-indigenous vegetation. T_1, T_2, T_3 represent time one, two and three, respectively. The overall change column represents the difference between T_3 (2009 – 2016) and T_1 (1944 – 1978). T_2 represents years 1981-1992.

ID	T_1 (ha)	T_2 (ha)	T_3 (ha)	Overall Change
1	0.73 Natural	0.87 Natural	1.18 **Mixed**	↑
2	1.64 Natural	2.25 **Mixed**	1.52 **Mixed**	↓
3	0.39 Natural	1.59 **Mixed**	0.68 **Mixed**	↑
4	1.40 Natural	1.23 Natural	1.87 **Mixed**	↑
5	0.45 Natural	0.37 Natural	0.20 Natural	↓
6	0.10 Natural	1.38 **Mixed**	1.09 **Mixed**	↑
7	0.80 Natural	2.09 Natural	1.69 Natural	↑
8	0.19 Natural	0.47 Natural	0.47 Natural	↑
9	0.88 Natural	0.89 Natural	1.16 Natural	↑
10	2.03 **Mixed**	0.68 **Mixed**	0.93 **Mixed**	↓

Perimeter-Area Ratio

Perimeter- Area ratio for all 10 one-hectare patches had a slight decreasing trend over the three time periods (Figure 3) indicating that forests were becoming larger and more spread-out from time 1 to time 3.

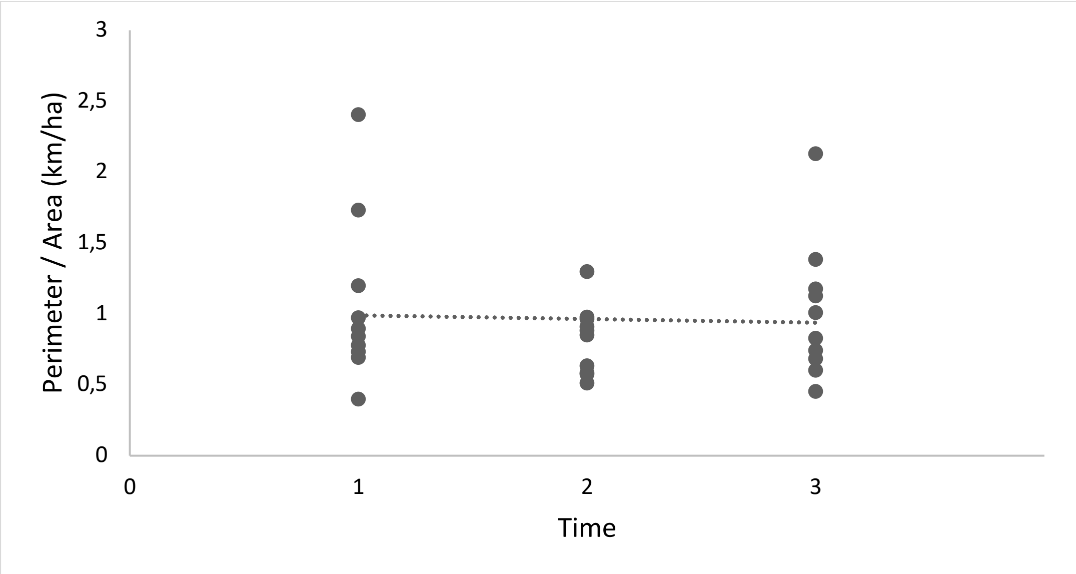

Figure 3: Perimeter- area ratio in kilometre per hectare of 10 one-hectare Afrotemperate Forest patches over three time periods in KwaZulu-Natal, South Africa. Time one represents years 1944 - 1978, time two 1981 – 1992 and time three 2009 – 2016. R- squared value: 0.0022, p-value = 0.002 and F-value = 3.13.

Medium forests

Forest Extent

Medium forest patches (approximately between 1 and 1000 hectares) in this study had a general decrease in extent over the three-time periods when excluding patch five (an outlier as it was the only data point that lied outside of the overall pattern) (Table 4). There was a general increase in forest extent between period one and two by 2.01% (± 16.86), a general decrease between period two and three by 5.92% (± 22.24) and a general decrease between period one and three by 3.76% (± 27.59) (Table 4).

Forest Composition

Overall, 60% of the patches remained natural and decreased by 9.49% (± 21.85) (Table 4). Mixed patches had a variable response but on average increased by 28.88% (± 53.20) (Table

4). Forest one that became mixed in time two decreased by 25.46% from time one and forests that became mixed in time three increased by 22.75% (± 41.88) from time one (Table 4). Patch five that remained mixed throughout the study increased in size by 92.38% in time three from time one (Table 4).

Table 4: Forest extent in hectares and composition of 10 100-hectare Afrotemperate Forest patches in hectares in KwaZulu-Natal, South Africa. Natural patches represent patches that are made up of indigenous trees and mixed patches are in bold and represent forest patches that are composed of indigenous trees and exotic vegetation. T_1, T_2, T_3 represent time one, two and three, respectively. The overall change column represents the difference between T_3 (2009 – 2016) and T_1 (1937 – 1972). T_2 represents years 1975-2000.

ID	T_1	T_2	T_3	Overall Change
1	25.55 Natural	26.60 **Mixed**	19.83 **Mixed**	↓
2	3.48 Natural	3.72 Natural	5.31 **Mixed**	↑
3	60.44 Natural	40.96 Natural	29.13 Natural	↓
4	18.00 Natural	14.93 Natural	16.59 Natural	↓
5	248.84 **Mixed**	296.22 **Mixed**	478.72 **Mixed**	↑
6	5.71 Natural	6.71 Natural	5.34 Natural	↓
7	17.55 Natural	18.76 Natural	18.31 Natural	↑
8	38.06 Natural	44.02 Natural	35.45 **Mixed**	↓
9	53.85 Natural	53.07 Natural	51.15 Natural	↓
10	67.81 Natural	79.90 Natural	74.52 Natural	↑

Perimeter-Area Ratio

Perimeter- Area ratio for nine 100-hectare patches has a general decreasing trend over the three time periods (Figure 4) indicating that forests were becoming more spread-out from time 1 to time 3.

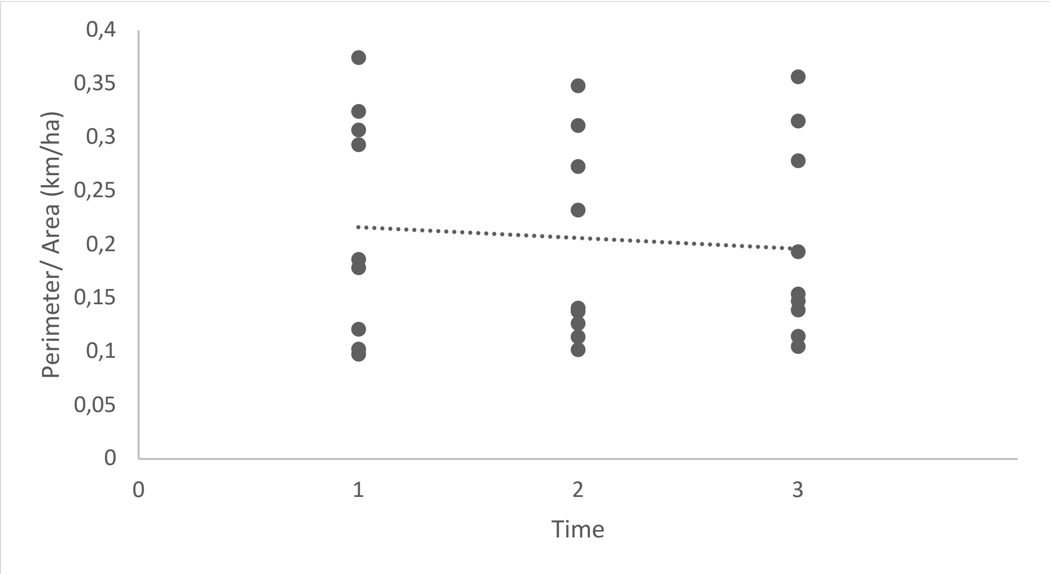

Figure 4: Perimeter- area ratio in kilometre per hectare of nine 100-hectare Afrotemperate Forest patches over three time periods in KwaZulu-Natal, South Africa. Time one represents years 1937 - 1972, time two 1975 – 2000 and time three 2009 – 2016. This graph excluded the outlier and the R-squared value: 0.0079, p value < 0, F value = 85.93.

Large Forests

Forest Extent

Large forest patches (approximately 1000 hectares) had a general increase in extent over the three-time periods (Table 5). There was a general increase in forest extent between time one and two by 2.26 % (±5.04), a general increase between time two and three by 12.76% (± 20.58) and a general increase between time one and three by 15.22% (± 21.23) (Table 5).

Forest Composition

Overall, 80% of the large patches remained natural (Table 5). Forest 10 that became mixed in time two increased by 34.65% from time one and forest 9 that became mixed in time three increased by 20.94 % from time one (Table 5). All other forests that remained natural, increased on by 12.08% (± 22.58) in extent (Table 5).

Table 5: Forest extent in hectares and composition of 10 1000-hectare Afrotemperate Forest patches in hectares in KwaZulu-Natal, South Africa. Natural patches represent patches that are made up of indigenous trees and mixed patches are in bold and represent forest patches that are composed of indigenous trees and exotic vegetation. T_1, T_2, T_3 represent time one, two and three, respectively. The overall change column represents the difference between T_3 (2006 – 2016) and T_1 (1944 – 1972). T_2 represents years 1978-2000.

ID	T_1	T_2	T_3	Overall Change
1	519.99 Natural	510.68 Natural	530.51 Natural	↑
2	366.24 Natural	372.39 Natural	416.51 Natural	↑
3	139.17 Natural	143.39 Natural	144.65 Natural	↑
4	115.60 Natural	118.62 Natural	192.42 Natural	↑
5	164.60 Natural	155.87 Natural	156.50 Natural	↓
6	123.45 Natural	140.62 Natural	131.71 Natural	↑
7	233.19 Natural	238.95 Natural	240.12 Natural	↑
8	671.88 Natural	670.01 Natural	710.54 Natural	↑
9	76.65 Natural	80.79 Natural	92.70 **Mixed**	↑
10	214.38 Natural	216.30 **Mixed**	288.67 **Mixed**	↑

Perimeter- Area Ratio

Perimeter- Area ratio for all 10 1000-hectare patches had a general decreasing trend over the three time periods (Figure 5) indicating that forests were becoming larger and more spread-out from time 1 to time 3.

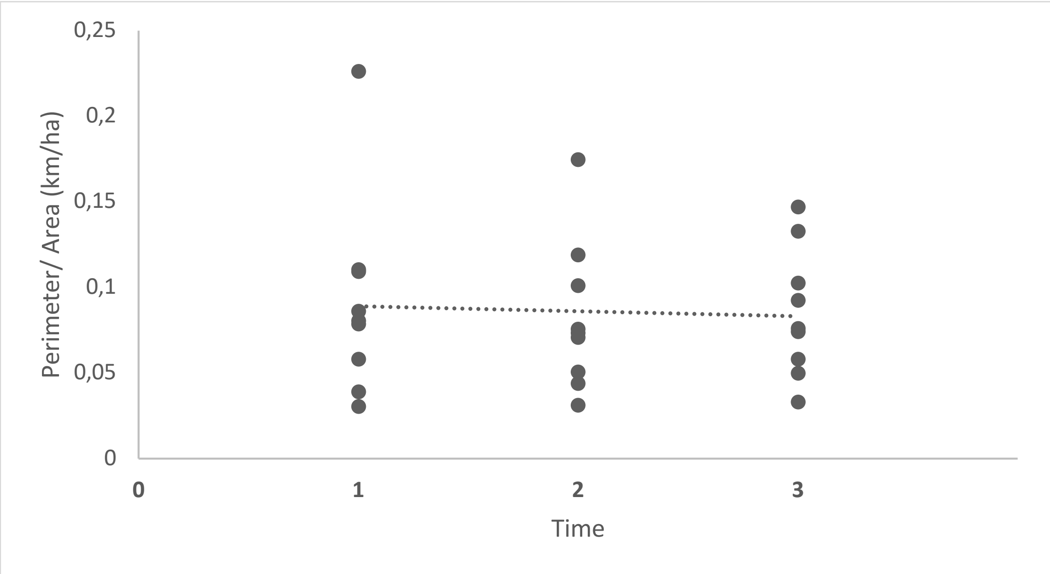

Figure 5: Perimeter- area ratio in kilometre per hectare of 10 1000-hectare Afrotemperate Forest patches over three time periods in KwaZulu-Natal, South Africa. Time one represents years 1944 - 1972, time two 1978 – 2000 and time three 2006 – 2016. R squared value: 0.0001, p value < 0, F value = 61.80

Objective 2 Results: *Identify the impact the surrounding land cover has on forest dynamics and vulnerability.*

Land cover

Small Forests

The dominant class for the small forests in period one was herbaceous vegetation which made up 80.71% (SE ± 24.53) of the composition of the matrix. This decreased to 55.63% (SE ± 21) in the third period (Figure 6). Non-indigenous vegetation made up 10.98% (SE ± 19.93) of the surrounding matrices' composition in period one and increased to 26.17% (SE ± 23.11), in period three (Figure 6).

Net land cover change between period one and two was higher (43.2%) than net land cover changes that occurred between period two and three (26.76%) (Table 6). Non-indigenous vegetation had the highest gain (18.36%) and herbaceous vegetation had the highest loss

(16.66%) out of all the land cover classes (Table 6). Urban cover had overall net increase of 3.92%.

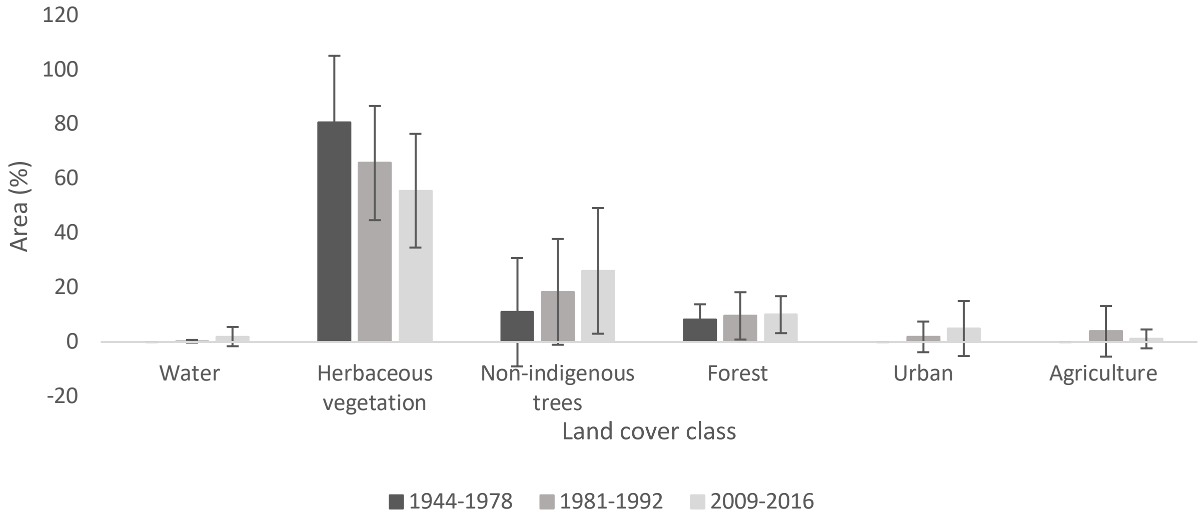

Figure 6: Mean area and standard error of the percentage contribution of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 small Afrotemperate forests (~ 1 ha) in KwaZulu-Natal, South Africa over three time periods (1944-1978, 1981-1992 and 2009-2016). Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

Table 6: Average gains and losses of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 small Afrotemperate forests (~ 1ha) in KwaZulu-Natal, South Africa, between 1944 and 1992 and between 1981 and 2016. Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

	1944-1992			**1981-2016**		
Land Cover	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**
Water	0.28	0	0.28	2.43	0.11	2.32
Herbaceous vegetation	14.04	27.30	13.26	16.66	13.16	3.5
Non-indigenous vegetation	18.36	6.32	12.04	6.35	2.88	3.47
Forest	4.94	2.32	2.62	4.65	4.70	0.05
Urban	1.89	0	1.89	7.70	1.89	5.81
Agriculture	13.11	0	13.11	4.55	16.16	11.61
Total	**52.62**	**35.94**	**43.2**	**42.34**	**38.9**	**26.76**

Medium Forests

The dominant class for the medium forests in period one was herbaceous vegetation which made up 84.48% (SE ± 8.29) of the composition of the matrix. This decreased to 56.10% (SE ± 18.90) in the third period (Figure 7). Non-indigenous vegetation made up 2.63% (SE ± 4.43) of the surrounding matrices' composition in period one and increased to 26.65% (SE ± 13.08), in period three (Figure 7).

Net land cover change between period one and two was higher (36.73%) than net land cover changes that occurred between period two and three (19.42%) (Table 7). Non-indigenous vegetation had the highest gain (21.42%) and herbaceous vegetation had the highest loss (21.68%) out of all the land cover classes (Table 7). Agricultural cover had overall net increase of 2.25% and water of 4.12% (Table 7).

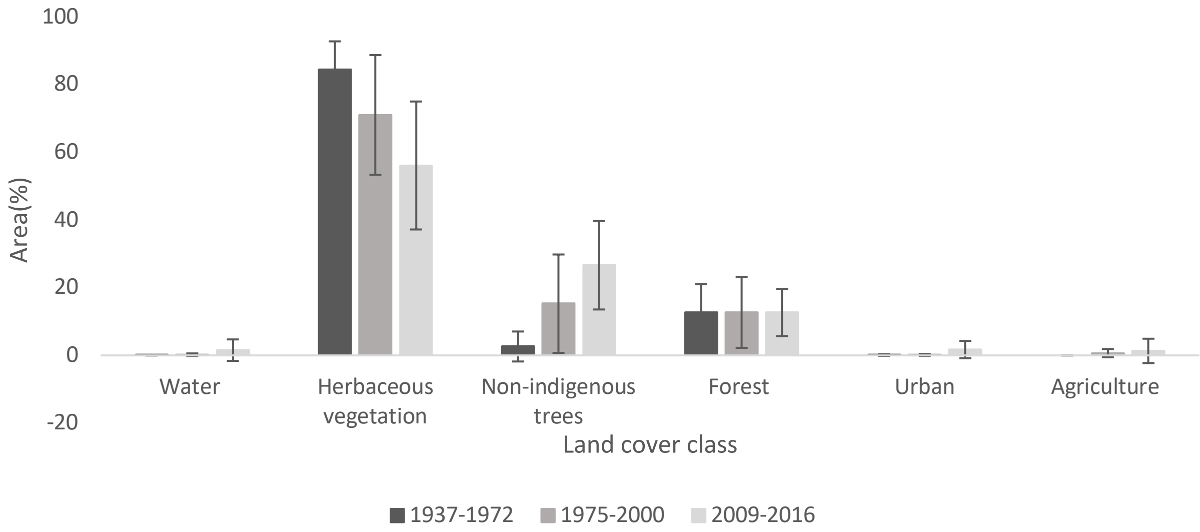

Figure 7: Mean area and standard error of the percentage contribution of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 medium Afrotemperate forests (~ between 1 and 1000 ha) in KwaZulu-Natal, South Africa, over three time periods (1937-1972, 1975-2000 and 2009-2016). Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

Table 7: Average gains and losses of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 medium Afrotemperate forests (~ 100ha) in KwaZulu-Natal, South Africa, between 1937 and 2000 and between 1975 and 2016. Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

	1937-2000			**1975-2016**		
Land Cover	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**
Water	0.19	0	0.19	4.88	0.57	4.31
Herbaceous vegetation	4.24	17.86	13.62	0.80	21.68	20.88
Non-indigenous vegetation	21.42	1.02	20.4	13.30	6.06	7.24
Forest	5.60	5.77	0.17	6.82	6.88	0.06
Urban	0.65	0.50	0.15	4.15	0.79	3.36
Agriculture	2.20	0	2.20	6.65	2.20	4.45
Total	**34.3**	**25.15**	**36.73**	**36.6**	**38.18**	**19.42**

Large Forests

The dominant class for the large forest in period one was herbaceous vegetation which made up 70.56% (SE ± 16.84) of the composition of the matrix. This decreased to 48.02% (SE ± 21.77) in the third period (Figure 8). Non-indigenous vegetation made up 15.62% (SE ± 16.43) of the surrounding matrices' composition in period one and increased to 28.35% (SE ± 11.99), in period three (Figure 8).

Net land cover change between period one and two was higher (53.94%) than net land cover changes that occurred between period two and three (16.62%) (Table 8). Non-indigenous vegetation had the highest gain (22.21%) and herbaceous vegetation had the highest loss (26.51%) out of all the land cover classes (Table 8). Agricultural cover had a net gain of 17.96% in between period one and two but a net loss of 6.69% between period two and three and water had a net increase of 3.85% (Table 8)

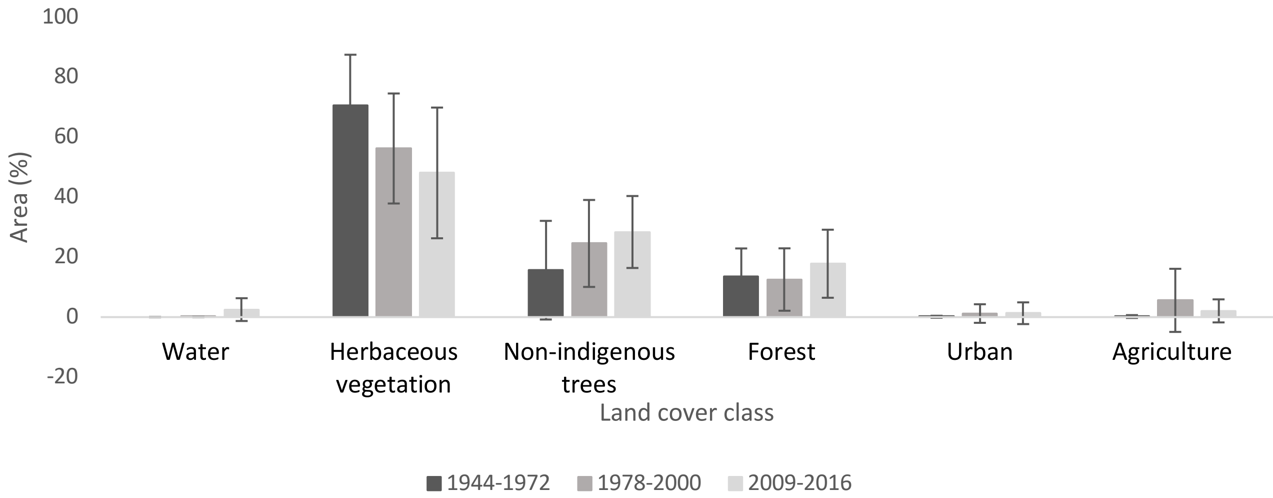

Figure 8: Mean area and standard error of the percentage contribution of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 large Afrotemperate forests (~ 1000 ha) in KwaZulu-Natal, South Africa, over three time periods (1944-1972, 1978-2000 and 2009-2016). Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

Table 8: Average gains and losses of different land cover classes in the surrounding matrix (one kilometre buffer area) of 10 large Afrotemperate forests (~ 1000 ha) in KwaZulu-Natal, South Africa, between 1944 and 2000 and between 1978 and 2016. Herbaceous vegetation represents barren land and all vegetation smaller than trees. Non-indigenous forests include plantations and all other exotic vegetation.

	1944-2000			**1978-2016**		
Land Cover	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**	**Gain (%)**	**Loss (%)**	**Absolute value of net change (%)**
Water	0.16	0	0.16	4.13	0.12	4.01
Herbaceous vegetation	3.77	26.51	22.74	13.74	14.88	1.14
Non-indigenous vegetation	22.21	14.72	7.49	11.55	7.80	3.75
Forest	8.97	10.99	2.02	12.52	11.66	0.86
Urban	3.57	0	3.57	0.71	0.54	0.17
Agriculture	17.96	0	17.96	7.50	14.19	6.69
Total	**56.64**	**52.22**	**53.94**	**50.15**	**49.19**	**16.62**

Fire

Fire Extent

The annual patterns of fire extent of all the forest patches can be divided into two main trends: pre-2013 and post-2013. Pre-2013s (Landsat 5 and 7 images), the mean fire extent for all forests was 23.16% and had high variability (SE $\pm$ 10.28) whereas post-2013 (Landsat 8 and Sentinel-2 images) showed more of a stable pattern (SE $\pm$ 3.13) with a mean of 23.87 % (Figure 9). Overall the small forests had a slightly higher fire extent (mean: 24.69 %, SE $\pm$ 10.19) than the medium (mean: 23.13 %, SE$\pm$ 8.15) and the large forests (mean: 22.33 %, SE$\pm$7.56). Smaller forests also had higher peaks in comparison to the medium and large patches (Figure 9).

There were significant peaks in fire extent in 1992, 1995, 1998 and 2007 and significant troughs in 1993, 1999, 2003 and 2013 (Figure 9) for all forests. Standard deviations of individual fire extent graphs can be found in Appendices D, E and F.

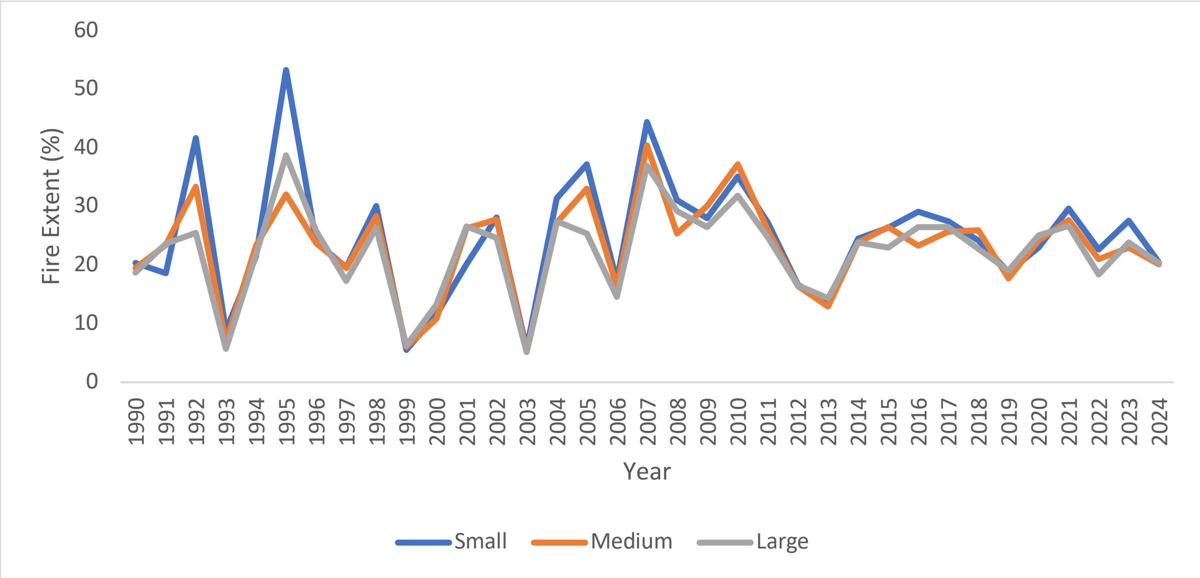

Figure 9: Mean fire extent in percentage of 30 Afrotemperate forests of various sizes in KwaZulu-Natal, South Africa, from 1990 to 2024. The blue line represents small forests (~1 ha), orange represents medium forests (~ between 1 and 1000 ha) and the grey line represents large forests (~1000 ha).

Fire Frequency

All forests and their matrices burnt at least once during the 34 years of this study but some areas burnt more regularly than others (Figures 10, 11 and 12). All forest patches burnt less in comparison to their surrounding matrix (Figures 10, 11 and 12). Overall the medium forest patches matrices burnt the most frequently (Figure 10) and the small patches burnt the least frequently (Figure 11). All of the medium and large patches burnt for less than five seasons whereas the smaller forests burnt between 11 and 20 seasons (Figures 10, 11 and 12).

The matrix of patch one burnt most frequently and the matrix of patch 10 the least frequently for the small forests (Figure 10). The matrix of patch eight burnt the most frequently and the matrix of patch two the least frequently for the medium forests (Figure 11). The matrix of patch six's burnt most frequently and the matrix of patch 10 burnt the least frequently for the large patches (Figure 12).

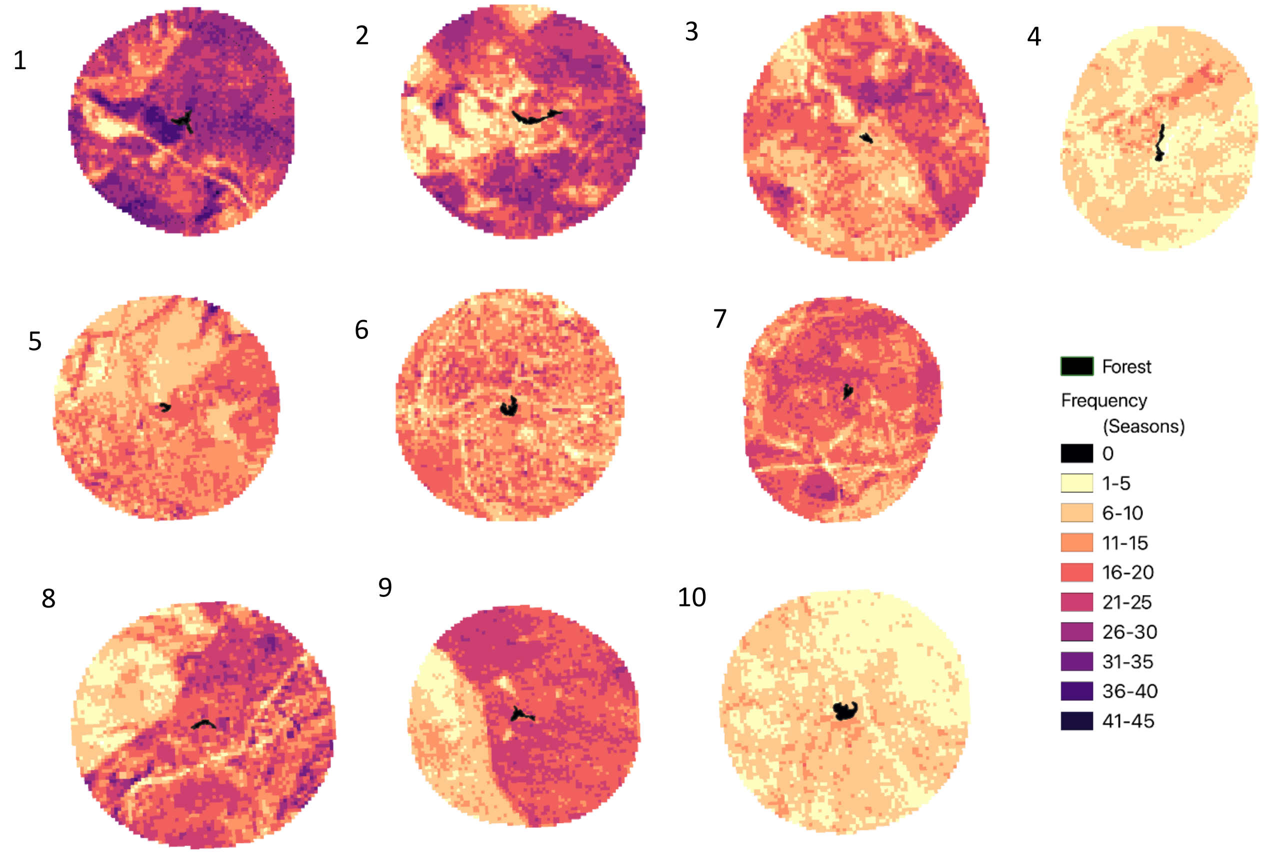

Figure 10: Seasonal fire frequency patterns, excluding the low severity class, around 10 small forests (~1 ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.

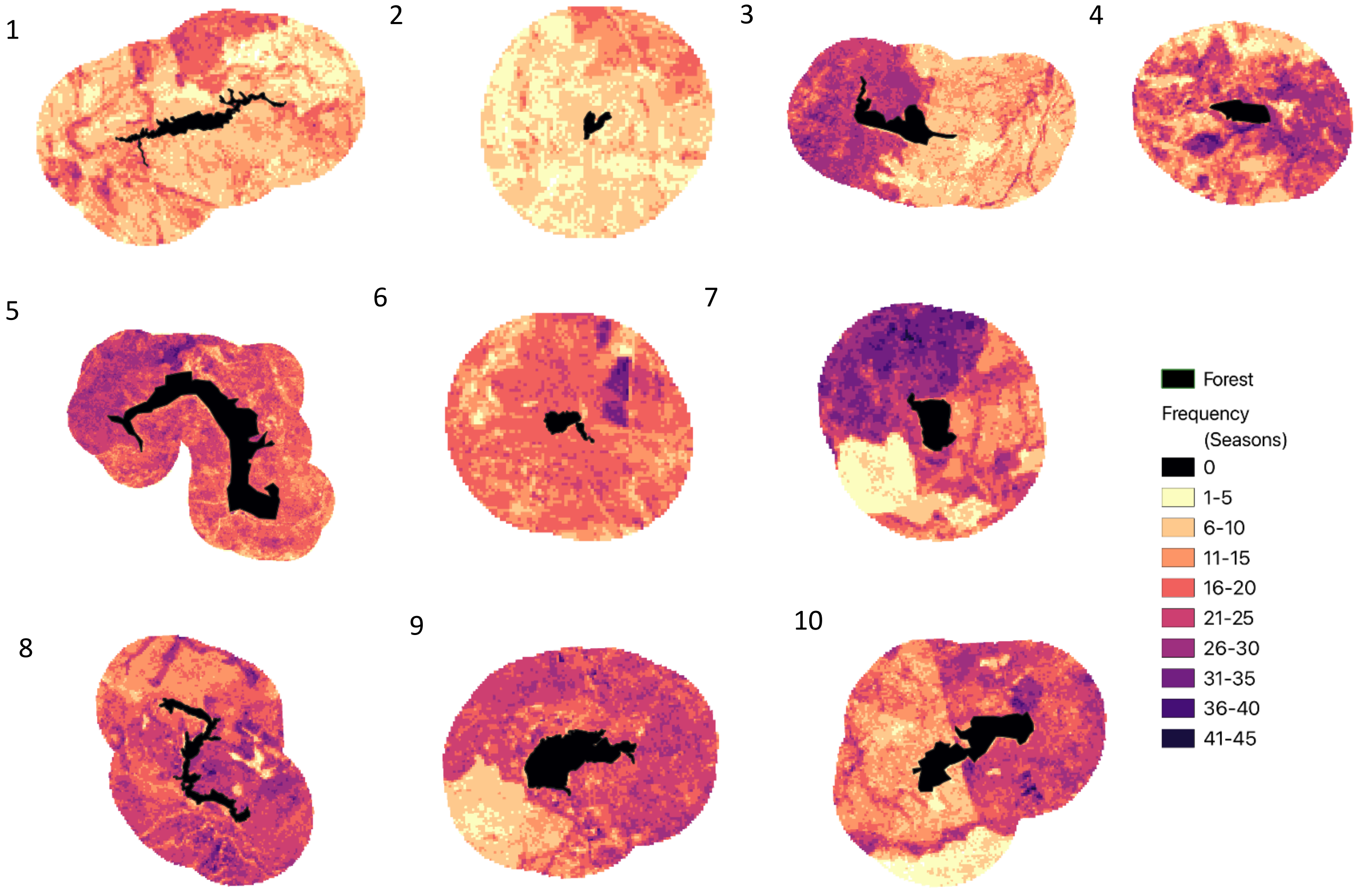

Figure 11: Seasonal fire frequency patterns, excluding the low severity class, around 10 medium forests (~ between 1 and 1000 ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.

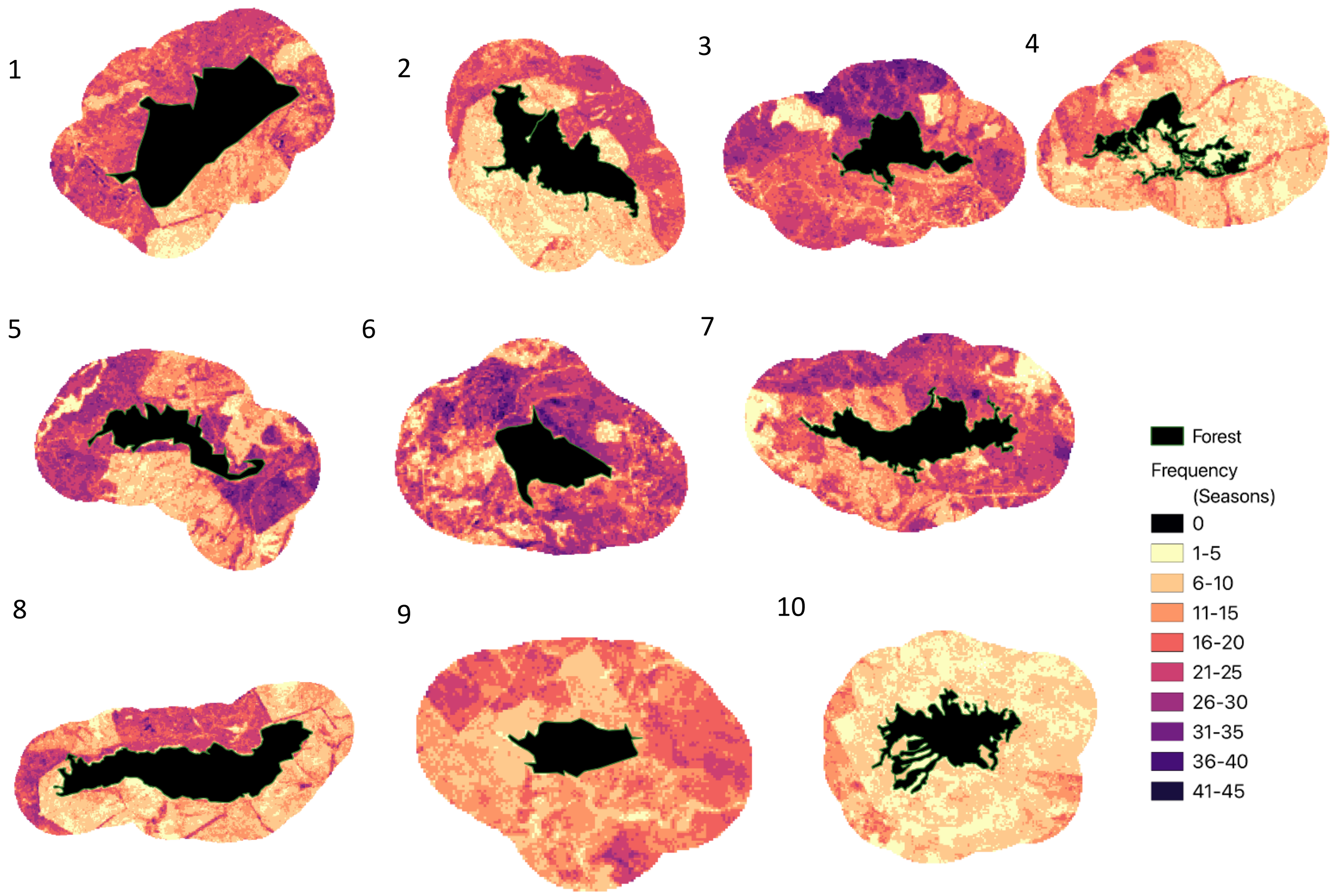

Figure 12: Seasonal fire frequency patterns, excluding the low severity class, around 10 large forests (~1000 ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.

Forest Change: GLMMs

Forest and land cover

There was significant relationship between the extent of land cover of the matrix and land cover type with changes in forest extent ($p < 0.05$). Any changes in land cover ($p = 0.04$) and land cover's extent ($p= 0.03$) were significant for changes in large forests' extent. Changes in the herbaceous vegetation land cover was significant for changes in forest extent for small ($p= 0.03$) and large forests ($p= 0.001$) and changes in water was significant for the changes in forest extent of medium ($p= 0.02$) and large forests ($p=0.003$). Further information about the statistical results can be found in Table 9.

Forest and fire

The relationship between changes in forest extent (dependent variable) and fire (independent variable) was significant for the small forests ($p = 0.01$) and large forests ($p = 0.001$). There was no significant relationship found for the medium forests ($p > 0.05$). Further information about the statistical results can be found in Table 9.

Fire and land cover

No significant statistical relationship was found between changes in fire (dependent variable) and land cover (independent variable) for any of the forest patches ($p > 0.05$). However, when doing an ecological analysis, I found that patches that had the highest fire extent and frequency tended to have herbaceous vegetation as the dominant land cover class in the surrounding matrix and patches that had the lowest fire extent or frequency had non-indigenous or indigenous vegetation as the dominant land cover class in the surrounding matrix for all forest patches (Table 10). Further information about the statistical results can be found in Table 9.

Forest, fire and land cover

No significant statistical relationship was found between forest extent (dependent variable), fire and land cover (independent variables) for any of the forest patches ($p > 0.05$). Ecologically, small and large patches increased in size irrespective of high or low fire extent and frequency or if they were surrounded by herbaceous or non-indigenous vegetation (Table 10). Medium patches had differing responses to fire frequency and extent with patches increasing in extent at low and high fire extents but decreasing in extent for low and high frequencies (Table 10). Further information about the statistical results can be found in Table 9.

Table 9: Statistical results from the Generalised Linear Mixed Models (GLMM) of forest extent, land cover and fire extent of 10 small (~ 1 ha), 10 medium (~ between 1 and 1 000 ha) and 10 large (~ 1000 ha) Afrotemperate forests in KwaZulu-Natal, South Africa. The results reported are linked to the intercept of each model.

Model variables	**Coefficient Estimate**	**Standard Error**	**Degrees of freedom**	**T value**
Small forests				
Land cover + forest extent	1.01	0.16	13.14	6.13
Fire + forest extent	-0.24	0.21	13.38	-1.14
Fire + land cover	-0.22	0.24	11.46	-0.92
Forest + fire + land cover	-0.26	0.08	3.43	0.51
Medium forests				
Land cover + forest extent	61.59	31.69	9.57	1.94
Fire + forest extent	3.34	0.40	9.09	8.35
Fire + land cover	3.34	0.40	9.09	8.35
Forest + fire + land cover	3.29	0.10	0.003	31.9
Large forests				
Land cover + forest extent	271.63	62.08	9.05	4.38
Fire + forest extent	5.39	0.21	9.11	25.31
Fire + land cover	5.39	0.22	9.52	25.01

Forest + fire + land cover	5.41	0.06	0.04	84.61

Table 10: Ecological analysis of changes in forest extent land cover of small (~ 1 ha), medium (~ between 1 and 1000 ha) and large (~ 1000 ha) forest patches with the lowest and highest fire extent and frequency in KwaZulu-Natal, South Africa.

Size category	**Dominant land cover class of patch with highest fire frequency and extent**	**Dominant land cover class of patch with lowest fire frequency and extent**	**Overall forest extent change of patch with the highest fire frequency and extent**	**Overall forest extent change of patch with the lowest fire frequency and extent**
Small	Herbaceous vegetation	Non-indigenous vegetation	↑	↑
Medium	Herbaceous vegetation	Herbaceous vegetation and non-indigenous vegetation	↑ Fire Extent ↓ Fire Frequency	↑ Fire Extent ↓ Fire Frequency
Large	Herbaceous vegetation	Indigenous forest	↑	↑

Discussion

Small forest patches had the highest increase in size (Table 3), the most exotic vegetation invasions (Table 3) and became the most spread out (Figure 3) of all the forests. The small forests also had the highest fire extent (Figure 9) and the forest patches burnt the most (Figure 10). The medium forests were the only forests with an overall decrease in size (Table 4) and had an intermediate response in composition and shape in comparison to the other size categories of forests. Medium patches had the highest increase in non-indigenous vegetation in their surrounding matrix (Figure 7). The medium forest's matrices burnt the most frequently (Figure 1). The large patches remained the most natural (Table 5) and remained the most compact (Figure 5) out of all the forests. The large patches also had the lowest increase in non-indigenous vegetation (Figure 8). The large patches also had the lowest fire extent (Figure 9). All forests had a reduction in herbaceous vegetation extent and an increase in non-indigenous trees extent in their surrounding matrix (Figures 6, 7 and 8) and most changes in land cover occurred from period one (1937 – 1978) to period two (1975-2000) for all patches (Tables 5, 6 and 7).

Both small and large forests increased in forest extent; however, the amount and composition of their expansions was different (Tables 3 and 5). The small forests increased in extent almost nine times more than the larger forests (Tables 3 and 5). Sixty percent of the smaller forests' composition changed from natural to mixed whereas only 20% of the large forest's composition changed from natural to mixed (Tables 3 and 5). The smaller forests were infiltrated with non-indigenous vegetation which causes forest degradation (Pamla et al., 2024). Many developing countries are struggling with the degradation of natural forests and the loss of ecosystem services that these forests provide (Mengist et al., 2021). Invasive vegetation such as Bugweed is one of the main invasive species found in South Africa and is known to transform habitats (Pamla et al., 2024; Witkowski and Garner, 2008). Alien plants outcompete indigenous vegetation for sunlight and water (Mavimbela et al., 2018; Meijninger and Jarmain, 2014), which increases non-indigenous vegetation's growth and distribution. This can alter many ecological processes that occur within the natural forests such as pollination and seed dispersal as non-indigenous vegetation may fruit earlier and in larger amounts (Pamla et al., 2024). An example of this can be seen with the samango monkey (*Cercopithecus mitis),* which is an endemic species to South Africa and is an important seed disperser in Afrotemperate forests (Linden et al., 2015). Samango monkeys are habitat generalists and are able tolerate a large range of habitat conditions (Castley and Kerley, 1996). Therefore, when non-indigenous vegetation is present, samango monkeys are more likely to reside in non-indigenous vegetation instead of indigenous forests and eat the fruit from non-indigenous vegetation, spreading non-indigenous fruits' seeds (Castley and Kerley, 1996). A disruption in the natural animal-plant dispersal mutualism found in Afrotemperate forests may further reduce the indigenous Afrotemperate forest's distribution and the biota that rely on this habitat for survival (Castley and Kerley, 1996; Linden et al., 2015; Pamla et al., 2024). A study done by Kotze and Lawes (2007) found that ecosystem processes that are essential to habitat functionality such as tree recruitment, litter decomposition and regeneration were possible in indigenous forests irrespective of their size but not in woodlands with non-indigenous vegetation (Kotze and Lawes, 2007). Species such as the Natal midland dwarf chameleon (*Bradypodion thamnobates)* or the Cape Parrot (*Phoicephalus robustus*) rely on intact Afrotemperate forests for survival are threatened when the natural vegetation and functionality of Afrotemperate forests change (Castley and Kerley, 1996). Changes in Afrotemperate forests' composition can also have implications on the livelihood of people that rely on Afrotemperate forests for food, medicine, fodder for livestock and cultural values (Guillozet et al., 2014; Mengist et al., 2021). Therefore, while small forests increased in extent, the ecosystem processes and biodiversity in these forests may have changed due to alien plant invasion which puts these forests at high risk of forest degradation.

Expansion of large forests was predominately natural as only 20% of the forest patches were infiltrated with exotic vegetation (Table 5). The surrounding matrix of the large forests also had the largest increase in agricultural land out of all the forests, between periods one and two (Table 8) and then a 37% reduction in agricultural land between the second and third period (Table 8). Evidence from the Wild Coast has shown that in former Transkei there has been a reduction in cultivated land by 79% from 1961 to 2009 which has been replaced by natural forest and woodland cover (Hoffman, 2014; Shackleton et al., 2013). This could explain why the larger forests did not only increase in extent in this study but also the fact that the forest vegetation found in their surrounding matrix also increased between the second and third time period (Table 8), potentially contributing to bush encroachment in the area (Hoffman, 2014; Shackleton et al., 2013). Natural forest expansion is due to a reduction in overgrazing, logging for firewood or timber, repeated fires, diseases or parasites and natural disasters (such as cyclones) in forests (Contreras-Hermosilla, 2000). Exotic plantations can also assist in nursing the establishment and succession of natural forest species as these trees can buffer the variable and extreme conditions of the surrounding matrix (Geldenhuys et al., 2017). The means that large forests being surrounded by non-indigenous plantations in their surrounding matrix could be a potential reason for the increase in the large forests' extent. The correct implementation of conservation management policies such as the National Forests Act (1998), National Veld and Forest Fire Act (1998) and the National Environmental Protection Act (1998) can assist in the reduction of overgrazing, logging and fires (Geldenhuys, 2002; Mensah et al., 2020; Republic of South Africa, n.d.) which can contribute to forest expansion. Globally, most funding initiatives towards forest conservation such as Reducing Emissions from Deforestation and Forest Degradation (REDD+) and the Clean Development Mechanism (CDM) focus on conserving larger and mostly intact forests (Guillozet et al., 2014) and ignoring smaller forests (Kotze and Lawes, 2007). Therefore, an increase in forest conservation could be the reason for the large forests' expansion however these factors were not studied in this study. While the implementation of these acts could be a possible explanation of the increases of forest extent seen in large forests, a further reduction in grazing and logging is still needed as they both are still major sources of land use change which impact forest extent. Larger forests are also more diverse and have larger population sizes therefore are more resilient to changes which might also be an explanation for the increase in the large forests' extent (Thompson et al., 2009).

The medium sized forests were the only forest category that overall, decreased in extent (Table 4). Medium forest patch also had the greatest net increase of 27.64% in non-indigenous vegetation in their surrounding matrix (Table 7). In the mid-1930s, South Africa had over 40, 000 ha of exotic plantations planted for timber, with the largest private tree plantations of Black Wattle being found in the former Natal province (Bennett and Kruger, 2013). These plantations included trees such as *Acacia, Hakea, Eucalyptus and Pinus* which had long term environmental impacts on the natural ecosystems. One of the ecosystems that were negatively impacted by exotic plantation were indigenous forests' habitat, structure and hydrology (Bennett and Kruger, 2013). Both *Eucalyptus* and *Acacia* have been found to reduce water supplies and have potentially contributed to the declining surface runoff, rainfall, desertification and desiccation in South Africa (Bennett and Kruger, 2013; Hoffman, 2014). Surveys done by the Agricultural Research Council on alien invasive plants found that KZN is one of the most invaded areas in South Africa with vegetation such as Bugweed (Kotzé et al., 2010) which may provide an explanation for the large amounts of alien plants found in this study. Alien invasive vegetation has invaded many of the savannas found in

KZN and in turn the matrices of many of the Afrotemperate forests (Hoffman, 2014), a possible explanation for the increase in non-indigenous vegetation in the surrounding matrix of the medium sized forests. Therefore, the 0.57% (Table 7) reduction in water availability due to the increasing presence of non-indigenous vegetation may be causing the death of the indigenous forests trees, decreasing forest extent. Due to forests having a long regeneration time and lifespan, the impacts of any of the measures that may cause forest reduction or expansion (besides physical removal measures such as fire or logging) may be time-lagged (Thijs et al., 2014). This could be a possible explanation why the highest net gain (20.4%) of the non-indigenous vegetation in the surrounding matrix of the medium patches (Table 7) happens between period one and two whereas the decrease in the water in the surrounding matrix (Table 7) and medium forests' extent by 5.92%, occurred between the second and third period (Table 4).

Smaller forests had the smallest decrease in perimeter-area ratio indicating that the smaller Afrotemperate forests in this study became more spread out with larger edges (Figure 3). An increase in a perimeter-area ratio causes an increase in forest edge. Increases in forest edges have been attributed to the effects of forest harvesting and natural disasters such as fire (Dupuch and Fortin, 2013; Pöpperl and Seidl, 2021). This could be a possible explanation for why smaller forests had the highest frequency of fire (Figure 10). Forest edges have different microclimatic conditions such as changes in light intensity, soil moisture and exposure to wind from the forest interior (Cardelús et al., 2013; Kotze and Samways, 2001; Lehouck et al., 2009). Forest edge effects can cause changes in the soil and foliar nutrients and the diversity of flora and fauna in the forest interior which can disrupt processes such as seed dispersal, recruitment and germination, potentially causing further reduction and degradation of indigenous forests (Cardelús et al., 2013; Lehouck et al., 2009). The differences in climatic conditions of the forest edge and forest interior may inhibit the expansion and regeneration of indigenous forest tree species (Lehouck et al., 2009). Forest edges tend to have a higher species richness in flora and fauna than forests interiors (Erdős et al., 2013; Krüger and Lawes, 1997), however a larger forest edge can cause a reduction in the biodiversity and ecosystem functioning of the forest interior (Castley and Kerley, 1996; Kotze and Lawes, 2007). The differences in the conditions of the indigenous forests caused by increased forest edges can aid the invasion of the non-indigenous vegetation into these forests as invasive species are more prominent on forest edges (McDonald and Urban, 2006) which would explain why the smaller forests had a more mixed composition (Table 3) in comparison to the other forest types .

Overall, the largest changes in land cover class of the matrix was the reduction of barren land and grasses and the increase in alien invasive plants and exotic plantations for all forests (Figure 6, 7 and 8). These results are similar to findings of a study done on Afrotemperate forests in Ethiopia by Mengist et al. (2021), which found that the greatest land cover conversions were found in forests, grassland and wetlands, and naturally most Afrotemperate forests are found in a grassland matrix. Most of the land cover conversion that occurred during my study occurred during the first (1937 – 1978) and second time period (1975-2000) (Tables 5, 6 and 7) which matches the time periods of other reports of landscape changes due to timber extraction and clearing of forest areas and the conversion of grasslands for agriculture and forestry from the 1960s to the 1990s (Castley and Kerley, 1996; Hill and Southworth, 2016). While statistically my study indicates that changes in herbaceous extent is more significant around larger forests than smaller forests, literature indicates that larger forests may be able to persist better against changes in land cover in their matrices due to being able to buffer the impacts of land use practices and disturbance regimes better in comparison to smaller forests (Castley and Kerley, 1996). Therefore, smaller forests are also more likely to be affected by the conversion of herbaceous vegetation into other land cover classes.

The small forests had highest net increase in urban cover (3.92%) out of all the forests (Table 6). Urbanisation tends to have large environmental consequences on forests that can lead to their decline and degradation (Pregitzer et al., 2020). Studies in Ethiopia and Zimbabwe found that rapid population growth was one of the main reasons for decline and degradation of natural forests and woodlands (Mengist et al., 2021). The habitat alterations caused by urbanisation can be in the form of increasing forest exposure to human-induced fire and the facilitation of invasion by alien species into forests (Referowska-Chodak, 2019). The changes in the habitat conditions due to the introduction of atmospheric and soil pollutants can cause changes in species diversity and aid in the invasion of non-indigenous vegetation (Referowska-Chodak, 2019). These changes can explain the high invasion of non-indigenous plants into and surrounding indigenous forests' canopies (Table 3). The higher fire extent (Figure 9) of the smaller forests can be as a result of urbanisation in combination with the effects of drier conditions due to climate change and the degradation of these forests through harvesting of non-timber forests products and the invasion of non-indigenous species, making the forests more vulnerable to fires (Powell et al., 2023).

Overall, there were two main trends found for the fire results of this study: pre-2013 and post-2013 (Figure 9). Data on meteorological droughts and floods indicate that there were fluctuations in dry and wet years from 1990 to 2013 which would explain the high variation found in this time period (Appendices D, E and F) (Buthelezi et al., 2016). KwaZulu-Natal was reported to be drier from the years 2013 to 2020, a possible explanation why there is a 0.71% higher fire extent in comparison to pre-2013 (Figure 9) (Buthelezi et al., 2016; Ndlovu et al., 2021). The high peaks in fire extent in 1993, 1995, 1998 and 2007/2008 can be explained by the moderate or extreme drought conditions during these years or the year before them (Buthelezi et al., 2016). Drought conditions alter the micro-climate of savannas, grasslands and forests which can cause the desiccation of the vegetation in these habitats, providing large amounts of flammable biomass which promotes the spread of fire (Giddey et al., 2022). This would explain the large increases in fire extent for all sizes of forests during for those years. The troughs in fire extent for years 1993, 1999 and 2013 can be explained by these years being wetter in comparison to their previous year (Buthelezi et al., 2016). Wetter conditions reduced the flammability of the fuel which would reduce fire's ability to spread, reducing the fire extent for that year (Giddey et al., 2022). The trough seen in 2003 is an outlier as it was a dry year therefore, we would expect an increase in fire extent. However, the reduction in fire extent was as a result of missing images from that year which skewed the results. It is also important to note that dNBR is not a direct measure of fire occurrence therefore the results presented may be as a result of changes in the vegetation's productivity and not necessarily fire occurrence.

Twenty three percent of the surrounding land cover of all the forests burnt during this study (Figure 9). Pre-1900, intense and severe wildfires were the main causes of forest destruction; however, there have been strategies put in place since then to reduce and control the number of intense and severe wildfires (Castley and Kerley, 1996). Post-1900, there has been an increasing record of woody biomass expansion across South Africa, but particularly in KZN, likely due to a reduction in fire severity and frequency (Hoffman, 2014). Increased fire suppression allows for woody plants to colonise surrounding savanna or grassland matrix, leading to bush encroachment (Abiem et al., 2020). While severe and intense fires may have decreased since pre- 1900, low severity fires are still commonly found in the surrounding grassland and savanna matrices of Afrotemperate forests which can creep into the forest interior and burn the thin leaf litter layers found on the ground without impacting the trees (Eriksson et al., 2003; Giddey et al., 2022; Kigomo et al., 2024).

The smaller forests had the highest peak in fire extent 1992 in comparison to the other forest types (Figure 9), which would indicate a severe fire occurred during 1992. While severe wildfires are less common post-1990, they still occur and can have large impacts on forest health and their ability to regenerate when their forest canopies are burnt and the trees' reproductive parts are destroyed (Kigomo et al., 2024). The destruction of the reproductive parts of trees can reduce the ability of trees to regrow and inhibit future recruitment (Giddey et al., 2022). This can increase the risks of creating non-natural canopy gaps in the forest interior which can disturb the natural canopy gap dynamics, risking further forest degradation (Giddey et al., 2022; Kigomo et al., 2024). The natural, closed canopy of indigenous Afrotemperate forests provides an microclimate that prevents the invasion from alien plants (Giddey et al., 2022). However, when gaps are present in the forest interior, there is a change in microclimate which can promote the invasion of alien plants such as *Acacia, Pinus,* and *Eucalyptus* species (Giddey et al., 2022). This is a possible explanation for the change in composition of the small forests in this study, as smaller forests are more vulnerable to the effects of fire. Forest patches that are frequently burnt can further degrade as there is not enough time for regeneration and sprouting of the natural vegetation before the next fire event (Giddey et al., 2022; Kigomo et al., 2024).

All forests had a minimum of 12% increase in the coverage of non-indigenous vegetation in the surrounding matrix (Figures 6, 7 and 8) during the time the study was done. Some of the common tree species found in exotic plantations such as *Pinus radiata, Eucalyptus camaldulensis* and Black Wattle, and other invasive vegetation such as Bugweed are known to be highly flammable (Giddey et al., 2022; Kraaij et al., 2024; Okoro et al., 2019). Exotic plantations in South Africa use lots of water and are highly-flammable which has contributed to many of the catastrophic fires that have occurred in the country (Kraaij et al., 2024). The flammability of these exotic plantations may be a contributing factor to the high fire extent and frequency in the surrounding matrices of some of the forests (Figures 11, 12 and 13). A study that looked at Afrotemperate forests in Kenya, found that areas that had low and high severity fires had a high prevalence of black wattle (Kigomo et al., 2024), possibly indicating that the presence of Black Wattle increases the number and amount of fire in the landscape. However in the current study, patches with the lowest fire frequency and extent tended to be surrounded by non-indigenous vegetation and forest vegetation whereas forests with the highest fire extent and frequency were surrounded by herbaceous vegetation (Table 10). This may be due to the implementation of firebreaks around plantations which prevents fire from entering the exotic plantations (Bartlett, 2012). The medium forests also had the lowest net reduction of herbaceous vegetation (39.54%) (Table 7) which could be a contributing factor to the high fire frequency in the surrounding matrix (Figure 11). Herbaceous vegetation is usually considered a fine fuel and allows fire to spread more easily in comparison to woody species (Wragg et al., 2018). Therefore, while non-indigenous vegetation is flammable and can contribute to the spread of fire, herbaceous vegetation is more flammable and is also likely to spread fire.

Individually land cover and fire impacted forest extent; however, not together. This could be due to there only being one set of land cover data that could be compared with the 34 years of fire data. A larger sample of land cover maps from 1990 to 2024 (fire data date range) could help in determining whether fire and land cover have a statistically significant relationship. Images from Landsat 5 and 7 images had reduced visibility which could be a potential reason for the high variability in the pre-2013 fire extent. Many of the Landsat 5 images had gaps in them due to poor image quality and the Landsat 7 images had gaps in them due to a fault in the Scan Line Corrector from May of 2003 (Ramjeawon et al., 2020; Robson, 2005). The data lost in those gaps could be skewing the data which could provide an explanation for the lack of a significant relationship between fire and land cover. While there may not be a statistical relationship between these variables there was a ecological one found between fire and land cover (Table 10). Medium forests, which had the highest amount of non-indigenous vegetation surrounding them, decreased in forest extent. Their reduction in extent could be due to non-indigenous vegetation using the surrounding water causing forest degradation. However it is also important to note that a decrease in forest extent could not be as a result of loss of indigenous tree cover but the clearing of non-indigenous trees within or around forests. Small forests had the highest increase in forest extent and the highest invasion of non-indigenous vegetation into the forests. Small forests were the only forests with a significant increase in urban cover which could be a contributing factor to high non-indigenous vegetation's invasion into the forests. Small forests also had the highest fire extent, which could be as a result of the fast growing, flammable invasive species that invaded these forests and their surrounding matrix. Lastly, the larger forest were the forests that remained the most natural and their matrices remained the most unchanged and had fire impact them the least in comparison to the other forests. However scale may be an important contributing factor to the differences seen between the small and large forests, as patterns at smaller scales are more easily detected in comparison to patterns at larger scales (Mogonong et al., 2023), which is a factor to consider for future studies.

Conclusion

The KwaZulu-Natal's Afrotemperate forests in this study have changed in extent and composition from the 1940s to current day. The type of change was dependent on the original size of the forests, with smaller and larger forests becoming larger in extent and medium forests decreasing in extent over time. The expansion of the smaller forest was predominately due to the invasion of non-indigenous vegetation while larger forests' expansion was mostly by indigenous vegetation. Smaller forests had the largest perimeter-area ratio, the highest fire extent and their forest patches burnt the most frequently. Increased exposure to fire can increase the gaps found within forest canopies and forest edge which alters the microclimate within the canopy and increase chances of invasion by exotic plants. The matrix of the medium forests' had the highest amount of non-indigenous vegetation and burnt the most frequently. An increase in non-indigenous vegetation can decrease the available water in the area which can contribute to the degradation and destruction of forests. The large forests remained the most natural and compact in size, had the lowest amount of non-indigenous vegetation in their surrounding matrix and the lowest fire extent. The larger forests also had the largest increase in agricultural cover in the surrounding matrix between period one and two but a decrease between period two and three. It has been found that abandoned agricultural land in KZN has seen an expansion of forest and woodlands which contributes to bush encroachment. Larger forests are less impacted by changes in fire and land cover which is a possible explanation for their mostly natural expansion. Statistically, changes in forest extent were attributed to changes in fire and land cover independently; however, not by the relationship between both fire and land cover. All forests had an increase in non-indigenous vegetation in the surrounding land cover. While non-indigenous vegetation is more flammable than indigenous vegetation, the use of fire breaks by plantation managers may explain why patches that were the least frequently burnt had non-indigenous vegetation as the dominant land cover class. Small and medium sized Afrotemperate forests are under a larger threat of forest loss and degradation in comparison to larger forests. Both small and medium Afrotemperate forests provide important ecosystem services and biodiversity and therefore will require more attention from conservationists.

For future studies, I recommend investigating the medium sized forests as two separate categories: ~10 ha and ~100ha, as these forests had mostly an intermediary response to changes in extent and composition (60% decreased and 40% increased in extent; and 60% mixed composition and 40% of them remained natural). From the results of this study, patches that were approximately 10 ha decreased in size whereas patches approximately between 1 and 1000 ha had an intermediary response. The patches that were approximately between 1 and 1000 ha remained predominately natural throughout the study. Having two medium sized category may help in understanding the size category of forests which are increasing and decreasing, which can be important for conservation efforts. I would also recommend using other fire detection indices such as Normalised Difference Fire Index (NDFI) in conjunction with NBR to determine if the overestimation of low severity fires is an issue solely linked to NBR or just fire detection around the Afrotemperate forests of KZN. A potential solution for over or under classification would be to calibrate the classifier to the environment that is being classified as was done by Fraser et al. (2017) who used Composite Burn Index (CBI) to calibrate their classifier (Fraser et al., 2017). Composite Burn Indices are commonly used in the field to asses burn severity in forests (Fraser et al., 2017). Therefore for future studies it would be advisable to incorporate ground truth data to increase the accuracy of the dNBR classifier. Lastly, I would recommend having more frequent time periods with fewer years in between them in order to have a better idea of when exactly forests increased or decreased in extent, exotic invasion occurred and the surrounding land cover changed. The incorporation of all of these suggestions can aid in providing more information for where forest conservation efforts should be focused.

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Appendices

Appendix A: Collection dates for aerial photos used for forest extent, composition and land cover classification. **S**= Small forests (~1 ha), **M=** medium forests (~between 1 and 1000 ha) and **L=** large forests (~1000 ha). The number after the forest size code refers to the forest number of that size category. $\mathbf{T_1}$**=** first time period, $\mathbf{T_2}$**=** second time period and $\mathbf{T_3}$= third time period.

Forest Code	Flight Year	Forest Code	Flight Year	Forest Code	Flight Year
S_1T_1	1978	M_1T_1	1944	L_1T_1	1966
S_1T_2	1992	M_1T_2	1978	L_1T_2	1981
S_1T_3	2016	M_1T_3	2016	L_1T_3	2006
S_2T_1	1944	M_2T_1	1937	L_2T_1	1967

S_2T_2 S_2T_3	1981 2016	M_2T_2 M_2T_3	1981 2016	L_2T_2 L_2T_3	1981 2016
S_3T_1 S_3T_2 S_3T_3	1944 1981 2016	M_3T_1 M_3T_2 M_3T_3	1944 1991 2016	L_3T_1 L_3T_2 L_3T_3	1962 1981 2009
S_4T_1 S_4T_2 S_4T_3	1944 1981 2016	M_4T_1 M_4T_2 M_4T_3	1972 2000 2016	L_4T_1 L_4T_2 L_4T_3	1944 1978 2016
S_5T_1 S_5T_2 S_5T_3	1944 1981 2009	M_5T_1 M_5T_2 M_5T_3	1967 1981 2009	L_5T_1 L_5T_2 L_5T_3	1963 1984 2016
S_6T_1 S_6T_2 S_6T_3	1944 1981 2009	M_6T_1 M_6T_2 M_6T_3	1937 1975 2016	L_6T_1 L_6T_2 L_6T_3	1972 2000 2016
S_7T_1 S_7T_2 S_7T_3	1963 1991 2009	M_7T_1 M_7T_2 M_7T_3	1953 1981 2009	L_7T_1 L_7T_2 L_7T_3	1967 1981 2010
S_8T_1 S_8T_2 S_8T_3	1953 1981 2009	M_8T_1 M_8T_2 M_8T_3	1962 1981 2009	L_8T_1 L_8T_2 L_8T_3	1955 1981 2013
S_9T_1	1967	M_9T_1	1966	L_9T_1	1962

S_9T_2 S_9T_3	1981 2016	M_9T_2 M_9T_3	1981 2016	L_9T_2 L_9T_3	1981 2009
$S_{10}T_1$ $S_{10}T_2$ $S_{10}T_3$	1967 1981 2016	$M_{10}T_1$ $M_{10}T_2$ $M_{10}T_3$	1955 1981 2013	$L_{10}T_1$ $L_{10}T_2$ $L_{10}T_3$	1944 1981 2016

Appendix B: Number of points used for accuracy assessment of land cover classification for buffered regions surrounding digested Afrotemperate forests in KwaZulu-Natal, South Africa. **S**= small forests (~1 ha), **M=** medium forests (~ between 1 and 1000 ha) and **L=** large forests (~1000 ha). The number after the forest size code refers to the forest number of that size category **T_1=** first time period, **T_2=** second time period and **T_3**= third time period.

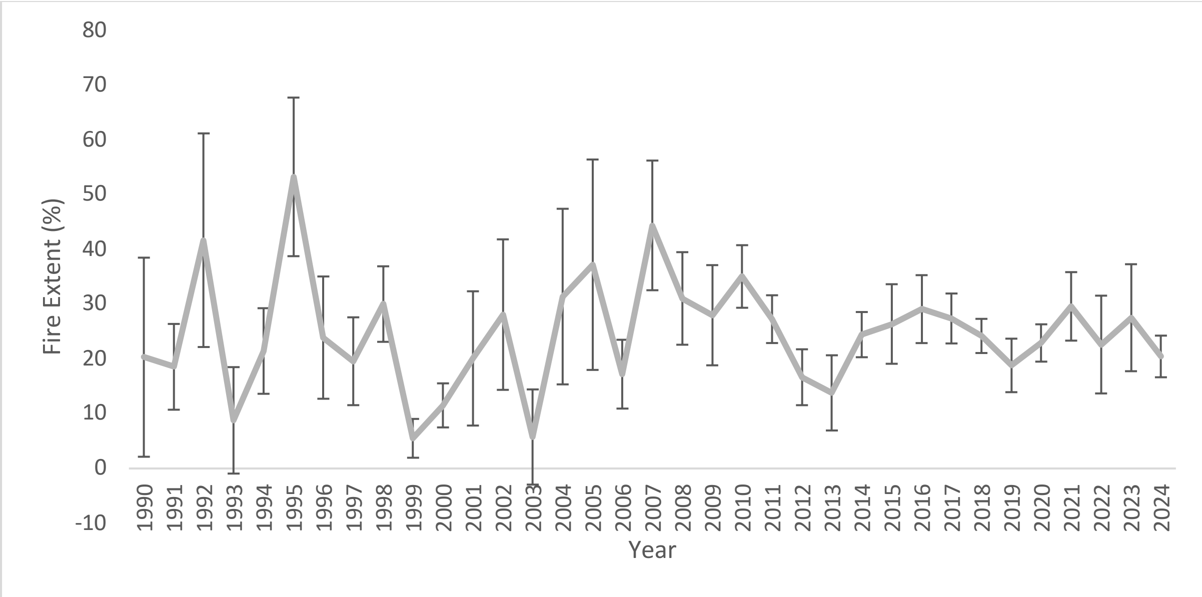
Forest Code	Number of Points	Forest Code	Number of Points	Forest Code	Number of Points
S_1T_1 S_1T_2 S_1T_3	112 115 102	M_1T_1 M_1T_2 M_1T_3	108 118 110	L_1T_1 L_1T_2 L_1T_3	106 105 100
S_2T_1 S_2T_2 S_2T_3	108 128 102	M_2T_1 M_2T_2 M_2T_3	100 100 105	L_2T_1 L_2T_2 L_2T_3	100 100 100
S_3T_1 S_3T_2 S_3T_3	111 127 121	M_3T_1 M_3T_2 M_3T_3	108 109 102	L_3T_1 L_3T_2 L_3T_3	100 109 125

S_4T_1	104	M_4T_1	115	L_4T_1	103
S_4T_2	107	M_4T_2	100	L_4T_2	110
S_4T_3	106	M_4T_3	103	L_4T_3	100
S_5T_1	110	M_5T_1	118	L_5T_1	109
S_5T_2	115	M_5T_2	124	L_5T_2	101
S_5T_3	109	M_5T_3	101	L_5T_3	129
S_6T_1	118	M_6T_1	100	L_6T_1	118
S_6T_2	100	M_6T_2	119	L_6T_2	109
S_6T_3	100	M_6T_3	136	L_6T_3	106
S_7T_1	106	M_7T_1	100	L_7T_1	100
S_7T_2	109	M_7T_2	100	L_7T_2	99
S_7T_3	115	M_7T_3	99	L_7T_3	141
S_8T_1	100	M_8T_1	119	L_8T_1	115
S_8T_2	103	M_8T_2	105	L_8T_2	118
S_8T_3	100	M_8T_3	115	L_8T_3	100
S_9T_1	103	M_9T_1	100	L_9T_1	123
S_9T_2	104	M_9T_2	100	L_9T_2	100
S_9T_3	102	M_9T_3	109	L_9T_3	101
$S_{10}T_1$	101	$M_{10}T_1$	100	$L_{10}T_1$	100
$S_{10}T_2$	100	$M_{10}T_2$	119	$L_{10}T_2$	110
$S_{10}T_3$	100	$M_{10}T_3$	110	$L_{10}T_3$	104

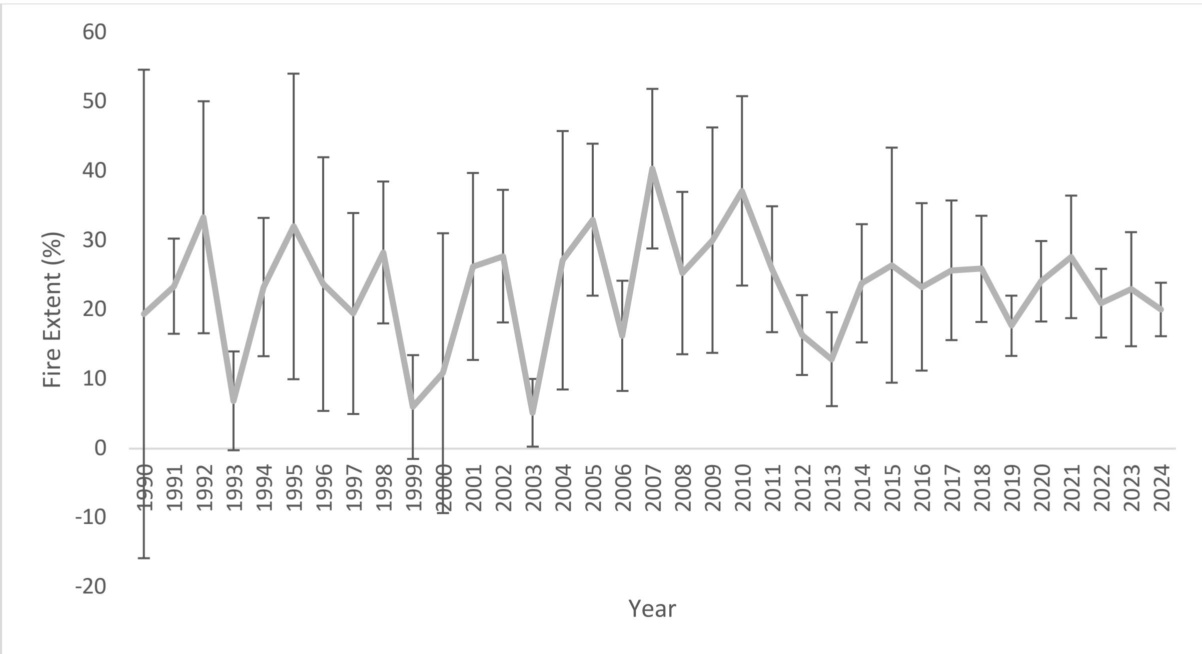
Appendix C: Kappa coefficient and producers (P) and user (U) accuracy for land cover classification of one kilometre buffer region surrounding digitised Afrotemperate forests in KwaZulu-Natal, South Africa. **S**= small forests (~1 ha), **M**= medium forests (~ between 1 and 1000 ha) and **L**= large forests (~1000 ha). The number after the forest size code refers to the forest number of that size category. $\mathbf{T_1}$= first time period, $\mathbf{T_2}$= second time period and $\mathbf{T_3}$= third time period.

Forest Code	**Kappa Coefficient**	**P and U Accuracy**	**Forest Code**	**Kappa Coefficient**	**P and U Accuracy**	**Forest Code**	**Kappa Coefficient**	**P and U Accuracy**
S_1T_1	0.83	0,93	M_1T_1	0,81	0,94	L_1T_1	0,72	0,82
S_1T_2	0,74	0,88	M_1T_2	0,83	0,88	L_1T_2	0,72	0,83
S_1T_3	0.75	0.89	M_1T_3	0,80	0,87	L_1T_3	0,71	0,81
S_2T_1	0,89	0,96	M_2T_1	0,75	0,9	L_2T_1	0,79	0,86
S_2T_2	0,83	0,88	M_2T_2	0,81	0,88	L_2T_2	0,83	0,89
S_2T_3	0,73	0,82	M_2T_3	0,74	0,82	L_2T_3	0,70	0,80
S_3T_1	0,72	0,91	M_3T_1	0,80	0,93	L_3T_1	0,94	0,98
S_3T_2	0,71	0,81	M_3T_2	0,82	0,89	L_3T_2	0,81	0,89
S_3T_3	0,81	0,93	M_3T_3	0,84	0,91	L_3T_3	0,75	0,86
S_4T_1	0,90	0,96	M_4T_1	0,85	0,90	L_4T_1	0,85	0,94
S_4T_2	0,81	0,91	M_4T_2	0.72	0,82	L_4T_2	0,78	0,85
S_4T_3	0,77	0,89	M_4T_3	0.81	0,86	L_4T_3	0,72	0,81
S_5T_1	0,73	0,93	M_5T_1	0,73	0,86	L_5T_1	0,76	0,86
S_5T_2	0,82	0,91	M_5T_2	0,77	0,85	L_5T_2	0,75	0,86

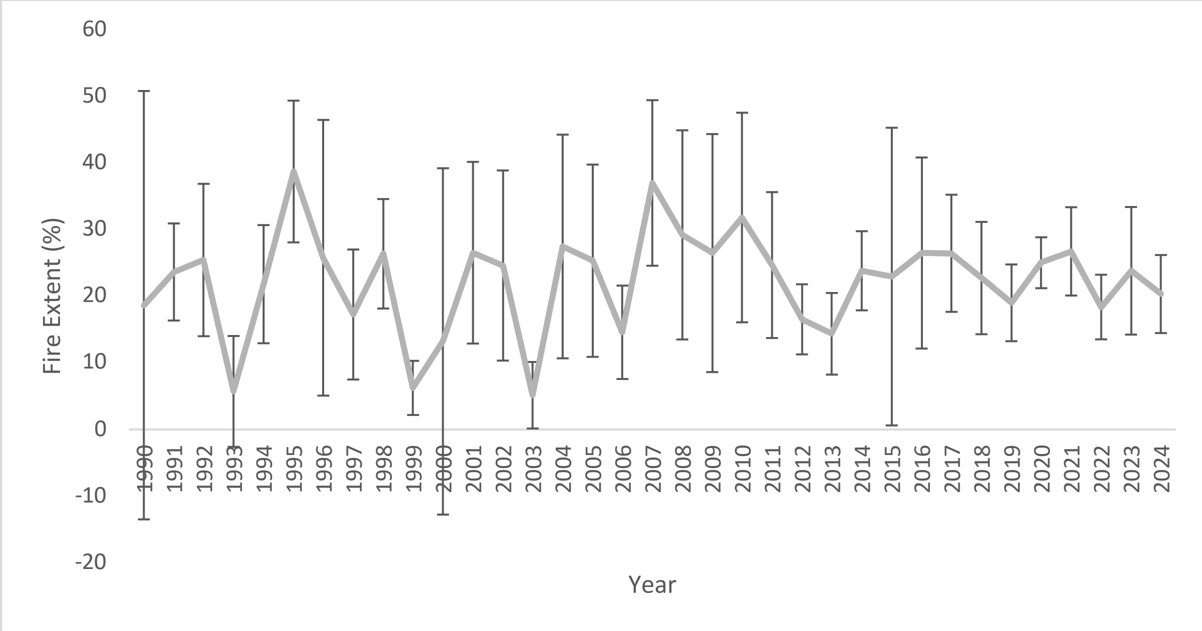
S_5T_3	0,86	0,92	M_5T_3	0,78	0,85	L_5T_3	0,72	0,80
S_6T_1	0,97	0,99	M_6T_1	0,71	0,95	L_6T_1	0,82	0,89
S_6T_2	0,74	0,82	M_6T_2	0,86	0,94	L_6T_2	0,71	0,82
S_6T_3	0,87	0,93	M_6T_3	0,75	0,84	L_6T_3	0,70	0,80
S_7T_1	0,73	0,96	M_7T_1	0,97	0,99	L_7T_1	0,88	0,93
S_7T_2	0,77	0,92	M_7T_2	0,79	0,87	L_7T_2	0,75	0,83
S_7T_3	0,94	0,97	M_7T_3	0,87	0,93	L_7T_3	0,79	0,84
S_8T_1	0,84	0,97	M_8T_1	0,87	0,95	L_8T_1	0,85	0,89
S_8T_2	0,80	0,91	M_8T_2	0,86	0,92	L_8T_2	0,72	0,77
S_8T_3	0,79	0,88	M_8T_3	0,82	0,89	L_8T_3	0,73	0,82
S_9T_1	0,86	0,95	M_9T_1	0,77	0,91	L_9T_1	0,76	0,85
S_9T_2	0,82	0,94	M_9T_2	0,88	0,95	L_9T_2	0,76	0,82
S_9T_3	0,81	0,91	M_9T_3	0,82	0,91	L_9T_3	0,77	0,83
$S_{10}T_1$	0,76	0,89	$M_{10}T_1$	0,74	0,92	$L_{10}T_1$	0,72	0,87
$S_{10}T_2$	0,77	0,84	$M_{10}T_2$	0,76	0,82	$L_{10}T_2$	0,71	0,76
$S_{10}T_3$	0,82	0,89	$M_{10}T_3$	0,92	0,95	$L_{10}T_3$	0,71	0,81



Appendix D: Mean and standard deviation of fire extent, excluding the low severity class, of percentage of 10 small Afrotemperate forests (~1ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.



Appendix E: Mean and standard deviation of fire extent, excluding the low severity class in percentage of 10 medium Afrotemperate forests (~ between 1 and 1000 ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.



Appendix F: Mean and standard deviation of fire extent excluding the low severity class in percentage of 10 large Afrotemperate forests (~1000 ha) in KwaZulu-Natal, South Africa, from 1990 to 2024.