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The impact of pine plantations on fynbos above-ground vegetation and soil seed bank composition

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ABSTRACT

Pine plantations and pine invasions have numerous impacts on native ecosystems in the Fynbos biome of South Africa. The severity of these impacts greatly determines the extent of potential ecosystem recovery after the pines are felled. The recovery potential of fynbos after felling of pine plantations of varying longevity and the subsequent application of ecological burns was investigated in the Helderberg Nature Reserve, Western Cape Province, South Africa. Above-ground vegetation, soil seed bank and abiotic variables were sampled across three treatments (reference fynbos and sites that had been under pines for 30 and 50 years respectively) using 1 m^2 quadrats placed along 50 m line transects. The soil seed bank samples were smoke treated and then monitored in a greenhouse to determine the soil seed bank species and growth form composition. Areas previously under 30 year old pine plantations had high native species and growth form density (number of species/growth forms per unit area) and similar plant density (number of individuals per unit area) to the reference fynbos areas. Conversely, areas previously under 50 year old pine plantations had significantly lower native species and growth form density and plant density than the reference fynbos and were dominated by alien species. In addition, areas previously under 50 year old pine plantations had lower species diversity than the reference fynbos areas and areas previously under 30 year old pine plantations which were found to be similar to one another. Felled pine plantations were shown to minimally impact on soil abiotic variables, with only soil temperature and pH showing significant differences. Therefore, areas previously under 30 year old pine plantations have higher recovery potential following pine removal than 50 year old plantations, owing to the depleted native soil seed bank in the latter. Consequently, active restoration may be needed to re-introduce the missing long-lived growth forms and to prevent soil erosion. Pine plantation and invasion management in the Fynbos biome should aim to fell pines before the native seed bank is depleted to maintain the recovery potential of fynbos and prevent the need for active restoration.

1. Introduction

Pine species are renowned invaders in the southern hemisphere having been introduced by humans for timber and other uses such as wind breaks (Richardson et al., 1994; Richardson and Higgins, 1998). Pines have invaded from these initial plantations (Richardson et al., 1994; Richardson, 1998; Richardson and Higgins, 1998; McConnachie et al., 2015). Alien tree invasions in South Africa, specifically across the Fynbos biome, are a serious problem that threatens both native biodiversity and water security (Le Maitre et al., 1996, 2002; Latimer et al., 2004; Richardson and Van Wilgen, 2004; Van Wilgen et al., 2008).

Invasive alien plants, including some pine species, can reduce ecosystem resilience of the invaded area by causing biotic and/or abiotic thresholds to recovery to be passed that may lead to a biotic/abiotic feedback threshold where the invader dominates the ecosystem (Gaertner et al., 2012, 2014). Pines can have a significant impact on abiotic variables such as fire severity and soil pH (Gaertner et al., 2012; Van Wilgen and Richardson, 2012; Mostert et al., 2016; Taylor et al., 2017). Pines can form dense stands which can survive for many years whilst excluding native plant species; this may also result in the native seed bank declining over time (Richardson and Higgins, 1998; Holmes and Marais, 2000; Gaertner et al., 2012). Pine trees are able to outcompete native vegetation for sunlight due to their large stature which leads to the gradual exclusion of native plant species beneath the pine tree canopy over time (Richardson and Van Wilgen, 1986; Maccherini and De Dominicis, 2003; Gaertner et al., 2012). Leaf litter produced by mature pine trees is far higher than that produced by fynbos which

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also contributes to the suppression of vegetation surviving in the understorey (Richardson and Van Wilgen, 1986; Gaertner et al., 2014). However, these pine-related impacts on the ecosystem are considered less severe compared with other invasive alien trees (e.g. *Acacia* species) and therefore areas invaded by pine species are believed to be easier to restore (Mostert et al., 2016).

When pine trees are cut down and the area burnt, these bare sites are often left to restore passively which can potentially result in soil erosion and reinvasion by alien plants (Holmes and Richardson, 1999; Holmes et al., 2000). Restoration of these bare sites depends on the presence of native seeds which have been naturally dispersed and persisted in the soil seed bank (Holmes and Newton, 2004; Heelemann et al., 2013). If these two sources of plant propagules are insufficient to restore the native species richness and functioning of the native ecosystem then these sites will need active restoration to re-introduce crucial native plant functional types (Holmes and Richardson, 1999; Montoya et al., 2012).

Much research has been undertaken on the impacts of other alien invaders (e.g. *Acacia saligna*) in the Fynbos biome on the species/guild richness of fynbos vegetation and seed banks. Pine tree effects on fynbos above-ground vegetation have been studied, but there has been little research concerning their effects on fynbos soil-stored seed banks (Richardson and Van Wilgen, 1986; Holmes et al., 2000). As current restoration practices mostly rely upon soil seed bank recruitment, it is crucial to understand when an area invaded by pines or under a pine plantation can be successfully restored passively and when active intervention (i.e. re-introduction of native species through either sowing or planting) is needed. This is especially important in the Western Cape where many pine plantations are being phased out as economically unviable, and the forestry industry is progressing towards more sustainable pine plantation management (Van Wilgen, 2015; Stehle, 2016). Mountain catchments are also extensively invaded by pines which need to be felled and the areas restored.

The overall aim of this study was to determine the recovery potential of fynbos when subjected to extended periods under pine plantations. The key study questions, which entail comparing fynbos sites and sites that were under pine plantations, were:

- Are there differences in native and alien species diversity, diversity by growth form and density among sites?
- Are there differences in surface cover and abiotic variables among sites?
- Are there differences in species and plant density between the above-ground vegetation and the soil seed banks among sites?

2. Materials and methods

2.1. Study area and sites

The study area was in the lower portion of the Helderberg Nature Reserve in the City of Cape Town, Western Cape, South Africa (34° 03′ 54.35″ S, 18° 52′17.79″ E; see supplementary material – Fig. S1). The vegetation type of the study area is endangered Cape Winelands Shale Fynbos (Rebelo et al., 2006). Fynbos is a fire-driven ecosystem that relies on fire for stimulating seedling recruitment (Kraaij and Van Wilgen, 2014). Fires in the southwestern Cape usually take place during the hot and dry summer months (Kraaij and Van Wilgen, 2014). An area that included fynbos and sites previously under pine (*Pinus radiata*) plantations were chosen for this study. The only recorded incidence of alien trees in the reference fynbos sites were small patches of *Eucalyptus* which were felled between 1992 and 1994. The pine plantations were planted in the 1960s with some sites being felled 30 years later between 1992 and 1994. These sites have subsequently had 20 years of recovery time whilst other sites were only felled in the winter of 2014 and thus were under pine plantations for about 50 years. All of the sites (reference fynbos and the sites previously under pines for 30 and 50 years) were burnt a year before the study, during the autumn of 2015. Scattered invasive alien species such as wattles, hakeas and pines are under maintenance control.

At each site, native and alien species composition was measured by sampling the above-ground vegetation and soil seed bank together with the abiotic variables. Possible edge effects were avoided by sampling more than 5 m away from the edge of each site. Field work was performed during January 2016 (mid-summer in South Africa) so that the condition of the stored soil samples could imitate the dry topsoil conditions experienced in the field during summer (Holmes and Cowling, 1997). The post-fire age of the vegetation was 8 months.

Each treatment (reference fynbos and the sites previously under pines for 30 and 50 years respectively) was replicated at 3 sites each. At each of the 9 sites, three parallel 50 m line transects (approximately 5 m apart) were used to position quadrats for sampling of the above-ground vegetation and soil seed bank. Three 1 m^2 quadrats were randomly positioned along each transect. To sample abiotic variables at each site, two transects were used, with 3 quadrats along the first transect and 2 quadrats along the second. The direction of each transect was arranged in accordance with the longest edge of each area sampled. Random numbers from a random number table were used to determine the distance among the quadrats that were placed along each transect (Holmes and Cowling, 1997). Google Earth was used to randomly pinpoint the starting points for each of the transects, with each of the end points being placed 50 m away conforming with the shape of the site. The areas sampled at each site were chosen to ensure that the sampling areas were between 150 m and 200 m apart to maintain independence among the sites.

2.2. Above-ground vegetation survey

Each quadrat was used to determine the above-ground species and growth form density and plant density, and projected canopy cover (Pierce and Cowling, 1991; Heelemann et al., 2013). Plant taxonomy followed that of Manning and Goldblatt (2012). The different species in each quadrat were identified and number of individuals counted. Each new plant species encountered was sampled and pressed in the field for identification using relevant field guides (Bean and Johns, 2005; Manning and Paterson-Jones, 2007; Bromilow, 2010; Manning and Goldblatt, 2012; Fish et al., 2015). In each quadrat, the projected plant cover for each species and the proportion of bare ground, rock, litter and pine debris was recorded using the Braun-Blanquet scale (Braun-Blanquet, 1964). Due to the logistics of this study, sampling took place during the dry summer months which may have resulted in an underestimate of annual plant species which usually emerge during the wetter winter months – these species were subsequently identified in the soil seed bank analysis.

2.3. Soil seed bank sampling

Using a manual metal soil corer, four soil core samples (50 mm in diameter and 100 mm in depth) were taken from each of the corners of the 1 m^2 quadrats and bulked together to cater for patchy seed distributions and to minimise variation (Vosse et al., 2008). The bulked soil cores from each quadrat were individually stored in a brown paper bag in a dry place at ambient temperatures for two months until mid-March 2016. As fynbos seedling recruitment occurs naturally during the wet winter months, the soil seed bank was assessed using the seedling emergence approach over this same time period.

These soil samples were then transported to a greenhouse with transparent roofing and shade-netting sides at the Department of Forest and Wood Science at Stellenbosch University. A basal layer of newspaper and then river-washed sand was placed into 81 seedling trays (27 cm long × 31 cm wide) with the bulked soil samples placed on top and spread to a depth of approximately 5 cm. An additional three seedling trays were filled with river-washed sand only to serve as control trays to detect any wind-borne seeds that may have germinated