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ORIGINAL PAPER

A landscape-scale assessment of the long-term integrated control of an invasive shrub in South Africa

Karen J. Esler · Brian W. van Wilgen · Kerry S. te Roller · Alan R. Wood · Johannes H. van der Merwe

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Abstract The control of invasive alien plants often involves the integration two or more approaches, including mechanical clearing, the application of herbicides, burning, and biological control. More than one species of invasive plant can threaten the same area, which necessitates prioritization in the allocation of scarce resources to support the control of different species. This paper describes the integrated control of the invasive shrub *Hakea sericea* over four decades in South Africa. The species is widespread across an area of approximately 800 × 200 km, and occurs mainly in rugged, inaccessible and fire-prone mountain areas. The species is serotinous, and produces copious amounts of seed that are wind dispersed after fires. We present a brief history of the control measures which included a combination of felling and burning, augmented by biological control. We used data from two surveys, 22 years apart, to assess changes in distribution and density of the species. The assessment suggested that the overall distribution of the species was reduced by 64%, from ~530,000 to ~190,000 ha between 1979 and 2001. The species either decreased in density, or was eliminated from 492,113 ha, while it increased in density, or colonised 107,192 ha. We conclude that initial programs of mechanical clearing were responsible for reducing the density and extent of infestations, and biological control was largely responsible for the failure of the species to re-colonize cleared sites, or to spread to new areas following unplanned wildfires. We propose that a significant portion of the resources used for clearing *Hakea* in the past can be reallocated to mechanical control efforts against other invasive species (such as alien pines) for which effective biological control options are not available, provided that sufficient resources are allocated to ensure the widespread and effective implementation of all biological control agents to maintain the advances reported on here.

Keywords Biological control · Mechanical control · Fire · Fynbos · *Hakea sericea* · Invasion management · Pines

K. J. Esler (✉) · K. S. te Roller
Centre for Invasion Biology, Department of Conservation Ecology and Entomology, University of Stellenbosch, Private Bag X1, Matieland 7602, South Africa
e-mail: kje@sun.ac.za

B. W. van Wilgen
Centre for Invasion Biology, CSIR Natural Resources and the Environment, P.O. Box 320, Stellenbosch 7599, South Africa

A. R. Wood
ARC-Plant Protection Research Institute, Private Bag X5017, Stellenbosch 7599, South Africa

J. H. van der Merwe
Department of Geology, Geography and Environmental Studies, University of Stellenbosch, Private Bag X1, Matieland 7602, South Africa

Introduction

The large-scale invasion of otherwise pristine ecosystems by alien plants is a growing global phenomenon (Mooney et al. 2005). These invasions can have serious impacts, and have often been subjected to prolonged attempts at control, which vary widely in their effectiveness. It is usually difficult to gauge the success of control efforts over the entire range of an invasive species over a long time, as control efforts are typically fragmented, discontinuous, and poorly recorded. Assessments of the effectiveness of control are nonetheless needed to guide ongoing control projects. Managers of such projects need to know whether their efforts are keeping the spread and density of invasive plants within acceptable limits, whether ongoing control should be continued, or whether scarce resources should be re-directed to address the control of other species where returns on investment may be greater.

This paper focuses on the outcome of a concerted effort to control an aggressively invasive shrub, *Hakea sericea* Schrader (Proteaceae), over millions of hectares in a control program that spanned four decades. *H. sericea* is a shrub or a small tree 2–5 m in height (Gordon 1993). It is native to south-eastern Australia, and became naturalized in South Africa following its introduction in 1833 (Fugler 1983; Neser and Fugler 1983). It invades the Cape fynbos shrublands, a species-rich, fire-prone vegetation type typical of the sandstone mountains and lowlands of the region. Where it invades, *H. sericea* forms dense impenetrable stands (van Wilgen and Richardson 1985), and causes several negative impacts. These include a significant reduction in indigenous species diversity (Macdonald and Richardson 1986), reductions in surface water resources (Versfeld and van Wilgen 1986; Le Maitre et al. 1996), and increases in biomass, fuel loads and the intensity of wildfires (van Wilgen and Richardson 1985).

The success of *H. sericea* as an invader can be attributed to its prolific seed production and serotinous habit (Richardson et al. 1987; Morris 1989; Kluge and Neser 1991). All of the seeds of *H. sericea* are retained in pairs in tough woody follicles which accumulate throughout the life of the plant. These seeds can occur at densities reaching 7,500 per m^2 in ash beds following a fire (Gordon 1993; Kluge and Richardson 1983). The wind-dispersed seeds are released *en masse* following the death of parent plants in fire. The seeds usually fall within tens of meters of the parent plants (Richardson et al. 1987), but some may be blown for much greater distances, establishing new foci and complicating control (Moody and Mack 1988). In the same habitats, invasive alien pines (such as *Pinus radiata*) have seeds that are morphologically similar to *Hakea* species, and these can disperse more than 1 km from source (Richardson and Brown 1986). The control of *H. sericea* in South Africa has involved a range of approaches, including mechanical and biological control, and burning (see below).

In this paper, we report on the history of integrated control efforts aimed at the reduction of *H. sericea* and its impacts, and we compare the extent of invasion between two surveys 22 years apart, to assess the effectiveness of the control efforts. Our goal was to assess, if possible, the contribution of integrated control, as well as to make recommendations on the future of the control program.

A brief history of the control of *Hakea sericea*

The importance of *H. sericea* as a weed has long been recognised (Wicht 1945; Stirton 1978), and it was declared as a "noxious weed" in terms of South African legislation in 1937. In early control attempts, invading plants were cleared by felling. As the species does not re-sprout after felling, herbicides were not required to treat the cut stumps. After felling, the follicles open to release the seeds, which either germinate or are eaten by rodents. If the area is burnt 12–18 months after felling, the crop of seedlings is destroyed. While this method has proved very effective, it was not used consistently until the mid 1970s. At that time, the then Department of Forestry embarked on an ambitious campaign to clear infestations from the mountain areas under its control (Fenn 1980). The exact area treated was never recorded, but the goal was to treat all mountain areas between Cape Town and Port Elizabeth (an area of approximately 800 × 200 km) using the fell-and-burn approach over the next 15 years. The clearing program was integrated with a policy that sought to divide the entire area into blocks, which were scheduled for prescribed burning on a 12 year cycle (Bands 1977). The plan was closely adhered to during the late 1970s, but the impetus was later lost in the 1980s as a result of declining funding, as well as stricter restrictions on the use of prescribed burning (van Wilgen et al. 1997). Later (in 1996), projects to clear invasive plants were resurrected under the banner of the Working for Water program (van Wilgen et al. 1998, 2002). These projects, however, focussed on other invasive alien species such as Australian wattles and boreal pines. For example, 35 and 10% of the national budget of the Working for Water program was spent on wattles and pines, respectively, compared to <2% on *Hakea* species (Marais et al. 2004).

Biological control of *H. sericea* began in 1962 with the introduction of seed-attacking insects from Australia (Kluge and Neser 1991). These insects were *Erytenna consputa*, a weevil which destroys developing fruits and *Carposina autologa*, a moth which destroys the seeds within mature, woody fruits. In 1979, the weevil *Cydmaea binotata*, which targets growing vegetative plant parts was also introduced. This weevil failed to establish at most sites where it was released, and failed to have any noticeable impact where it did (Kluge and Neser 1991). It has apparently died out, except possibly at one site (Gordon 1999). In addition, the bark-attacking indigenous fungus *Colletotrichum acutatum* (Lubbe et al. 2004) was noticed killing plants in 1969, and was subsequently developed as a bioherbicide to be applied to either mature plants (Morris 1983) or seedlings (Morris 1989). The success of the fungus has not been evaluated, though a number of workers have recorded high levels of mortality in stands of *H. sericea* caused by this fungus with natural infection levels (Fugler 1983; Richardson and Manders 1985; Gordon 1993). Two further insect agents have been recently released, the stem boring longhorn beetle *Aphanasium australe* in 2001 (Gordon 2003), and the flowerbud-feeding weevil *Dicomada rufa* in 2005 (Kluge and Gordon 2004). It is anticipated that these last two agents will contribute further to long-term control.

Methods

Comprehensive data on the distribution and density of *H. sericea* infestations were collected during two surveys 22 years apart. The first survey took place between 1976 and 1978, when Fugler (1979) mapped the distribution and density of *H. sericea* across its known distribution range, using 1:500,000 topographical maps. He divided his study area into one minute squares (approximately 1.6 × 1.6 km) and assigned a density category to each square. Values were assigned to squares based on a questionnaire-based survey involving all managers of mountain land throughout the area; an earlier survey completed by the Department of Forestry in 1977, in which all mountain slopes were surveyed using binoculars to identify infestations and scattered plants; and information collected from colleagues and members of the Mountain Club of South Africa. The categories used in Fugler's (1979) survey were: absent; scattered individual plants; a thicket of plants covering <1 ha; and a thicket of plants covering >1 ha. The recorded grid-cell value therefore represents the densest stand encountered within each ~256 ha grid cell.

The second survey took place between 1991 and 2001, when data on the distribution of all species of Proteaceae were collected for the Protea Atlas Project. The Protea Atlas Project database contains a total of 673 taxa in the family Proteaceae, and included indigenous and alien species. The data were collected by 478 volunteers over 10 years and include over 250,000 species occurrence records. The database is considered to be a nearly complete inventory of Proteaceae in the Region (Grand et al. 2007). Data included the GPS coordinates of a randomly selected central point in a *H. sericea* stand representing a circular area with radius of 500 m (~79 ha), as well as an assigned density category. The density categories were: Absent (0); dead plants only; 1–10 individuals recorded as the actual number; 10–100 individuals; 100–10,000 individuals; and >10,000 individuals. Considering that the Protea Atlas Project had many contributors over 10 years, points overlapped and an irregular point-data network resulted, with density values different from the earlier data set.

To facilitate comparison between the two surveys, the later density estimates had to be standardized in both measurement units and in spatial resolution to that of the earlier set. Consequently, the 2001 data were equated as shown in Table 1. A dense *H. sericea* stand would consist of individual plants growing not more than ~1 m apart. Therefore, 1 ha would support 10,000 or more individuals, equivalent to Fugler's (1979) densest category. These data were

Table 1 Density categories used to map invasions by *Hakea sericea* in two studies in 1979 and 2001, and corresponding densities assigned for comparative purposes in this study

Density categories used for comparative purposes in this study	Density categories used in 1979	Density categories used in 2001
High	Stands >1 ha	>10,000 individuals
Moderate	Stands <1 ha	100–10,000 individuals
Low	Scattered individuals	1–10, or 10–100 individuals
Absent	Absent	Dead shrubs only, or absent

Table 2 Values assigned to grid cells in a GIS analysis to indicate the degree of change in density classes mapped in two surveys in 1979 and 2001

		Comparative density recorded in 2001			
		Absent	Low	Moderate	High
Comparative density recorded in 1979	Absent	0	−1	−2	−3
	Low	1	0	−1	−2
	Moderate	2	1	0	−1
	High	3	2	1	0

Positive values indicate declines, and negative values indicate increases in density, respectively

used to create a new points-data layer that could be compared directly in terms of absence, presence and abundance. Spatial standardization of the 2001 (finer-scale) data was obtained by converting the point data to polygon grid-format by overlaying the ~256 ha grid-cell structure over the irregular point-data set in GIS. The point with the highest density recording value was propagated as density attribute to the containing grid cell, assuring the most conservative invasion density estimate. The degree of difference in density between the two time periods was assessed as the difference in number of density categories, as shown in Table 2 (these changes could be either positive or negative depending on the direction of change).

Results

Our analysis suggested large changes in the overall distribution of *H. sericea* between the two surveys.

Table 3 The total area (ha) occupied by *Hakea sericea* infestations in three density categories (see Table 1) estimated in two surveys in 1979 and 2001

Density	1979	2001
Low	254,789	140,521
Moderate	154,419	43,291
High	121,736	4,500
Total	531,229	191,094

Table 4 Area experiencing varying degrees of change in density between two surveys of *Hakea sericea*

Degree of change (number of classes difference between 1979 and 2001)	Possibilities	Area (ha)
3	High to absent	98,067
2	Moderate to absent or high to low	131,983
1	Low to absent; moderate to low; high to moderate	262,063
0	No change	31,929
−1	Absent to low, low to moderate, moderate to high	84,839
−2	Absent to moderate or low to high	19,912
−3	Absent to high	2,441

The species was recorded on ~190,000 ha in 2001, a reduction of 64%, compared to more than 530,000 ha in 1979 (Table 3). In 1979, high density infestations were recorded from ~120,000 ha, and this decreased by 96% to only 4,500 ha in 2001, while the corresponding decreases in the moderate and low categories were 55 and 28%, respectively (Table 3). The species either decreased in density, or was eliminated from ~490,000 ha between 1979 and 2001 (Table 4). About half of this decrease (~230,000 ha) represented a shift of two or more classes, in other words from either high or moderate levels of infestation to absent, or from high to low levels of infestation. The species was also recorded as either appearing in new areas, or increasing in density over ~107,000 ha (Table 4). About 80% of this was an increase of only one density class. A small amount of land (~32,000 ha) showed no change. Overall, our comparison suggests that *H. sericea* became less widespread and less dense over most of its range between the two surveys.

Discussion

The effectiveness of mechanical clearing

The substantial decreases of two or more density categories observed between 1979 and 2001 would almost certainly have come about largely as a result of active clearing and burning, rather than from biological control. The insect biological control agents that have been released would have reduced the seed loads of living plants, but would not have killed adult plants. Observations from several sites suggest strongly that the fell-and-burn technique is extremely effective, resulting in large reductions in the density of invasive *Hakea* shrubs at sites (van Wilgen and Kruger 1981; Holmes et al. 2000). On the other hand, much smaller areas experienced an increase in the density of invasion, with 2,500 ha going from absent to high, and ~20,000 ha going from either absent to moderate, or low to high. We assume that this would have happened in areas where no control effort had been made (either remote areas, or privately-owned land where government clearing programs were not active). Our results support the observation that the initial clearing programs in the late 1970s and early 1980s contributed to sharp declines in density of *H. sericea* stands over almost half of its range.

In the mid-1980s, the prescribed burning program, a key part of the *Hakea* control program, started to fall behind schedule. For example, 39 and 41 prescribed block burns were carried out in 1981 and 1982, respectively, compared to 10 and 11 in 1987 and 1988, respectively (van Wilgen et al. 1990). The two reasons behind this were declining funds, and policies that were introduced to limit prescribed burning to a very narrow seasonal window for ecological reasons (van Wilgen et al. 1990, 1997). Over the past 20 years, fire regimes in the area have been dominated by wildfires, rather than prescribed burns. During this time, wildfires burnt 90% of the area, mostly in a relatively small number of very large fires (Forsyth and van Wilgen 2007). Management (felling followed by prescribed burning) was thus not a dominant feature of the landscapes infested by *H. sericea* over the past two decades.

The effectiveness of introduced seed-attacking insects

Hakea sericea is well adapted to survive repeated fires, and aggressive re-invasion of the cleared range, or even expansion beyond the 1979 range could have been expected when the rate of mechanical clearing slowed considerably. The widespread presence of seed-attacking insects provides a logical explanation for this not having happened (Table 3) despite significant declines in the intensity of the clearing program in the late 1980s (van Wilgen et al. 1990, 1997). A simulation model developed to examine the effects of biological control on the dispersal of *H. sericea* in fynbos areas predicted reduced population growth rates, reduced maximum seed dispersal distances and a reduction in the formation of new invasion foci (Le Maitre et al. 2008). The time between surveys in 1979 and 2001 (22 years) would be sufficient for at least one, and possibly two fires to have occurred (mean fire return periods in fynbos vegetation are around 12–15 years). In the absence of seed-attacking insects, populations could have been expected to (1) increase in density after each fire in areas where they already occurred, and (2) have established new foci in adjacent areas (within 1–2 km) through the dispersal of a small number of seeds to these new sites. The number of seeds available for both processes would have been orders of magnitude less in the presence of seed-attacking insects, than if they had been absent. Le Maitre et al. (2008) found that in mature (10 years post-fire) *H. sericea* plants, 99% of young follicles were destroyed by *Erytenna consputa*, while 80% of mature follicles were destroyed by *Carposina autologa*. Overall, seed production was reduced in these stands by around 95%. Their model predicted that this level of predation would have reduced population growth rates, maximum seed dispersal distances and the formation of new invasion foci, as noted above. In the virtual absence of mechanical control programs after the mid-1980s, the presence of seed-attacking insects provides an explanation for the failure of this previously aggressive invading species to expand its range.

The effectiveness of fungal infections

The impact of the indigenous pathogenic fungus *Colletotrichum acutatum* on populations of *H. sericea* has never been accurately assessed. Earlier attempts to model the impact of this fungus predicted 82% mortality in infected stands after 10 years, based on observations in a single stand (Richardson and Manders 1985). However, whether or not this level of mortality is currently achieved by natural spread is unclear, and the impact of the fungus appears to vary considerably across the climatic range of *H. sericea* in South Africa. The bioherbicides that were developed based on this fungus have not been used to any large extent following an initial wide dispersal effort. Nonetheless, the fungus maintains fairly high levels of infection in populations of *H. sericea* across a wide range, often resulting in considerable mortality of adult plants (ARW and BvW personal observation). These levels of infection presumably affect the ability of the plant to produce copious amounts of seed as well as thinning plant populations to some extent. These effects, when added to those of seed-attacking insects, would presumably have further reduced the rates of spread of *H. sericea* after fires. Active implementation, as required for bioherbicides, would increase the contribution of the fungus to the overall control of the weed.

Implications for management

The ability of *H. sericea* to increase in numbers, spread, and to establish distant foci, has been severely limited through biological control (Le Maitre et al. 2008). This, in turn, should significantly reduce the costs associated with both the clearing of sites prior to fire, as well as the follow-up operations that are necessary to clear seedlings after unplanned wildfires. The initial clearing of dense stands of *H. sericea* (75–100% canopy cover) costs about 70 US $/ha compared with about 7 US $/ha for sparse stands with 1–5% cover (Marais et al. 2004). The reductions in the density of *H. sericea* reduce clearing costs by 50–90% (Le Maitre et al. 2008; Moran et al. 2004). If managers were to continue to clear and burn areas invaded by *H. sericea* in future, considerable savings in cost and effectiveness could therefore be expected. However, given the observation that aggressive spread of the weed appears to have been arrested by seed-feeding insects, possibly augmented by an indigenous pathogenic fungus, managers should seriously consider whether further mechanical control would be necessary at all. In some cases, especially where there is competition for scarce resources to control a number of different invasive species, it may well be a viable option to leave the control of *H. sericea* to biological control alone. The species-rich fynbos areas that are invaded by *H. sericea* are simultaneously under threat from invasion by alien pine trees (*Pinus* species). Pines have similar ecological traits to hakeas, in that they retain seeds in serotinous cones, and in that seeds are released after fire and dispersed by wind. They also have very similar impacts on the environment. Pine species were considered at one stage as possible targets for biological control (Moran et al. 2000). This project had to be abandoned, however, as pines have significant commercial value, and the introduction of cone-feeding insects was considered a possible risk due to their potential as vectors for the spread of pitch canker (caused by *Fusarium circinatum*). The danger in fire-prone fynbos areas is that pines may simply replace *H. sericea* as the dominant invasive species, as the latter is brought under control. We would propose, therefore, that managers seriously consider the option of the mechanical control of *H. sericea* in some cases. In such cases, for example where funds are limiting, and *H. sericea* infestations are widespread and difficult to access, it may be effective to rely on biological control alone, and to focus any mechanical control effort on pines. Where invasive pines and hakeas co-occur (which is in most areas), managers could simply fell only the pines prior to burning, as this would reduce the costs of pre-fire treatment considerably, and in the long term could make a bigger contribution to the long-term conservation of these areas. A co-ordinated, comprehensive and long term control program maximising the impact of all the biological control agents needs to be implemented against *H. sericea*, to ensure that the level of control reported on here is maintained and improved. Some limited supplementary mechanical control would still be required in such a program, yet the overall cost will be reduced allowing the above suggested targeting of other species for mechanical control. Although limited in extent, the observed increase in density and invasion of previously clear areas, provides a warning that if this program is not implemented, *H. sericea* will become a serious weed again.

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