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Historical costs and projected future scenarios for the management of invasive alien plants in protected areas in the Cape Floristic Region

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ABSTRACT

Scarce funds for conservation need to be optimally used, yet there are few studies that record the costs and projected outcomes of major conservation efforts. Here we document the historical costs and extent of efforts to control invasive alien plants in the protected areas of the Cape Floristic Region of South Africa, a biodiversity hotspot of global importance. We also estimate the resources that would be needed to bring the problem under control within a reasonable timeframe, under a range of scenarios of funding, rate of spread, and management effort. Trees and shrubs in the genera *Pinus*, *Acacia*, *Eucalyptus*, *Hakea*, *Leptospermum* and *Populus* were estimated to cover >66% of 750 000 ha at various densities in 2014. Historical costs of attempts to control these invasions over the past 20 years amounted to ZAR 564 million (~38 million US$), most of which (90%) was expended on *Acacia*, *Pinus* and *Hakea* in that order. The estimated cost to bring remaining invasions under control was between ZAR 170 and 2608 million (~1.3 and 174 million US$), depending on the scenario. Only substantial increases in annual funding under a scenario of low spread (4%), and removal of some taxa from the control programme, would allow for control to be achieved in <20 years. Even with increased spending, control would probably not be achieved under less favourable but more probable scenarios. Our findings suggest that, unless bold steps are taken to improve management, then a great deal of money would have been, and will continue to be, wasted. The essential element of an improved management approach would be to practice conservation triage, focusing effort only on priority areas and species, and accepting trade-offs between conserving biodiversity and reducing invasions.

1. Introduction

It is well known that the many needs for conservation action cannot be met by available resources (Murdoch et al., 2011), and conservation actions therefore need to be prioritized (Wilson et al., 2007). Prioritization alone is also not sufficient to ensure optimal outcomes, and conservation scientists need to shift some of their attention towards the design of effective policies and frameworks for action. In addition, there is a growing realization that using funds to set aside land in protected areas will not in itself achieve goals unless a sufficient proportion of the available funds are utilized to reduce threats, including legal and illegal harvesting of natural resources, pollution, climate change and invasion by alien species (Wilson et al., 2007). Moreover, we may need to practice conservation triage to achieve effective outcomes, by focusing sufficient resources on those priority areas where goals can be achieved. Following the basic principles of conservation triage should not be seen as a defeatist conservation ethic, but rather as being no more than the efficient allocation of resources, and that by failing to follow the basic principles of triage, we would simply be wasting resources (Bottrill et al., 2008). Finally, although many existing conservation frameworks claim to emphasize efficiency or wise investment, few have examined the actual costs of interventions, leading to calls for conservation biologists to make a major effort to include and record the costs of conservation actions, so that returns on investment can be demonstrated (Murdoch et al., 2011).

The establishment and management of protected areas are key components of global strategies to conserve biodiversity. South Africa's Cape Floristic Region (CFR) is one of the planet's recognised biodiversity hotspots (Mittermeier et al., 2011), and detailed plans have been developed to expand the network of protected areas in the CFR, to capture and conserve a representative sample of the region's biodiversity (Cowling et al., 2003; South African Government, 2008). However, once proclaimed, protected areas need to be actively managed if the biodiversity of these areas is to survive the multiple threats that they face. In the CFR in particular, invasive alien species are arguably the largest of

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these threats (van Wilgen, 2013). Over 1000 indigenous plant species are threatened by invasive alien species in the CFR (Raimondo et al., 2009), and if invasions were to reach the full extent of their potential distribution, overall biodiversity (expressed as a biodiversity intactness index, Scholes and Biggs, 2005) in the region could be reduced by as much as 40% (van Wilgen et al., 2008). In addition, most of the region's watersheds lie within protected areas, where ongoing invasion by trees and shrubs threatens to reduce surface water runoff by as much as 36% (if allowed to reach the full extent of their potential distribution), with substantial economic impacts (van Wilgen et al., 2008).

In response to concerns about the loss of water resources and biodiversity, the South African Department of Water Affairs launched a large programme to clear invasive alien plants in 1995 (Koenig, 2009). This programme, Working for Water, operates at a national scale, and within the CFR it provides funding for the control of invasive alien plants both inside and outside of protected areas. In places where the programme has been active in the CFR, there are indications that the area occupied by invasive alien plants has been reduced by almost 50% (McConnachie et al., 2016), but the programme has only reached a small proportion (4–13%) of the total invaded area (van Wilgen et al., 2012). Importantly, at the scale of the CFR's protected areas, there has been no attempt to date to accurately quantify the magnitude of the problem, or the cost of control, nor has it been possible to assess progress towards reducing invasions due to the lack of a monitoring programme (van Wilgen and Wannenburgh, 2016). The study described here therefore set out to assess these issues. We sought to quantify the magnitude of the invasive alien plant problem in the major protected areas of the CFR; to document the extent and costs of substantial control efforts over the past two decades, and to estimate the resources that would be needed to reduce the problem to a maintenance level at which it could be managed sustainably (see Section 2.5 for a definition of maintenance level). We use the findings to support suggestions for changes that should improve the effectiveness of management.

2. Methods

2.1. Study sites

Our study was conducted in 25 protected areas (3 National Parks and 22 Provincial Nature Reserve complexes) covering approximately 750 000 ha in the CFR (Table 1; Fig. 1). The Nature Reserves are managed by the provincial authority (CapeNature), and the National Parks by South African National Parks (SANParks). The natural vegetation is dominated by fynbos shrublands that vary according to substrate (sandstone, granite, limestone or shale), as well as other shrubland types (renosterveld and strandveld). There are also smaller areas of Afro-temperate forest; these are not extensive except in the Garden Route National Park. The topography varies from relatively flat (mainly coastal) areas, to rugged mountainous areas, and all are invaded to a lesser or greater degree by invasive alien trees and shrubs (Fig. 2). Alien plant control programmes were initiated in these areas in the 1970s (Fenn, 1980) or earlier (Macdonald et al., 1989), and in 1995 they were substantially expanded with the initiation of the Working for Water programme, in response to growing concerns about impacts on water resources and biodiversity. Working for Water provides management capacity and labour to control invasive alien plants in protected areas, in collaboration with the responsible authorities, and with the dual goals of managing invasive alien plants and creating employment opportunities (van Wilgen and Wannenburgh, 2016).

2.2. Extent of alien plant invasions

The Nature Reserves managed by CapeNature are divided into management units of between 5 and 200 ha. In each management unit, we estimated the cover of invasive alien trees and shrubs in the genera *Pinus* (pine trees introduced from North America and Europe), *Acacia* (Australian wattle trees), *Eucalyptus* (Australian gum trees), *Hakea* (Australian shrubs), *Leptospermum* (Australian myrtle trees) and *Populus* (North American poplar trees) in 2014. These six genera account for almost all of the invasive alien plant cover in the protected areas assessed here. We estimated the percentage cover of each genus in each management unit in collaboration with experienced reserve staff, using a range of products, including high-resolution satellite imagery, aerial photography, and Google Earth. In some cases, where there was uncertainty about the estimates, they were verified in the field. Similar procedures were used to estimate cover in the Table Mountain and Agulhas National Parks, except that management units were larger (up to 1250 ha) in some cases. In the Garden Route National Park, we used alien plant cover data collected by Vromans et al. (2010), who divided the area into homogenous vegetation units, using 1:10 000 orthophoto maps as a base. The percentage

Table 1
Salient features of 25 protected areas in the Cape Floristic Region, South Africa.

Protected area	Area (ha)	Centre point	Location and topography	Dominant vegetation (after Mucina and Rutherford, 2006)
Agulhas National Park	21 693	34° 48′ S; 19° 59′ E	Coastal	Strandveld; sandstone fynbos
Cederberg Nature Reserve	33 717	32° 30′ S; 19° 00′ E	Inland; mountainous	Sandstone fynbos
De Hoop Nature Reserve	34 151	34° 28′ S; 20° 30′ E	Coastal	Limestone fynbos; dune strandveld
Gamkaberg Nature Reserve	39 307	33° 40′ S; 22° 00′ E	Inland; mountainous	Sandstone fynbos
Garden Route National Park	115 782	34° 00′ S; 24° 00′ E	Coastal; mountainous	Sandstone fynbos; southern coastal forest
Genadendal Nature Reserve	26 619	34° 00′ S; 19° 30′ E	Inland; mountainous	Sandstone fynbos; shale fynbos
Goukamma Nature Reserve	2282	34° 10′ S; 22° 50′ E	Coastal	Southern Cape dune fynbos
Grootvadersbosch Nature Reserve	26 044	33° 55′ S; 20° 50′ E	Inland; mountainous	Sandstone fynbos; southern Afrotemperate forest
Groot Winterhoek Nature Reserve	27 512	33° 00′ S; 19° 10′ E	Inland; mountainous	Sandstone fynbos
Hottentots-Holland Nature Reserve	30 519	34° 10′ S; 19° 10′ E	Inland; mountainous	Sandstone fynbos; shale fynbos
Jonkershoek Nature Reserve	15 397	34° 00′ S; 19° 00′ E	Inland; mountainous	Sandstone fynbos; granite fynbos
Kammanassie Nature Reserve	27 056	33° 35′ S; 22° 51′ E	Inland; mountainous	Sandstone fynbos; shale fynbos
Keurbooms Nature Reserve	898	33° 58′ S; 23° 25′ E	Coastal	Sandstone fynbos; southern Afrotemperate forest
Kogelberg Nature Reserve	24 508	34° 16′ S; 19° 00′ E	Coastal; mountainous	Sandstone fynbos
Limietberg Nature Reserve	44 804	33° 31′ S; 19° 09′ E	Inland; mountainous	Sandstone fynbos; granite fynbos
Marloth Nature Reserve	13 752	34° 00′ S; 20° 20′ E	Inland; mountainous	Sandstone fynbos; shale fynbos
Matjiesrivier Nature Reserve	12 806	32° 25′ S; 19° 20′ E	Inland	Quartzite fynbos
Outeniqua Nature Reserve	38 902	33° 52′ S; 22° 36′ E	Coastal; mountainous	Sandstone fynbos
Riverlands Nature Reserve	1716	33° 30′ S; 18° 40′ E	Inland	Granite fynbos; dolerite renosterveld
Robberg Nature Reserve	186	34° 08′ S; 23° 25′ E	Coastal	Sand fynbos; seashore (azonal) vegetation
Swartberg Nature Reserve	131 557	33° 21′ S; 22° 19′ E	Inland; mountainous	Sandstone fynbos; shale renosterveld
Table Mountain National Park	26 554	34° 09′ S; 18° 23′ E	Coastal; mountainous	Sandstone fynbos; granite fynbos
Vrolijkheid Nature Reserve	1963	33° 50′ S; 19° 55′ E	Inland	Shale renosterveld
Walker Bay Nature Reserve	8647	34° 30′ S; 19° 20′ E	Coastal	Dune strandveld
Waterval Nature Reserve	32 044	33° 21′ S; 19° 05′ E	Inland; mountainous	Sandstone fynbos

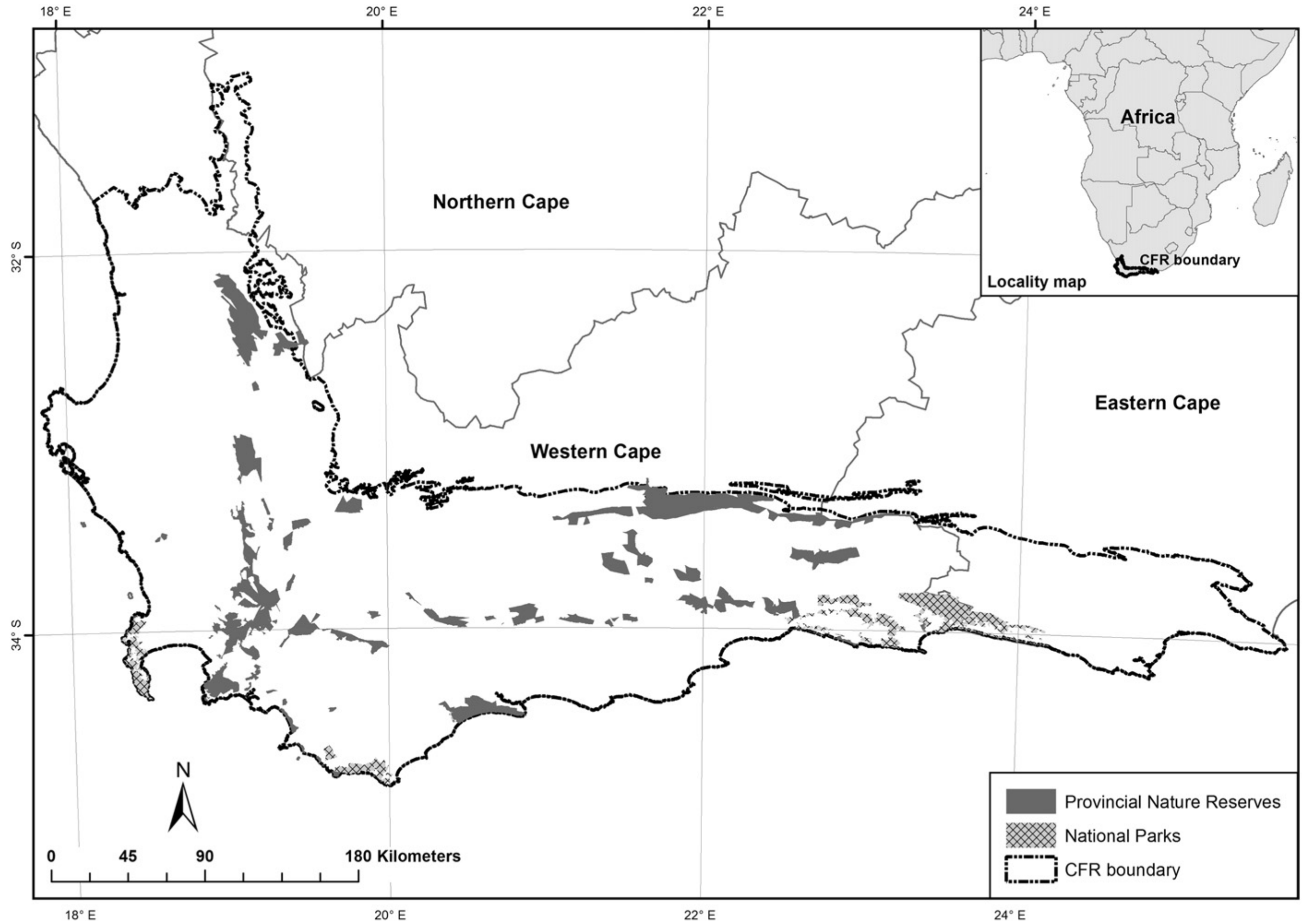

Fig. 1. Location of 25 protected areas in which invasive alien plant management scenarios were assessed in the Cape Floristic Region.

cover of each genus in each homogenous unit was estimated, either from the ground (for the more accessible parts of the park), or from the air, using helicopters. The Garden Route National Park's management units were then overlaid on Vromans et al.'s (2010) database to obtain estimates of alien plant cover per management unit. In the Garden Route National Park, areas covered by Afro-temperate forest (about 30% of the park) were excluded from the survey.

Fig. 2. Mountainous protected area in the Genadendal Nature Reserve. The trees in the foreground are invasive Australian wattles (*Acacia mearnsii*). Trees in the background are invasive Monterey pines (*Pinus radiata*), showing scattered trees spreading from the source population (photograph: B.W. van Wilgen).

2.3. Historical costs of control

The costs of control have been recorded by both CapeNature and SANParks in Working for Water's spatially-explicit database (Marais et al., 2004). These records were initiated at different times in each protected area, between 2002 and 2014, and normally covered 8 years or more. All of the control work was carried out by contractors, and the records include the species treated and the direct costs paid out to contractors for labour and herbicides. We estimated the amounts spent on control prior to the initiation of detailed records using the total national expenditure by the Working for Water programme since its initiation in 1995 (van Wilgen and Wannenburgh, 2016). We assumed that the funds expended between 1995, and the initiation of records in each protected area, were allocated to the individual protected areas in the same proportion of the national total as funds expended after records began. This exercise was necessary because of a lack of detailed records, and probably resulted in an underestimate of the earlier amounts spent in the CFR. For example, the annual budget for all projects grew from 25 million South African rands (ZAR) in 1995 to ZAR 432 million in 2003 (1 US$ ~ 15 ZAR in 2015), but the proportion spent in the CFR fell from an initial 58% to 20% of the total as new projects were initiated outside of the CFR (Working for Water annual reports). We further assumed that these funds were allocated to the control of individual genera in the same proportions as funds expended after records began. We then used the consumer price index to inflate all costs to 2015 ZAR to account for inflation. These costs include labour and herbicides, but do not include the costs of overheads such as transport, equipment, supervision and administration. Both CapeNature and

SANParks levy a mean of 32.5% on direct costs, across all protected areas, to cover overheads. We therefore added 32.5% to our estimates of direct clearing costs to account for overheads.

2.4. Projected future control costs

Managers of protected areas use norms and standards (Neethling and Shuttleworth, 2013) to estimate the cost of labour and herbicides that would be required to treat a management unit (either initial felling of invasive trees and shrubs, or conducting follow-up clearing). The effort (person-days) and methods are listed for different taxa, and for different cover classes. The effort required is then adjusted to account for time required to reach the site (by road and then on foot), and for slope, where costs increase with slope, with multiplication factors ranging from one (no adjustment) for flat areas to 2 (double) for slopes exceeding 50°. All protected area management units were available as spatial data layers, and we used the distance between the centre point of the management unit and the closest road to estimate walking time (assuming a walking speed of 3 km/h), and between the road and the closest town to estimate driving time (assuming a driving speed of 60 km/h).

We divided each management unit into two slope classes based on a digital elevation model (DEM) at a resolution of 20 m: "relatively flat" areas (<40° slope) and "steep" areas (>40° slope). We assumed that alien plant cover was distributed between these two slope classes in proportion to contribution of the slope class to the area of the management unit.

We used the norms and standards to estimate the costs of an initial clearing effort and three follow-up treatments (see Section 2.5). Cost estimates were adjusted for slope using the modal slope for the relatively flat and steep areas respectively (based on the 20 m DEM). For steep areas, we also adjusted the estimates upwards by an additional factor of 1.5 to account for the need for trained teams (who receive higher wages, and allowances for camping out overnight), special equipment (which included ropes and harnesses, and other safety equipment) and transport (sometimes including airlifting using helicopters). Costs were estimated for each management unit separately, and summed for all management units in a protected area.

As above (Section 2.3), we added 32.5% to account for overheads (transport, equipment, supervision and administration) that are not accounted for in the norms and standards.

We estimated the future costs for a range of scenarios to cater for different levels of annual funding, rates of spread, number of follow-up treatments required, and the mix of species to be controlled (see Section 2.5 below). We first estimated the funding that would be required to clear remaining invasions of a particular mix of species, and to carry out the requisite number of follow-up treatments, using the methods described above (the "base costs"). To achieve control at the base cost would require all invasions to be cleared, and followed up, in one year. For each scenario, we reduced the base cost by the annual level of funding expended on control and follow-up in that scenario, and then inflated the remainder of the base cost by the rate of spread associated with the scenario, in annual time-steps. This was repeated until the funds required reached zero.

2.5. Scenarios for future control

A fundamental assumption of our approach was that the reduction of alien plant invasions to a "maintenance level" could be achieved by conducting one initial felling (or in some cases ring-barking) treatment, and three follow-up treatments to deal with any subsequent regeneration. The concept of a maintenance level recognises that alien species, once well-established, cannot be eradicated, but that they can be reduced to a low level of invasion that could be contained at a relatively low cost in perpetuity. This was also defined by Goodall and Naude (1998) as "the systematic reduction of the major invasive alien plant species in defined tracts of land to a level where they no longer present a problem". Currently, follow-up treatments are carried out annually for all species, starting one year after initial felling.

We investigated the outcomes of three funding scenarios, first that funding would continue at current levels, second that it would be double the current funding levels, and third that funding would be reduced to 75% of current levels. The rationale for these scenarios was that (1) current funding levels reflect the status quo; (2) doubling the funding to protected areas could potentially be achieved by re-directing funds from other projects in the CFR, outside of protected areas, to ensure sufficient funds to achieve sustainable control inside protected areas; and (3) funding is probably more likely to decline due to the prevailing slow economic growth prospects in the country.

We used two scenarios for spread rates by assuming that alien plants in parts of the protected area that were not treated in a given year would continue to spread at either 4% or 8% per year. Spread rates of invasive alien plants are difficult to estimate over large areas, and our selection of spread rates was based on a limited number of studies where some estimates were available (see Section 2.6).

We used the assumption that three follow-up treatments would be sufficient to reduce invasions to sustainably manageable levels, as it is widely used by Working for Water as a guideline. However, there are many uncertainties, discussed in more detail below (Section 4.1). For example, many more follow-up treatments are used for *Pinus* and *Hakea* than would be required if follow-up treatments were linked to a programme of prescribed burning, see Section 4.1. Managers of protected areas also felt that three follow-up treatments would often not be sufficient to reduce invasions of *Acacia* species to manageable levels, because areas invaded by *Acacia* species are characterised by the presence of large reserves of long-lived, viable seeds in the soil (see Richardson and Kluge, 2008). These seeds germinate constantly, necessitating regular and repeated removal. We therefore added two additional management scenarios in which five and seven follow-up treatments were applied to *Acacia* invasions (but not to the other genera). Finally, we investigated a scenario in which all available funds were used to treat only *Pinus* and *Hakea*, leaving other invasive species untreated (see Section 4.2 for an explanation of this scenario).

2.6. Estimation of spread rates

We used limited available information to estimate plausible rates of spread for invasive alien plants in the CFR, using *Pinus* species, and *Acacia mearnsii* as indicator taxa. In the case of *Pinus*, Higgins et al. (2000) used successive aerial photographs to estimate the rate of spread of *Pinus pinaster* at four sites. They used a linear regression of the natural logarithm of the area versus time, and estimated doubling times at between 10 and 30 years, or 3–8% per year. Moeller (2010) used a time series of aerial photographs in the Garden Route National Park to assess the spread of *Pinus*. She found that the trees had increased from 6.5 ha in 1986 to 14.1 ha in 2007, a rate of increase of 3.75%. Richardson and Brown (1986) estimated the number of squares (50 × 50 m) that were occupied at different times by invasive *Pinus* trees of "moderate" density, again using aerial photographs. Higgins et al. (2000) used this data to estimate an annual spread rate of 7.9%.

In the case of *Acacia mearnsii*, Rebelo et al. (2013) used historical aerial photographs to estimate spread rates in a river floodplain and on adjacent slopes. The invaded area increased from 1440 ha in 1954 to 3134 ha in 2007 or about 2.1% per year, but this estimate was affected by clearing operations from 1996 onwards. Between 1954 and 1969 the rate of increase was 4.8% per year and from 1969 to 1986 it was about 1.8% per year, indicating that invasion rates were slowing down as more of the river system became invaded. Assuming that the first introduction was in 1880, and 1 ha was established at one location, then a spread rate of 10% per year would result in

Table 2
Area (ha) occupied by six invasive alien tree and shrub genera in five cover classes in 25 protected areas in the Cape Floristic Region.

Invasive alien plant genus	Cover class					Total
	Dense (>50%)	Medium (26–50%)	Low (6–25%)	Scattered (0.5–5%)	Rare (<0.5%)	
Pinus	9706	20 345	67 538	175 686	207 053	480 331
Acacia	12 934	15 721	66 311	139 191	71 862	306 020
Hakea	5370	10 828	41 763	188 477	143 238	389 677
Leptospermum	95	86	498	3937	48 957	53 573
Eucalyptus	85	908	5622	34 199	70 775	111 590
Populus	789	561	718	2513	57 946	62 528
Total	29 005	49 820	186 496	568 614	687 660	

about 1440 ha of invasion by 1954. Our selection of 4 and 8% for annual rates of spread were based on these estimates.

3. Results

3.1. Extent of alien plant invasions

Three genera (*Pinus*, *Hakea* and *Acacia*) accounted for the bulk of invasive alien plant cover in protected areas (Table 2). *Pinus* was the most widespread of these, with invasive stands of >25% cover occurring on approximately 30 000 ha, with a further 450 000 ha supporting *Pinus* invasions at lower levels of cover. Thus about 64% of the protected area estate is invaded to some degree by alien pine trees. The most severely-affected areas were in the mountainous portions of the Garden Route, Outeniqua, Hottentots-Holland and Jonkershoek reserves (where pine-based plantation forestry is practiced on a large scale on land adjacent to protected areas, see Kraaij et al., 2011; McConnachie et al., 2015). *Acacia* trees and shrubs were also widespread, with invasive stands of >25% cover occurring on approximately 29 000 ha, with a further 277 000 ha at lower levels of cover. Thus about 40% of the protected area estate is invaded to some degree by *Acacia* trees. The most severely-affected areas were along the coast (where *Acacia* species had been aggressively planted as part of a programme to stabilize mobile dunes along the coast, see Lubke, 1985). The coastal protected areas most affected by *Acacia* include De Hoop, Table Mountain, Agulhas and Walker Bay. *Hakea* shrubs were also prominent, with invasive stands of >25% cover also occurring on approximately 16 000 ha, with a further 373 000 ha at lower levels of cover. Thus over half of the protected area estate is invaded to some degree by *Hakea* shrubs. The areas most affected were in mountainous topography, with substantial invasive populations in the Garden Route, Outeniqua, Hottentots-Holland, Jonkershoek, Waterval and Limietberg reserves. The remaining three genera (*Eucalyptus*, *Leptospermum* and *Populus*) were less widespread, although where they do occur, they can form dense stands. Dense stands (>25% cover) were found on about 2500 ha, or approximately 0.3% of the protected area estate.

Approximately 20% of the total protected area fell into the steep category (>40° slope) where control would be more expensive (Section 2.4). *Pinus* and *Hakea* were approximately twice as prevalent in steep areas as *Acacia* (93 000 and 76 000 ha for *Pinus* and *Hakea*, vs. 44 000 ha for *Acacia*).

3.2. Historical and projected future costs of control

An estimated total of ZAR 564 million (approximately US$38 million) has been spent on the control of invasive alien plants in CFR protected areas over the past 20 years (Table 3). More than half of this was spent on the three National Parks (Agulhas, Garden Route and Table Mountain), with the largest portions having been spent on the Garden Route National and Table Mountain Parks (ZAR 164 and 135 million respectively). Almost half (48.6%) of the funding (ZAR274 million) was spent on the control of *Acacia* species (Table 3). Even though *Pinus* species were more widespread than *Acacia* species (Table 2), the amount spent on control of *Pinus* was about half of that spent on *Acacia* (ZAR 154 million, or 27% of the total). The historical focus on *Acacia* has come about partly because these trees usually occur in flatter areas, and are concentrated along access roads or drainage lines from which seeds have spread, making access to many invaded areas much easier. *Pinus* and *Hakea*, on the other hand, spread by means of wind-dispersed, winged seeds, released from serotinous cones or follicles after fires; these species occur as widespread populations in rugged areas far from access roads. Consequently, access to these populations has been difficult, and they have received less attention in the past. ZAR 78 million and ZAR 34 million were spent on *Hakea* and *Eucalyptus* respectively, with only ZAR 22 million (<4% of the total) spent on the remaining two genera (*Leptospermum* and *Populus*). The estimated cost to instantaneously treat all alien plants at current levels of invasion (initial clearing and three, five or seven follow-up treatments, the “base costs”) amounted to between ZAR 762 and 897 million (approximately US$ 51–60 million), considerably more than has already been spent over the past 20 years (Table 3). Under a scenario where only *Pinus* and *Hakea* are treated, the projected future base cost was less than the historical expenditure (ZAR 477 vs. 564 million, Table 3). However, the real cost would differ from the base cost estimate, depending on the management scenario (Section 3.3).

3.3. Scenarios for future control

The projected future cost of control ranged from ZAR 170 million to ZAR 2608 million, depending on the management scenario (Table 4). For those scenarios where all six invasive genera were subjected to control, the estimated total cost that would have to be spent (in

Table 3
Estimated historical (1995–2015) and projected future base costs of control of six invasive alien tree and shrub genera for a range of scenarios in protected areas in the Cape Floristic Region. Data are in millions of 2015-equivalent South African rands (ZAR; 1 US$ ~ ZAR15). See Section 2.5 for a detailed description of the scenarios.

Scenario	Invasive alien plant genus						Total cost (ZAR)
	Pinus	*Eucalyptus*	*Hakea*	*Acacia*	*Leptospermum*	*Populus*	
Historical cost	154.7	34.6	78.7	274.7	13.9	8.1	564.7
All genera subjected to three follow-up treatments	273.1	20.6	204.2	253.3	1.1	9.8	762.1
Acacia subjected to five follow-up treatments	273.1	20.6	204.2	320.7	1.1	9.8	826.7
Acacia subjected to seven follow-up treatments	273.1	20.6	204.2	388.1	1.1	9.8	897.0
Only *Pinus* and *Hakea* treated	273.1	0	204.2	0	0	0	477.3

Table 4
Projected future costs (millions of 2015-equivalent South African rands, ZAR; 1 US$ = 15 ZAR) of reducing alien plant invasions to sustainably manageable levels in 25 protected areas in the Cape Floristic Region under different management scenarios (scenarios are listed in ascending order of future costs).

Management scenario	Invasive alien plant genus						Total (ZAR × 1 000 000)	Time to reach sustainably manageable levels (years)
	Acacia	*Pinus*	*Hakea*	*Eucalyptus*	*Leptospermum*	*Populus*		
Funding at double current levels 4% annual rate of spread Three follow-up treatments *Pinus* and *Hakea* only	–	137	33	–	–	–	170	7
Funding at current levels 4% annual rate of spread Three follow-up treatments *Pinus* and *Hakea* only	–	147	36	–	–	–	183	15
Funding at 75% of current levels 4% annual rate of spread Three follow-up treatments *Pinus* and *Hakea* only	–	154	38	–	–	–	192	21
Funding at double current levels 8% annual rate of spread Three follow-up treatments *Pinus* and *Hakea* only	–	157	38	–	–	–	195	8
Funding at current levels 8% annual rate of spread Three follow-up treatments *Pinus* and *Hakea* only	–	225	55	–	–	–	280	23
Funding at double current levels 4% annual rate of spread Three follow-up treatments	567	216	52	41	33	2	911	11
Funding at double current levels 4% annual rate of spread Five follow-up treatments on *Acacia*	670	255	62	48	39	3	1077	13
Funding at double current levels 8% annual rate of spread Three follow-up treatments	773	294	71	56	45	3	1242	15
Funding at double current levels 4% annual rate of spread Seven follow-up treatments on *Acacia*	773	295	71	56	45	3	1243	15
Funding at double current levels 8% annual rate of spread Five follow-up treatments on *Acacia*	927	353	86	67	54	4	1491	18
Funding at current levels 4% annual rate of spread Three follow-up treatments	979	373	91	71	57	4	1575	38
Funding at current levels 4% annual rate of spread Five follow-up treatments on *Acacia*	979	373	91	71	57	4	1575	38
Funding at current levels 4% annual rate of spread Seven follow-up treatments on *Acacia*	979	373	91	71	57	4	1575	38
Funding at double current levels 8% annual rate of spread Seven follow-up treatments on *Acacia*	1185	451	110	86	69	5	1906	23
Funding at 75% current levels 4% annual rate of spread Three follow-up treatments on *Acacia*	1623	618	150	117	94	6	2608	84
Funding at 75% current levels 4% annual rate of spread Five follow-up treatments on *Acacia*	1623	618	150	117	94	6	2608	84

2015-value ZAR) was up to 4.6 times greater than the amount spent over the past 20 years (ZAR 564 million spent to date vs. up to 2608 million projected). The projected cost of controlling *Acacia* was more than double that of *Pinus* and *Hakea* combined (ZAR 567–979 million vs. ZAR 268–464 million respectively). The estimated future costs for controlling the remaining three genera (*Eucalyptus*, *Leptospermum* and *Populus*) were one to two orders of magnitude less than for *Acacia*, *Pinus* and *Hakea*. The scenarios listed in Table 4 include only those that could possibly lead to a reduction in the extent of invasions; the remaining scenarios would see a growth in invaded area despite sustained spending (Fig. 3). At a 4% rate of spread, only a doubling of funding would achieve control within 12–15 years (Fig. 3A, C and E). At an 8% rate of spread, control would also only be achieved if funding were to be doubled, but it would take 15–23 years (Fig. 3B, D and F). If control was to focus only on *Pinus* and *Hakea*, then control would be possible within 7–21 years, depending on the level of funding (Fig. 3G and H). A reduction in funding (a more probable scenario) would only result in successful control if rates of spread remained low, and if three follow-up treatments were effective, or if only *Pinus* and *Hakea* were treated, but this would take >80 years. Under other scenarios (for example 8% spread and current or reduced funding, Fig. 3B, D and F) the invaded area would continue to grow, despite significant spending.

4. Discussion

4.1. Prognosis for control

Our analysis suggests that the achievement of control to a maintenance level (that could be sustainably contained at a relatively low

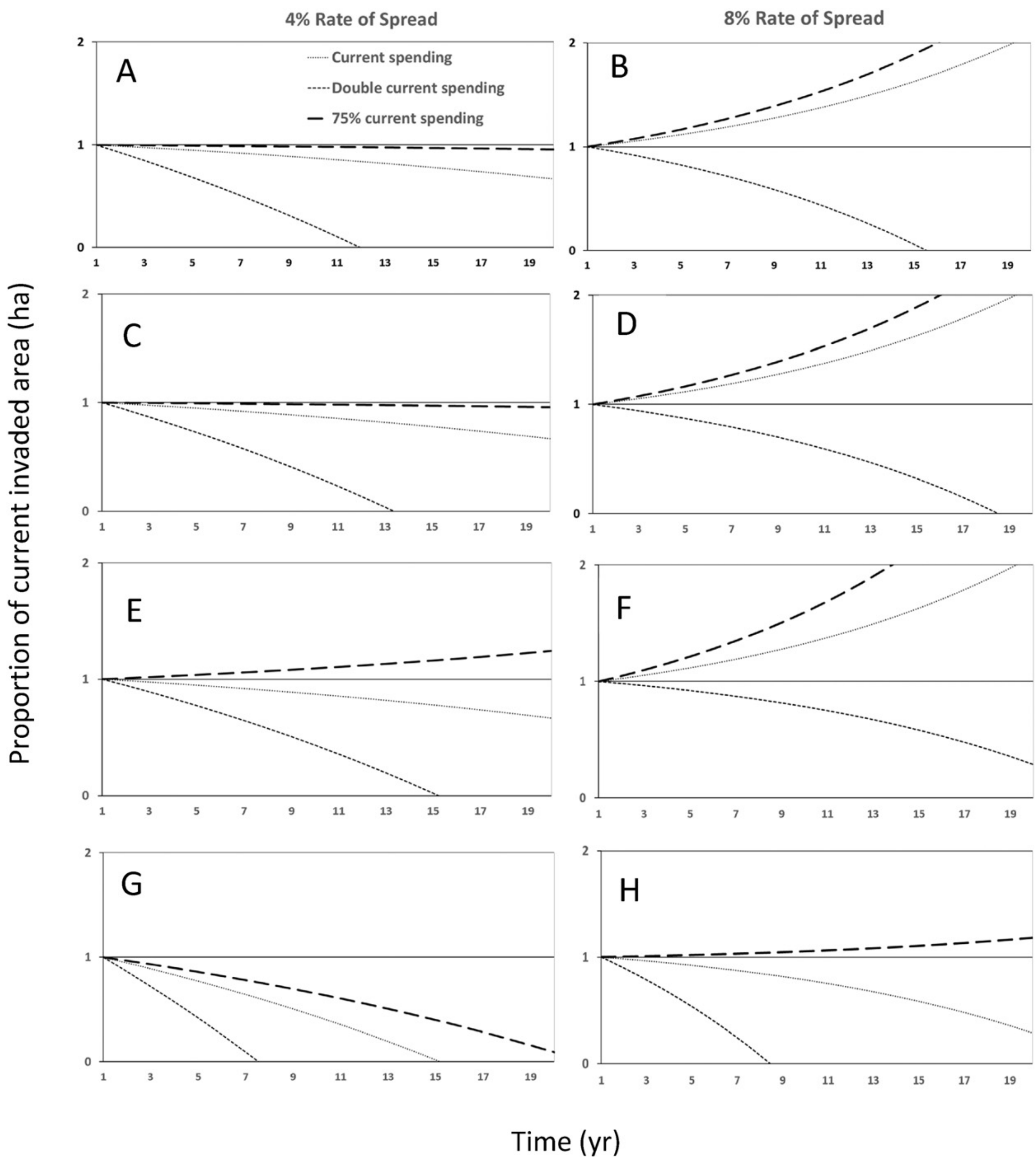

Fig. 3. Estimated future cover of invasive alien plants under different funding scenarios, spread rates and management regimes in 25 protected areas in the Cape Floristic Region (see text). Estimated future cover is shown as a proportion of estimated cover in 2014. The scenarios are: (A) and (B), all genera receive an initial clearing and three follow-up treatments; (C) and (D), *Acacia* receives an initial clearing and five follow-up treatments, and the remaining genera receive an initial clearing and three follow-up treatments; (E) and (F), as for (C) and (D) but *Acacia* receives an initial clearing and seven follow-up treatments; and (G) and (H), an initial clearing and three follow-up treatments is applied to *Pinus* and *Hakea* only.

cost) in all of the protected areas of the CFR would only be achieved if funding were to be substantially increased, or if control were to focus on selected genera only. The more likely scenarios (Fig. 3E and F) are those where the current levels of funding are reduced, spread rates are between 4 and 8%, and many follow-up treatments are needed for *Acacia* (because of the very large soil-stored seed banks that constantly produce new seedlings). Here, it is unlikely that a maintenance level will ever be achieved. There is a further assumption behind these predictions, and that is that control will be carried out using best-practice approaches based on decades of research and experience, and that contractors will apply treatments diligently, and repeatedly achieve the short-term goals that underlie the norms and standards on which contracts are awarded. Best practice should include the integration of fire in control programmes, particularly in the case of *Pinus* and *Hakea*. These trees are killed by fire and spread over considerable distances by means of winged seeds that germinate in the post-fire environment. Control is possible through pre-fire felling (after which seeds are released close to the ground, and do not disperse) and burning after 1–2 years, which kills any resultant alien seedlings before they can mature. This approach also allows the fire-adapted indigenous plants to regenerate normally (see Holmes et al., 2000). Currently, however, prescribed burning is seldom if ever used (van Wilgen et al., 2010), and follow-up weeding is done manually, which is expensive and much less effective than destroying alien seedlings with fire. In addition, most fires in the

CFR (almost 90% of area burnt, van Wilgen et al., 2010) are unplanned wildfires. Follow-up weeding should ideally focus on these burnt areas to remove post-fire seedlings before they grow to a size where removal becomes very expensive. To do this, management would have to be flexible, and able to re-direct resources to recently-burnt areas as these opportunities arise. Unfortunately, the control programmes operate in an extremely bureaucratic and rule-bound environment from which such flexibility is essentially absent (van Wilgen and Wannenburgh, 2016). Research has also clearly demonstrated that control efforts would be much more effective if they focused on areas with lower densities of young alien plants (as opposed to clearing dense invasions, see Higgins et al., 2000), but this advice is certainly not rigorously followed in the current operating environment where longer-term planning is for all practical purposes virtually absent (van Wilgen and Wannenburgh, 2016). In addition, the presence of large blocks of pine plantations immediately adjacent to protected areas provides a ready source of seeds from which cleared areas can be re-invaded (McConnachie et al., 2015), and although there are ongoing discussions about how to best address this problem, no real sustainable solutions have been found to date (van Wilgen, 2015). Finally, the possibility always exists that new invasive species could replace the current suite should existing invasions be substantially reduced. However, we believe that this particular risk has been reduced by new legislation that would prevent the indiscriminate introduction and widespread planting of new alien species, a practice that spawned the current problems. This brief discussion indicates that there are several factors that complicate the management of alien plant invasions which, if taken together, suggest that our projections may be over-optimistic.

4.2. Appropriate management responses

Alien plant control operations are typically embedded within complex social-ecological systems characterised by different perceptions, and multiple and sometimes conflicting objectives, in which the goals of management can become particularly challenging to achieve (Xiang, 2013). Shackleton et al. (2016) identified over 100 barriers that were relevant to the management of the invasive genus *Prosopis* (mesquite trees) in South Africa, most of which have direct relevance to our study. In the case of our study, the most important of barriers are both biophysical (for example plant species that are difficult to control, and rugged terrain) and social (for example a lack of adequate planning, monitoring and evaluation, and the need to meet the competing goals of maximising both employment and management efficiency, Shackleton et al., 2016; van Wilgen and Wannenburgh, 2016).

Despite these barriers, we believe that much can be done to improve the chances of achieving sustainable outcomes in priority areas by practicing conservation triage. Our analysis also suggests that if this is not done, then a great deal of money would have been, and will continue to be, wasted. The essential elements of the triage approach (Bottrill et al., 2008) would include a focus on priority areas and species, and accepting a trade-off between conserving biodiversity and reducing the extent of invasions. In addition, an improved adherence to best practice could potentially improve efficiencies.

We believe that conservation triage needs to be considered seriously, as the consequences of not doing so could be substantial. The three most important consequences would be: (1) a loss of up to 36% of the water resources of major towns and cities in the CFR (Le Maitre et al., 1996; van Wilgen et al., 2008), severely constraining the growth of local economies and impacting on quality of life; (2) a substantial loss of biodiversity in a recognised hotspot (Raimondo et al., 2009; van Wilgen et al., 2008); and (3) an increase in fire severity and resultant damage arising from increased fuel loads associated with invaded areas (Nel et al., 2014; van Wilgen and Scott, 2001). The idea of conservation triage in the CFR is not new, and Wicht (1945) advised that “it would be a more practical and realistic policy to destroy them [invading alien plants] only on selected areas, such as proclaimed Nature Reserves, and to take no action elsewhere.” The most important changes that would be needed are summarised below.

Focus on priority areas: Some areas have a higher priority for invasive alien plant clearing than others (Forsyth et al., 2012). Historic practice has often spread available funding across many projects, with the result that most projects are inadequately funded and often fail to make adequate progress (McConnachie et al., 2012; van Wilgen et al., 2012; Kraaij et al., 2016). Funding that is available for the CFR is currently also being invested into many projects outside of protected areas, and a scenario in which these external projects are phased out, and funds are re-directed into protected areas could be considered. This would be the only way in which the funding levels for protected areas could be doubled, and it could be justified by arguing that protected areas should be prioritized over areas that are not formally protected, and are thus at higher risk of future conversion to other land uses.

Focus on priority species: Clearly, three genera (*Pinus*, *Hakea* and *Acacia*) dominate the invasive alien plant problem in the CFR's protected areas. Historically, *Acacia* has received more funding than the other two genera, even though it has invaded a smaller area. We included a scenario in which funding would be directed towards *Pinus* and *Hakea* only (Fig. 3G and H) as these genera pose a far greater threat to a larger area if allowed to spread, because the habitat suitable for invasion by these species is much greater, and currently less densely invaded. We believe it would be a viable option to leave *Eucalyptus* and *Leptospermum* untreated (as they are relatively minor problems at present), as well as *Acacia* species because they have effective seed-reducing biological control agents (Moran and Hoffmann, 2012). Moreover, *Acacia* spp. could be also harvested in some areas by commercial woodcutters who could fell the stands under supervision and sell the wood as firewood.

Accept trade-offs between biodiversity and effective alien plant management: Experience with invasive alien plant management worldwide has shown clearly that invasive alien species, once they have become established and widespread, cannot be eradicated. It may therefore be necessary to accept some limited biodiversity loss in exchange for reducing alien plant invasions. For example, burning prescriptions for fynbos shrublands call for the use of fire to be restricted to certain seasons and frequencies to prevent the loss of sensitive native species (van Wilgen, 2009), but deviations from these restrictions in some areas might be the only effective way to bring invasions under control. Currently, fires for ecological management are only allowed if they are applied outside of the peak fire season (for safety reasons) and not in winter or spring (for ecological reasons). This leaves a very narrow window of opportunity to apply fires (van Wilgen and Richardson, 1985), and is one of the major constraints to the use of fire for ecosystem management (van Wilgen, 2013). Fire prescriptions that allow for burning in spring, for example, would create opportunities for much more widespread integration of fire and alien plant control operations. The alternative to relaxing burning prescriptions, which would be to insist on strictly promoting a fire regime that would best suit pristine fynbos, would result in fewer opportunities to burn, and consequently a greater area burnt in wildfires. Ultimately, a wildfire-driven fire regime would result in greater levels of invasion, and a greater loss of biodiversity, than judicious burning to contain rampant invasion.

Use ‘best practice’ methods: We believe that the effectiveness of management could be substantially improved by insisting on the use of best practice methods. As discussed above (Section 4.1), the control of *Pinus* and *Hakea* would be much more efficient if the use of fire was integrated into the control programme. A deliberate strategy of prioritizing clearing in sparsely-invaded areas (in the case of *Pinus* and *Hakea*), and on the upper reaches of drainage lines (in the case of *Acacia*) would further increase efficiency. We are also of the opinion that the current practice of using manual labour, hand-saws and axes (favoured as this increases employment, requires less safety precautions, and reduces the need for training) should be replaced with a greater use of power tools such as chainsaws and brushcutters. In addition, the timing of follow-up operations should be based on ecological considerations, that suggest treatments every four years for *Pinus* and *Hakea* and every two years for *Acacia* (Kraaij et al., 2016), and not annually, which is the current practice. Finally, biological control should be developed and used against as many of the aggressive invasive species as possible, as it often offers the only real way in which gains made by mechanical clearing can be maintained sustainably (Moran and Hoffmann, 2012).

We recognise that there may be substantial practical constraints to implementing some of our recommendations, and in some ways it may even be naïve to suggest far-reaching changes to an entrenched government programme. However, we are not aware of any other studies that have assessed the long-term likelihood of achieving the goals of invasive alien plant control over a large area as we have done here. Without such assessments, there is a risk that very large sums of money could be spent on such programmes without taking the likely outcomes into account (Bottrill et al., 2008). On the one hand, our study has a positive outcome in that it appears that control may be achievable under certain scenarios, provided that changes are made to the overall approach, as outlined above. The levels of investment being made, and the potential consequences of success or failure, make it important to articulate these points so that the estimates can be used to inform any debate on future management policies.

Conflict of interest

The authors declare that they have no potential conflicts of interest regarding the work reported here.

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References

Bottrill, M.C., Joseph, L.N., Carwardine, J., Bode, M., Cook, C., Game, E.T., Grantham, H., Kark, S., Linke, S., McDonald-Madden, E., Pressey, R.L., Walker, S., Wilson, K.A., Possingham, H.P., 2008. Is conservation triage just smart decision making? Science & Society http://dx.doi.org/10.1016/j.tree.2008.07.007.

Cowling, R.M., Pressey, R.L., Rouget, M., Lombard, A.T., 2003. A conservation plan for a global biodiversity hotspot-the Cape Floristic Region, South Africa. Biol. Conserv. 112, 191–216.

Fenn, J.A., 1980. Control of *Hakea* in the western Cape. In: Neser, S., Cairns, A.L.P. (Eds.), Proceedings of the Third National Weeds Conference of South Africa. Balkema, Cape Town, South Africa, pp. 167–173.

Forsyth, G.G., Le Maitre, D.C., van Wilgen, B.W., O'Farrell, P.J., 2012. The prioritisation of invasive alien plant control projects using a multi-criteria decision model informed by stakeholder input and spatial data. J. Environ. Manag. 103, 51–57.

Goodall, J.M., Naude, D.C., 1998. An ecosystem approach for planning sustainable management of environmental weeds in South Africa. Agric. Ecosyst. Environ. 8, 109–123.

Higgins, S.I., Richardson, D.M., Cowling, R.M., 2000. Using a dynamic landscape model for planning the management of alien plant invasions. Ecol. Appl. 10, 1833–1848.

Holmes, P.M., Richardson, D.M., van Wilgen, B.W., Gelderblom, C., 2000. The recovery of South African fynbos vegetation following alien tree clearing and fire: implications for restoration. Austral Ecology 25, 631–639.

Koenig, R., 2009. Unleashing an army to repair alien-ravaged ecosystems. Science 325, 562–563.

Kraaij, T., Cowling, R.M., van Wilgen, B.W., 2011. Past approaches and future challenges to the management of fire and invasive alien plants in the new Garden Route National Park. S. Afr. J. Sci. 107 (9/10). http://dx.doi.org/10.4102/sajs.v107i9/10.633 (Art. #633, 11 pages).

Kraaij, T., Baard, J.A., Rikhotso, D.R., Cole, N., van Wilgen, B.W., 2016. Assessing the Effectiveness of Invasive Alien Plant Management in a Large Fynbos Protected Area. African Biodiversity and Conservation, Bothalia (in review).

Le Maitre, D.C., van Wilgen, B.W., Chapman, R.A., McKelly, D., 1996. Invasive plants and water resources in the Western Cape Province, South Africa: modelling the consequences of a lack of management. J. Appl. Ecol. 33, 161–172.

Lubke, R.A., 1985. Erosion of the beach at St Francis Bay, Eastern Cape, South Africa. Biol. Conserv. 32, 99–127.

Macdonald, I.A.W., Clark, D.L., Taylor, H.C., 1989. The history and effects of alien plant control in the Cape of Good Hope Nature Reserve, 1941–1987. S. Afr. J. Bot. 55, 56–75.

Marais, C., van Wilgen, B.W., Stevens, D., 2004. The clearing of invasive alien plants in South Africa: a preliminary assessment of costs and progress. S. Afr. J. Sci. 100, 97–103.

McConnachie, M., Cowling, R.M., van Wilgen, B.W., McConnachie, D.A., 2012. Evaluating the cost-effectiveness of invasive alien plant control: a case study from South Africa. Biol. Conserv. 155, 128–135.

McConnachie, M., van Wilgen, B.W., Richardson, D.M., Ferraro, P.J., Forsyth, T., 2015. Estimating the effect of plantations on pine invasions in protected areas: a case study from South Africa. J. Appl. Ecol. 52, 110–118.

McConnachie, M., van Wilgen, B.W., Ferraro, P.J., Forsyth, A.T., Richardson, D.M., Gaertner, M., Cowling, R.M., 2016. Using counterfactuals to evaluate the cost-effectiveness of controlling biological invasions. Ecol. Appl. 26, 475–483.

Mittermeier, R.A., Turner, W.R., Larsen, F.W., Brooks, T.M., Gascon, C., 2011. Global biodiversity conservation: the critical role of hotspots. In: Zachos, F.E., Habel, J.C. (Eds.), Biodiversity Hotspots. Springer-Verlag, Berlin Heidelberg, pp. 3–22.

Moeller, J., 2010. Spatial Analysis of Pine Tree Invasion in the Tsitsikamma Region, Eastern Cape, South Africa: A Pilot Study (Honours dissertation) Department of Geography, Rhodes University, Grahamstown.

Moran, V.C., Hoffmann, J.H., 2012. Conservation of the fynbos biome in the Cape Floral Region: the role of biological control in the management of invasive alien trees. BioControl 57, 139–149.

Mucina, L., Rutherford, M.C., 2006. The vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19. South African National Biodiversity Institute, Pretoria.

Murdoch, W., Polasky, S., Wilson, K.A., Possingham, H.C., Kareiva, P., Shawe, R., 2011. Maximizing return on investment in conservation. Biol. Conserv. 139, 375–388.

Neethling, H., Shuttleworth, B., 2013. Revision of the Working for Water Workload Norms. Forestry Solutions, White River, South Africa (https://sites.google.com/site/wfwplanning/implementation).

Nel, J.L., Le Maitre, D.C., Nel, D.C., Reyers, B., Archibald, S., van Wilgen, B.W., Forsyth, G.G., Theron, A.K., O'Farrell, P.J., Kahinda, J.-M.M., Engelbrecht, F.A., Kapangaziwiri, E., van Niekerk, L., Barwell, L., 2014. Natural hazards in a changing world: a case for ecosystem-based management. PLoS ONE 9 (5), e95942. http://dx.doi.org/10.1371/journal.pone.0095942.

Raimondo, D., Von Staden, L., Foden, W., Victor, J.E., Helme, N.A., Turner, R.C., Kamundi, D.A., Manyama, P.A., 2009. Red list of South African plants. Strelitzia 25. South African National Biodiversity Institute, Pretoria.

Rebelo, A.J., Le Maitre, D.C., Esler, K.J., Cowling, R.M., 2013. Are we destroying our insurance policy? The effects of alien invasion and subsequent restoration: a case study of the Kromme River System, South Africa. In: Fu, B., Jones, K.B. (Eds.), Landscape Ecology for Sustainable Environment and Culture. Springer, Dordrecht, Netherlands, pp. 335–364.

Richardson, D.M., Brown, P.J., 1986. Invasion of mesic mountain fynbos by *Pinus radiata*. S. Afr. J. Bot. 52, 529–536.

Richardson, D.M., Kluge, R.L., 2008. Seed banks of invasive Australian *Acacia* species in South Africa: role in invasiveness and options for management. Perspectives in Plant Ecology, Evolution and Systematics 10, 161–177.

Scholes, R.J., Biggs, R., 2005. A biodiversity intactness index. Nature 434, 45–49.

Shackleton, R.T., Le Maitre, D.C., van Wilgen, B.W., Richardson, D.M., 2016. Identifying barriers to effective management of widespread invasive alien trees: *Prosopis* species (mesquite) in South Africa as a case study. Glob. Environ. Chang. 38, 183–194.

South African Government, 2008. National Protected Area Expansion Strategy for South Africa 2008: Priorities for Expanding the Protected Area Network for Ecological Sustainability and Climate Change Adaptation. Department of Environmental Affairs, Pretoria.

van Wilgen, B.W., 2009. The evolution of fire management practices in savanna protected areas in South Africa. S. Afr. J. Sci. 105, 343–349.
van Wilgen, B.W., 2013. Fire management in species-rich Cape fynbos shrublands. Front. Ecol. Environ. 11, e35–e44. http://dx.doi.org/10.1890/120137 (Online Issue 1).
van Wilgen, B.W., 2015. Plantation forestry and invasive pines in the Cape Floristic Region: towards conflict resolution. S. Afr. J. Sci. 111 (7/8). http://dx.doi.org/10.17159/sajs.2015/a0114 (Art. 114, 2 pages).
van Wilgen, B.W., Richardson, D.M., 1985. Factors influencing burning by prescription in mountain fynbos catchment areas. South. Afr. For. J. 134, 22–32.
van Wilgen, B.W., Scott, D.F., 2001. Managing fires on the Cape Peninsula: dealing with the inevitable. Journal of Mediterranean Ecology 2, 197–208.
van Wilgen, B.W., Wannenburgh, A., 2016. Co-facilitating invasive species control, water conservation and poverty relief: achievements and challenges in South Africa's Working for Water programme. Curr. Opin. Environ. Sustain. 19, 7–17.
van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M., Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. J. Environ. Manag. 89, 336–349.
van Wilgen, B.W., Forsyth, G.G., de Klerk, H., Das, S., Khuluse, S., Schmitz, P., 2010. Fire management in Mediterranean-climate shrublands: a case study from the Cape fynbos, South Africa. J. Appl. Ecol. 47, 631–638.
van Wilgen, B.W., Forsyth, G.G., Le Maitre, D.C., Wannenburgh, A., Kotzé, J.D.F., van den Berg, L., Henderson, L., 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biol. Conserv. 148, 28–38.
Vromans, D.C., Maree, K.S., Holness, S., Job, N., Brown, A.E., 2010. The Garden Route Biodiversity Sector Plan for the George, Knysna and Bitou Municipalities. Supporting Land-use Planning and Decision-making in Critical Biodiversity Areas and Ecological Support Areas for Sustainable Development. Garden Route Initiative. South African National Parks, Knysna.
Wicht, C.L., 1945. Report of the committee on the preservation of the vegetation of the South Western Cape. Special Publication of the Royal Society of South Africa (Cape Town).
Wilson, K.A., Underwood, E.C., Morrison, S.A., Klausmeyer, K.A., Murdoch, W.W., Reyers, B., Wardell-Johnson, G., Marquet, M.A., Rundel, P.W., McBride, M.F., Pressey, R.L., Bode, M., Hoekstra, J.M., Andelman, S., Looker, M., Rondinini, C., Kareiva, P., Shaw, M.R., Possingham, H.P., 2007. Conserving biodiversity efficiently: what to do, where, and when. PLoS Biol. 5, e223. http://dx.doi.org/10.1371/journal.pbio.0050223.
Xiang, W., 2013. Working with wicked problems in socio-ecological systems: awareness, acceptance, and adaptation. Landsc. Urban Plan. 110, 1–4.