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An extinction-risk assessment tool for flora threatened by *Phytophthora cinnamomi*

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Abstract. A risk-assessment tool was used to investigate the risk of extinction from disease caused by *Phytophthora cinnamomi* to 33 taxa from the Stirling Range National Park, Western Australia. Criteria used to score risk of extinction were the direct impact of *P. cinnamomi* on taxa, number of extant or extinct populations, percentage of populations infested by *P. cinnamomi*, proximity and topographical relationship of populations to *P. cinnamomi*, proximity of populations to tracks and the number of additional threatening processes. Direct impact scores were derived from mortality curves determined from the survival of taxa after soil inoculation with *P. cinnamomi* in a shade-house environment. On the basis of the total extinction risk score, nine taxa had a 'very high', five had a 'high', six a 'moderate', eight a 'low', four a 'very low' and one 'no' risk of extinction. Whereas the methodology confirmed the current threatened status of nine taxa, it also identified five taxa, not currently listed, to be at 'high' risk of extinction. Other threatening processes identified included fire, herbivory, aerial canker disease and climate change. These combine with *P. cinnamomi* to push taxa further towards extinction. Quantification of risk of extinction identifies taxa at risk and allows for prioritisation of management actions for currently threatened flora. This risk-assessment methodology combined glasshouse inoculation with habitat and ecological data, current *in situ* disease impact and proximity to disease and vectors, to enable a more comprehensive assessment of extinction risk and may be used in other areas with endemic flora threatened by *P. cinnamomi*.

Introduction

Phytophthora dieback resulting from the introduced root and collar-rot pathogen, *Phytophthora cinnamomi* Rands, is listed as a key threatening process under the Commonwealth of Australia's *Environment Protection and Biodiversity Conservation Act 1999*. Increased knowledge regarding the extent of threat of *P. cinnamomi*, the susceptibility of threatened flora and habitats and an increased predictive capacity are critical for the management of threatened flora (Commonwealth of Australia 2001). Species risk-assessment models are a useful tool to assist with prioritisation of taxa threatened by *P. cinnamomi* (CPSM 2005; Wilson *et al*. 2005). Flora at particular risk are those with geographically restricted ranges and naturally fragmented and disjunct distributions. The Stirling Range National Park (SRNP) is an ideal study area for the development of a predictive *P. cinnamomi* risk-assessment tool. The park contains more than 1500 of the 5710 described plant species of the South-West Botanical Province (Paczkowska and Chapman 2000), at least 82 of which are endemic to the SRNP. Many taxa have narrow range specificity and small population size. Disease caused by *P. cinnamomi* is considered to be the foremost threatening process for rare and endemic flora of the SRNP (Wills 1993; Wills and Keighery 1994; Barrett 1996, 2000, 2005; Barrett and Gillen 1997). Some 36% of the Stirling Range flora is estimated to be susceptible to *P. cinnamomi*, of which 10% is highly susceptible (Shearer *et al*. 2004*a*). The impact of the disease in the SRNP has been well documented with more than two-thirds of the park currently estimated to be infested (Wills 1993; Barrett 1996; Grant and Barrett 2003; Crane and Shearer 2007; Shearer *et al*. 2007*a*).

Disease spread is a key aspect when considering risk of infestation. *Phytophthora cinnamomi* may spread autonomously by the direct growth of the mycelium through root-to-root contact of infected plants, through the dispersal of motile zoospores in surface or subsurface water flow or by the transport of infested soil by animal or human vectors (Wills 1993; Shearer *et al*. 2007*a*). Disease spread varies considerably with climatic conditions, soil type and topography. Autonomous rates of spread on flat sites on deep sand may range from 0.7 to 2.0 m per annum (Grant and Barrett 2003). However, in mountain areas, once introduced high in the landscape, the pathogen may spread rapidly downslope through dispersal of zoospores (Weste and Law 1973; Grant and Barrett 2003). Upslope autonomous spread of *P. cinnamomi* is slower, with rates of 1.5 m per annum recorded on lateritic soils in the jarrah forest bioregion compared with 9.5 m cross-slope (Streilein *et al*. 2006).

Multiple processes threaten many taxa and communities and may interact with *Phytophthora* dieback to increase extinction risk. Fire has the capacity to increase the extinction

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vulnerability of many narrow range endemics (Keith 1996; Yates *et al*. 2003). The interaction between fire and *Phytophthora* dieback has only recently been explored with increased disease impact documented after fire (Moore 2006). Herbivory also increases extinction vulnerability and is particularly evident in the early years after fire. Summer drought, which is predicted to be exacerbated by climate change (Indian Ocean Climate Initiative 2002), is considered to be a significant threat to flora associated with refugial mountain habitat and is also an important factor in the death of *P. cinnamomi*-infested hosts (Shearer and Tippett 1989; Barker and Wardlaw 1995).

Previous studies have assessed the risk of extinction or disease impact owing to *P. cinnamomi* for several threatened or endemic flora in the Victorian ranges of eastern Australia (Peters and Weste 1997; Hewett 2001; Reiter *et al*. 2004). This paper tests a methodology that incorporates ecological and habitat data to assess the extinction risk of conservation-listed and endemic taxa in general, specifically assessing 33 taxa from the SRNP.

Materials and methods

Flora investigated

Thirty-three plant taxa of conservation significance were assessed (Table 1). Threatened flora were ranked *Critically Endangered*, *Endangered* and *Vulnerable* nationally under the *Environmental Protection and Biodiversity Conservation Act* (1999) and in Western Australia (WA) by using IUCN criteria (IUCN 2001). Poorly known or *Priority flora* are data-deficient species (IUCN 2001) and are categorised in WA as Priority 1, 2, 3 or 4 (Atkins 1998, 2006). Twenty-three taxa are endemic to the SRNP; another three taxa (*Lambertia ericifolia* R.Br., *Kunzea montana* (Diels) Domin and *Eucalyptus ligulata* subsp. *stirlingica* D.Nicolle) each have a single outlying population in close proximity to the park, and these were included in the study. Seven taxa have significant outlying populations disjunct from the park. These were *Banksia brownii* R.Br., *Daviesia obovata* Turcz., *Banksia pseudoplumosa* A.S. George, *Adenanthos pungens* Meisn. subsp. *pungens, Sphenotoma drummondii* (Benth.) F.Muell., *Andersonia echinocephala* (Stschegel.) Druce and *Calothamnus affinis*

Table 1. Conservation status of flora investigated

Threatened flora were ranked as *Critically Endangered*, *Endangered*, and *Vulnerable* in Australia (CR, E and V) and Western Australia (cr, e and v). Priority flora in Western Australia were ranked P1, taxa known from few (generally <5) populations which are under threat; P2, taxa known from few (generally <5) populations at least some of which are not under immediate threat; P3, taxa known from several populations, species not believed to be under immediate threat; P4, taxa considered to have been adequately surveyed and although rare are not currently threatened

Plant taxon	Endemic to SRNP	Plant family	Status	Ranking
Acacia awestoniana	Y	Mimosaceae	Threatened	V, v
Acacia veronica	Y	Mimosaceae	Priority	P3
Adenanthos pungens subsp. *pungens*	N	Proteaceae	Threatened	V, v
Andersonia echinocephala	N	Epacridaceae	Priority	P3
Banksia anatona	Y	Proteaceae	Threatened	E, cr
Banksia aculeata	Y	Proteaceae	Priority	P2
Banksia brownii	N	Proteaceae	Threatened	E, cr
Banksia concinna	Y	Proteaceae	Priority	P3
Banksia hirta	Y	Proteaceae	Priority	P3
Banksia montana	Y	Proteaceae	Threatened	E, cr
Banksia pseudoplumosa	N	Proteaceae	Threatened	e
Banksia rufa subsp. *pumila*	Y	Proteaceae	Priority	P2
Banksia solandri	Y	Proteaceae	Priority	P4
Calothamnus affinis	N	Myrtaceae	Priority	P4
Calothamus crassus	Y	Myrtaceae	Priority	P4
Darwinia collina	Y	Myrtaceae	Threatened	E, e
Darwinia leiostyla	Y	Myrtaceae	Priority	P4
Darwinia squarrosa	Y	Myrtaceae	Threatened	V, v
Daviesia glossosema	Y	Papilionaceae	Threatened	CR, cr
Daviesia obovata	N	Papilionaceae	Threatened	e
Daviesia pseudaphylla	Y	Papilionaceae	Threatened	E, cr
Deyeuxia drummondii	Y	Poaceae	Threatened	E, v
Eucalyptus ligulata subsp. *stirlingica*	N	Myrtaceae	Priority	P4
Gastrolobium leakeanum	Y	Papilionaceae	Priority	P2
Gastrolobium vestitum	Y	Papilionaceae	Priority	P2
Isopogon latifolius	Y	Proteaceae	Priority	P3
Kunzea montana	N	Myrtaceae	None	
Lambertia ericifolia	N	Proteaceae	None	None
Lambertia fairallii	Y	Proteaceae	Threatened	E, cr
Leucopogon gnaphalioides	Y	Epacridaceae	Threatened	E, cr
Persoonia micranthera	Y	Proteaceae	Threatened	E, cr
Sphenotoma drummondii	N	Epacridaceae	Threatened	E, e
Velleia foliosa	Y	Goodeniaceae	Priority	P4

Turcz. These taxa were assessed in terms of their extinction risk within the park only. Plant nomenclature follows (Paczkowska and Chapman 2000; Mast and Thiele 2007).

Plant source, propagation and inoculation

Plants were propagated from seed collected from the SRNP for long-term storage in the Department of Environment and Conservation's (DEC) Threatened Flora Seed Centre (Cochrane and Coates 1994). The exceptions were *Persoonia micranthera* P.H Weston and *Leucopogon gnaphalioides* Stschegel. These taxa were propagated from cuttings taken from natural populations because of the lack of availability of seed. Cuttings were propagated as described by Close *et al*. (2006). Germinated seedlings of the remaining 31 taxa were transplanted into 15-cm-diameter pots containing a potting mix soil of 1 part German peat to 3 parts river-washed sand mixture sterilised by methyl bromide fumigation. The pots were fertilised at the time of transplanting with 3 g of Yates Nutricote No. 10 (Chisso Asahi Fertiliser Co. Ltd, Tokyo) controlled release fertiliser (20N : 0P : 10.8K). Pots were inoculated with *P. cinnamomi* as described by Shearer *et al*. (2004*a*, 2007*b*).

Analysis of disease susceptibility

Percentage plants with collar lesion or percentage mortality was calculated for the assessment times after inoculation. The upper asymptote K_{max}, lag time $t_{1/2K}$ and intrinsic rate of increase (r) were calculated from mortality curves (Shearer *et al*. 2007*b*). The susceptibility score of each taxon was then calculated as follows: if $t_{1/2K} = 0$, then susceptibility $score_{taxon} = 0$; if $t_{1/2K} > 0$, then susceptibility $score_{taxon} = K_{max} + (100 - t_{1/2K}) + (100 \times r)$. The direct impact score was derived from the susceptibility score as follows: direct impact score = (susceptibility $score_{taxon}$/ susceptibility $score_{max}$) $\times 10$ (Shearer *et al*. 2007*b*).

Assessment of natural populations

Data on the number of extant and extinct populations, habitat type and condition, life history characteristics, threatening processes and the condition of threatened and priority flora populations were collated from the DEC threatened flora database and records, interim recovery plans (Holland *et al*. 2001; Phillimore and Brown 2001*a*, 2001*b*, 2001*c*, 2002; Phillimore *et al*. 2001; Stack and Brown 2003; Gilfillan and Barrett 2005; Hartley and Barrett 2005*a*, 2005*b*; Hartley *et al*. 2005) and Florabase (Western Australian Herbarium 1998–). The presence of *P. cinnamomi* in populations, the location of the nearest infestation adjacent to populations and the topographical position of infestations in relation to populations were determined from DEC records for threatened and priority flora populations, survey and mapping of *P. cinnamomi* within the park from 1986 to 2000 (Barrett 1996; Grant and Barrett 2003), mapping of the eastern Stirling Range Montane Heath and Thicket threatened ecological community (TEC) (Barrett 2000) and Montane Mallee Thicket of the Stirling Range TEC (Barrett 2005) and extensive flora survey and monitoring in the SRNP from 1999 to 2007. The presence of *P. cinnamomi* on site was confirmed where necessary by sampling and isolation of *P. cinnamomi* from indicator species. Otherwise, the presence or absence of *P. cinnamomi* was interpreted in the field by the pattern of plant death of highly susceptible species. Distance of populations from the nearest known *P. cinnamomi* infestation and nearest track, firebreak or road was calculated with the geographic information system program ARCGIS9 (ESRI 2005).

Risk of extinction

Extinction risk was evaluated and scored on the basis of the following parameters: number of extant populations, whether population extinction in association with *P. cinnamomi* had already occurred, the predicted direct impact of *P. cinnamomi* on the basis of susceptibility scores from pot trials, percentage of extant populations infested by *P. cinnamomi*, the percentage of populations less than 500 m from (i) a *P. cinnamomi* infestation and (ii) the nearest track, firebreak or road, the topographical relationship between healthy populations and the nearest *P. cinnamomi* infestation and the presence of other significant threatening processes (Table 2).

The species direct impact score was weighted more heavily than the potential or current impact of *P. cinnamomi* on habitat. In the case of a resistant species in a non-susceptible habitat (*Acacia awestoniana* R.S.Cowan & Maslin), the scoring process was not continued further. Proximity of populations to *P. cinnamomi* was considered a risk regardless of topography. However, the relatively lower risk for populations occurring upslope by at least 100 m of *P. cinnamomi* infestations was allowed for by allocating a negative score to species that met these criteria. Although uphill spread of *P. cinnamomi* may be in the order of

Table 2. Method of scoring risk of extinction

Direct impact score (0–10)	Number of extant populations (0–5)	Population extinctions have already occurred (0–5)	Percentage of extant populations infested by *Phytophthora cinnamomi* (0–5)	Percentage of extant populations <500 m from *Phytophthora cinnamomi* (0–5)	Percentage of populations at least 100 m upslope of *Phytophthora cinnamomi* (0–3)	% of extant populations <500 m from track (0–3)	Number of additional threatening processes (1–3)	Extinction risk (Total score 0–36)
Table 4	1–5 = 5	Yes = 5	>75 = 5	>75 = 5	None = 0	>60 = 3		≥30 = very high (VH)
	6–10 = 3	No = 0	>50–75 = 3	>50–75 = 3	0–20 = –1	>20–60 = 2		≥25–29 = high (H)
	11–20 = 2		>25–50 = 2	>25–50 = 2	>20–60 = –2	1–20 = 1		≥20–24 = moderate (M)
	20+ = 0		1–25 = 1	1–25 = 1	>60 = –3	0 = 0		≥15–19 = low (L)
			0 = 0	0 = 0				≥1–14 = very low (VL)

1 m per annum, downhill spread may be over 100 m per annum, with up to 400 m per annum recorded (Weste and Law 1973). Proximity to tracks and other access points was given a lower weighting than proximity to *P. cinnamomi* as the introduction and spread of *P. cinnamomi* within the SRNP has not been limited to vectoring along access tracks. The presence of other significant threatening processes known to be either currently or potentially affecting populations was also scored for each species. These threats were high-frequency fire, herbivory, aerial canker disease (Shearer 1994) and drought. Species that were considered to be most threatened by frequent fire were obligate reseeders with soil or canopy-stored seed banks, with moderate to long juvenile periods of more than 4 years. Although population extinctions may be associated with a range of fire regimes, in the context of the recent fire history of the SRNP, too frequent fire is considered to constitute the greatest driver of population decline and extinction (Barrett 2005). Fire-response categories were modified from Gill and Bradstock (1992). Juvenile period was based on the time from germination for 50% of a population to reach reproductive maturity. Where species occurred on upper and lower slopes, the juvenile period used was the greater time period of the upland populations. Drought associated with climate change was considered to be a significant threat to those species that were associated with refugial habitat but this threat was also based on direct observations of drought death. Risk of extinction was based on the total score and was ranked from very low to very high (Table 2).

Results

The 33 taxa investigated represented 17 genera from seven families and were all woody shrubs, with the exception of the graminoid *Deyeuxia drummondii* (Steud.) Vickery (Table 1). The majority were obligate reseeders (Table 3) and of these, 13 had a canopy-stored seed bank. Three taxa were capable of resprouting after fire; two of these (*Calothamnus crassus* (Benth.) Hawkeswood and *D. obovata*) were facultative seeder–resprouters (Table 3). Twenty-two taxa had a primary juvenile period of more than 4 years (Table 3); 10 had juvenile periods of 6 years or more. All occurred in susceptible habitat with the exception of *A. awestoniana* which grows in wandoo (*Eucalyptus wandoo* Blakely subsp. *wandoo*) woodland, considered non-interpretable for the presence of *P. cinnamomi* because of the absence of susceptible indicator species. Twenty-seven taxa were associated with upland habitat, with 25 recorded within either of the two montane TECs (Table 3). Five taxa were restricted to lowland flats or low ridges in mallee–heath or shrubland habitat. Seven taxa occurred in upland habitat as well as valley flats.

Direct impact scores ranged from 0 (four taxa) to the maximum score of 10 for *Isopogon latifolius* R.Br. (Table 4). Fourteen taxa had scores greater than 7 and all of these were members of the families Proteaceae, Papilionaceae and Epacridaceae. Taxa with a direct impact score of 0 included the two members of the Mimosaceae, the single Poaceae and one member of the Proteaceae, *Banksia aculeata* A.S.George. Of the seven members of the Myrtaceae assessed, all had scores between 0 and 7. Twelve taxa (excluding *A. awestoniana*) had less than six extant populations within the park and of these 10 were listed as *Threatened*. One taxon, *Daviesia pseudaphylla* Crisp, was known from only a single population. Previous population extinctions were recorded in 14 taxa, including 10 *Threatened* and three *Priority flora*. Eight taxa had undergone more than one population extinction. In the case of 28 of the 32 taxa with susceptible habitat assessed, more than 50% of each taxon's populations were infested by *P. cinnamomi*. Of the 32 taxa with susceptible habitat, 30 had more than 50% populations within 500 m of a *P. cinnamomi* infestation whereas 15 taxa had no populations at least 100 m upslope of a *P. cinnamomi* infestation. For 19 taxa, more than 60% of populations were less than 500 m from tracks or roads, and for all taxa, more than 20% of populations were less than 500 m from access tracks.

Of the four significant additional threatening processes identified, one taxon was subject to four and nine taxa to three threatening processes, whereas another 9 and 13 taxa were subject to two and one threatening processes, respectively (Table 3). Aerial canker fungi were reported in two species, *B. brownii*, (*Zythiostroma* sp.) and *P. micranthera* (*Truncatella* sp.). Twenty-nine taxa were associated with what was considered to be refugial habitat and therefore threatened by climate change. Twenty-one taxa were threatened by inappropriate fire regimes. Herbivory impacts were documented in nine taxa and were largely attributed to grazing by introduced *Oryctolagus cuniculus* (rabbit) and/or the native marsupial *Setonix brachyurus* (quokka).

On the basis of the total extinction risk score, nine taxa had a 'very high', five a 'high', six a 'moderate', eight a 'low', four a 'very low' and one (*A. awestoniana*) 'no' risk of extinction resulting from *P. cinnamomi* infestation(Table 5).

Discussion

This extinction-risk tool suggested a 'very high' risk of extinction within the SRNP for nine taxa and concurs with their current threatened status with seven ranked *Critically Endangered*. As seven of these taxa are endemic to the park they are also at very high risk of species extinction. The tool confirmed the known status of these taxa, and it also identified taxa, currently considered to be relatively abundant, as at risk of extinction also. Five SRNP endemic taxa were determined to have a 'high' extinction risk although none is currently listed as *Threatened* in WA. High susceptibility combined with incremental loss of habitat has resulted in few secure refuges for these plant taxa. The management of flora and plant communities becomes increasingly difficult as the area of healthy habitat and population size decline, and early identification of species at risk and the plant communities in which they occur is essential.

For non-endemic taxa, the tool is still valuable in highlighting the risk of loss of genetic diversity. Although *D. obovata* in this study has two small disjunct and healthy populations outside of the SRNP, the loss of genetic diversity through local extinction within the park would be considered highly significant. In the case of *B. brownii*, the tool results in a 'very high' risk of species extinction if applied to all populations outside the SRNP also.

Table 3. Life-history characteristics, juvenile period, habitat, landform, threatened ecological community and threatening processes other than *Phytophthora cinnamomi* for taxa investigated

OSS, obligate reseeder with soil-stored seed bank; OSC, obligate reseeder canopy-stored seedbank; FSR, facultative seeder-resprouter; and RS, resprouter (basal or epicormic). Juvenile period, time to first flowering after germination for 50% of population; MT, eastern Stirling Range Montane Heath and Thicket threatened ecological community; MMT, Montane Mallee Thicket of the Stirling Range threatened ecological community

Plant taxon	Life-history characteristics	Juvenile period (years)	Habitat	Landform	Threatened ecological community	Threatening processes
Acacia awestoniana	OSS	≤4	Woodland	Flat		
Acacia veronica	OSS	≤4	Woodland, mallee–heath/ thicket, thicket	Upper to lower slope	MMT	Climate change
Adenanthos pungens subsp. *pungens*	OSS	>4	Mallee–heath/thicket	Midslope		Fire, climate change
Andersonia echinocephala	OSS	>4	Mallee-heath/thicket	Lower to upper slope, summit, flat	MT, MMT	Fire, climate change, herbivory
Banksia aculeata	OSC	>4	Mallee–heath	Flat		Fire
Banksia anatona	OSC	>4	Mallee–thicket	Lower to midslope, valley flat		Fire, herbivory, climate change
Banksia brownii	OSC	>4	Mallee–heath/thicket, thicket	Mid- to upper slope, summit	MT, MMT	Fire, climate change, aerial canker disease
Banksia concinna	OSC	>4	Mallee–heath, thicket	Mid- to upper slope, summit	MT, MMT	Fire, herbivory, climate change
Banksia hirta	OSC	>4	Mallee–heath/thicket	Lower to upper slope, summit	MMT	Fire, climate change
Banksia montana	OSC	>4	Thicket	Summit	MT	Fire, herbivory, climate change
Banksia pseudoplumosa	OSC	>4	Mallee–heath, thicket	Ridge		Fire
Banksia rufa subsp. *pumila*	OSC	>4	Mallee–heath/thicket	Mid to upper slope	MMT	Fire, climate change
Banksia solandri	OSC	>4	Mallee–heath/thicket, thicket	Mid- to upper slope, summit	MT, MMT	Fire, herbivory, climate change
Calothamus crassus	FSR	>4	Mallee–heath/thicket, thicket	Summit	MT, MMT	Climate change
Calothamnus affinis	OSC	>4	Woodland, mallee–heath	Lower slope, ridge		Fire
Darwinia collina	OSS	>4	Thicket	Summit	MT	Fire, climate change
Darwinia leiostyla	OSS	≤4	Mallee–heath/thicket, thicket	Lower to upper slope, summit	MT, MMT	Climate change
Darwinia squarrosa	OSS	≤4	Mallee–heath/thicket, thicket	Mid- to upper slope, summit	MT, MMT	Climate change
Daviesia glossosema	OSS	≤4	Mallee–heath	Flat		Climate change
Daviesia obovata	FSR	>4	Mallee–heath/thicket	Mid- to upper slope, summit	MT, MMT	Fire, herbivory, climate change
Daviesia pseudaphylla	OSS	≤4	Mallee–heath	Flat		Climate change
Deyeuxia drummondii	OSS	≤4	Thicket, scrub	Summit	MT, MMT	Climate change
Eucalyptus ligulata subsp. *stirlingica*	RS		Mallee–heath/thicket	Lower to upper slope, summit	MMT	Climate change
Gastrolobium leakeanum	OSS	≤4	Mallee–heath/thicket, thicket	Lower to upper slope, summit	MT, MMT	Herbivory, climate change
Gastrolobium vestitum	OSS	≤4	Woodland, thicket	Mid- to upper slope, summit	MMT	Climate change
Isopogon latifolius	OSC	>4	Mallee–heath/thicket, thicket	Mid- to upper slope, summit	MT, MMT	Fire, climate change
Kunzea montana	OSS	>4	Mallee–heath/thicket, thicket	Mid- to upper slope, summit	MT, MMT	Fire, herbivory, climate change
Lambertia ericifolia	OSC	>4	Mallee–heath/thicket	Lower to upper slope, summit, flat	MMT	Fire, climate change
Lambertia fairallii	OSC	>4	Mallee–thicket/heath	Midslope	MMT	Fire, climate change
Leucopogon gnaphalioides	OSS	>4	Thicket, scrub	Summit	MT, MMT	Fire, herbivory, climate change
Persoonia micranthera	OSS	>4	Thicket	Summit	MT	Aerial canker disease, fire, herbivory, climate change
Sphenotoma drummondii	OSS	>4	Scrub	Summit	MT, MMT	Fire, climate change
Velleia foliosa	OSS	≤4	Thicket, scrub	Summit	MT, MMT	Climate change

Table 4. Mortality-curve parameters of species from the Stirling Range National Park with *Phytophthora cinnamomi*, following soil inoculation in a shade-house environment and susceptibility and direct impact ratings calculated from parameters

Mortality-curve parameters are K_{max}, the upper asymptote, with $t_{1/2K}$, lag time to half maximum and r, the intrinsic rate of increase of the logistic model. Taxa ordered from the greatest to least susceptibility score

Taxon	Mortality-curve parameters			Susceptibility	Direct impact
	K_{max} (%)	$t_{1/2K}$ (days)	r (% per day)	Score	Score
Isopogon latifolius	95.0	20	0.39	214	10.0
Banksia montana	55.0	31	0.76	200	9.3
Banksia brownii	100.0	28	0.22	194	9.1
Adenanthos pungens subsp. *pungens*	97.6	38	0.27	187	8.7
Banksia rufa subsp. *pumila*	100.0	31	0.16	185	8.7
Leucopogon gnaphalioides	100.0	35	0.20	184	8.6
Daviesia glossosema	94.7	33	0.22	184	8.6
Banksia anatona	98.0	30	0.15	183	8.6
Banksia pseudoplumosa	97.0	30	0.11	178	8.3
Lambertia fairallii	94.7	30	0.16	181	8.4
Banksia solandri	87.5	36	0.19	171	8.0
Lambertia ericifolia	93.0	36	0.13	170	7.9
Banksia hirta	85.8	38	0.13	160	7.5
Gastrolobium vestitum	80.0	37	0.10	153	7.1
Daviesia obovata	73.5	43	0.20	150	7.0
Banksia concinna	72.9	42	0.11	142	6.6
Darwinia collina	72.8	47	0.15	141	6.6
Sphenotoma drummondii	65.0	42	0.13	135	6.3
Persoonia micranthera	70.1	46	0.10	134	6.3
Gastrolobium leakeana	59.7	43	0.13	130	6.1
Kunzea montana	26.3	30	0.17	113	5.3
Andersonia echinocephala	76.6	82	0.05	99	4.6
Velleia foliosa	7.8	19	0.08	96	4.5
Daviesia pseudaphylla	33.0	50	0.09	92	4.3
Calothamnus affinis	13.3	48	0.10	75	3.5
Darwinia squarrosa	30.9	62	0.05	74	3.5
Calothamnus crassus	18.0	56	0.04	66	3.1
Eucalyptus ligulata subsp. *stirlingica*	6.1	53	0.06	59	2.7
Darwinia leiostyla	40.0	86	0.04	58	2.7
Acacia awestoniana	0.0	0	0.00	0	0.0
Acacia veronica	0.0	0	0.00	0	0.0
Banksia aculeata	0.0	0	0.00	0	0.0
Deyeuxia drummondii	0.0	0	0.00	0	0.0

Some limitations to the model may apply where taxa with low direct impact scores may be at risk of indirect impacts of *Phytophthora* dieback. Taxa such as *Darwinia squarrosa* (Turcz) Domin with a 'low' direct impact score could undergo a slow process of attrition during a prolonged period of time that might escape detection by monitoring across shorter timeframes. However, in terms of prioritisation of extinction risk, this model would appear to have allocated an appropriate ranking, i.e. a 'moderate' risk of extinction, as sudden rapid population decline is unlikely. Similarly, species may also be vulnerable to the indirect impact of *P. cinnamomi* if the disease results in a change in community structure and canopy cover unfavourable to the species (Wills 1993; Shearer *et al.* 2007*a*). However, this tool still gives weight to potential indirect impacts of the pathogen by scoring the extent of infestation of populations and their proximity to infestations. Other factors may influence persistence and decline such as life history and site characteristics. It has been proposed that species that recruit in high densities after fire may 'escape' disease at least in the short term through the maintenance of a soil-stored seed bank (Podger *et al*. 1996). This hypothesis requires further testing. Other flora may benefit from reduced competition such as occurs in the early years after fire, and may not be vulnerable to the indirect impact of the disease. The adverse impact of *P. cinnamomi* on plant reproduction as reported by Peters and Weste (1997) is poorly understood but may further influence ongoing survival of species with low to moderate susceptibility. Site differences in disease impact are also poorly understood and certain soils may have more disease-suppressive qualities (Shearer and Crane 2003). The long-term impact of the disease on species with low susceptibility and indirect impacts of *P. cinnamomi* requires further research. Long-term studies of species persistence or elimination in long-infested sites (e.g. Weste *et al*. 2002) are also critical for understanding the full impact of *P. cinnamomi* on the flora.

In the present study, 93% of taxa investigated had more than 50% of their populations within 500 m of a known infestation, clearly demonstrating the extent of the current *P. cinnamomi* infestation after a long history in the SRNP (Wills 1993). Remoteness from tracks and firebreaks has not resulted in

Table 5. Risk of extinction caused by *Phytophthora cinnamomi*

Plant taxon	Direct impact score (0–10)	Habitat susceptible (Y/N)	Number of extant populations (0–5)	Population extinctions known to have already occurred (0–5)	Percentage of extant populations infested by *Phytophthora cinnamomi* (0–5)	Percentage of extant populations <500 m from *Phytophthora cinnamomi* (0–5)	Percentage of populations at least 100 m upslope of *Phytophthora cinnamomi* (0–3)	% of extant populations <500 m from track (0–3)	Number of additional threatening processes (1–3)	Extinction risk (0–36) VL–VH
Banksia montana	9.3	Y	5	5	5	5	0	3	3	35.3 VH
Banksia anatona	8.6	Y	5	5	5	5	0	2	3	33.6 VH
Persoonia micranthera	6.3	Y	5	5	5	5	0	3	4	33.3 VH
Banksia brownii	9.1	Y	3	5	5	5	0	3	3	33.1 VH
Daviesia glossosema	8.6	Y	5	5	5	5	0	3	1	32.6 VH
Lambertia fairallii	8.4	Y	5	5	5	5	0	2	2	32.4 VH
Darwinia collina	6.6	Y	5	5	5	5	0	3	2	31.6 VH
Leucopogon gnaphalioides	8.6	Y	3	5	5	5	–1	3	3	31.6 VH
Daviesia obovata	7.0	Y	5	5	5	5	–1	2	3	31.0 VH
Banksia rufa subsp. *pumila*	8.7	Y	5	0	5	5	0	2	2	27.7 H
Banksia concinna	6.6	Y	2	5	5	5	–1	2	3	27.6 H
Banksia solandri	8.0	Y	0	5	5	5	–1	2	3	27.0 H
Gastrolobium vestitum	7.1	Y	5	0	5	5	0	3	1	26.1 H
Isopogon latifolius	10.0	Y	0	5	3	5	–1	2	2	26.0 H
Darwinia squarrosa	3.5	Y	2	5	5	5	0	3	1	24.5 M
Gastrolobium leakeanum	6.1	Y	3	0	5	5	0	3	2	24.1 M
Lambertia ericifolia	7.9	Y	0	5	3	5	–2	3	2	23.9 M
Daviesia pseudaphylla	4.1	Y	5	0	5	5	0	3	1	23.1 M
Sphenotoma drummondii	6.3	Y	2	0	5	5	–1	3	2	22.3 M
Banksia hirta	7.5	Y	2	0	5	5	–1	2	2	22.5 M
Banksia pseudoplumosa	8.3	Y	5	0	0	2	0	3	1	19.3 L
Velleia foliosa	4.3	Y	2	0	5	5	–2	3	1	18.3 L
Calothamnus crassus	3.1	Y	2	0	5	5	–1	3	1	18.1 L
Deyeuxia drummondii	0.0	Y	3	0	5	5	0	3	1	17.0 L
Eucalyptus ligulata subsp. *stirlingica*	2.7	Y	2	0	5	5	–1	2	1	16.7 L
Adenanthos pungens subsp. *pungens*	8.7	Y	5	0	0	0	–3	3	2	15.7 L
Andersonia echinocephala	4.6	Y	0	0	3	5	–2	2	3	15.6 L
Kunzea montana	5.3	Y	0	0	3	5	–2	2	3	15.3 L
Acacia veronica	0.0	Y	2	0	3	5	0	3	1	14.0 VL
Darwinia leiostyla	2.7	Y	2	0	3	5	–2	2	1	13.7 VL
Calothamnus affinis	3.5	Y	2	0	2	3	–3	2	1	13.5 VL
Banksia aculeata	0.0	Y	3	0	1	5	–1	3	1	12.0 VL
Acacia awestoniana	0.0	N								0.0

greater protection from *Phytophthora* dieback. Thirteen taxa had at least 40% of their populations more than 500 m removed from tracks or roads; however, seven of these had a 'very high' or 'high' risk of extinction. The appearance of apparently recent (less than 0.25 ha) mountain-top disease centres, upslope from valley infestations, would seem unrelated to human vectoring along tracks and are not readily explained. These new infestations have occurred with no established walk trails and where access has been strictly regulated since the mid-1990s (Department of Conservation and Land Management 1999). Animal vectors may be implicated in the spread of disease; alternatively, the pathogen may have been introduced in earlier times and only recently manifested itself as a result of disturbance such as fire (Moore 2006) or the onset of more conducive environmental conditions. Increased knowledge of vectoring mechanisms, in particular for the introduction of new disease centres, would greatly assist in disease management and risk assessment. However, the inclusion of proximity to tracks as a risk factor is still important as there is a well established association between vehicular movement and disease spread, and walk trails may facilitate both human and animal vectoring.

Although family and genus may be poor predictors of susceptibility (Shearer *et al*. 2007*a*), the present study supports the bias towards high susceptibility of members of the Proteaceae, Papilionaceae and Epacridaceae and the lower susceptibility of the Myrtaceae and Mimosaceae (Wills 1993; Shearer *et al*. 2004*a*). However, in the present study *B. aculeata* had an unanticipated 'zero' direct impact score and was presumed susceptible. Although lower susceptibility has been noted in prostrate *Banksia* species such as *B. gardneri* A.S.George, resistance in larger shrub species is uncommon. Artificial inoculations in controlled environments are therefore critical to clarify species' susceptibility (Shearer *et al*. 2004*a*, 2007*b*). Such testing under controlled conditions imposes favourable conditions for pathogen infection and host response, minimises disease escape and provides good correlation between relative susceptibilities in controlled and natural environments (Shearer *et al*. 2007*b*). Furthermore, the use of the logistic model of disease progress allows the positioning of a plant's response to infection on the resistance–susceptibility continuum (Shearer *et al*. 2007*b*).

For many taxa that are primarily threatened by *P. cinnamomi*, the additional impacts of fire and post-fire grazing act to push these taxa further towards extinction. For obligate-seeder species with long juvenile periods, a short fire interval in conjunction with depleted population size owing to *Phytophthora* dieback results in accelerated population decline (Barrett 2005). More recent research suggests that fire also promotes conditions more conducive for *P. cinnamomi* (Moore 2006). Although the impacts of climate change are unclear, the predicted reduction in rainfall for this area (Hope *et al*. 2006) will be an additional stressor.

Although rare species in south-western WA have evolved with drought, the recent changes in temperature and rainfall present far more stressful conditions than these plants will have tolerated historically. It is possible that overall lower annual rainfall could reduce pathogen activity. However, any associated increase in summer rainfall events is likely to be highly conducive for the sporulation and dispersal of the pathogen on the southern coast of WA as this region receives regular summer precipitation and soils retain considerable levels of inoculum in this period.

The methods used in the present study draw on previous methodologies by scoring disease susceptibility (Reiter *et al*. 2004; CPSM 2005), the number of extant populations and the percentage infested by *P. cinnamomi* (Department of the Environment and Water Resources 2007), as well as the proximity of tracks and topography (Reiter *et al*. 2004). The present study did not, however, attempt to quantify plant numbers as was done by CPSM (2005). Although the number of individuals in populations and their areas of occupancy are key aspects of rarity (IUCN 2001), sufficient data were not available for accurate assessment of all the taxa. An improved methodology would ideally include 'area of occupancy' as this is a more critical parameter than population size in relation to disease spread as *P. cinnamomi* can rapidly move through a large population in a small area. In addition, the current methodology scored multiple threats, previous population extinctions and used a direct impact score that provides a more accurate reflection of where the flora sits on the 'resistant–susceptible continuum'. CPSM (2005) used a distance of 1–2 km from a population to the nearest infestation as indicative of a high chance of infestation within the next 10 years; however, in the present study this distance was reduced to 500 m. With many populations within 2 km of *P. cinnamomi* infestations in the SRNP, a finer level of assessment was required to enable the flora to be adequately compared. Ongoing monitoring of currently healthy areas and populations is essential as changes in disease status will also affect species extinction risk. The development of remote-sensing techniques to map occurrences has been recommended as a priority (Department of the Environment and Water Resources 2007), and increased data gathering on the rate of disease spread, up, across and down slope, will increase predictive capacity.

Quantifying the risk of extinction enables management actions such as regulation of access, hygiene, application of the fungicide phosphite and *ex situ* conservation of germplasm and reintroductions to be prioritised (Wilson *et al*. 2005). Since 1997, aerial application of the fungicide phosphite in the SRNP has been effective in slowing population decline (Guest and Grant 1991; Aberton *et al*. 1999; Hardy *et al*. 2001; Barrett 2003; Shearer *et al*. 2004*b*, 2006; Shearer and Fairman 2007) and 'buying time' for other conservation initiatives.

In conclusion, the extinction-risk methodology trialled in the present study effectively identified and prioritised those species at risk of extinction owing to the root pathogen *P. cinnamomi*. By combining glasshouse inoculation with habitat and ecological data, current *in situ* disease impact and proximity to disease and vectors, a more comprehensive assessment of the risk to extinction to significant populations of a taxon or the taxon as a whole can be determined. This risk-assessment methodology lends itself for use in other areas with short-range endemics and multiple species threatened by *P. cinnamomi*.

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