

## Modelling the impact of *Hieracium* spp. on protected areas in Australia under future climates

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Anthropogenically-induced climate change is one of the most important global threats to biodiversity. Understanding its impact on the distribution of exotic plant species is critical for developing effective adaptation and management strategies. However, there is insufficient information currently available on the biodiversity at risk from 1) exotic plant invasions, 2) climate change, and 3) the interaction between these two major threats, to develop such strategies. We use ecological niche models as a first step to identify zones inside and outside Australian protected areas that may be most at risk from invasions of three species of *Hieracium* (hawkweeds) under current and future (2030 and 2070) climate scenarios, should current control and eradication methods fail. These perennial herbs are native to Europe and invasive to New Zealand and North America. Naturalised in Australia, hawkweeds threaten native tussock grasslands and the grazing industry, and have been placed on the National Alert List. Using eight ecological niche models currently available in the software package BIOMOD, we found that these species have yet to realize the extent of their climatic distribution under present day climate in Australia. As climate change accelerates, the climatic range of hawkweeds was projected to contract overall. However, much of the Australian Alps, which contain large contiguous tracts of reserves and many endemic species, will continue to retain climatically suitable areas for hawkweeds through to 2070. These results emphasise the need for ongoing monitoring as well as focused control to minimize the likelihood of hawkweeds realizing their invasive potential in protected areas and beyond.

Reserve systems are important repositories of native biodiversity. They also afford a level of protection that is critical for the ongoing provision of ecosystem processes and services. Currently, over 8000 individual areas occupying around 88 million ha of Australia, or 11.5% of the continent, have been set aside as protected areas (Dunlop and Brown 2008). These areas are highly valued for their conservation, recreational and cultural significance. The integrity of current and future reserves must be managed in the face of a number of threats, including climate change, encroachment of exotic species and the potential synergism of the two. Within New South Wales, Australia, ca 45% of 945 species listed under the Threatened Species Conservation Act 1995 are currently threatened by weeds (Courtts-Smith and Downey 2006). As such, invasive species management is a high priority within conservation reserves with around \$AUD20 million invested annually in controlling infestations of invasive plant species (Sinden et al. 2004).

While the impact of invasive plants on protected areas has long been documented (Obermeijer 1937, MacDonald et al.

1989, MacDonald 1994, Freitag-Ronaldson and Foxcroft 2003, Foxcroft and Downey 2008), understanding of the potential interactions between invasive species and climate change on reserve values is very limited (MacDonald 1994). Prioritising invasive species according to the threat they pose is central to effective allocation of resources. In Australia, the formation of the Weeds of National Significance and National Alert List of species has underpinned an integrated response to weed management across all states and territories (Sheppard et al. 2007), and models have been developed for a select number of invasive plant species to illustrate their impacts on biodiversity (Downey 2008). To sustain the success of these programs into the future the potential effects of climate change on invasive species needs to be incorporated into management strategies.

Climate change may enhance the capacity of many exotic species to invade new areas, while simultaneously decreasing the resistance of natural ecosystems to invasion (Thuiller et al. 2007). Among the major concerns is the poleward and/or elevational shift in climatic zones due to

warmer temperatures and the ability of organisms, both native and exotic, to track spatial changes in optimum environmental conditions. While the capacity of species to respond to climate change will be highly idiosyncratic, some plant species possess traits such as short generation times and wind or avian dispersal modes that increase their ability to migrate across landscapes. These species have an increased likelihood of keeping pace with shifts in climatic zones associated with warming temperatures. Unfortunately, the traits they exhibit are synonymous with those of numerous invasive species and increase the probability that invasive species will rapidly invade new areas in future climates (Rejmánek and Pyšek 2000, Rejmánek et al. 2005, Pyšek and Richardson 2007).

To effectively target exotic plant species in protected areas it is important to pre-emptively investigate the areal extent of such areas at risk of invasions under current and future climates. Ecological niche models (ENMs) are popular tools for generating hypotheses of potential changes in the distributional range of an exotic species. ENMs can provide a preliminary understanding of the magnitude of the threat posed. These models associate the known locations of a species to a set of environmental variables that are thought to influence the species' distribution. The central premise of this approach is that the best indicator of the climatic requirements of a species is its current distribution. ENMs assume that species distributions are in equilibrium with current climate (Guisan and Zimmermann 2000, Guisan and Thuiller 2005), that species will only be able to establish populations in areas with the same climatic conditions as their native range, and that niche conservatism is maintained in both space and time (Peterson and Holt 2003). The models have been used to investigate the distributions of a number of different exotic species, including plants (Welk et al. 2002, Kriticos et al. 2003, Peterson et al. 2003, Rouget et al. 2004, Thuiller et al. 2005, Baret et al. 2006, Mau-Crimmins et al. 2006), invertebrates (McKenney et al. 2003, Sutherst and Maywald 2005, Estrada-Peña et al. 2007, Fitzpatrick et al. 2007) and vertebrates (Sutherst et al. 1995, Peterson et al. 2006, Ficitola et al. 2007, Urban et al. 2007), and, less frequently, to assess the interaction between exotic species and climate change (Broennimann and Guisan 2008, Beaumont et al. 2009).

The genus *Hieracium* (Family Asteraceae: hawkweeds) contains around 9000 species, most of which are native to temperate and montane areas of the northern Hemisphere. Hawkweeds can be extremely serious weeds, having invaded New Zealand, North America, Japan and Patagonia. Adapted to poor soils (Rose et al. 1998), they can tolerate heat, frost and snow (Barker et al. 2006), and spread both vegetatively and via seeds. They are highly fecund, producing up to 40 000 seeds  $m^{-2}$  of cover per year (Burton and Dellow 2005). Four species of hawkweeds have become naturalised in Australia; *Hieracium aurantiacum*, *H. murorum*, *H. pilosella* and *H. praealtum*. Although the distribution of these species in Australia is currently limited, with *H. pilosella* thought to be eradicated (Williams and Holland 2007), their potential threat has resulted in them being identified as "sleeper weeds" and they have been placed on the National Alert List. Sleeper weeds are defined as invasive plants that have naturalised in a region but not yet increased

their population size exponentially (Brinkley and Bomford 2002, Groves 2006). Hawkweeds potentially threaten tussock grasslands and tablelands in alpine and temperate regions of the eastern states of Australia (Barker et al. 2006), but have also been found in *Eucalyptus pauciflora* woodlands and alpine heathlands (Williams and Holland 2007). They have been identified as emerging weeds of significance to the grazing industry in Australia because members of the genus compete aggressively with native species, agricultural crops and pastures to form dense monocultures (Barker et al. 2006).

Given the potential threat of hawkweed species to Australia's biodiversity, the aims of this study were to 1) identify the distribution of climatically suitable habitat for *H. pilosella*, *H. aurantiacum* and *H. murorum* under current and future climate scenarios (2030 and 2070) and 2) assess the extent to which their projected ranges coincide with areas currently set aside for the conservation of fauna and flora.

## Methods

### Climate and species location data

Data collection and ecological niche models used in this study are described in detail in Beaumont et al. (2009). We summarise these methods below.

Six climate variables were obtained from WorldClim (ver. 1.4, Hijmans et al. 2005): annual precipitation (AP), mean temperature of the coldest quarter (MTCQ), maximum temperature of the hottest month (MTHM), mean diurnal range (MDR), precipitation of the driest month (PDM) and precipitation seasonality (PS). These were selected as they provide a mix of means, extremes and seasonality that influence the distribution of the species. Four climate scenarios for two periods (2030 and 2070) were extracted from OzClim (v 2.0, <[www.cmar.csiro.au/ozclim/index.html](http://www.cmar.csiro.au/ozclim/index.html)>). These were derived from two atmosphere-ocean general circulation models (CSIRO Mk2.0 and NCAR CCSM3) and two emissions scenarios (B1 with low climate sensitivity and A1F with high climate sensitivity). CSIRO Mk2 and NCAR estimate an increase in Australia's annual mean temperature of 0.48 and 0.47°C, respectively, under the B1 low climate sensitivity scenario by 2030, and 1.00 and 0.97°C under the A1F high climate sensitivity scenario.

Occurrence records from the native and invasive ranges of *H. aurantiacum*, *H. murorum*, and *H. pilosella* were downloaded from a variety of online databases, including the Global Biodiversity Information Facility (GBIF, <[www.gbif.org/](http://www.gbif.org/)>), Australia's Virtual Herbarium (AVH, <[www.flora.sa.gov.au/avh/](http://www.flora.sa.gov.au/avh/)>), Arctos (<[arctos.database.museum/home.cfm](http://arctos.database.museum/home.cfm)>), Eunis (<[eunis.eea.europa.eu/](http://eunis.eea.europa.eu/)>) and E-Flora BC (<[www.eflora.bc.ca/](http://www.eflora.bc.ca/)>). Insufficient distribution records were available to model *H. praealtum*. In total, 1088, 1281 and 2721 records for *H. aurantiacum*, *H. murorum* and *H. pilosella* were used to model distributions. Of these 505, 33 and 285, respectively, came from invasive populations.

Natural history collections, such as those used in this study, typically contain presence-only records. As binary

data were required for the ENMs, pseudo-absences were generated from sites lying outside of the species' realized climatic envelopes, as defined by the six climate variables (LeMaitre et al. 2008). This approach reduces the selection of false absences – i.e. locations within the climatic tolerance of the species but where the species has not been recorded due to insufficient surveying, dispersal limitations or biotic constraints (Austin and Meyers 1996). The geographic distribution of each species' climatic envelope was mapped in ArcGIS (ESRI 2006) and locations outside of this region were randomly selected as pseudoabsences, where the number of pseudoabsences equalled ten times the number of presence records for each species. Ten alternate sets of pseudoabsences were selected for each species to ensure that results were not dependent upon a single selection.

## Ecological niche models

While a growing number of ENMs are available (Elith et al. 2006) there is no “best” method, and it is recognised that the use of a single modelling technique does not provide any information on whether that method gives the best predictive accuracy for the data set used (Thuiller 2004, Araújo and New 2007). As an alternative, an ensemble of simulations from multiple ENMs can be used to explore the range of projections (Thuiller 2004, Araújo and New 2007). We created ensembles consisting of projections derived from eight statistical models available in the BIOMOD package (v 2006.01.26) developed by Thuiller (2003, 2004). These consisted of two regression methods (GAM: general additive model, GLM: general linear model), three machine learning methods (ANN: artificial neural network, GBM: generalised boosting model, RF: random forest), two classification methods (CTA: classification and regression tree, MDA: mixture discriminant analysis) and a surface range envelope (SRE). To assess whether the disparity in the number of records from native versus invasive regions influenced model results, we weighted occurrence data such that both native and invasive regions were equally represented and ran the CTA, GAM, GBM and GLM models ten times.

Probabilities of occurrence derived from all models (except SRE) were transformed into presence/absence data using the area under the ROC curve (AUC). Ensembles of model projects were created in ArcGIS using the “frequency histogram” approach with models being given equal weights (Araújo and New 2007, Marmion et al. 2009). We then explored areas of majority consensus (i.e. areas classified as suitable by a minimum of five models). Analyses were conducted at the resolution of 10 km<sup>2</sup> to mask potential inaccuracies in the locations given on herbarium data.

An assumption of ENMs is that the realized climatic niche of a species is conserved spatially and temporally. However, a species may realize a different climatic niche in non-indigenous regions (Broennimann et al. 2007, Fitzpatrick et al. 2007, Loo et al. 2007). We have previously found significant differences in the climate niches occupied by native and invasive populations of hawkweeds (Beaumont et al. 2009). Modelling the distributions of these species using data from only their

native range failed to identify much of the known invasive distribution of these species in North America and Australia. For *H. aurantiacum* and *H. pilosella*, invasive populations in western North America, eastern North America and Australia/New Zealand realized climatic niches that only partly overlap with each other (insufficient data was available for *H. murorum* to assess similarity of climatic niches among invasive populations in different geographic areas). While most closely aligned with native ranges, the climatic niche of populations in Australia/New Zealand also partly overlaps with those in North America, and expands into climatic space not occupied by populations on other continents. It is unlikely, therefore, that modeling the distributions of hawkweeds based solely on native range data or invasive range data will accurately represent the invasive potential of these species. Hence, we calibrated ENMs on data from species entire ranges (i.e. native and invasive).

Presence and pseudo-absence data were randomly split into two subsets with 70% of records used to calibrate the ENMs and 30% used to evaluate the predictive power of each model using the area under the receiver operating characteristic curve (AUC) and Cohen's Kappa statistic. The calculation of the AUC traditionally requires both presence and absence data. If true absence data are unavailable and random or background points are used in their place (as with this study) then the maximum achievable AUC is <1 (Wiley et al. 2003, Phillips et al. 2006). Probabilities of occurrence derived from all models were transformed into presence and absence using the area under the ROC curve (AUC) (except for SRE which provides presence/absence of the species, not probability values: accuracy of these models were assessed only with Cohen's Kappa statistic). We applied a conservative classification where AUC <0.8 is a poor or null model; 0.8 <AUC <0.9 is fair; 0.9 <AUC <0.95 is good and 0.95 <AUC 1.0 is an excellent model (Thuiller et al. 2006a). Models with a Kappa <0.6 were classified as poor or null; 0.6 <Kappa <0.8 as fair; 0.8 <Kappa <0.9 as good and >0.9 as excellent.

## Protected areas

The boundaries of areas set aside for the conservation of native flora and fauna were obtained for each Australian state and territory, and included National Parks, wildlife sanctuaries, conservation parks, natural feature reserves and nature reserves. The extent to which current and future predicted distributions were contained within the boundaries of these reserved areas was calculated in ArcGIS (ESRI 2006).

## Results

### Climatically suitable habitat under current and future climates

This study used eight ENMs to assess the potential current and future distributions of *H. aurantiacum*, *H. murorum* and *H. pilosella*. Model accuracy was consistently high, with most AUC and Kappa scores exceeding 0.95 and 0.9, respectively (exceptions were ANN and MDA which scored

Table 1. AUC and Kappa scores indicating the average accuracy of eight ecological niche models for three species of *Hieracium* (standard deviations in brackets). To assess the extent to which the selection of random pseudoabsences may influence model results ten sets of pseudoabsences were generated and each ENM was run ten times. As SRE models do not derive probabilities of occurrences only Kappa scores were generated.

	ANN	CTA	GAM	GBM	GLM	MDA	RF	SRE
<i>H. aurantiacum</i>								
AUC	0.980(0.018)	0.992(0.005)	0.999(0.000)	0.997(0.002)	0.999(0.000)	0.988(0.003)	0.993(0.028)	–
Kappa	0.889(0.098)	0.975(0.008)	0.988(0.004)	0.957(0.009)	0.985(0.006)	0.897(0.027)	0.989(0.005)	0.965(0.010)
<i>H. murorum</i>								
AUC	0.989(0.008)	0.991(0.004)	0.999(0.000)	0.997(0.001)	0.999(0.001)	0.996(0.002)	0.995(0.003)	–
Kappa	0.902(0.055)	0.976(0.010)	0.990(0.003)	0.964(0.007)	0.986(0.004)	0.956(0.007)	0.993(0.003)	0.977(0.006)
<i>H. pilosella</i>								
AUC	0.992(0.006)	0.992(0.003)	0.999(0.000)	0.993(0.002)	0.999(0.000)	0.989(0.007)	0.994(0.003)	–
Kappa	0.907(0.060)	0.981(0.004)	0.996(0.002)	0.981(0.004)	0.993(0.002)	0.905(0.014)	0.997(0.002)	0.975(0.005)

“good” in several runs), irrespective of the selection of pseudoabsence points (Table 1) or the difference in the number of native versus invasive records (AUC scores usually varied by  $< \pm 0.01$ , except for GBM where scores differed by up to 0.03). Variability, however, occurred among the output of the alternative models (Fig. 1). This is primarily the result of commission error (over-prediction) by a few models, notably ANN and RF. Although these models correctly identified known locations of each species in Australia and within their native range, their projections incorporated parts of Australia that are unlikely to be suitable ecologically, e.g. large areas of central New South Wales (Fig. 1).

The majority consensus (agreement across five or more of the eight models) identifies climatically suitable habitat for the three hawkweed species extending across south-eastern Australia (Fig. 1) and including high altitude regions. These projections suggest that the three hawkweed species can occupy larger geographical areas than in which they are currently found.

As climate change progresses, the potential distributions of the three species are projected to decrease. By 2030 areas of majority consensus are projected to decrease on average 18, 11 and 24%, for *H. aurantiacum*, *H. murorum* and *H. pilosella*, respectively (Fig. 1). By 2070, these areas are projected to decrease by 46, 32 and 40%, respectively. Core

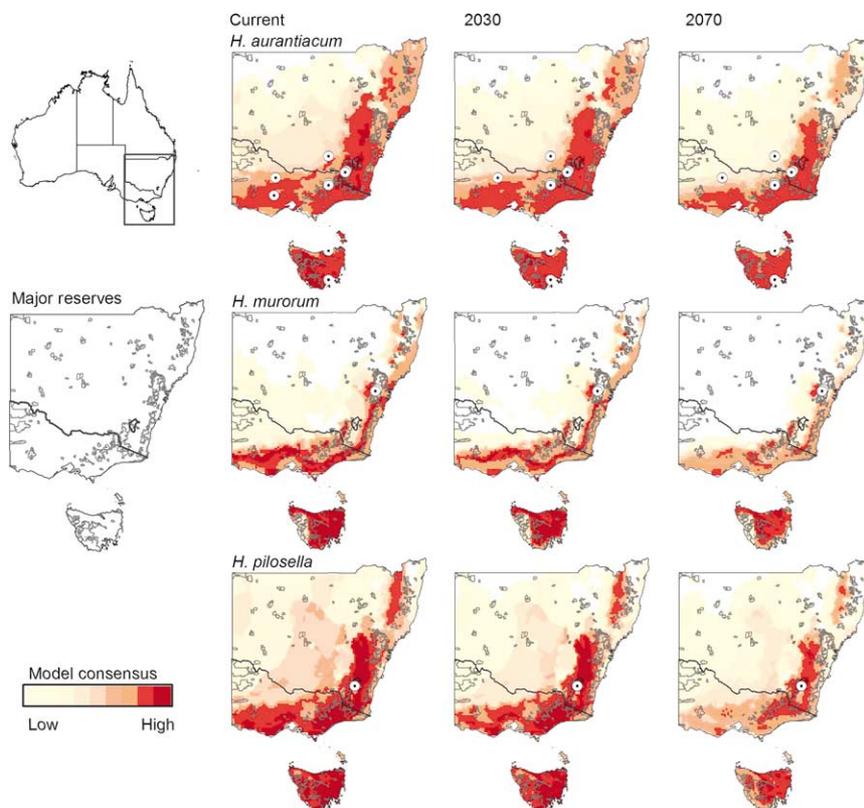


Figure 1. Current and future distributions of three species of *Hieracium* across New South Wales, Australia. The maps show agreement (consensus) across eight alternate ENMs and four climate scenarios for both 2030 and 2070. Circles indicate known locations of the three species.

regions will remain relatively stable, while mainland range margins will contract. Projections for Tasmania remain similar across the three time periods, although model consensus decreases with time, indicating that some models project smaller distributions on this island.

### Hawkweed distributions in protected areas

A key aim of this study was to determine the extent to which biodiversity in protected areas may be exposed to hawkweed infestations as climate changes. We overlaid the projected current and future distributions of the three hawkweed species on Australian protected areas using a GIS. Of the area projected as suitable by the majority of ENMs (i.e. five or more) under current climate, 16, 21 and 18% is contained within the protected areas examined (for *H. aurantiacum*, *H. murorum*, and *H. pilosella*, respectively) (Fig. 2). As the severity of climate change increases, the absolute size of species ranges contained within the parks decreases while the proportion of the majority consensus area within reserves increases slightly (2070: 20, 22 and 21%, *H. aurantiacum*, *H. murorum* and *H. pilosella*). This reflects the contraction of range margins from areas with few reserves to high altitude regions encompassing some of Australia's largest reserves.

### Discussion

Projections of the current distributions of three hawkweed species indicate that they have yet to realize the full extent of climatically suitable habitat available to them under present day climate. This may be a function of their relatively short residence time – *H. aurantiacum* was established in Tasmania prior to 1963 (Williams and Holland 2007) and in Victoria in the early 1990s (Faithful 2000), *H. murorum* in New South Wales in 1998 and *H. pilosella* in Tasmania in 2001 (Williams and Holland 2007) – and the success of control and eradication programs. At present hawkweeds are known from a small number of sites in the central highlands and southern midlands of Tasmania, the Blue Mountains, south coast and southern highlands of New South Wales and alpine regions of Victoria (Fig. 1). Our modelling suggests that climatically-suitable habitat currently occurs across much of south-eastern New South Wales, southern South Australia and Victoria and most of Tasmania. The ability of hawkweeds to realise their full geographic potential under current and future climates may be limited by dispersal. Hawkweed seeds are wind dispersed, with most seeds falling within 2 m of the parent plant, and long-distance events are rare (Williams and Holland 2007, and references therein). Several infestations of hawkweed in Australia are the focus of control programs, and while it is hoped that *H. pilosella* has been successfully eradicated it is possible that undetected populations may still exist (Williams and Holland 2007).

As climate change accelerates, some areas from where the hawkweed species have recently been recorded will no longer remain climatically suitable, and model consensus indicates an overall potential loss of range for these species (Fig. 1). While this is encouraging, the loss of climatically

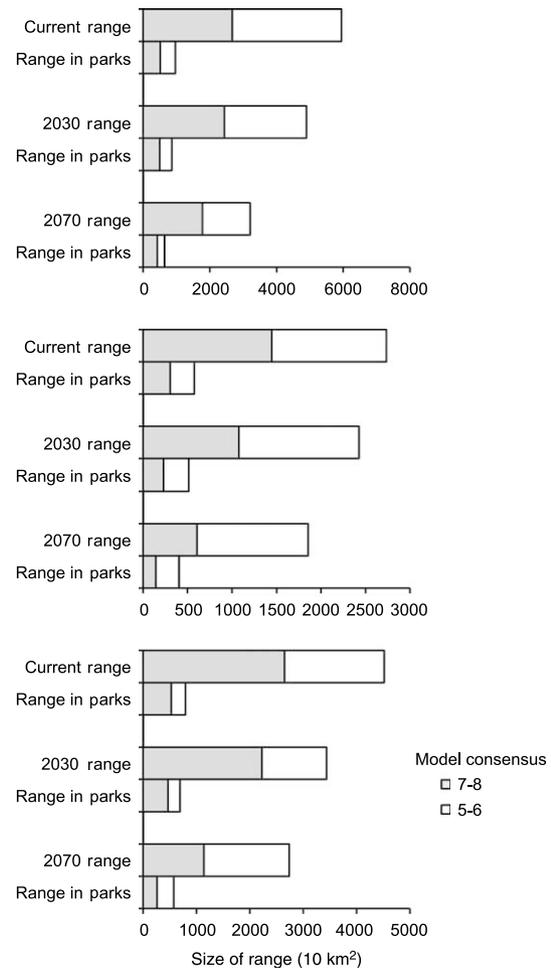


Figure 2. Projected total size of current and future distributions of three species of *Hieracium* and the size of each species potential range that is contained within the boundaries of protected areas. These results represent consensus across 5–6 (white) and 7–8 (grey) models (i.e. areas classified as suitable by 1–4 models are not shown).

suitable habitat primarily occurs at lower elevations. Contraction of range margins towards the core of species' ranges means that much of the contiguous area of sub-alpine and alpine habitat contained within protected areas in Australia's south-east, will remain climatically suitable through to at least 2070; this includes high altitude areas where populations of *H. aurantiacum* and *H. pilosella* currently occur (or are the focus of control programs) (Fig. 1). Considering the richness of endemic species in Australia's alpine zone, this could result in a disproportionately high number of rare and threatened species being at risk from hawkweeds in the future. There are currently 16 plant species listed as vulnerable or endangered under the NSW Threatened Species and Conservation Act (1995) that occur in the Alpine Complex Vegetation Type present within these reserves (Keith 2004). The functional similarity of a number of these native species to the hawkweeds may intensify competitive interactions to their detriment. For example, Max Mueller's burr daisy *Calotis pubescens* plus four other listed species are also members of the daisy family (Asteraceae). This endangered daisy shares a range of

traits with *H. aurantiacum*, such as a perennial habit, hirsute leaves of similar size and a tendency to form thick, sprawling mats at ground level (Harden 1993). Functional similarity has been implicated as an important factor for predicting the impact of species' invasions due to the constraints imposed by life history trade-offs in comparable environments (Forys and Allen 2002, Thuiller et al. 2006b). However, the role of functional similarity is equivocal and there is some evidence that communities may also be able to repel the invasion of functionally similar life-forms (Symstad 2000) and that the impact upon ecosystems varies substantially within functional types (Yelenik et al. 2007).

*Hieracium pilosella* has a number of characteristics that may provide it with a competitive advantage over some native species in Australia. This hawkweed is efficient at absorbing phosphorous, an attribute which may have contributed to its widespread invasion of tussock grasslands in New Zealand (Davis 1997). This ability would be advantageous in soils with low P availability, such as those that characterize much of the Australian landscape. Further, *H. pilosella* is able to extract a major portion of the moisture and nutrients from the soil surrounding it, resulting in a "halo" of soil that is drier, more acidic and less favourable for the development of competing plant species (McIntosh et al. 1995, Boswell and Espie 1998). Glasshouse experiments have recorded a significant CO<sub>2</sub> fertilisation effect on *H. pilosella*, suggesting that the competitive ability of this species may be further enhanced as climate change progresses (Leadley and Stöcklin 1996).

There are several other traits that suggest that climate change may ultimately be detrimental to hawkweeds. While most hawkweed seeds can germinate immediately upon release, late maturing seeds are enhanced by cold treatment, and seedlings are known to be highly susceptible to drought stress (Williams and Holland 2007). Thus with an increase in minimum daily temperature range such "cold treatments" may become rarer events. Better understanding of such traits and their incorporation into future modeling exercises will strengthen future projections and increase our capacity to make effective management decisions.

## Limitations of ENMs

Modelling studies are a useful tool to provide general indications of potential range expansions or contractions of invasive species, although limitations do occur. Consistent with other studies (Thuiller 2004, Araújo et al. 2005, Lawler et al. 2006, Araújo and New 2007) we found substantial differences among the projections of current and future distributions made by the eight ENMs. This was primarily a result of over-predictions made by ANN and RF. In contrast, there was general agreement in the output of GAM, GBM, and GLM. Our modeling illustrates how the use of a poor predictive model could result in a misleading distribution pattern and how "consensus" outputs can be used to make more informed management decisions. A further difficulty is that ENMs do not provide information as to whether a species will realize its current or future potential distribution, as this will depend upon other factors including dispersal, spatial and temporal stochastic

events, competitive abilities and interactions with native species, which may also change under altered climatic regimes. Thus, studies such as these can only provide a first approximation as to the impact of climate change on the species in question.

## Conclusion

As climate change progresses, climatically suitable habitat available to hawkweeds in Australia is projected to decline, although suitable areas will remain in sub-alpine and alpine regions until at least 2070. However, hawkweeds are yet to realise their geographic range in Australia. Even though much of the alpine region of Australia is contained within the national reserve system, this area is small, harbours numerous endemic species and is already at risk from climate change. Our finding that areas surrounding infestations currently occurring in high altitude regions continue to remain suitable under future climate scenarios until at least 2070 highlights the need for ongoing control and monitoring to minimize the likelihood of hawkweeds realizing their full invasive potential.

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