# Human activities, propagule pressure and alien plants in the sub-Antarctic: Tests of generalities and evidence in support of management

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## ABSTRACT

Despite concerns about the richness of plant invaders on islands, and their likely effects on local systems, impacts of these species seem to be small. However, this may be due to an absence of information on impacts, including changing species occupancy and forecast occupancy, rather than lack of impact per se. Here we use the plant invaders on the sub-Antarctic Prince Edward Islands (PEIs) and spatially explicit modeling of presence–absence survey data to demonstrate that the geographic extent of many invasives is increasing and is forecast to lead to occupancy of >60% of the islands' surface area by 2060, with ongoing climate change. In keeping with theory, proximity to human activity, neighboring populations (i.e. propagule pressure) and residence time, along with more minor contributors such as elevation, explain >50% of the variation in the occupancy of each of the six main invasive species on the islands. Human disturbance and changing climates seem to have led to recent increases in the rate of range expansion. Our results suggest that impacts of island plant invaders may be more significant than previously estimated, largely owing to prior data deficiency. More specifically they also suggest that control plans for the PEI (and other Southern Ocean Islands, SOIS) should first target less widely distributed species, which are invasive elsewhere. They also indicate that for the other SOIS, and for Antarctica, surveillance and anticipatory control plans should be in place.

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## 1. Introduction

The introduction of non-indigenous species is a major cause of biodiversity change, especially on islands. These species modify diversity, alter ecosystem functioning, and often cause population declines (D'Antonio and Dudley, 1995; Blackburn et al., 2004; Pyšek et al., 2012). However, plant invaders are generally thought to be having less significant an impact on island systems than others such as mammals (compare Davis, 2003; Sax and Gaines, 2008 with Blackburn et al., 2004). Indeed, urgent calls for further investigation of island plant invaders have been made (e.g. Sax and Gaines, 2008). Vascular plants are the most species rich of the groups that have been introduced to and become invasive on the islands of the Southern Ocean, which are globally significant by virtue of their location, endemicity, and seabird populations (Convey

and Lebouvier, 2009). Nonetheless, the extent of the impacts of invasive plants, by comparison with those of other taxa such as mammals, appears relatively minor (Frenot et al., 2005), and mostly indirect (Frenot et al., 2001; but see also Gremmen et al., 1998). Either the plant invasives are only having a minor impact, or the situation is a consequence of limited explicit investigation of the impacts of invasive plants across the region (for discussion see Vilà et al., 2010; Hulme et al., in press).

If impact for a given area is considered the product of the number of species that has undergone a transition from established to invasive, the range size (or average density or biomass) of each species, and effect per individual or unit biomass (see Parker et al., 1999; McGeoch et al., 2010), few assessments for the Southern Ocean Islands (SOIS) have indeed been made. Investigations of the number and identity of introduced vascular plant species per island are common (Bergstrom and Smith, 1990). However, assessments of changes in the status of species and in their geographic range are less widely undertaken (e.g. Gremmen and Smith, 1999; Scott and Kirkpatrick, 2005). For only one of the 25 SOIS

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has a spatially explicit approach been used to document both gains and losses in area of occupancy (Frenot et al., 2001). Investigations of abundance and local impacts are less common still (Gremmen et al., 1998; Scott and Kirkpatrick, 2005).

Remedying this situation is of considerable conservation significance. If invasive plants are having minimal impact across the region, conservation spending might better be directed at other taxa, such as mammals, which do have significant impacts (Frenot et al., 2005). However, if the purported absence of impact is a consequence of limited investigation, impacts will accumulate until they can no longer be ignored, by that time likely being irreversible (Vilà et al., 2011; McConnachie et al., 2012). Perhaps for this reason, most SOIS management plans currently take a precautionary approach, recommending eradication of alien plants where feasible (de Villiers et al., 2005). However, control of plant invaders across the SOIS is uncommon, and has mostly been piecemeal, largely because the information required for prioritization and planning is absent (Grant et al., 2012). For the same reason, few generalities concerning the mechanisms underlying changes in invasion status (e.g. propagule pressure, residence time) have been developed across the region (unlike the case elsewhere – Křivánek et al., 2006; Gravuer et al., 2008), so further hampering invasion management (for general discussion see McGeoch et al., 2012), and failing to capitalize on an opportunity to test theory (e.g. Wilson et al., 2007) in a relatively remote system. Overall, the situation is not conducive to efficient conservation management, especially given forecasts of increasing invasion impacts for the SOIS (Frenot et al., 2005), and for Antarctica, which is prone to invasion by the same suite of species (Chown et al., 2012a).

Here we quantify the spatial distribution and local species richness patterns of introduced plant species at the sub-Antarctic Prince Edward Islands, as an exemplar system for the SOIS region. We model the factors explaining the current distribution and local richness of these species and, in conjunction with data from previous studies, estimate maximum rates of change in distributions on the islands. We also estimate: (1) the extent to which residence time has played a role in determining the current extent of invasion; (2) the likely equilibrium ranges of the species; and (3) the expected distributions of these species in 50 years' time given current rates of spread of each species and realized climate change (le Roux and McGeoch, 2008). In doing so we follow the useful development of such approaches applied to alien plant invasions in both continental and island settings (e.g. Schussman et al., 2006; Senan et al., 2012).

## 2. Methods

### 2.1. Site, species and residence time

The Prince Edward Islands (46°54′S, 37°48′E) comprise Marion Island (MI: 293 km$^2$, 1 230 m elevation) and Prince Edward Island (PEI: 45 km$^2$, 672 m elevation). They have a cool, oceanic climate and a tundra-type indigenous vegetation, varying from tussock grassland and mires in the lowlands to an impoverished polar desert at the higher elevations (Chown and Froneman, 2008). The islands were discovered in 1673, with the first documented landing in 1803/1804 (Chown and Froneman, 2008). Non-indigenous plants were first recorded in 1873 (*Cerastium fontanum* on MI; Moseley, 1874). In 1947 the islands were annexed by South Africa, with a continuous scientific presence since then on MI (Chown and Froneman, 2008). The first comprehensive survey of the vascular flora took place in 1965/1966 (Huntley, 1971), and biological research has since been continuous (Chown and Froneman, 2008), with the development of a network of field accommodation (huts) around MI in the 1970s (Fig. 1). Human activity on MI peaked during the feral cat, *Felis catus* L., eradication pro-

gramme (1986–1991), involving many field workers moving between the huts and the research station. Prince Edward Island is less commonly visited and now has a higher conservation status than MI (de Villiers et al., 2005).

Seventeen non-indigenous vascular plant species have been introduced to and are currently established on MI, although some are of uncertain status (Table 1). Only three non-indigenous species have been recorded on PEI, all of which are still present (Ryan et al., 2003). Since Huntley's (1971) first assessments, comprehensive surveys of the alien vascular plant species on MI have been undertaken on a sporadic basis (Gremmen, 1975, 1981; Bergstrom and Smith, 1990; Gremmen and Smith, 1999), with fewer assessments for PEI (Bergstrom and Smith, 1990; Ryan et al., 2003; Table A1). Several of the previous studies have been comprehensive, with the authors suggesting that the spatial distributions of the species were well documented (Gremmen and Smith, 1999: 407; Ryan et al., 2003: 556).

To assign a year of first detection to each species we examined the literature on non-indigenous vascular plants recorded on the islands, and confirmed previous herbarium records (at Royal Botanical Gardens, Kew, UK). However, for most species, the first year of detection may not reflect the year of establishment owing to the absence of previous surveys (e.g. species recorded by Huntley, 1971) and to the taxonomic difficulty of some groups (Gremmen and van der Meijden, 1995). Thus, residence times (first year of this study (2006) minus year of first detection) should be considered a minimum estimate (Wonham and Pachepsky, 2006).

### 2.2. Surveys

Data from previous surveys (Table A1) were considered representative of the occurrence of the species unless the authors of the works either suggested that this was not the case or indicated that previous studies may have been so compromised (see Gremmen and Smith, 1999). In previous studies, the occurrence data for many of the species were either described by location or presented as mapped extents of occurrence (*sensu* Gaston, 1990) (e.g. Bergstrom and Smith, 1990; Ryan et al., 2003). Locality data from these studies were digitized (ARcGIS 9.3.1, ESRI, California) as presence data at a 0.5 × 0.5 min (hereafter half-minute) grid resolution (∼926 × 635 m). This was done separately for each species, each of the islands and each of the years for which data were available (Table A1). Although the more cryptic species (e.g. *Agrostis castellana*) may have gone undetected, we made the assumption that the previous data represented a reasonable estimate of true absence. That is, at least since Gremmen's (1975) assessment, the now digitized data could be considered a minimum assessment of area of occupancy (Gaston, 1990) at a half-minute grid resolution for each species. Although such an assumption is complicated by the fact that the original occurrence maps may have included some half-minute grids unoccupied by the species concerned, the relationship between area of occupancy and extent of occurrence is typically strong (Gaston, 1990). Moreover, we selected the half-minute resolution to minimize this likely problem, acknowledging that resolution has an effect on estimates of occupancy (McGeoch and Gaston, 2002).

For the current survey, the half-minute resolution grid was retained for Marion Island. Over a period of 1 year (April 2006–May 2007) the center point of each grid was visited (below 500 m a.s.l., see Fig. 1 and Appendix 1) and a survey made of all non-indigenous vascular plant species within an 8 × 8 m square. Species absences were formally recorded, as was altitude, substrate type, vegetation type, aspect, and slope (see Chown and Froneman, 2008 for descriptions of substrate and vegetation types). As part of the survey protocol, all alien vascular plant species were also recorded *en route* to these sites. These 'ad hoc' re-