Animal introductions to southern systems: lessons for ecology and for policy

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Numerous animal species have been introduced to areas from which they were previously absent, and many of these have become invasive, with substantial impacts. However, in other cases, impacts are assumed from theory. Empirical demonstrations are uncommon, making evidence-based conservation policy difficult to achieve. Here we review the broader ecological and conservation lessons from recent work on non-indigenous species in two southern systems, the policy implications thereof, and the subsequent changes to policy as a result of this work. First, we discuss invasions in the Antarctic region. Strong relationships exist between numbers of animal invasions and numbers of human visitors to Southern Ocean Islands, abiotic factors are often limiting for introduced species, homogenization across islands differs among taxonomic groups, and control actions can rapidly result in unintended consequences. This knowledge has influenced national policy and decisions within the Antarctic Treaty System. Second, we discuss ungulate introductions and translocations, both in South Africa and elsewhere. We show that substantial homogenization has resulted from both processes. However, firm evidence for impacts of ungulate introductions and translocations is sometimes difficult to find, despite the theoretical likelihood thereof. Such a lack of information may have profound consequences for the effective implementation of policy.

Key words: barriers, biological invasions, legislation, mammals, propagule pressure, Marion Island, South Africa.

INTRODUCTION

Biological invasions constitute a significant threat to biodiversity, ecosystem functioning, and ecosystem service provision. Recognition of these threats stretches back to before Elton's (1958) seminal work on introduced species, and indeed to before Darwin's (1859) clear recognition that non-indigenous species pose significant risks to indigenous ones. A wide variety of both strictly scientific and more policy-orientated studies has now demonstrated the substantial ecological and economic impacts that result from the introduction of non-indigenous species which subsequently become invasive (e.g. Mack *et al.* 2000; Sala *et al.* 2000; Millennium Ecosystem Assessment 2005).

An increasing number of studies is also revealing the extent to which the impacts of invasive alien species might be indirect, further facilitate invasion, and later lead to unexpected outcomes of management interventions. For example, the introduction and subsequent spread of European earthworms in North American forests has caused changes to nutrient cycling that have led to elevated levels of plant invasibility in the forest floor environment under invasive, but not under indigenous trees (Belote & Jones 2009). On Christmas Island, the introduction of the invasive ant *Anoplolepis gracilipes* facilitated the invasion of scale insects (Abbott & Green 2007) and resulted in a reduction in the abundance of the red land crab, causing changes to nutrient cycling and habitat structure (O'Dowd *et al.* 2003). The control of particular invasive species has likewise had complex and unanticipated outcomes. For example, in Australia, the introduction of a highly specific weed biocontrol agent (the tephritid fly *Mesoclanis polana* to control the South African daisy *Chrysanthemoides monilifera*) has been associated with declines in the abundance of indigenous insect species via apparent competition (Carvalheiro *et al.* 2008). As a consequence, it is now widely acknowledged that control programmes must be undertaken within a context which explicitly recognizes the potential for substantial ecosystem change during and after a control intervention (e.g. Zavaleta *et al.* 2001; Simberloff 2005).

By contrast, several factors that are key to understanding the likelihood of invasion and the consequences thereof remain poorly investigated. For example, successful dispersal, the first stage of the multi-step sequence which leads to invasion,

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has been much less comprehensively examined than have other stages (Puth & Post 2005; Hulme *et al.* 2008; Blackburn *et al.* 2009). Similarly, the likelihood of particular kinds of impacts of non-indigenous species is frequently assumed, in the absence of studies documenting them. Whilst a precautionary application of information derived from theory, or from empirical evidence from other systems, is a useful first step in anticipating and preventing impacts, and is often the only option available, an evidence-based conservation approach (Pullin *et al.* 2004) is often much more defensible. It is also especially valuable where several sectors of society might be affected by a decision (Pullin *et al.* 2004; Sutherland *et al.* 2004). Indeed, in the absence of an evidence-based evaluation, conservation decisions may face substantial challenges, especially where conservation and economic activities are in conflict, as they often are in the trade and movement of species that are of interest to aquaculture (Arndt *et al.* 2002), angling, or game farming (Castley *et al.* 2001; Butler *et al.* 2005) (see also Hulme *et al.* 2008). As a consequence, legislated policy may prove ineffective in addressing the conservation goals for which it was originally developed. Thus, a considerable need exists for the evaluation of the evidence for particular kinds of impacts attributed to invasive alien species, and the incorporation of these research outcomes into policy.

Here, we illustrate how research into two very different systems, one regional and the other taxon-based, has provided a range of well-supported information that has not only contributed to the establishment and testing of ecological generalities, but that has also provided evidence that can be used to inform policy. We also show how this evidence was readily adopted by policy-makers in one instance, but was adopted only after a substantial period in another. Based on these examples, we also offer perspectives on what might be expected by and required of researchers who wish to enter the policy arena.

NON-INDIGENOUS SPECIES IN THE ANTARCTIC REGION

The scientific evidence

Although they are among the most pristine regions on the planet, Antarctica and its surrounding Southern Ocean Islands (SOI) are nonetheless home to a wide variety of non-indigenous marine, freshwater and terrestrial species (Frenot *et al.* 2005; Aronson *et al.* 2007). Several of these species have become invasive and have had dramatic impacts, not only on indigenous species, but also on ecosystem functioning, especially on the SOI (Bonner 1984; Crafford & Scholtz 1987; Leader-Williams 1988; Chown & Smith 1993; Chapuis *et al.* 1994; Gremmen *et al.* 1998; Copson & Whinam 2001; Bester *et al.* 2002; Frenot *et al.* 2005; Chown *et al.* 2007, 2008). On the continent itself, the non-indigenous *Poa annua* is spreading on King George Island (Chwedorzewska 2008), and recent records have revealed the survival of non-indigenous marine species in the region (Clayton *et al.* 1997; Thatje & Fuentes 2003; Tavares & de Melo 2004; Lee & Chown 2007).

Early reviews recognized the importance of invasive species as significant conservation concerns in the region and made several recommendations for prevention of the further introduction of non-indigenous species (Carrick 1964; Walton 1975; Dingwall 1987; Cooper & Condy 1988). Many of these recommendations concerned the prevention of further intentional introductions, which have largely now ceased (Bergstrom *et al.* 2006; de Villiers *et al.* 2006). However, many introductions, especially to the SOI, have clearly been accidental, especially in the case of rodents and invertebrates (Chown *et al.* 2002; Frenot *et al.* 2005), and are ongoing (see e.g. Lee *et al.* 2007).

Recent work has revealed that across three major taxonomic groups – insects, mammals, and vascular plants – the strongest predictors of non-indigenous species variation across the SOI are the annual numbers of human occupants present on the islands, and mean annual ambient temperature (Chown *et al.* 1998, 2005). Moreover, the number of human occupants is also related to ambient temperature. Thus, warm islands tend to have higher numbers of human occupants than cold ones, but even taking this relationship into account, the former also tend to have high numbers of non-indigenous species. The management implications of these results are immediately apparent. Visitor numbers to islands that are to be kept pristine should be kept low, the pathways for accidental introduction need to be further understood and narrowed for islands with high visitor numbers, and ongoing climate change across many areas in the region (e.g. Bergstrom & Chown 1999; Convey 2006; le Roux & McGeoch 2008) will mean that areas in which alien propagules previously tended not to establish, owing to severe climatic conditions, may become more accessible through time.

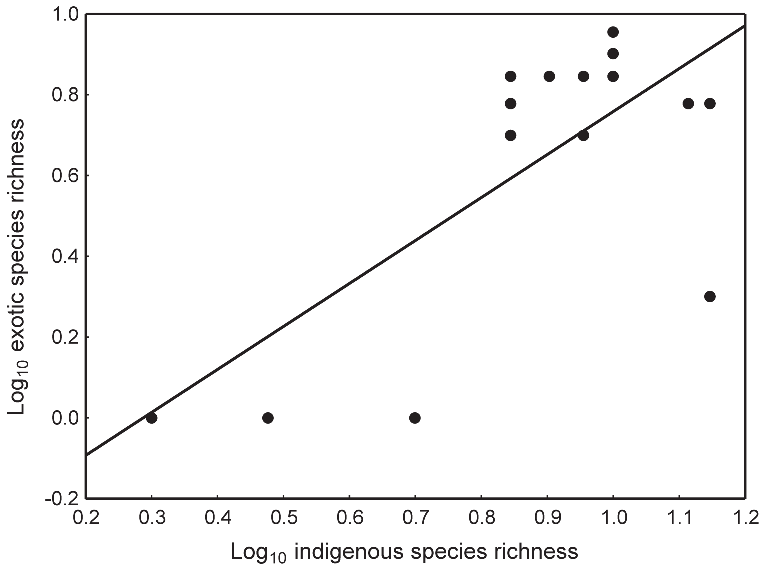

Fig. 1. Plot of indigenous insect and springtail species richness against exotic insect and springtail species richness for 18 habitats on sub-Antarctic Marion Island. The fitted line indicates a significant, positive relationship (redrawn from Chown *et al.* 2005).

This work also provided support for ecological generalities emerging elsewhere. First, it showed that both indigenous and non-indigenous species richness responds positively to energy availability, and that a combination of the productive and ambient energy hypotheses (see Evans *et al.* 2004) likely explains the covariation. Second, in keeping with Stohlgren *et al.*'s (1999 – see also Stohlgren *et al.* 2008) 'rich get richer' hypothesis, the relationship between indigenous and non-indigenous species richness is positive, both at large spatial scales across the islands (Chown *et al.* 2005), and at small spatial scales within particular islands (Fig. 1). Third, assessment of nestedness of the vascular plant, insect and land bird assemblages indicated significant nestedness of both indigenous and non-indigenous species, but significantly more nested assemblages in the case of the latter (Greve *et al.* 2005). This outcome suggested that a group of 'weedy' European species dominated the non-indigenous species being introduced across the region, a view also held by other authors (Walton 1975; Frenot *et al.* 2005).

The nestedness assessment also suggested that one of the more subtle impacts of the introduction of non-indigenous species is likely to be homogenization of assemblages across the islands (see McKinney & Lockwood 1999; Olden 2006 for a general discussion of homogenization by both introductions of non-indigenous species and extinction of indigenous ones). Explicit examination thereof in insects (for the first time in any system) and vascular plants revealed that homogenization across the SOI was taking place for insects, but not for vascular plants (Shaw *et al.* 2009). Close investigation of the assemblage data revealed that common groups of typically Holarctic vascular plant and insect species were being introduced across the region, but that indigenous insect assemblages differed much more substantially among the islands than did indigenous vascular plant assemblages, so leading to homogenization of the insect assemblages and differentiation of the plant assemblages. Further analysis revealed considerable variation in patterns of assemblage change among archipelagos spanning the region, so adding generality to a finding previously made explicitly for birds (Cassey *et al.* 2007).

Much of the early research on the impacts of particular invasive species suggested that considerable conservation benefits would be gained by eradication of one or more of these species (Carrick 1964; van Aarde 1979; Leader-Williams 1985; Dingwall 1987; Cooper & Condy 1988; see also Angel *et al.* 2009). Indeed, eradication or control of several of these species have now either been completed, or have been implemented and are ongoing (Chapuis *et al.* 2001; Copson & Whinam 2001; Bester *et al.* 2002; Towns & Broome 2003). Typically, these interventions have had the positive impacts that were intended (see e.g. Chapuis *et al.* 2001; Ryan & Bester 2008). However, in addition to the positive impact, in one case eradication has led to a trophic cascade that was largely unanticipated in its extent and impact. Following the eradication of feral cats from Macquarie Island, the population density of the invasive rabbit increased substantially, with landscape-wide impacts on the vegetation (Bergstrom *et al.* 2009a). Although the vegetation impacts could have been predicted from previous such impacts on the island (in the 1950s and 1960s), the substantial nature of the increase in the rabbit population and its subsequent impacts was largely unexpected, likely because the extent of top-down control of rabbits by cats was not fully appreciated, for a variety of reasons. Although the extent of this top down control has been questioned (Dowding *et al.* 2009), the original evidence and subsequent analyses continue to show that the removal of top-down control resulted in a trophic cascade, which has had landscape-wide effects (Bergstrom *et al.* 2009b). From a management perspective, the sequence of events illustrates the importance of developing integrated management plans that take into account the likely impacts of control options, as has widely been recommended (e.g. Zavaleta *et al.* 2001; Simberloff 2005).

Evidence-based policy in the Antarctic region

Much of the research on the impacts of invasive species, the relationships of non-indigenous richness with visitor numbers and abiotic conditions, and the importance of particular vectors and pathways has seen relatively rapid take-up into Antarctic conservation policy (e.g. IUCN 1991; Antarctic Treaty Consultative Parties 1991; Mansfield & Gilbert 2008). The outcomes of this work have led to the development of management policies explicitly preventing the introduction of non-indigenous species, both to SOI (Cooper & Ryan 1993; Australian Antarctic Division 2005; Chown *et al.* 2006; de Villiers *et al.* 2006; Parks and Wildlife Service 2006; Davies *et al.* 2007) and to the Antarctic continent (Antarctic Treaty Consultative Parties 1991). Moreover, further recognition of the sheer magnitude of the non-indigenous species problem for some islands (e.g. Gough Island, where 71 of the 99 insect species are non-indigenous, Gaston *et al.* 2003, see also Jones *et al.* 2003), has prompted either revision of some of these plans, or additional measures to prevent introductions (e.g. Davies *et al.* 2007). This rapid take-up of research outcomes into policy can be attributed to the close involvement of many of those doing the scientific research with the development of policy, and to the clear conduit for science into the policy arena via, *inter alia,* the Antarctic Treaty System (for an overview of the Antarctic Treaty System see Stokke & Vidas 1996).

The Scientific Committee on Antarctic Research (SCAR) is permanently represented as an observer providing independent scientific advice (Stokke & Vidas 1996) at the meetings of the Antarctic Treaty Consultative Parties, and is able to introduce Working Papers at the meetings both of the Treaty, and more recently at the Committee for Environmental Protection. By the rules of procedure of the Antarctic Treaty, these Working Papers must be given an opportunity for substantive discussion (unlike Information Papers, which need not be formally introduced and sometimes are not discussed at all) (see Stokke & Vidas 1996). SCAR also enjoys a formal working relationship with the Antarctic Treaty Parties and NGO Observers, having co-hosted or participated in a range of workshops to discuss these and other conservation issues (e.g. IUCN 1991; Rogan-Finnemore 2008). In consequence, independent scientific advice can inform policy in a relatively straightforward manner (although for many issues not without subsequent disagreement on political grounds among Parties).

Because the SOI are not governed by an international treaty, as is the area south of 60°S (i.e. the area over which the Antarctic Treaty, signed in Washington D.C. in 1959, applies), but instead are managed and governed by those countries that have annexed them, the situation is more complex. Nonetheless, a close relationship between scientists and policy makers also exists here. For example, the management plan for South Georgia (Pasteur & Walton 2006), which includes a substantial focus on the management and prevention of biological invasions, was co-written by a scientist long involved in understanding the impacts of invasive alien species in the region (e.g. Walton 1975), and who has spent much time conveying the outcome of science to policy makers. Similarly, the Prince Edward Islands Management Plan, both in its original form (Anon. 1996) and its revision (see Chown *et al.* 2006; Davies *et al.* 2007), has been drafted by, or has had major contributions from, scientists investigating biological invasions in the region. This is true also of the Gough Island

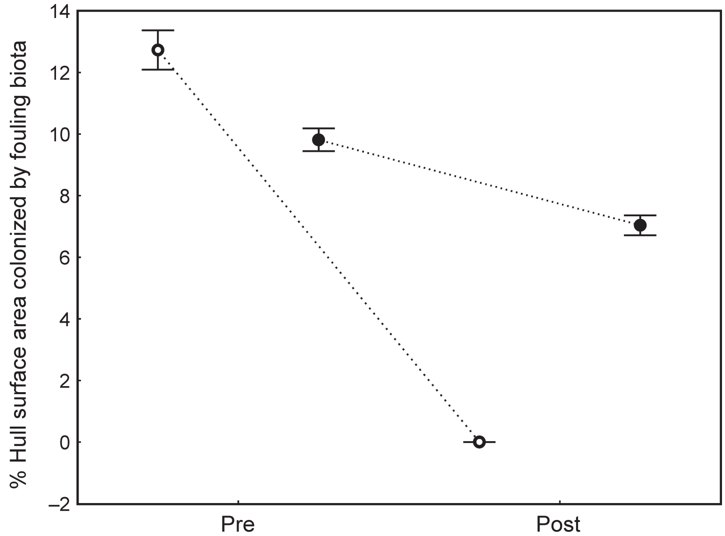

Fig. 2. Mean (±S.E.) fouling cover on the South African re-supply vessel *SA Agulhas* before and after travel though sea ice (**O**) and travel not involving sea ice (●) (redrawn from Lee & Chown 2009a).

Management Plan (Cooper & Ryan 1993). In all of these cases the relationship between researchers and policy makers has been open, and, if not always straightforward, then certainly reasonably effective in the outcomes and in most cases the duration over which it has taken these to be finalized.

An excellent illustration of the nature of the flow of scientific information into policy in the Antarctic region is the recent focus on the vectors and pathways of introductions of non-indigenous species. Early research indicated that accidental introductions are one of the major avenues for the establishment of invasive alien species in the region (Gremmen 1975; Bergstrom & Smith 1990; Jones *et al.* 2003), and it is clear that such accidental introductions are ongoing, often despite substantial management interventions to limit them (see Slabber & Chown 2002; Hughes *et al.* 2005, 2009; Lee *et al.* 2007). In consequence, investigation of vectors and pathways rapidly became a priority. Initially, the focus was on the introduction of diseases into bird and seal colonies (Gardner *et al.* 1997; Kerry *et al.* 1999), but has now shifted to the introduction of non-indigenous species more generally (see Rogan-Finnemore 2008). In the marine realm, work has dealt largely with the risks associated with hull fouling assemblages (Lewis *et al.* 2003, 2004, 2006; Lee & Chown 2007). It has shown that whilst areas surrounded by sea-ice are at low risk of introductions associated with hull fouling assemblages (except those that travel in sea chests – Lee & Chown 2007), substantial fouling assemblages are routinely transported to sub-Antarctic islands where the thermal conditions may allow their establishment (Lee & Chown 2009a) (Fig. 2). Logistics operations are also a pathway for the introduction of propagules into the terrestrial realm and expeditioner luggage, cargo operations and construction activities have been shown to transport large numbers of propagules, some of which are known aliens, into the Antarctic (Whinam *et al.* 2005; Hughes *et al.* 2009; Lee & Chown 2009b, c). Closer examination of this pathway indicates that items such as day packs, camera bags and clothing items which are not issued by the National Antarctic Programme present the highest risk (Fig. 3), and therefore should be the target of management interventions.

The outcomes of this work rapidly made their way into the policy arena, with at least 12 Working Papers (Table 1) on the topic being presented at the Antarctic Treaty's Committee for Environmental Protection over the last 10 years, typically providing explicit details of outcomes and making recommendations for the ways in which pathways can be narrowed and vectors reduced in their efficacy. Many of these recommendations are now seeing implementation by a variety of nations operating in the area, and the International Association of Antarctica Tour Operators (IAATO) (www.iaato.org). For example, it is mandatory for vessels operated by IAATO members to adhere to

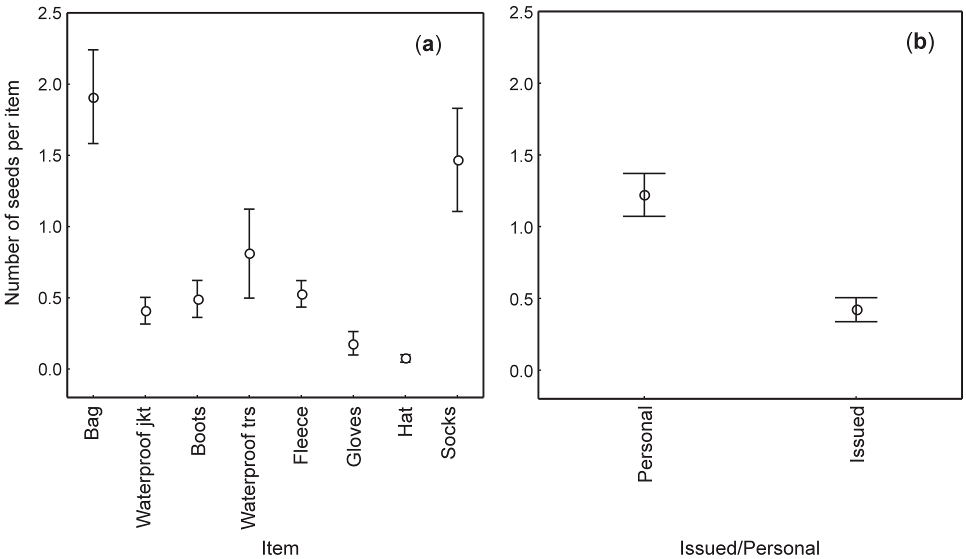

Fig. 3. Mean (±S.E.) number of seeds found per item of field clothing on expeditioners travelling to South African stations on Marion Island, Gough Island and the South African National Antarctic Expedition station, Western Dronning Maud Land, continental Antarctica (**a**) across all clothing items surveyed (**b**) on items issued by the National Operator and participants own clothing (redrawn from Lee & Chown 2009b).

the operational guidelines, which include boot, clothing and equipment decontamination procedures. These guidelines are regularly updated and outline the requirements for boot washing (including a suitably recommended disinfectant) and the checking of pockets prior to disembarkation. IAATO advises that all passengers and small boats are rinsed off prior to re-boarding the ship, highlighting the risk of inter-landing contamination. In addition to IAATO, New Zealand and Australian authorities responsible for regulating tourist permits and visits provide staff representatives to supervise tourist activities, conduct on-board pre-arrival briefings, and to distribute brochures to ensure tourists are made aware of, and comply with, stringent quarantine requirements when visiting these islands.

The outcomes of recent research concerning pathways have also rapidly made their way into protocols implemented as part of the management of several SOI. For example, the clothing inspection (or 'boot-washing' ceremony as it is called.) in advance of arrival to Marion Island (part of the Prince Edward Island group) now focuses especially on high risk items (such as bags and socks) which it previously did not include (J. Cooper, pers. comm.). Moreover, much attention has been given by the Australian Antarctic Division to the design of field clothing and packaging that will help reduce the risks of propagule transfer (Chown 2003; Whinam *et al.* 2005; D. Bergstrom, pers. comm.).

Whether the close involvement of researchers with policy development will continue into the future is difficult to ascertain. Certainly from an Antarctic Treaty perspective this appears likely, especially given the establishment in several of the entities undertaking and managing research and logistics in the area (such as the British Antarctic Survey, Antarctica New Zealand, Australian Antarctic Division, and NSF Office for Polar Programmes) of specific environmental management sections. However, for some countries, the division between science and policy seems to be growing, as environmental management, and environmental policy development and implementation are increasingly separated from the research. In South Africa, this is a particular concern. Recommendations have been made for an integration of Antarctic activities under a single umbrella organization (such as a National Facility managed by the National Research Foundation) (see von Gruenewaldt 2009), but the final decision in this regard has yet to be taken.

NON-INDIGENOUS UNGULATES IN SOUTH AFRICA

The scientific evidence

Non-indigenous ungulates have substantial value as a source of food and recreation (Lever

Table 1. List of Working papers presented at ATCM/CEP meetings referring to non-native species.

Meeting	WP	Title	Presented by
ATCM XXIII (Lima, 1999)	WP 32	Report to ATCM XXIII on outcomes from the workshop on diseases of Antarctic Wildlife	Australia
SATCM XII (The Hague, 2000)	WP6	Diseases of Antarctic wildlife	Australia
ATCM XXIV (St Petersburg, 2001)	WP10	Report on the open-ended intersessional contact group on diseases of Antarctic Wildlife: report 1 – review and risk assessment	Australia
	WP11	Report on the open-ended intersessional contact group on diseases of Antarctic Wildlife: report 1 – practical measures to diminish risk (draft)	Australia
ATCM XXVIII (Stockholm, 2005)	WP28	Measures to address the unintentional introduction and spread of non-native biota and disease to the Antarctic Treaty area	Australia
ATCM XXIX (Edinburgh, 2006)	WP5	Practical guidelines for Ballast water exchange in the Antarctic Treaty area	U.K.
	WP13	Non-native species in the Antarctic. Report of a Workshop	New Zealand
ATCM XXXI (Kyiv, 2008)	WP16	Antarctic alien species database	Australia
ATCM XII (Baltimore, 2009)	WP23	Propagule transport associated with logistic operations: a South African appraisal of a regional issue	South Africa
	WP32	Procedures for vehicle cleaning to prevent transfer of non-native species into and around Antarctica	U.K.
	WP5	A work programme for CEP action on non-native species	Australia, New Zealand, France
	WP33	Review of provisions relating to non-native species introductions in ASPA and ASMA management plans	U.K.

1985). For example, large revenues are obtained annually from the hunting of non-indigenous ungulates (Hofer 2002; Loveridge *et al.* 2006; Lindsey *et al.* 2007). However, non-indigenous ungulates can also have substantial impacts on biodiversity. Like other alien species they alter habitat and the functioning of ecosystems (Wardle *et al.* 2001), and compete with (Baccus *et al.* 1985; Côté 2005), eat (Fordham *et al.* 2006), infect (Meltzer 1993) and/or hybridize with (Goodman *et al.* 1999) indigenous species. In consequence, understanding the invasion pathway in ungulates, and establishing evidence-based options for mitigation of impacts are of considerable economic and conservation significance.

Recent work has shown that globally Stohlgren *et al.'s* (1999) 'rich get richer' hypothesis does not apply for ungulates. That is, no significant relationship between the number of established non-indigenous ungulate introductions and indigenous ungulate species richness exists across countries, even when taking country area into account (Spear & Chown 2009a). At the country level, countries that are species rich (such as South Africa) have introduced large numbers of non-indigenous ungulates, but so have countries (such as New Zealand and Australia) with no indigenous ungulate species (Fig. 4a). By contrast, countries that are species rich are a source of non-indigenous species in proportion to their indigenous species richness (Fig. 4b). Globally, and in South Africa, the movement of ungulates has led to homogenization (see above) measured as a 2% increase in similarity of assemblages (Spear & Chown 2008). The outcome is similar to that for plants (Rejmánek 2000; McKinney 2004; Castro *et al.* 2007), fish (Taylor 2004) and insects (Shaw *et al.* 2009) at the resolution of aquatic ecoregions, territories, provinces, islands and states.

Within South Africa, differentiating between extra-regional species introductions (introductions from outside the natural distribution range of a species and outside geopolitical boundaries) and extra-limital species introductions (outside the natural distribution range of a species, but within a geopolitical unit, e.g. a country) reveals considerable subtlety to homogenization. Typically, the effects of extra-regional species are differentiation,

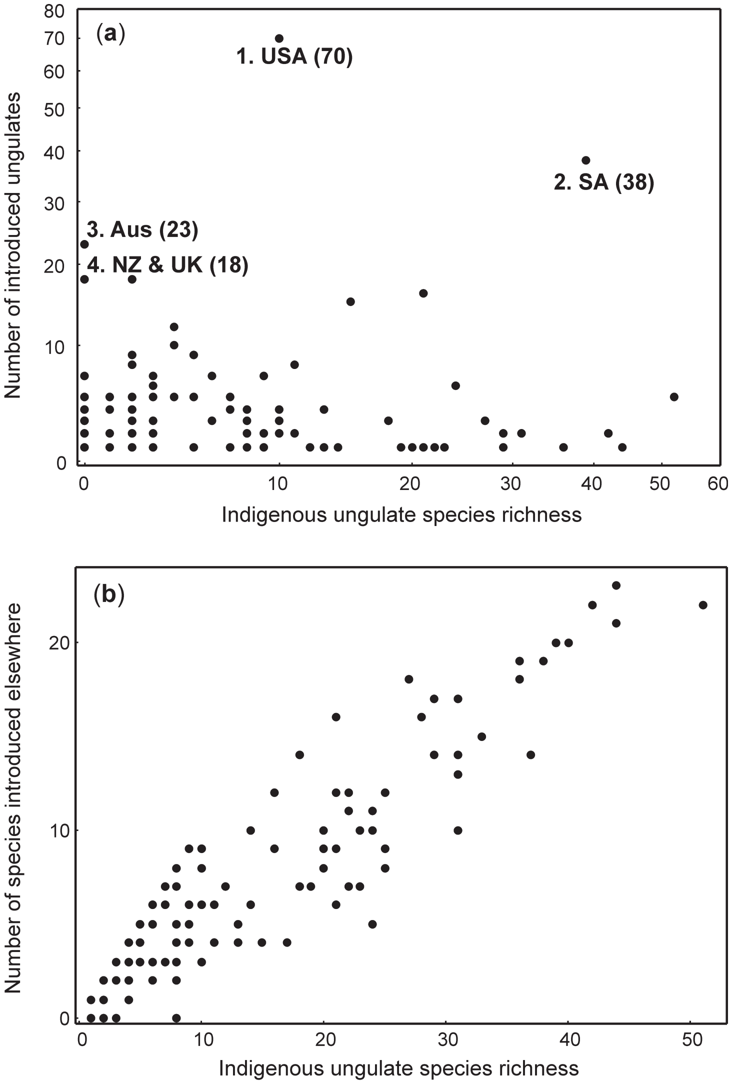

Fig. 4. **a**, The relationship between the number of introduced (non-indigenous) ungulates and indigenous ungulate species richness (both on a log scale) for countries globally. The top five countries for non-indigenous ungulate introductions are annotated with the number of ungulates in brackets. **b**, The relationship between indigenous ungulate species richness and the number of species indigenous to a country that have been introduced elsewhere (redrawn from Spear & Chown 2009a).

which has not changed over time. That is, extra-regional introductions differentiate assemblages (Fig. 5), suggesting that often different ungulates are preferred in different areas, even though certain species (e.g. fallow deer) are introduced quite broadly. By contrast, extra-limital introductions have been increasing the homogenization of assemblages over time (Fig. 5). Indeed, the range change of some species has been spectacular: as much as an 187% increase on average, and more than 800% maximum (for species) (Spear & Chown 2009a). These range expansions are far larger in terms of the percentage change, than the percentage change predicted for climate-change associated range contractions for the same region (Erasmus *et al.* 2002), illustrating the significant influence that trade in these species is having. Moreover, it is not only private landowners that are engaging in such trade, but zoos are also selling off the majority of their surplus stock to non-zoos (Spear & Chown 2009a).

Whilst homogenization is therefore clearly a conservation concern, much of the focus on the impacts of ungulates has not been on homogenization. Rather, the focus has been on impacts, such as those associated with hybridization, competition and herbivory (Castley *et al.* 2001, Butler *et al.* 2005). Given the substantial domestic, and indeed international, economic interest in these species (Hofer 2002; Loveridge *et al.* 2006; Lindsey *et al.* 2007), establishing the extent of global evidence for these impacts is of considerable importance, especially because recent legislation (e.g. for South Africa, see Anon. 2004, 2009) seeks to limit the movement of species.

Recent work by Spear & Chown (2009b) has ranked the requirements for establishing a given impact of ungulates as weak to strong, based on best practise in empirical ecology, and has examined the evidence for each of these impacts in the published Anglophone literature. The outcomes are striking. The published evidence for introgression resulting from hybridisation in the wild is weak, as it typically is for apparent and exploitative competition between introduced and indigenous ungulates. By contrast, strong evidence is available for the impact of non-indigenous ungulates on vegetation structure and composition. Where the published evidence is weak, this does not mean that the impacts might not be realized, but rather that strong evidence for them is often lacking, and this is especially true in the South African situation (Spear & Chown 2009a,b). Moreover, in South Africa the focus has also been on extra-limital species, rather than on extra-regional ones, despite the widespread introduction of the latter, and evidence of their impacts elsewhere (e.g. Wardle *et al.* 2001). This is a particularly problematic situation, given the considerable scope for conflict that exists between private landowners and conservation agencies (Veblen *et al.* 1992; Castley *et al.* 2001; Bradshaw *et al.* 2007).

Evidence-based policy for ungulates

The national legislation that regulates the movement of species, at least in the context discussed here, is the National Environmental Management: Biodiversity Act (NEM:BA, Anon. 2004, 2009), and specifically Chapter 5 thereof. Given that the Act was published in June 2004, much of the more recent research on ungulates reported above has not influenced the development of the legislation. However, it appears that previous research outcomes about the full range of invasive alien species (i.e. microbes, plants, invertebrates, and vertebrates) and their management were also not included, given substantial controversy in the public-comment process on the proposed Act, and how these comments were finally incorporated into the promulgated version of the Act. This controversy subsequently spilled over into the process for the development of regulations for the Act, because the Task Team responsible for the process was faced with the requirement of developing a set of regulations that were in keeping with current science, and which were practicable for the South African situation, on the basis of an Act which in several ways seemed to recognize neither. To a large extent this resulted in a considerable delay in the process, which was initiated in June 2004, and has not yet been completed (see Macleod 2006; Anon. 2009). More specifically, considerable disagreement arose between key individuals in the government department responsible for NEM:BA (at the time, Environmental Affairs and Tourism), and the scientists and policy advisors represented on the initial Task Team (S.L. Chown, pers. obs.). This disagreement added to the complexity of negotiations between the Task Team and stakeholders, so making the incorporation of sound scientific evidence into the regulations more difficult than it otherwise might have been, especially since consensus about scientific questions, such as species ranges, was reached relatively quickly in stakeholder workshops (S.L. Chown, pers. obs.).

The long delay in finalization of a draft set of regulations nonetheless had several significant benefits. Most notably, the revised draft regulations (as now published, Anon. 2009) include more current science, including aspects of the work reported here, and are largely much more practicable for most South Africans. Of course, these draft regulations (Anon. 2009) are still in the process of public comment, and given the interests of a wide range of economic sectors (angling, aquaculture, game farming, horticulture), may still be altered. However, it now seems that best scientific practise will continue to have an appropriate voice in a process that is complicated by a larger variety of stakeholders and issues than is typically the case for Antarctic policy (where only science, tourism and conservation are the key issues).

Nonetheless, the situation that developed regarding Chapter 5 of NEM:BA does raise the question of how current science can best be integrated into the policy environment. Clearly, some organizations established to do so have been effective, but in

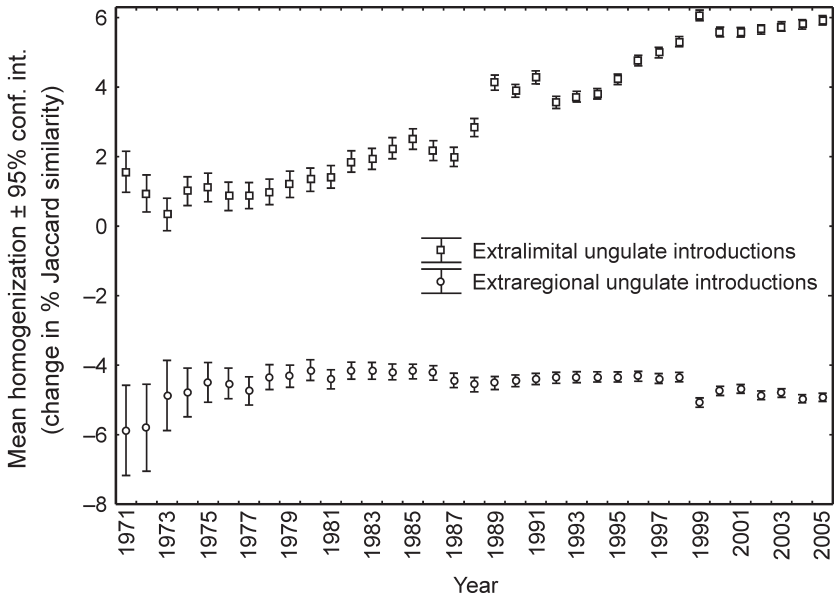

Fig. 5. Temporal trends in homogenization as a result of extraregional and extralimital introductions to South Africa, at the quarter-degree grid-cell resolution, between 1971 and 2005 (redrawn from Spear & Chown 2008).

other cases the situation is considerably more labile and the integration of science into policy more difficult. In South Africa, to date, greater success has perhaps been achieved on a regional scale (e.g. Cowling *et al.* 2003) than nationally. One reason for this may be the difficulty policy-makers face in keeping abreast of and extracting relevant science developments (the difficulty of staying abreast of the literature remains a considerable modern concern, even though it is venerable – see e.g. Krogh 1929). Another may be the tendency, owing to time constraints, of holding stakeholder meetings jointly with all stakeholders, including those undertaking science in the particular area of concern. This means that the opportunities for careful scrutiny of the available evidence may be lost in a time-constrained process. In consequence, decisions may, at times, be more opinion-based than evidence-based. Although opinion-based approaches can be useful (e.g. Sutherland *et al.* 2006, 2008), at times they may be substantially at odds with the available evidence (e.g. Chown *et al.* 2001). Moreover, useful ways of developing appropriate evidence, from similar situations elsewhere, or using appropriate information from the area of interest, are now available (see Pullin *et al.* 2004; Sutherland *et al.* 2004; Pullin & Stewart 2006). Perhaps one solution would be for those developing policy to more regularly hold a series of proactive science for policy round-ups, based on current evidence, and which identify substantial research gaps for the policy questions that might arise (see e.g. Spear & Chown 2009b). Importantly, for such an intervention to succeed, participation by the scientific community is required. Although such participation has, to date, been relatively simple to ensure, increasingly onerous intellectual property and other clauses in legal agreements that now seem set to be a feature of the 'science to policy' landscape in South Africa may limit the extent to which the best advice is made available.

CONCLUSIONS

Here we have demonstrated, using two very different situations, how both basic and applied ecology have provided a range of well-supported information that has not only contributed to ecological generalities, but has also informed policy. Whilst incorporating science outcomes into the policy environment is not always straightforward, and often requires substantial commitment, doing so is important because science is such an effective way of understanding our world and improving the way we interact with it (see Ziman 1978, 2007). From a conservation perspective this point has also made by Terborgh (1999) who concluded that 'Nature can be saved, but only through a thoughtful combining of good science and strong institutions', and Gore (2002) who argued 'I am resigned to the idea that even though we already know more than enough, we must also investigate any significant scientific uncertainty…' Indeed, Erlich's (1997) plea to ecologists that they cannot afford to devote their time only to questions that pique their intellectual curiosity, but rather that they should also be concerned with work which will help understand and mitigate the profound effects that humans are having on the living world, and with educating society in general about the need for environmental conservation, remains as important now as it was more than a decade ago.

This does not mean that engagement in the policy environment will be straightforward or frustration-free. Rather, the converse seems likely, and here we find two further sets of advice helpful. In the first, Dyson (2006) showed that '…three human weaknesses devilishly hamper our efforts … an inability to define or agree upon our objectives … an inability to raise sufficient funds … the fear of a disastrous failure…' and argued that one should act strategically to overcome these weaknesses. The second, from Winston Churchill, has a much more personal flavour, though we have each found it useful in a policy context: 'Never give in, never give in, never; never; never; never – in nothing, great or small, large or petty – never give in except to convictions of honour and good sense.'

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