# 5 The Role of Human Activities in the Introduction of Non-native Plants to Antarctic and Sub-Antarctic Islands

**Eduardo Fuentes-Lillo[1,2,3,4]* and Marely Cuba-Diaz[5,6]**
*[1]Universidad de Concepción, Chile; [2]Research Group on Plants and Ecosystems (PLECO), University of Antwerp, Belgium; [3]Instituto de Ecología y Biodiversidad (IEB), Santiago, Chile; [4]Adventist University of Chile, Chillán, Chile; [5]Universidad de Concepción, Los Ángeles, Chile; [6]Programa de Ciencia Antártica y Subantártica, Universidad de Concepción, Concepción, Chile*

**Abstract**
Antarctica and the sub-Antarctic islands have a series of climatic and geographic conditions that result in high levels of endemism and are of high priority for conservation. The synergies between climate change, globalization and biological invasions have made these ecosystems vulnerable to biodiversity loss. In Antarctic and sub-Antarctic ecosystems, human activities (industrial, scientific and tourism) have historically functioned as vectors for the transport of propagules of non-native species, but due to extreme climatic conditions the risk of establishment has been low compared with other ecosystems worldwide. Current trends show that over the next few decades there will be a significant increase in tourism in these ecosystems (*c*.100,000 tourists annually), which presents a greater risk of invasion associated with higher propagule pressure. These factors combined with the effect of climate change, will increase the establishment of non-native plants, increasing the risk of future impacts on the biodiversity of these ecosystems. This chapter reviews the most common non-native plants documented in Antarctic and sub-Antarctic ecosystems, assesses their impacts and determines the role of tourism in the transport, establishment and dispersal of non-native plants.

## 5.1 Introduction

Sub-Antarctic and Antarctic ecosystems are among the ecosystems least impacted by human activities, mainly due to the geographic isolation from population centres and the extreme climatic conditions that have limited the historical movement of people to these areas (Convey *et al*., 2012).

During the last 200 years, human activities in these ecosystems have changed. At the beginning of the 20th century, whaling and seal hunting were the main activities in the sub-Antarctic islands and maritime Antarctic, while today, scientific and tourist activities are dominant (Bergstrom and Selkirk, 2007; le Roux *et al*., 2013). One of the main impacts of human-associated activities in these areas has been the transport of non-native plants, where a minimum percentage has been established in the sub-Antarctic islands and some areas of the maritime Antarctica (Frenot *et al*., 2005; Chown

*Corresponding author: efuenteslillo@gmail.com



DOI: 10.1079/9781800620544.0005

*et al.*, 2012; Fuentes-Lillo *et al.*, 2017; Hughes *et al.*, 2020).

From an ecological point of view, Antarctic and sub-Antarctic ecosystems have low invasibility because they are characterized by extreme abiotic conditions (low temperatures, strong winds, low nutrient levels and low water availability) that function as a filter limiting non-native plant establishment (Galera *et al.*, 2018; Bazzichetto *et al.*, 2021). In addition to these conditions, the low propagule pressure due to the large distance from propagule sources (continent) and the low flow of human activities compared with other ecosystems in the world, are factors that reduce the invasibility of sub-Antarctic and Antarctic ecosystems (Galera *et al.*, 2018).

However, the current processes associated with global change, including an increase in regional temperatures and anthropogenic pressure (construction of scientific stations, increase of tourists, etc.), are expected to modify the factors that are now limiting the establishment of non-native plants in Antarctic ecosystems (Convey, 2011; Znój *et al.*, 2017; Duffy and Lee, 2019; Hughes *et al.*, 2020). Climatic models suggest that a 1.5°C increase in temperature would have implications for the expansion of ice-free areas, increasing the area of colonization for non-native plants, as well as facilitating the establishment of non-native plants due to increased water availability, nutrient availability and soil development (Amesbury *et al.*, 2017; Siegert *et al.*, 2019). It is also expected a significant increase in the number of people visiting both the sub-Antarctic islands and the maritime Antarctic, with predictions suggesting that close to 100,000 people will visit these ecosystems annually over the next 10 years (Bender *et al.*, 2016). This sustained visitor increase will significantly increase the propagule pressure of non-native plants and consequently their dispersal within and between biogeographic zones (Chown *et al.*, 2012; Kariminia *et al.*, 2013; Hughes *et al.*, 2019, 2020).

A significant increase in the number of non-native plants could bring a series of impacts on biodiversity both for native plants and for biotic interactions that depend on the presence of native vegetation inhabiting these ecosystems (Shaw *et al.*, 2010; Greve *et al.*, 2017; Molina-Montenegro *et al.*, 2019). Currently, although the impacts of non-native plants have not been investigated in depth, experimental and observational studies have observed that non-native species can negatively affect native species growth, modify the microclimatic conditions of invaded ecosystems and generate biotic homogenization (Greve *et al.*, 2017; Molina-Montenegro *et al.*, 2019). The objective of this chapter is to provide an overview of the non-native plants documented on the sub-Antarctic islands and Antarctic continent, assess their potential impacts and determine the role of tourism in the transport, establishment and dispersal of non-native plants.

## 5.2 Sub-Antarctic and Antarctic Ecosystems

The Antarctic region can be divided into three biogeographic zones: (i) the sub-Antarctic; (ii) maritime Antarctic; and (iii) continental Antarctic (Convey, 2017). Maritime and continental Antarctica are among the ecosystems that have historically shown the least human influence (Hughes *et al.*, 2019). This lower anthropogenic influence is determined by the remoteness of this continent from urban centres (which are between ~1000 km and 5000 km away, depending on which part of the continent is being considered) and extreme climatic conditions (Convey, 2011). These areas are mostly ice-covered (about ~99%) and feature about ~1% ice-free, where most terrestrial life develops (Duffy and Lee, 2019). Continental Antarctica is characterized by low temperatures ranging from -48°C to 9°C, low precipitation, strong winds and high salinity levels, all of which have limited the development of plant life (Convey, 2011, 2017). The maritime Antarctic has less severe climatic conditions particularly during the growing season, with temperatures ranging from -45°C to 15°C (Convey, 2011, 2017).

These extreme climatic conditions have shaped its vegetation, composed mainly of cryptogam species (mosses and lichens ~300 species) and only two vascular plants: *Colobanthus quitensis* (Kunth) Bartl. and *Deschampsia antarctica* Desv. (Convey, 2017). Most plant life develops in maritime Antarctica, near coastal areas where climatic conditions are less extreme and there is a higher percentage of ice-free areas (Convey, 2017; Duffy and Lee, 2019).

Compared with continental and maritime Antarctica, the sub-Antarctic islands have a greater diversity of vascular plant species, due to less severe climatic conditions and greater proximity to sources of propagules. Sub-Antarctic islands have temperatures in the growing season varying between -2°C and 25°C. These islands have a higher annual rainfall (~831 mm), compared with the maritime Antarctic where it does not exceed 300 mm a year (Convey, 2011, 2017). Geographic isolation, extreme climatic conditions, glaciation processes and low anthropogenic pressure (last ecosystems to be anthropogenically intervened) have resulted in many endemisms (Chau *et al*., 2020). Some examples of endemic plants are *Elaphoglossum randii* (Fée) Christ, *Colobanthus kerguelensis* Hook.f., *Poa cookii* Hook.f., *Pringlea antiscorbutica* Brown ex Hooker, *Ranunculus moseleyi* Hook.f., *Azorella macquariensis* Orchard, *Festuca kerguelensis* Hook.f., *Juncus scheuchzerioides* Gaudich (Chau *et al*., 2020). Not only do endemisms occur in plants but also in other taxonomic groups that play an important role in the sustainability and processes of these ecosystems, as in the case of bacteria (e.g. *Antarctobacter heliolhermus*), fungi (*Friedmanniomyces endolithicus*), microalgae (*Hemichloris antarctica*) and protozoa (*Notodendrodes antarctikos*) (Vyverman *et al*., 2010).

The level of endemism and the high biodiversity of native species make these ecosystems of high conservation value, so reducing anthropogenic impact is fundamental as it can negatively affect endemic and native flora (Bergstrom and Selkirk, 2007). The impacts of human activities include fuel pollution, which has impacted soil biota, limiting insect reproduction, inhibiting seed germination and plant growth (Errington *et al*., 2018). Another anthropogenic impact associated with tourism is intense trampling that directly affects the native flora, reducing species richness, size and vegetation cover (Convey, 2011). Finally, the transport of non-native plants to sub-Antarctic islands and maritime Antarctica is one of the most important human impacts in these ecosystems (Chown *et al*., 2012; Fuentes-Lillo *et al*., 2017; Hughes *et al*., 2019).

## 5.3 Human Activities in Sub-Antarctic and Antarctic Ecosystems

Human impacts on Antarctic and sub-Antarctic ecosystems date back over 200 years, and originally concentrated on marine mammal exploitation (seals and whales) (Tin *et al*., 2014). This activity promoted the development of areas of industrial concentration, including land-based facilities, throughout the South Shetland and South Orkney Islands and the northern Antarctic Peninsula (Tin *et al*., 2014). By the mid-19th (seals) and mid-20th (whales) centuries, rampant over-exploitation led to the collapse of these industries, though left a legacy of human disturbance and decaying infrastructure. The International Geophysical Year of 1957–1958 and the subsequent negotiation of the Antarctic Treaty, one of whose founding principles is the preservation of Antarctic ecosystems, ushered in a new era where scientific and tourist activities predominate. Technological developments in the latter part of the 20th century have allowed scientific stations and expeditions to reach the most remote areas of the sub-Antarctic and Antarctic ecosystems, challenging the definition of ‘pristine’. To date, it has been suggested that pristine areas are reduced to only 32% of Antarctica (Leihy *et al*., 2020). With the adoption of the Protocol for Environmental Protection to the Antarctic Treaty (negotiated in 1991, came into force in 1998) and through that the formation of the Antarctic Treaty Consultative Meetings (ATCMs) Committee for Environmental Protection, currently conservation and environmental protection are some of the highest priorities in the Antarctic Treaty System. Currently, scientific activities and tourism are mainly concentrated in the vicinity of the Antarctic Peninsula and South Shetland Islands ( COMNAP, 2017; IAATO, 2021).

Of the *c*.5000 scientists and support staff that work in Antarctica each year, a significant percentage use the Fildes Peninsula as a gateway, through both air and shipping operations. The tourism industry brings the highest number of people, with the last decade seeing a significant increase in the number of visits to Antarctica. For example, during 2003 about 15,000 people visited Antarctica, while in 2019 this number increased to 74,000 people, including people

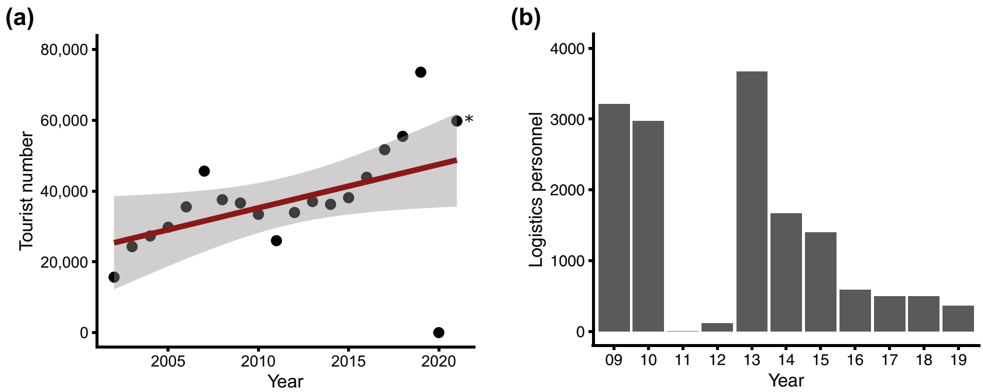


**Fig. 5.1.** (**a**) Number of tourists visiting Antarctic ecosystems annually from 2002 to 2020. *Year 2021 corresponds to 2021–2022 projection proposed by the International Association of Trade Training Organisations (IATTO). (**b**) Number of people visiting Antarctica for logistical activities (support for scientists, military activities, etc.) from 2009 to 2019. Authors' own work.

travelling both by air and by sea (Fig. 5.1a). Regarding logistical support, the data shows a decreasing trend in the number of people travelling to Antarctica, for example, while in 2009 about 3000 people travelled to support scientific activities, only 400 people travelled in 2019 (Fig. 5.1b). Although the current coronavirus disease 2019 (COVID-19) pandemic will lead to a drastic reduction in both national governmental and tourism operations (only 15 visitors during 2020, Fig. 5.1a) (Hughes *et al.*, 2019; IAATO, 2021), a return to the recent trends of increasing numbers of scientists and tourists (around 55,000 for the period 2021–2022, Fig. 5.1a) and diversification in the activities carried out, is expected (Tin *et al.*, 2014; Hughes *et al.*, 2020). For scientific activities, the number of visits to Antarctic ecosystems each year is variable and not very accurate since there is no recorded data over time and these numbers vary according to the season. In 2017, for example, about 1500 scientists visited Antarctica in the summer season, but in other years these numbers can reach approximately 5000 scientists during the summer season and about 1000 during the winter ( COMNAP, 2017).

Compared with Antarctica, the sub-Antarctic islands have less visitor use, mainly due to travel restrictions to these islands, less tourist interest and difficult access (Landan, 2007). For example, only ~1500 people have visited the sub-Antarctic islands of New Zealand between 1967 and 2015 (Stewart *et al.*, 2017). Among the sub-Antarctic islands, South Georgia Island receives the higher number of visitors with ~6000 people annually, while Crozet, Kerguelen, South Sandwich, Heard and Macquarie Islands receive only about 500 people annually (Tracey, 2007). These numbers may not be accurate because unlike visits to Antarctica, the International Association of Trade Training Organisations (IAATO) does not keep a record of visits to sub-Antarctic ecosystems, therefore the numbers may be larger and likely to increase. This increase is due to the growing interest in Antarctic tourism, as tour operators choose to visit various sub-Antarctic islands as a preliminary step to landing on the Antarctic Peninsula, to give an additional attraction to the trip (Stewart *et al.*, 2017).

## 5.4 Plant Introductions in Sub-Antarctic and Antarctic Ecosystems: an Overview

Unlike continental ecosystems, where non-native plants are common, Antarctic and sub-Antarctic ecosystems have historically shown low levels of non-native species introductions, including invasive species (Hughes and Convey, 2012). Natural dispersal processes (e.g. water, wind, ocean currents and birds) of non-native plants are infrequent between continental lands and Antarctica, while natural dispersal of non-native plants to sub-Antarctic islands is more frequent (Hughes and Convey, 2012; Hughes *et al.*, 2020). Most of the natural long-distance transport of non-native propagules to Antarctica corresponds to spores of bryophytes and lichens (Hughes and Convey, 2012), while the non-native plant *Ochetophila trinervis* Gillies ex Hook, has been observed to be transported to sub-Antarctic Marion Island from the Andean mountains by wandering birds (Kalwij *et al.*, 2019). In this context, human activities have historically been the main vector for the transport of non-native plants (Chown *et al.*, 2012; Hughes *et al.*, 2020).

Most of the introductions of non-native plants in sub-Antarctic islands occurred during the 19th century associated with the high anthropogenic disturbance caused in the sub-Antarctic islands from the exploitation of marine mammals (Convey, 2007; le Roux *et al.*, 2013; Tin *et al.*, 2014). Nowadays, the presence of scientific stations, coupled with visitor use and climate change have facilitated the dispersal of non-native plants from the points of introduction, driven by the presence of roads and trails (Bazzichetto *et al.*, 2021). During the last few years about 110 non-native plants have managed to become established in the sub-Antarctic and Antarctic ecosystems, but only ~5% (six species) of these have become invasive (e.g. *Poa annua* L, *Poa pratensis* L, *Cerastium fontanum* Baumg) (Williams *et al.*, 2013). A high percentage of these species are of European, Asian and North African origin and are common invaders in Arctic and high mountain ecosystems (Frenot *et al.*, 2005).

Of all the 110 non-native species recorded, Crozet (58), Kerguelen (53) and South Georgia (34) islands have the highest number of non-native species (Fig. 5.2). Of these, there are nine common plant species that have been established in both sub-Antarctic islands and maritime Antarctica, including the herbs *C. fontanum*, *Rumex acetosella* L, *Stellaria media* (L.) Vill., *Sagina procumbens* L and the grasses *P. annua* and *P. pratensis* (Table 5.1).

About 27% of the non-native species belong to the Poaceae family, 20% to Asteraceae, 7% to Brassicaceae and Juncaceae and 5% to the Fabaceae family. The remaining 33% is

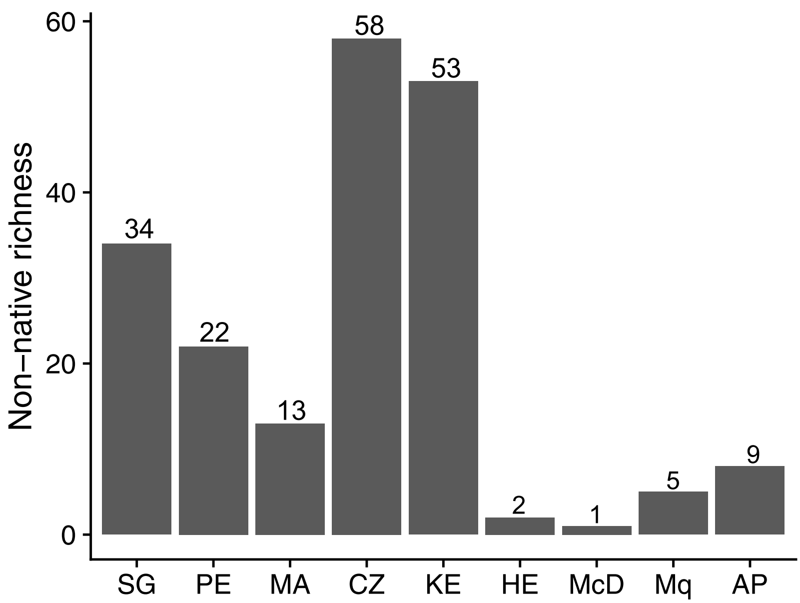


**Fig. 5.2.** Richness of established non-native species in sub-Antarctic islands and Antarctica Peninsula. SG, South Georgia; PE, Prince Edward; MA, Marion; CZ, Crozet; KE, Kerguelen; HE, Head; McD, McDonald; Mq, Macquarie; AP, Antarctic Peninsula. Authors' own work.

**Table 5.1.** Most common non-native species on the sub-Antarctic islands and the Antarctic Peninsula. Authors' own work.

| Species | Family | Biogeografic origin | Establishment zone | References |
|---|---|---|---|---|
| *Alopecurus geniculatus* L[a] | Poaceae | Eurasia | Prince Edward, Crozet, Kerguelen, Antarctic Peninsula | Greene and Walton (1975), Frenot *et al*. (2005), le Roux *et al*. (2013) |
| *Cerastium fontanum*[a] | Caryophyllaceae | Europe | South Georgia, Marion, Prince Edward, Crozet, Kerguelen, Macquarie, Antarctic Peninsula | Greene and Walton (1975), Frenot *et al*. (2005), le Roux *et al*. (2013), Greve *et al*. (2017), Sindel *et al*. (2018) |
| *Elymus repens* (L) Gould | Poaceae | Eurasia | South Georgia, Marion, Prince Edward, Crozet, Kerguelen | Greene and Walton (1975), Frenot *et al*. (2005), Greve *et al*. (2017) |
| *Plantago lanceolata* L | Plantaginaceae | Eurasia | Marion, Prince Edward, Crozet, Kerguelen | Greene and Walton (1975), Frenot *et al*. (2005), Greve *et al*. (2017) |
| *Poa annua*[a] | Poaceae | Eurasia | South Georgia, Marion, Prince Edward, Crozet, Kerguelen, Heard, Macquarie, Antarctic Peninsula | Greene and Walton (1975), Frenot *et al*. (2005), Williams *et al*. (2013), Greve *et al*. (2017), Sindel *et al*. (2018) |
| *Poa pratensis*[a] | Poaceae | Europe /Africa | South Georgia, Marion, Prince Edward, Crozet, Kerguelen, Antarctic Peninsula | Greene and Walton (1975), Frenot *et al*. (2005), Greve *et al*. (2017) |
| *Rumex acetosella* | Polygonaceae | Eurasia | South Georgia, Marion, Prince Edward, Crozet, Kerguelen | Greene and Walton (1975), Frenot *et al*. (2005), Greve *et al*. (2017) |
| *Sagina procumbens* | Caryophyllaceae | Eurasia | South Georgia, Marion, Prince Edward, Crozet, Kerguelen | Gremmen *et al*. (2001), Frenot *et al*. (2005), Greve *et al*. (2017) |
| *Stellaria media*[a] | Caryophyllaceae | Eurasia | Marion, Prince Edward, Crozet, Kerguelen, Macquarie, Antarctic Peninsula | Greene and Walton (1975), Frenot *et al*. (2005), le Roux *et al*. (2013) |

[a]Species eradicated on the Antarctic Peninsula.

subdivided into 19 taxonomic families. The data shows that only five families concentrate about 67% of the non-native species that have historically colonized Antarctic and sub-Antarctic ecosystems. These patterns of homogenization of taxonomic families are consistent with patterns found in cold ecosystems, including in mountains in the Arctic (Wasowicz *et al*., 2020). When analysing each sub-Antarctic island and the Antarctic Peninsula separately, there is a negative relationship between species richness and the number of taxonomic families, where areas of higher non-native species richness (Prince Edward, Crozet, Kerguelen, South Georgia) are represented by a few taxonomic families, for example the Poaceae family (Fig. 5.3).

Some of the species that invade sub-Antarctic ecosystems are being managed, with some non-native species eradicated on Macquarie Island including *P. annua*, *C. fontanum* and *S. media* (Sindel *et al*., 2018). On the other hand, the expansion of eight invasive species (i.e. *Luzula multiflora*, *Festuca rubra*, *R. acetosella* and some species of the genus *Agrostis*) has been limited on Prince Edward and Marion Islands,

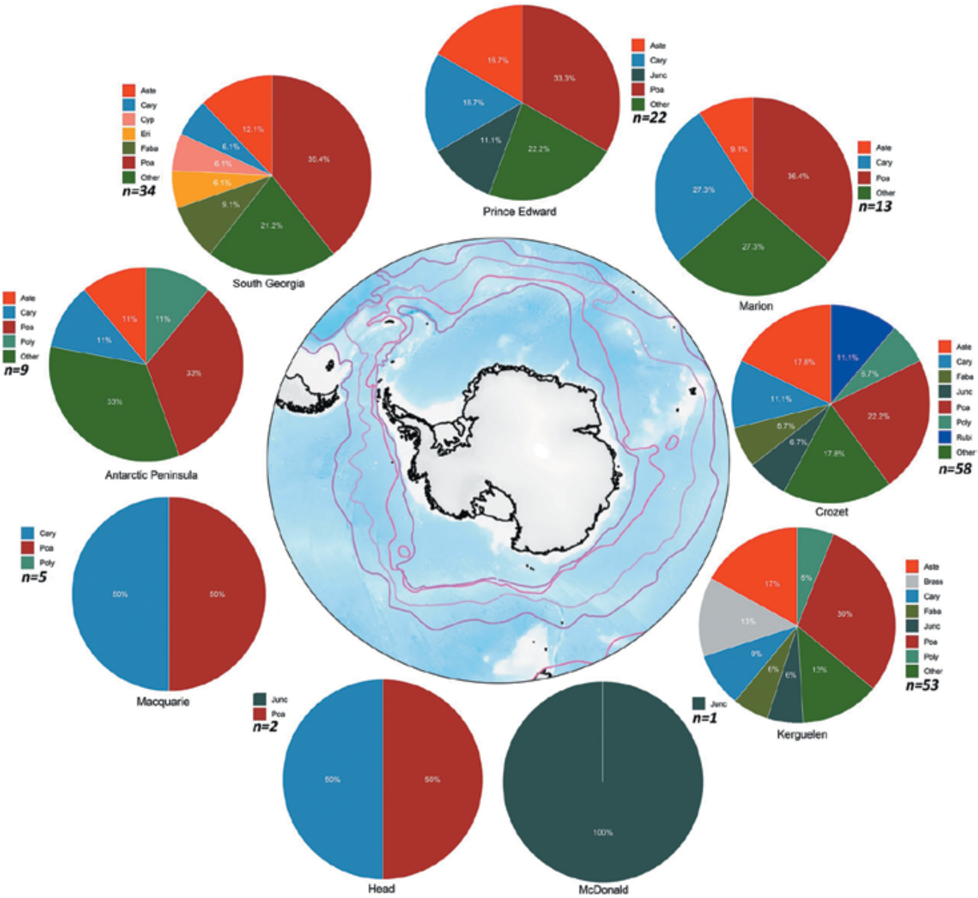


**Fig. 5.3.** Map of Antarctica and major sub-Antarctic islands, each pie chart represents the percentages of the most frequent taxonomic families for each sub-Antarctic Island and for the Antarctic Peninsula. Aste, Asteraceae; Brass, Brassicaceae; Cary, Caryophyllaceae; Cyp, Cyperaceae; Eri, Ericaceae; Faba, Fabaceae; Junc, Juncaceae; Poa, Poaceae; Poly, Polygonaceae; Rubi, Rubiaceae. *n* values correspond to the richness of non-native plants for each of the sub-Antarctic islands and the Antarctic Peninsula. Authors' own work.

using herbicides and manual removal as control methods (Greve *et al.*, 2017).

Recent studies have reported the transport of multiple non-native species to Antarctica by human vectors (Lee and Chown, 2009; Chown *et al.*, 2012; Hughes *et al.*, 2020). Construction of station facilities has been identified as an important route of introduction of non-native species (Lee and Chown, 2009). Lee and Chown (2009) examined building materials imported for station construction and identified 176 seeds, belonging to 14 families, of non-native species. In a similar study, Hughes *et al.* (2010) identified two plants of non-native species in soil imported on vehicles to a research station. The same imported soils were used for germination studies under controlled conditions and six non-native species germinated, with most of the non-native species already present on sub-Antarctic islands.

In addition to soils, local and regional studies have demonstrated that clothing and equipment can function as dispersal vectors for non-native species (Chown *et al.*, 2012). For example, in a study conducted on the Polish research station Arctowski on King George

Island, the number of seeds carried by scientists and logistics staff in their clothing was quantified, identifying more than 144 seeds of non-native species from 20 families, with an average of 1.7 seeds per scientist. Of the 144 seeds found, nine were of species already known to have invaded cold ecosystems such as in Arctic and sub-Antarctic regions (Lityńska-Zając *et al.*, 2012). Similar studies at a regional level, looking at different travel groups to Antarctica, identified that clothing from field scientists transport the highest percentage of seeds (Chown *et al.*, 2012). In another study conducted by Huiskes *et al.* (2014), they determined that visitors' bags and footwear carry the highest proportion of seeds of different species, recording 115 families of non-native species. In addition, these studies showed that transport vehicles, specifically aircraft and ships, contribute significantly to the transport of non-native species seeds. It has also been demonstrated that the transport of food is a significant contributor to the arrival of non-native species to the region (Hughes *et al.*, 2011). As is often observed in biological invasions, of the broad spectrum of propagules arriving in Antarctica, only a low proportion is able to establish due to the edaphoclimatic conditions of these ecosystems. For example, in a study of surface soil samples from the Fildes Peninsula, Fuentes-Lillo *et al.* (2017) found seeds of various non-native species, mainly from the families Asteraceae and Poaceae, mostly deposited in highly disturbed areas.

These studies have clearly shown that a considerable number of non-native species have overcome geographical and environmental barriers to dispersal both by direct human transport from other continents (Chown *et al.*, 2012) or by an intra-regional 'stepping stone' movement through the sub-Antarctic islands and maritime Antarctica (Lee and Chown, 2009). However, to date only a small number of these species have become established in Antarctic ecosystems. Further studies are clearly required to assess objectively the risk presented by human activities in relation to the establishment of non-native species.

Species that have successfully germinated on the Antarctic Peninsula are: *Puccinellia svalbardensis*, *Cerastium* sp., *Alopecuris geniculatus*, *Puccinellia distans*, *Rumex pulcher*, *Stellaria media* and *Chenopodium rubrum* (Table 5.1) (Hughes and Convey, 2012). Recently, the establishment of the species *Nassauvia magellanica* has been recorded on Deception Island (Hughes *et al.*, 2011), and *Juncus bufonius* in samples from the nearby Polish station Arctowski (Cuba-Díaz *et al.*, 2013). Two species of the genus *Poa* have been the most successful non-native species to establish: *P. pratensis and P. annua. P. pratensis* was introduced because of a transplant experiment at Cierva Point and has recently been eradicated from this area (Pertierra *et al.*, 2017). *P. annua* is invading the vicinity of the Polish station Arctowski following introduction associated with human activities carried out in the area (Olech and Chwedorzewska, 2011). As previously stated, few species are able to establish in Antarctica, however, no studies have yet addressed the key question of whether and what proportion of plant propagules arriving in Antarctica are viable.

The similarities on edaphoclimatic conditions with other cold ecosystems (Arctic, high mountain and sub-Antarctic) (Bazzichetto *et al.*, 2021), combined with regional climate change and direct human influence mean that ecosystems on the Antarctic Peninsula may be at similar risk of invasion as those already seen in Arctic and montane ecosystems (Duffy and Lee, 2019). Fildes Peninsula, for instance, that is of high use for logistics, scientific and tourist activities (Fuentes-Lillo *et al.*, 2017; Pertierra *et al.*, 2017), and where rapid climate change has been documented (Siegert *et al.*, 2019), provides a key location in which to study and unravel these complexities.

## 5.5 Biosecurity Protocol: Required to Prevent Plant Invasions

Biosecurity protocols applied to prevent the invasion of non-native plants in Antarctic and sub-Antarctic ecosystems are classified in three main groups: (i) prevention; (ii) monitoring; and (iii) response (Hughes and Pertierra, 2016). For the sub-Antarctic islands (above 60° latitude), protocols are applied by the countries that exercise sovereignty over each sub-Antarctic island. They are oriented to prevent the introduction (mediated by human vectors) of non-native plants (seeds, live plants), through quarantine periods and cleaning of cargo and equipment of any person visiting the sub-Antarctic islands (Potter, 2007).

Some examples of governance associated with applying biosecurity on sub-Antarctic islands (Head, McDonald, Macquarie) is the Australian Antarctic Program which is responsible for prevention, monitoring and applying management and/or eradication mechanisms for non-native species. Prevention protocols focus on searching for plants and seeds in cargo (i.e. food, construction materials, vehicles) and scientific equipment, applying cleaning and quarantine prior to entering the islands (Potter, 2007). The same protocols are applied for the Crozet and Kerguelen islands by the French Ministry of Overseas France, which is also responsible for species surveys and conservation plans, which prohibit generating any type of human disturbance that could facilitate the establishment of non-native plants (Lebouvier, 2007).

As for the response to established non-native plants, eradication has been the most widely used method, due to its efficiency and speed, mainly when the area of invasion is reduced (Raymond *et al.*, 2011). Currently on Macquarie Island, the eradication of the invasive species *P. annua*, *C. fontanum* and *S. media* has been carried out, with mechanical control the most used method, since compared with chemical control, it significantly reduces the impact on the native and endemic flora of the island (Sindel *et al.*, 2018).

In the case of continental, maritime Antarctica and sub-Antarctic islands south of latitude 60°, the guidelines for preventing the entry of non-native species are regulated by Article 4 (Annex II) of the Antarctic Treaty, which prohibits any entry of non-native plants in any form (Hughes and Pertierra, 2016). In this context, the biosecurity protocols are oriented to cleaning clothing, equipment and vehicles of tourists, scientists and logistic staff who carry out activities in Antarctica. Those in charge of applying the protocols in the case of tourists are the tour operators and in the case of scientists are the Antarctic programmes of each country (Convey *et al.*, 2012).

Regarding non-native plant monitoring, the protocol indicates that each country exercising sovereignty is responsible for monitoring the establishment of non-native plants in the areas where they carry out activities. For this reason, monitoring is generally low and in some cases non-existent, causing an underestimation of the real number of non-native plants that have been established in Antarctic ecosystems (Hughes and Pertierra, 2016).

In response to a successful establishment and/or invasion process, the protocol indicates that eradication will be applied for 3 months after detection of the non-native plant. This time generally varies significantly and is dependent on variables such as logistics and time of year, implying that a non-native plant can be established for a longer period. Currently about ten non-native plants have been eradicated mainly in the Antarctic Peninsula, among the most common are *P. annua*, *P. pratensis*, *S. media* and *Cerastium* sp. (Table 5.1) (Hughes and Pertierra, 2016).

## 5.6 Impacts

On the sub-Antarctic islands, the impacts of non-native plants remain understudied and are related to direct effects on native biota and microclimatic conditions (Frenot *et al.*, 2005; Greve *et al.*, 2017). Studies conducted on Macquarie Island have determined that the presence of *P. annua* generates a slight displacement of native species, which is not significant over time without the presence of anthropogenic disturbances (Frenot *et al.*, 2005). In Marion and Prince Edward Islands the presence of *Agrostis stolonifera* does not present a risk for the extinction of native species, but a significant increase in the abundance of this species can negatively affect the distribution of native species, in addition to modifying the microclimatic conditions (light availability) generating a change in the community structure of native mosses (Greve *et al.*, 2017).

Studies assessing the impact of non-native species on Antarctic ecosystems have been mainly experimental under both controlled conditions and field experiments. These studies have observed that increased abundance of *P. pratensis* and *P. annua* can displace native plants *D. antarctica* and *C. quitensis*, mainly associated with reduced biomass and reduced photosynthetic yield and decreased survival of native species (Molina-Montenegro *et al.*, 2012, 2019; Pertierra *et al.*, 2017). Other experimental studies include assessing the impacts of non-native species found in Antarctic ecosystems but with no stable populations in these ecosystems. Under this context, it was observed that *J. bufonius* has an impact on biomass production and increased mortality of *C. quitensis* and *D. antarctica* species (Cuba-Díaz *et al.*, 2013, unpublished results).

## 5.7 Final Remarks

Human activities are the main dispersal vector of non-native plants in Antarctic and sub-Antarctic ecosystems. In the case of the sub-Antarctic islands, human activities in the early 20th century were responsible for their introduction, while at present tourist activities contribute to the dispersal of non-native plants (Bazzichetto *et al.*, 2021). In Antarctica, tourism and scientific activities are the main dispersal vector of non-native plant propagules (Chown *et al.*, 2012). Although the establishment success of non-native plants in Antarctic ecosystems is low, the synergy between climate change and tourism increase could enhance the invasibility of these ecosystems, particularly in the most visited areas.

Regarding biosecurity protocols, it is necessary to strengthen monitoring actions to assess the establishment success of non-native plants. As previously stated, monitoring is almost nonexistent in Antarctica and the sub-Antarctic islands, so the real number of non-native plants that are likely to be established in these ecosystems is underestimated. In this context, frequent visits to the sites that receive the largest arrival of tourists and scientists to search new establishments of non-native plants would contribute to develop a more efficient response to the invasion process. This is particularly important because early detection and response in the initial stages of invasion reduces eradication efforts, as well as minimizing the impact of invasive species on ecosystems.

It is important to understand the role of other non-native species (biotic interactions) in the invasion process (e.g. fungi, insects, worms, bacteria) as they may play an important role in the success of non-native plant establishment. Preliminary studies have indicated that the presence of the invasive midge *Eretmoptera murphyi* generates changes in nutrient availability, increasing by three to five times the nitrate content in nutrient-poor Antarctic soils, which would imply an increase in the invasibility of Antarctic ecosystems and a possible successful establishment of non-native plants (Bartlett, 2019). This process, known as invasional meltdown, may be key to understanding the process of non-native plant invasion in sub-Antarctic and Antarctic ecosystems.

Finally, studies that examine the impact of non-native plants should be strengthened, since most of the studies are experimental under controlled conditions and mainly report the effect of competition on native plants. Therefore, it is important to add more studies on other impacts including the effects on microclimatic variables, soil biota, pollination and endemic fauna. This will provide a more general overview of the real impacts of non-native species on Antarctic and sub-Antarctic ecosystems.

## Acknowledgements

Eduardo Fuentes Lillo acknowledges the support of the ANID/BASAL FB210006 project.

## References

Amesbury, M.J., Roland, T.P., Royles, J., Hodgson, D.A., Convey, P. *et al*. (2017) Widespread biological response to rapid warming on the Antarctic Peninsula. *Current Biology* 27(11), 1616–1622. DOI: 10.1016/j.cub.2017.04.034.

Bartlett, J.C. (2019) Ecophysiology and ecological impacts of an Antarctic invader: the chironomid, *Eretmoptera murphyi*. Doctoral dissertation, University of Birmingham, Birmingham, UK.

Bazzichetto, M., Massol, F., Carboni, M., Lenoir, J., Lembrechts, J.J. *et al*. (2021) Once upon a time in the far south: influence of local drivers and functional traits on plant invasion in the harsh sub-Antarctic islands. *Journal of Vegetation Science* 32(4), e13057. DOI: 10.1111/jvs.13057.

Bender, N.A., Crosbie, K. and Lynch, H.J. (2016) Patterns of tourism in the Antarctic Peninsula region: a 20-year analysis. *Antarctic Science* 28(3), 194–203. DOI: 10.1017/S0954102016000031.

Bergstrom, D.M. and Selkirk, P.M. (2007) Human impacts on sub-Antarctic terrestrial environments. *Papers and Proceedings of the Royal Society of Tasmania* 141(1), 159–167. DOI: 10.26749/rstpp.141.1.159.

Chau, J.H., Mtsi, N.I.S., Münbergová, Z., Greve, M., le Roux, P.C. *et al*. (2020) An update on the indigenous vascular flora of sub-Antarctic Marion Island: taxonomic changes, sequences for DNA barcode loci, and genome size data. *Polar Biology* 43(11), 1817–1828. DOI: 10.1007/s00300-020-02747-7.

Chown, S.L., Huiskes, A.H.L., Gremmen, N.J.M., Lee, J.E., Terauds, A. *et al*. (2012) Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. *Proceedings of the National Academy of Sciences of the United States of America* 109(13), 4938–4943. DOI: 10.1073/pnas.1119787109.

Convey, P. (2007) Influences on and origins of terrestrial biodiversity of the sub-Antarctic islands. *Papers and Proceedings of the Royal Society of Tasmania* 141(1), 83–93. DOI: 10.26749/rstpp.141.1.83.

Convey, P. (2011) Antarctic terrestrial biodiversity in a changing world. *Polar Biology* 34(11), 1629–1641. DOI: 10.1007/s00300-011-1068-0.

Convey, P. (2017) Antarctic ecosystems. In: Scheiner, S.M. (ed.) *Encyclopedia of Biodiversity*, Vol. 1. Elsevier, Amsterdam, pp. 179–187. Available at: https://doi.org/10.1016/B978-0-12-809633-8.02182-8

Convey, P., Hughes, K.A. and Tin, T. (2012) Continental governance and environmental management mechanisms under the Antarctic Treaty System: sufficient for the biodiversity challenges of this century? *Biodiversity* 13(3–4), 234–248. DOI: 10.1080/14888386.2012.703551.

Council of Managers of National Antarctic Programs (2017) *Antarctic Station Catalogue*. COMNAP, Christchurch, New Zealand.

Cuba-Díaz, M., Troncoso, J.M., Cordero, C., Finot, V.L. and Rondanelli-Reyes, M. (2013) *Juncus bufonius*, a new non-native vascular plant in King George Island, South Shetland Islands. *Antarctic Science* 25(3), 385–386. DOI: 10.1017/S0954102012000958.

Duffy, G.A. and Lee, J.R. (2019) Ice-free area expansion compounds the non-native species threat to Antarctic terrestrial biodiversity. *Biological Conservation* 232, 253–257. DOI: 10.1016/j.biocon.2019.02.014.

Errington, I., King, C.K., Wilkins, D., Spedding, T. and Hose, G.C. (2018) Ecosystem effects and the management of petroleum-contaminated soils on subantarctic islands. *Chemosphere* 194, 200–210. DOI: 10.1016/j.chemosphere.2017.11.157.

Frenot, Y., Chown, S.L., Whinam, J., Selkirk, P.M., Convey, P. *et al*. (2005) Biological invasions in the Antarctic: extent, impacts and implications. *Biological Reviews of the Cambridge Philosophical Society* 80(1), 45–72. DOI: 10.1017/s1464793104006542.

Fuentes-Lillo, E., Cuba-Díaz, M., Troncoso-Castro, J.M. and Rondanelli-Reyes, M. (2017) Seeds of non-native species in King George Island soil. *Antarctic Science* 29(4), 324–330. DOI: 10.1017/S0954102017000037.

Galera, H., Chwedorzewska, K.J., Korczak-Abshire, M. and Wódkiewicz, M. (2018) What affects the probability of biological invasions in Antarctica? Using an expanded conceptual framework to anticipate the risk of alien species expansion. *Biodiversity and Conservation* 27(8), 1789–1809. DOI: 10.1007/s10531-018-1547-5.

Greene, S.W. and Walton, D.W.H. (1975) An annotated check list of the sub-Antarctic and Antarctic vascular flora. *Polar Record* 17(110), 473–484. DOI: 10.1017/S0032247400032459.

Gremmen, N., Barendse, J. and Orr, I. (2001) Invasion and eradication of *Sagina procumbens* L (Procumbent pearlwort) on Gough Island. *Aliens* 14, 19–20.

Greve, M., Mathakutha, R., Steyn, C. and Chown, S.L. (2017) Terrestrial invasions on sub-Antarctic Marion and Prince Edward Islands. *Bothalia* 47(2), 1–21. DOI: 10.4102/abc.v47i2.2143.

Hughes, K.A. and Convey, P. (2012) Determining the native/non-native status of newly discovered terrestrial and freshwater species in Antarctica - current knowledge, methodology and management action. *Journal of Environmental Management* 93(1), 52–66. DOI: 10.1016/j.jenvman.2011.08.017.

Hughes, K.A. and Pertierra, L.R. (2016) Evaluation of non-native species policy development and implementation within the Antarctic Treaty area. *Biological Conservation* 200, 149–159. DOI: 10.1016/j.biocon.2016.03.011.

Hughes, K.A., Convey, P., Maslen, N.R. and Smith, R.I.L. (2010) Accidental transfer of non-native soil organisms into Antarctica on construction vehicles. *Biological Invasions* 12(4), 875–891. DOI: 10.1007/s10530-009-9508-2.

Hughes, K.A., Lee, J.E., Tsujimoto, M., Imura, S., Bergstrom, D.M. *et al*. (2011) Food for thought: risks of non-native species transfer to the Antarctic region with fresh produce. *Biological Conservation* 144(5), 1682–1689. DOI: 10.1016/j.biocon.2011.03.001.

Hughes, K.A., Convey, P., Pertierra, L.R., Vega, G.C., Aragón, P. *et al*. (2019) Human-mediated dispersal of terrestrial species between Antarctic biogeographic regions: a preliminary risk assessment. *Journal of Environmental Management* 232, 73–89. DOI: 10.1016/j.jenvman.2018.10.095.

Hughes, K.A., Pescott, O.L., Peyton, J., Adriaens, T., Cottier-Cook, E.J. *et al*. (2020) Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. *Global Change Biology* 26(4), 2702–2716. DOI: 10.1111/gcb.14938.

Huiskes, A.H.L., Gremmen, N.J.M., Bergstrom, D.M., Frenot, Y., Hughes, K.A. *et al*. (2014) Aliens in Antarctica: assessing transfer of plant propagules by human visitors to reduce invasion risk. *Biological Conservation* 171, 278–284. DOI: 10.1016/j.biocon.2014.01.038.

International Association of Antarctica Tour Operators (2021). Available at: https://iaato.org/ (accessed 12 July 2022).

Kalwij, J.M., Medan, D., Kellermann, J., Greve, M. and Chown, S.L. (2019) Vagrant birds as a dispersal vector in transoceanic range expansion of vascular plants. *Scientific Reports* 9(1), 1–9. DOI: 10.1038/s41598-019-41081-9.

Kariminia, S., Ahmad, S.S., Hashim, R. and Ismail, Z. (2013) Environmental consequences of Antarctic tourism from a global perspective. *Procedia - Social and Behavioral Sciences* 105, 781–791. DOI: 10.1016/j.sbspro.2013.11.081.

Landan, D. (2007) Tourism in polar regions and the sub-Antarctic islands. *Papers and Proceedings of the Royal Society of Tasmania* 141(1), 173–179. DOI: 10.26749/rstpp.141.1.173.

Lebouvier, M. (2007) Conservation and management in the French sub-Antarctic islands and surrounding seas. *Papers and Proceedings of the Royal Society of Tasmania* 141(1), 23–28. DOI: 10.26749/rstpp.141.1.23.

Lee, J.E. and Chown, S.L. (2009) Quantifying the propagule load associated with the construction of an Antarctic research station. *Antarctic Science* 21(5), 471–475. DOI: 10.1017/S0954102009990162.

Leihy, R.I., Coetzee, B.W.T., Morgan, F., Raymond, B., Shaw, J.D. *et al*. (2020) Antarctica's wilderness fails to capture continent's biodiversity. *Nature* 583(7817), 567–571. DOI: 10.1038/s41586-020-2506-3.

le Roux, P.C., Ramaswiela, T., Kalwij, J.M., Shaw, J.D., Ryan, P.G. *et al*. (2013) Human activities, propagule pressure and alien plants in the sub-Antarctic: tests of generalities and evidence in support of management. *Biological Conservation* 161, 18–27. DOI: 10.1016/j.biocon.2013.02.005.

Lityńska-Zając, M., Chwedorzewska, K., Olech, M., Korczak-Abshire, M. and Augustyniuk-Kram, A. (2012) Diaspores and phyto-remains accidentally transported to the Antarctic Station during three expeditions. *Biodiversity and Conservation* 21(13), 3411–3421. DOI: 10.1007/s10531-012-0371-6.

Molina-Montenegro, M.A., Carrasco-Urra, F., Rodrigo, C., Convey, P., Valladares, F. *et al*. (2012) Occurrence of the non-native annual bluegrass on the Antarctic mainland and its negative effects on native plants. *Conservation Biology* 26(4), 717–723. DOI: 10.1111/j.1523-1739.2012.01865.x.

Molina-Montenegro, M.A., Bergstrom, D.M., Chwedorzewska, K.J., Convey, P. and Chown, S.L. (2019) Increasing impacts by Antarctica's most widespread invasive plant species as result of direct competition with native vascular plants. *NeoBiota* 51, 19–40. DOI: 10.3897/neobiota.51.37250.

Olech, M. and Chwedorzewska, K.J. (2011) Short note: the first appearance and establishment of an alien vascular plant in natural habitats on the forefield of a retreating glacier in Antarctica. *Antarctic Science* 23(2), 153–154. DOI: 10.1017/S0954102010000982.

Pertierra, L.R., Hughes, K.A., Tejedo, P., Enríquez, N., Luciañez, M.J. *et al*. (2017) Eradication of the non-native *Poa pratensis* colony at Cierva Point, Antarctica: a case study of international cooperation and practical management in an area under multi-party governance. *Environmental Science & Policy* 69, 50–56. DOI: 10.1016/j.envsci.2016.12.009.

Potter, S. (2007) The quarantine protection of Sub-Antarctic Australia: two islands, two regimes. *Island Studies Journal* 2(2), 177–192. DOI: 10.24043/isj.206.

Raymond, B., McInnes, J., Dambacher, J.M., Way, S. and Bergstrom, D.M. (2011) Qualitative modelling of invasive species eradication on subantarctic Macquarie Island. *Journal of Applied Ecology* 48(1), 181–191. DOI: 10.1111/j.1365-2664.2010.01916.x.

Shaw, J.D., Spear, D., Greve, M. and Chown, S.L. (2010) Taxonomic homogenization and differentiation across Southern Ocean Islands differ among insects and vascular plants. *Journal of Biogeography* 37(2), 217–228. DOI: 10.1111/j.1365-2699.2009.02204.x.

Siegert, M., Atkinson, A., Banwell, A., Brandon, M., Convey, P. *et al*. (2019) The Antarctic Peninsula under a 1.5°C global warming scenario. *Frontiers in Environmental Science* 7, 102. DOI: 10.3389/fenvs.2019.00102.

Sindel, B.M., Kristiansen, P.E., Wilson, S.C., Shaw, J.D. and Williams, L.K. (2018) Managing invasive plants on sub-Antarctic Macquarie Island. *The Rangeland Journal* 39(6), 537. DOI: 10.1071/RJ17073.

Stewart, E.J., Espiner, S., Liggett, D. and Taylor, Z. (2017) The forgotten islands: monitoring tourist numbers and managing tourism impacts on New Zealand's subantarctic islands. *Resources* 6(3), 38. DOI: 10.3390/resources6030038.

Tin, T., Lamers, M., Liggett, D., Maher, P.T. and Hughes, K.A. (2014) Setting the scene: human activities, environmental impacts and governance arrangements in Antarctica. In: Tin, T., Liggett, D., Maher, P.T. and M, L. (eds) *Antarctic Futures*. Springer, Dordrecht, the Netherlands, pp. 1–24.

Tracey, P. (2007) Tourism management on the southern oceanic islands. In: Snyder, J.M. and Sronehouse, B. (eds) *Prospects for Polar Tourism*. CAB International, Wallingford, UK, pp. 263–284.

Vyverman, W., Verleyen, E., Wilmotte, A., Hodgson, D.A., Willems, A. *et al*. (2010) Evidence for widespread endemism among Antarctic micro-organisms. *Polar Science* 4(2), 103–113. DOI: 10.1016/j.polar.2010.03.006.

Wasowicz, P., Sennikov, A.N., Westergaard, K.B., Spellman, K., Carlson, M. *et al*. (2020) Non-native vascular flora of the Arctic: taxonomic richness, distribution and pathways. *Ambio* 49(3), 693–703. DOI: 10.1007/s13280-019-01296-6.

Williams, L., Kristiansen, P., Shaw, J., Sindel, B. and Wilson, S.C. (2013) Weeds down under: invasion of the sub-Antarctic wilderness of Macquarie Island. *Plant Protection Quarterly* 28(3), 71–72.

Znój, A., Chwedorzewska, K.J., Androsiuk, P., Cuba-Diaz, M. and Giełwanowska, I. (2017) Rapid environmental changes in the western Antartic Peninsula region due to the climate change and human activity. *Applied Ecology and Environmental Research* 15(4), 525–539. DOI: 10.15666/aeer/1504_525539.