Universiteit
Antwerpen

This item is the archived peer-reviewed author-version of:

The role of roads and trails for facilitating mountain plant invasions

Reference:
Barros Agustina, Haider Sylvia, Müllerová Jana, Alexander Jake M., Alvarez María Alisa, Aschero Valeria, Daehler Curtis, Peyre Gwendolyn, Backes Amanda Ratier, Arévalo José Ramón, ...- The role of roads and trails for facilitating mountain plant invasions
Tourism, recreation and biological invasions / Barros, Agustina [edit.]; et al. [edit.] - ISBN 978-1-80062-054-4 - London, CABI, 2022, p. 14-26
Full text (Publisher's DOI): https://doi.org/10.1079/9781800620544.0003
To cite this reference: https://hdl.handle.net/10067/1980670151162165141

uantwerpen.be
Institutional repository IRUA

Please cite this chapter as follows:

Barros, A., Haider, S., Mullerova, J., Alexander J.M., Alvarez M.A., Aschero V., Daehler C., Peyre G., Ratier Backes A., Arévalo J.R., Cavieres L., Dar P., Fuentes-Lillo E., Liedtke R., McDougall K., Milbau A., Morgan J.W., Naylor B.J., Nuñez M.A., Pauchard A., Rashid I., Reshi Z.A., Rew L., Sandoya V., Seipel T., Vorstenbosch T., Vítková M., Neville W., Wedegärtner R.E. M., Zong S., Lembrechts J. J. 2022. The role of roads and trails for facilitating mountain plant invasions. In: Barros, A., Shackleton, R., Rew, L., Pauchard, A. (Eds.). Tourism, Recreation and Biological Invasions. CABI publishing, London. pp 14-26.

Pre-print version

Chapter 3: The role of roads and trails for facilitating mountain plant invasions

Agustina Barros [1x*], Sylvia Haider[2,3x], Jana Müllerová[4x], Jake M. Alexander[5], Maria Alisa Alvarez[6], Valeria Aschero [1, 7], Curtis Daehler[8], Gwendolyn Peyre [9], Amanda Ratier Backes[2,3], José Ramón Arévalo [10], Lohengrin Cavieres[11], Pervaiz Dar[12], Eduardo Fuentes-Lillo [13,14,15], Rebecca Liedtke[16], Keith McDougall [17], Ann Milbau [18], John W. Morgan[19], Bridgett J. Naylor[20], Martín A. Nuñez[21,22], Aníbal Pauchard [14], Irfan Rashid[12], Zafar A. Reshi [12], Lisa Rew[23], Veronica Sandoya [24], Timothy Seipel[23], Tom Vorstenbosch [25], Michaela Vítková [4], Neville Walsh[26], Ronja E. M. Wedegärtner [27], Shengwei Zong[28], Jonas J. Lembrechts[13x]

[x] Lead authors

[1] Instituto Argentino de Nivologia, Glaciologia y Ciencias Ambientales (IANIGLA), CONICET, CCT - Mendoza.

[2] Institute of Biology / Geobotany and Botanical Garden, Martin Luther University Halle-Wittenberg, Am Kirchtor 1, 06108 Halle (Saale), Germany

[3] German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Puschstraße 4, 04103 Leipzig, Germany

[4] Institute of Botany, The Czech Academy of Sciences, 25243 Průhonice, Czech Republic

[5] Plant Ecology, Institute of Integrative Biology ETH Zurich, Universitätsstrasse 16, CH-8092 Zürich, Switzerland

[6] Instituto Argentino de Investigaciones de las Zonas Áridas, Centro Científico y Tecnológico CONICET & Universidad Nacional de Cuyo, Mendoza, Argentina

[7] Facultad de Ciencias Exactas y Naturales, Universidad Nacional de Cuyo, Mendoza, Argentina

[8] School of Life Sciences, University of Hawaii at Manoa, Honolulu HI 96822 USA

[9] Department of Civil and Environmental Engineering, Universidad de los Andes, 111711 Bogota, Colombia

[10] Departamento de Botánica, Ecología y Fisiología Vegetal, Universidad de La Laguna, La Laguna 38206, Spain.

[11] Departamento de Botánica, Facultad de Ciencias Naturales y Oceanográficas, Universidad de Concepción & Instituto de Ecología y Biodiversidad, Concepción, Chile

[12] Department of Botany, University of Kashmir, Srinagar - 190006, Jammu and Kashmir,

India
[13] Research Group Plants and Ecosystems (PLECO), University of Antwerp, Belgium
[14] Laboratorio de Invasiones Biológicas (LIB), Facultad de Ciencias Forestales, Universidad de Concepción, Concepción, Chile
[15] School of Education and Social Sciences, Adventist University of Chile, Chillán, Chile
[16] Department of Botany and Biodiversity Research, University of Vienna, 1010, Vienna, Austria
[17] Department of Ecology, Environment and Evolution, La Trobe University, Bundoora, Victoria 3083 Australia.
[18] Research Institute for Nature and Forest – INBO, Brussels, 1000 Belgium25
[19] Research Centre for Applied Alpine Ecology, La Trobe University, Bundoora 3086, Victoria, Australia
[20] USDA Forest Service, Pacific Northwest Research Station, La Grande, OR 97850, USA
[21] Grupo de Ecología de Invasiones, INIBIOMA, CONICET, Universidad Nacional del Comahue, Quintral 1250, San Carlos de Bariloche, CP 8400, Argentina
[22] Department of Biology and Biochemistry, University of Houston, Houston, Texas 77204, USA
[23] Land Resources & Environmental Sciences Department, Montana State University, Bozeman, MT, 59717, USA
[24] Escuela de Ciencias Biológicas e Ingeniería, Universidad Yachay Tech, Urcuquí, Ecuador
[25] Bioinvasions, Global Change, Macroecology-Group, Department of Botany and Biodiversity Research, Division of Conservation, Vegetation and Landscape Ecology, University Vienna, Rennweg 14, 1030 Vienna, Austria
[26] National Herbarium of Victoria, Birdwood Avenue, South Yarra, Victoria 3141, Australia
[27] Department of Biology, Norwegian University of Science and Technology, 7491 Trondheim, Norway
[28] Laboratory of Geographical Processes and Ecological Security in Changbai Mountains, Ministry of Education, School of Geographical Sciences, Northeast Normal University, Changchun 130024, China

*Corresponding author: anaagustinabarros@gmail.com, abarros@mendoza-conicet.gob.ar

Abstract

Mountain ecosystems are comparatively less invaded by non-native plants than lowland ecosystems. However, climate change and current human use increase the risk of plant invasions. This includes risks posed by mountain tourism and recreation, and associated infrastructure. This chapter summarizes global data about non-native species patterns along mountain roads and trails obtained through standard protocols developed by the Mountain Invasion Research Network (MIREN). A total of 610 non-native species have been recorded along and in close proximity to mountain roads in 18 regions globally, with the highest numbers of non-natives in Australia and Hawaii. Less non-native species have been recorded along trails, 86 across 8 regions, with the highest numbers in South American countries. Non-native richness patterns were similar for roads and trails, showing a decrease in species number with increasing elevation and more non-natives on road- and trailsides compared to the adjacent natural vegetation. Despite these similarities, non-native plant invasion is far less advanced along mountain trails than along roads, possibly driven by the smaller disturbance effect of trails and lower propagule pressure. Nevertheless, given the potential of both roads and trails to promote plant invasions, it is important to implement management strategies to reduce propagule pressure and disturbance, particularly in high elevation environments of conservation significance.

3.1 Introduction

Plant invasions are considered a major cause of biodiversity loss worldwide (Ehrenfeld, 2010; Pyšek, *et al.*, 2011; IPBES, 2019). While mountain ecosystems have been relatively uninvaded (Paiaro *et al.*, 2011) the risk of plant invasions is increasing due to changes in climate, land use, and globalization (Pauchard et al., 2009; Dainese *et al.*, 2017). Steep environmental and climatic gradients make mountains especially sensitive to climate change, affecting plant phenology, competitiveness and species distributions (Teller *et al.*, 2016). For example, current evidence shows that ongoing warming can trigger the upward expansion of non-native plants to higher elevations (Dainese *et al.*, 2017; Alexander *et al.*, 2018).

Non-native species richness typically decreases strongly with elevation, sometimes with a richness peak in the lower part of the gradient (e.g. Seipel *et al.*, 2011; Barni *et al.*, 2012;

Haider *et al.*, 2018). This can be partially explained by "directional ecological filtering" of the lowland species pools (Alexander *et al.*, 2011); at high elevations, non-native plants are mainly constrained by cold temperatures and short growing seasons, and as stress-tolerant species are usually poorly represented among invaders, it is basically a small set of non-native climatic generalists, good dispersers or species with high adaptability that manage to spread from the lower elevations and colonize high-elevation environments (Alexander *et al.,* 2011; Petitpierre *et al.*, 2012; Marini *et al.* 2013; McDougall *et al.*, 2018). Even though environmental stress poses an important barrier for plant invasions, the low invasion status of mountain systems could additionally be caused by a time lag on the expansion after lowland introductions (Alexander *et al.*, 2018) or by less intensive human activities at high elevations; also shown to be a crucial driver of plant invasions (Marini *et al.*, 2009). Taken together, the invasibility of mountain ecosystems is likely to increase through the deliberate introduction of cold-adapted species and the enhanced propagule supply that generally coincide with human disturbances (Pauchard *et al.*, 2009; Rew *et al.*, 2020; Fuentes-Lillo *et al.*, 2021).

Tourism and recreation in mountains is increasing and consequently contributing to plant invasions in these ecosystems (Stöckli *et al.*, 2012) (Fig. 3.1). Cities and towns in the lowlands are important propagule sources for non-native plant species that are then transported towards higher elevations on people's clothing, recreation equipment, on the fur, hooves and in the dung of pack animals, and the vehicles they drive (e.g. Ansong and Pickering, 2013a; Ansong and Pickering, 2014; Rew *et al.* 2018, see Chapter 17 this volume). Agricultural crops and pasture areas that surround cities are also important propagule sources (Vila and Ibañez, 2011). Roads create specific environmental conditions in terms of microclimate (solar radiation, snow cover, temperatures and humidity) and soil conditions (road material), and alter water, nutrient and disturbance regimes (Barni *et al*., 2012), which makes them suitable for many non-native species (Seipel et al., 2012; McDougall *et al.,* 2018). To a lesser degree than roads, trail disturbance in mountains can also alter local environmental conditions and promote the establishment of trampling resistant species, including many ruderal non-native plants (Ballantyne and Pickering, 2015).

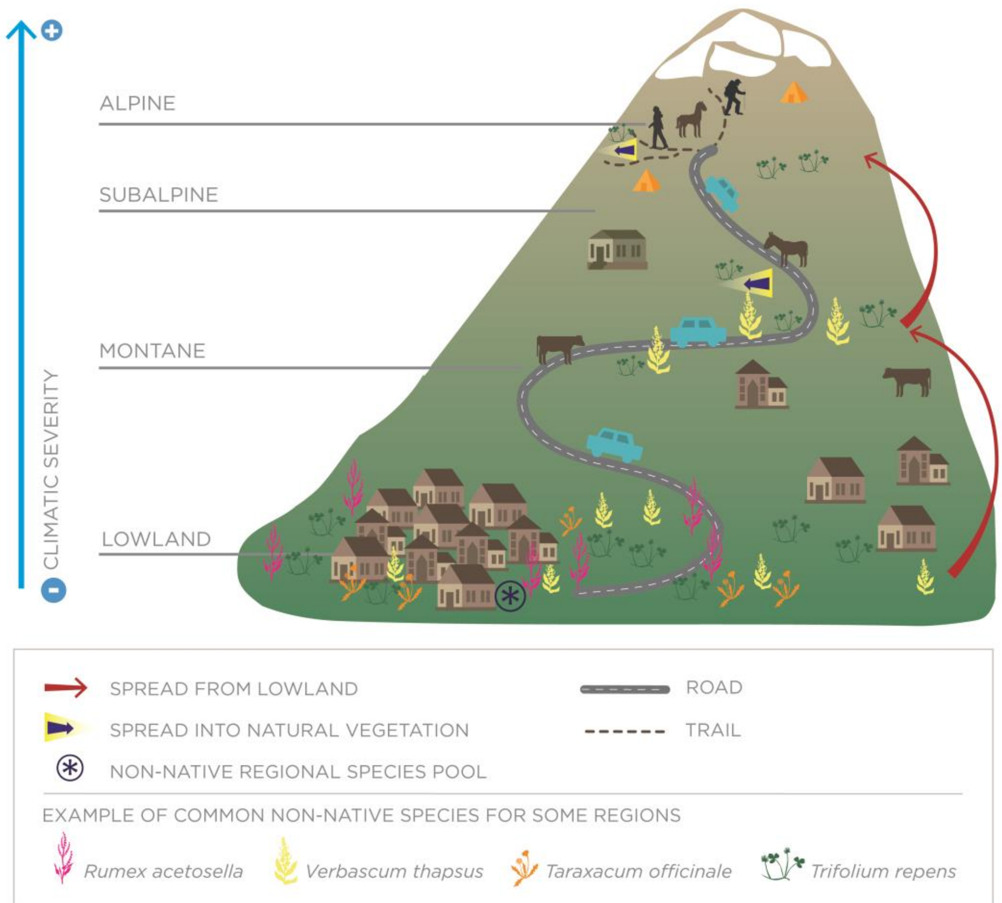

Figure 3.1. Examples of human factors affecting plant invasions along mountain roads and trails. Propagules arrive in mountain areas from lowland towns, tourism-associated infrastructure and other human activities such as agriculture. Roads and trails act as dispersal corridors through vehicles, hikers, and domestic animals. Recurrent human disturbance including road construction and maintenance, and domestic livestock grazing and trampling affect natural vegetation, further promoting non-native plant invasions. Adjacent natural vegetation can be secondarily invaded from road and trail sides, and trails have the potential to disperse non-native species further as they often reach higher elevations. The lower levels of plant invasions at higher elevations can be the result of increased climatic severity, a temporal lag phase for non-native plants in their dispersal from lowlands, lower propagule pressure or lower human disturbance compared to the lowlands. Author's own figure.

Once present along roads and trails, there is a risk that non-native plants will spread up the elevation gradient and into the adjacent, relatively undisturbed habitats (McDougall *et al*., 2018). In this chapter, we summarize the state-of-the-art on patterns of non-native plant species richness along mountain roads and trails as obtained through standard protocols developed by the Mountain Invasion Research Network (MIREN https://www.mountaininvasions.org/; Haider *et al.*, 2022; Liedtke *et al.*, 2020), a network that aims to understand the effects of global change on plant species' distributions and biodiversity in mountain areas.

3.2 Evidence of plant invasions along roads and trails

3.2.1 MIREN T-surveys

The MIREN road (Haider *et al.*, 2022) and trail (Liedtke *et al.*, 2020) protocols provide a standardized, long-term methodology to assess changes in non-native and native plant species distributions across space and time in mountain regions globally. One of their main objectives is to examine the role of disturbance from roads or trails on non-native plant establishment and spread across elevation gradients (Haider *et al.*, 2022). The protocol is repeated every five years by participating regions, and includes three sample roads or trails in each region. Along each road or trail, 'T-transects' (Fig. 3.2) are placed at equal elevation intervals, from the lowest point (e.g. start of the road or trail in the valley or catchment) to the highest point of the road or trail. For roads, the elevation range is divided into 19 elevation bands for a total of 20 sample sites per road. Along trails, which are usually shorter in length, elevation is divided into at least 9 elevation bands, for a total of at least 10 sample sites per trail (Fig. 3.2). At each sample site, three plots are laid out in the form of a T, with one plot parallel to the road or trail and two plots extending perpendicularly, the latter ones capturing the (semi-)natural vegetation of the region. The plot size for roads is 2 m x 50 m, so the T-transect extends up to 102 meters from the edge of the road surface, while for trails the plot size is 2 m x 10 m, extending the T-transect up to 22 meters from the trail (Fig. 3.2). Differences in plot sizes are due to the generally greater magnitude of disturbance created by roads compared to trails and the topography of trails that often prevents extending transects far from the trailside (Liedtke et al., 2020). In each plot, all vascular plant species (natives and non-natives) are recorded, vegetation cover and abundance is visually estimated for each species. To date, the protocols have been implemented on all continents except Antarctica.

For full methodological details, see Haider *et al.* (2022) for roads and Liedtke *et al.* (2020) for trails.

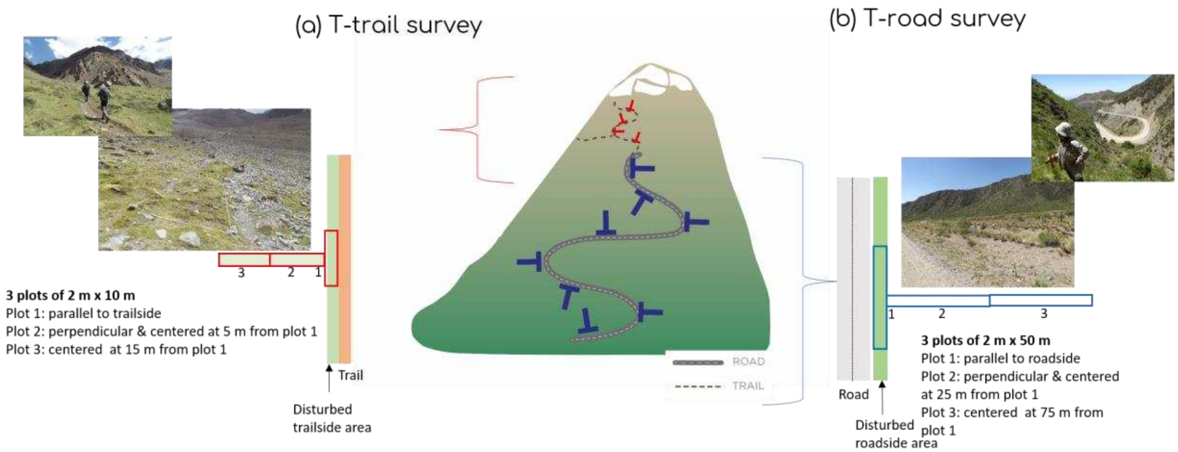

Figure 3.2. Layout of the MIREN T-survey design for a) roads and b) trails to assess changes in non-native and native plant species distributions across space and time in mountain regions across the globe. Transects are located at equal elevation intervals, with 20 sample sites along each mountain road and at least 10 sample sites along each trail. Each T-transect consists of three plots, the first one (plot 1) runs parallel to the road-/trailside, and the second and third plot are perpendicular and record the adjacent interior vegetation. Only a few T-transects are shown in the figure for clarity. Note that surveyed trails could, yet do not have to, be connected to surveyed roads. Author’s own figure.

3.2.2 Roads and plant invasions

The MIREN road survey has been conducted along 57 mountain roads in 18 regions. On average, surveyed mountain roads cover 1750 m in elevation, with the smallest elevational range in Norway (approx. 700 m) and the greatest in Hawaii (almost 4200 m). Overall, 610 non-native plant species were recorded in the survey, yet the number of non-native species differs substantially across regions and continents. Despite their suitable climate, mainland European regions had the lowest number of non-native species, with an average of 24 species per region (Fig. 3.3). In contrast, 139 non-native species were found in the Australian regions, and a total of 247 in Hawaii. Generally, there is a positive relationship between the length of the elevational gradient and the total number of non-native species. In contrast, the mean number of non-native species per plot is not related to the total number of non-native species found in the respective region.

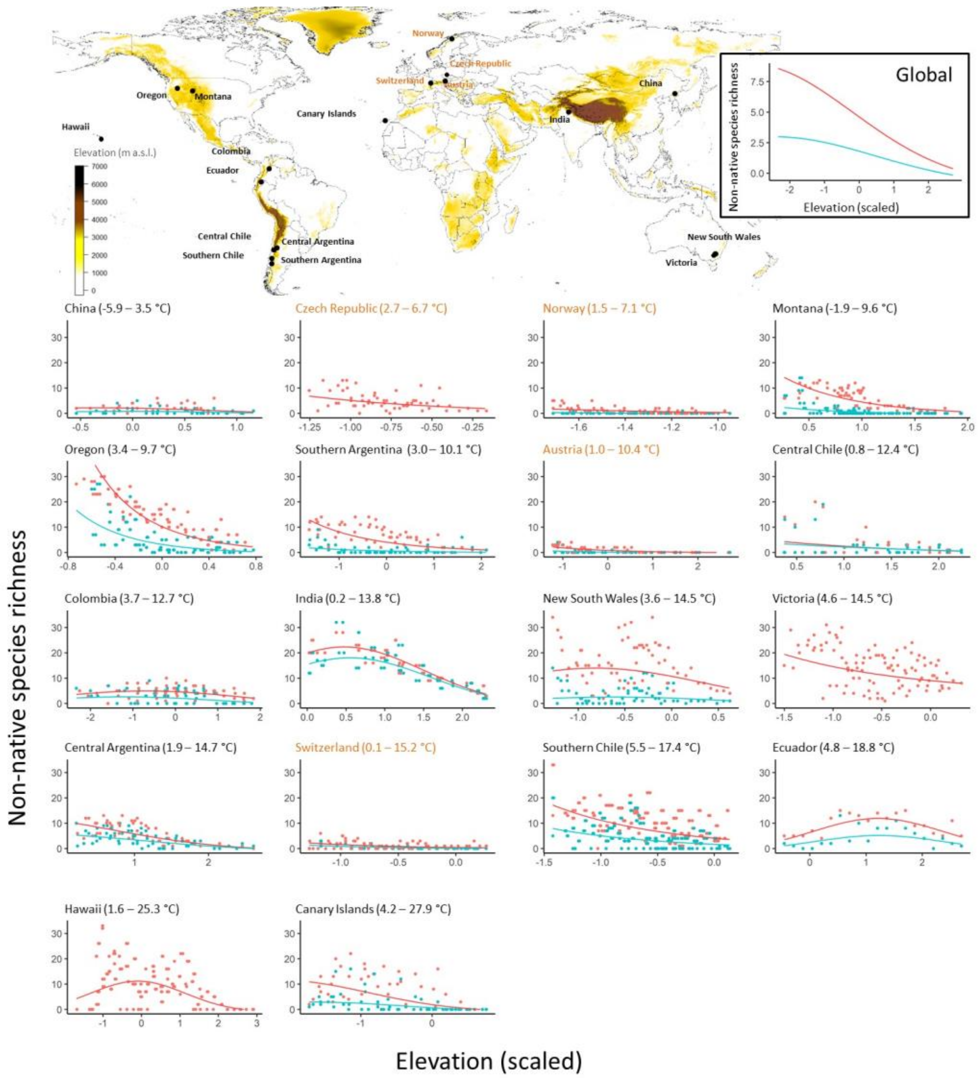

Figure 3.3. Non-native species richness along mountain roads (red lines and dots) and in the adjacent natural vegetation (turquoise lines and dots) in 18 mountain regions across the globe (see map on top); data from the Mountain Invasion Research Network. The global pattern is displayed in the inset at the top. Regions are ordered by the mean annual soil temperature of their warmest plot (extracted from Lembrechts et al., 2021 at a 1 km² resolution). The range between mean annual temperature of the coldest and warmest plot in a region is given in brackets. Mainland European region names are highlighted in orange in the map and the figure panels to highlight their comparably low non-native species richness. Lines depict predictions from a linear mixed model including elevation, its quadratic effect and plot type (roadside or adjacent natural vegetation) and their two-way interactions. For details on survey design see Haider et al. (2022). Note that in some regions, only roadside plots were surveyed (red lines and dots only). Author's own figure.

Non-native species richness decreases with increasing elevation in all regions, supporting the assumption that non-native species are primarily introduced at low elevations and spread upwards from there (Alexander *et al.*, 2011; Alexander *et al.*, 2016) (Fig. 3.3). However, some regions showed a peak of non-native species richness in the lower third of the elevational gradient (India) or at mid-elevations (Colombia, Ecuador, Hawaii) (Fig. 3.3). While temperature generally decreases along the elevation gradient and is seen as one of the main environmental drivers, other environmental factors, such as human disturbance or moisture conditions, can influence the shape of non-native species richness patterns across elevation gradients as well (Fuentes-Lillo *et al.*, 2021).

In all surveyed mountain regions, non-native species richness was at least slightly higher along roadsides than in the adjacent natural vegetation (plot 1 compared to plot 3 in Fig. 3.2) (see also Seipel *et al.*, 2012; McDougall *et al.,* 2018; Haider *et al.*, 2018) (Fig. 3.3). This observation supports the assumption that roads are the main dispersal corridors for non-native species first introduced in the lowlands (Fekete *et al.*, 2021). Studies have also shown that species found in adjacent natural vegetation have invaded from the roadside rather than from lower natural sites (McDougall *et al.*, 2018). This pattern demonstrates that disturbance and increased propagule pressure in roadsides raises a risk for plant invasions into mountains (Lembrechts *et al.*, 2016). Increased recreation and the associated infrastructure provides an enhanced risk of non-native species invasions at high-elevation sites.

Roadsides had a higher non-native species richness compared to the adjacent natural vegetation (566 spp. vs. 334 spp.) across all regions surveyed. Approximately 45% of the observed non-native species occurred only in roadsides. Among the 44 species observed only in natural vegetation were a number of woody species (e.g. *Pinus sp.*), while overall the non-native flora was dominated by herbaceous species. Only very few species occurred in more than half of the regions, the most frequent ones being *Hypochaeris radicata*, *Trifolium repens*, *Taraxacum* sp., *Rumex acetosella* and *Plantago lanceolata* (Fig. 3.4a). Some common species (e.g. *R. acetosella*, *Rosa rubiginosa*) had high occurrence in roadsides as well as in the adjacent interior vegetation, yet for most common species their frequency of occurrence was much higher in roadsides than in the natural vegetation. Almost half of all non-native species were found only in a single region, with a higher number of non-native species in countries in South and North America and Australia.

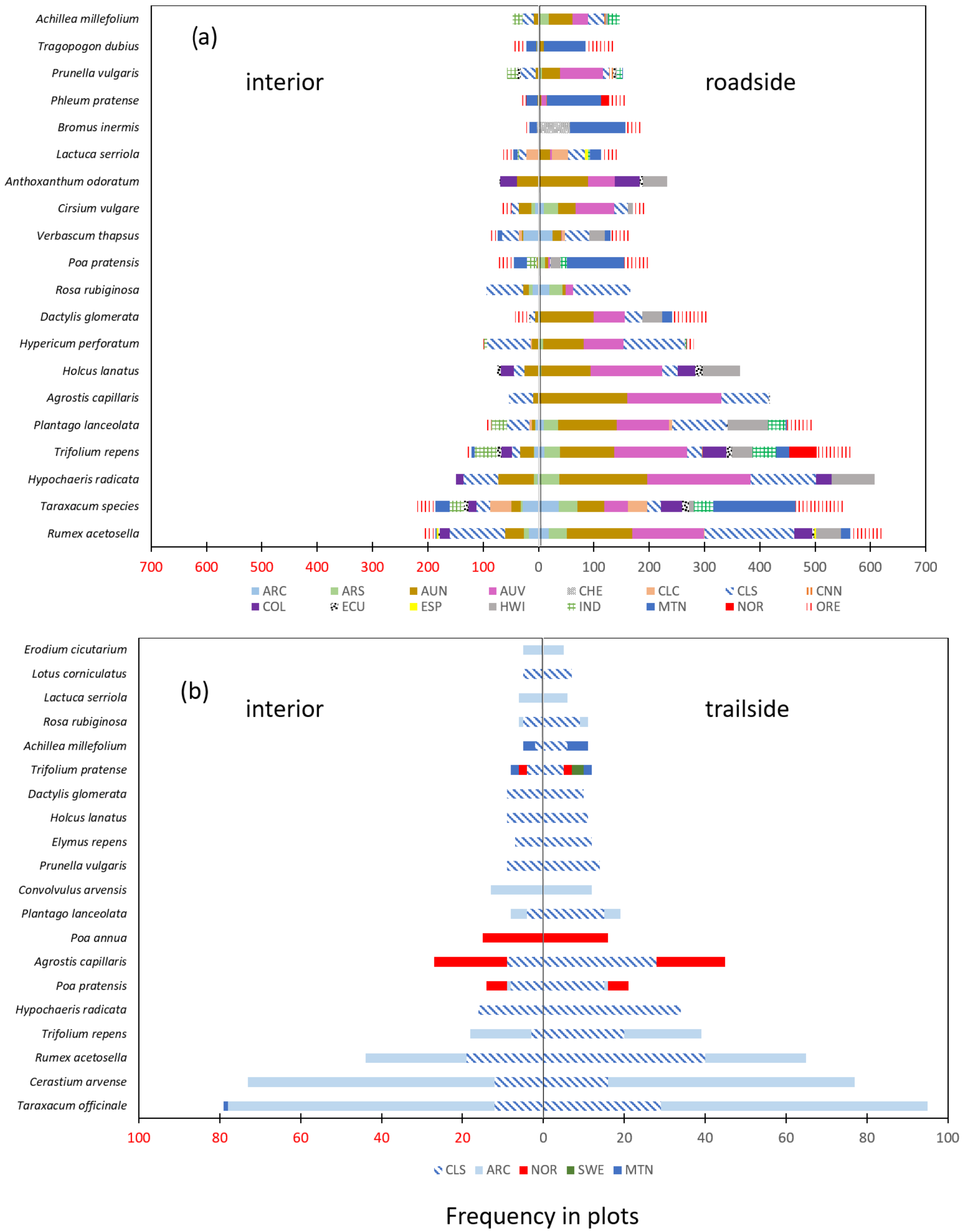

Figure 3.4. The 20 most frequent non-native species and the corresponding frequency of occurrence in the plots surveyed in the a) road survey and the b) trail survey, including road

and trailside plots and the adjacent interior vegetation. Figure 3.4a includes 16 regions, with data from the most recent survey (between 2012 and 2018). Figure 3.4b includes 5 regions surveyed between 2017 and 2019. Surveyed regions with a low number and frequency of the most common species (e.g. trails from China and the Czech Republic) are excluded here. ARC= Argentina Central, ARS=Argentina South, AUN=Australia New South Wales, AUV=Australia Victoria, AUT= Austria, CHE=Switzerland, CLC=Chile Central, CNN= China Northeast, COL=Colombia, ECU= Ecuador, ESP= Canary Islands, HWI= Hawaii, IND= India, MTN= USA Montana, ORE= USA Oregon, SWE= Sweden. Author's own figure.

3.3 Trails and plant invasions

The MIREN trail survey has been conducted in six regions from four continents, including countries in South America (Argentina and Chile), Europe (Sweden, Norway, Czech Republic) and Asia (China) (Fig. 3.5). Additionally, in two regions in the USA, a selection of most common non-natives were recorded. All together, 47 trails including over 1800 T-transects have been surveyed. Trails varied in length, ranging from 3 to 22 km (6.6 $\pm$ 3.7 km) and covered an average elevation gradient of 1500 meters (between 595 m in China and 1780 m in Chile). The surveys ranged from montane to alpine vegetation, including forests, shrublands, grasslands and tundra vegetation.

The number of recorded non-native species in the trail survey was remarkably smaller than that observed with the road survey. Most of the 86 non-native species found along trails originated from Europe and Asia, and to a lesser extent from the Americas. Most of the non-native species were recorded in the Andes of South America including Chile (44% of these 86 non-native species) and Argentina (41%). Far fewer non-natives occurred in the Northern Hemisphere: 10% in Norway, 7% each in Sweden and the Czech Republic, and 5% in China. The most frequent non-native species recorded along the trails (i.e. *Taraxacum officinale, Rumex acetosella, Trifolium repens* and *Achillea millefolium*; Fig. 3.4b) have also been reported as some of the most common non-native species along mountain roads (Seipel *et al.,* 2012; Fig. 3.4). The non-native species pool along mountain trails largely consists of a subset of lowland species and globally common mountain invaders, with most species occurring in both trailsides and adjacent vegetation (Fig. 3.4b).

At the global and regional scales, non-native richness along trails decreased with elevation (except in China, see Fig. 3.5). Across all regions, this decrease was significantly stronger along trailsides compared to the adjacent vegetation (plot 1 vs plot 3 in Fig. 3.2). The steeper decrease arises from the fact that at lower elevations non-native richness was greater along trailsides than in adjacent natural vegetation, while this difference disappeared at higher elevations (inset Fig. 3.5). This pattern could be due to the fact that trail disturbance favors non-natives by reducing competition at lower elevations, whereas under the harsher conditions in alpine environments, it may in contrast disrupt facilitative interactions between native and non-native species (Cavieres *et al.*, 2007). In the mountains of Central Chile, for example, studies have shown that cushion plants can create favourable microclimatic conditions in extreme environments, enhancing the establishment of both native and non-native plants at higher elevations into undisturbed vegetation (Cavieres et al., 2007).

Plot-level non-native richness was highest in Chile and Argentina, where mean annual soil temperatures were warmer compared to other regions (Fig. 3.5). Four of the six regions that conducted the complete survey (Norway, Czech Republic, Argentina and Chile) showed greater non-native richness at trailsides compared to adjacent natural vegetation (Fig. 3.5), yet differences were minor in all regions except for the lower half of the elevation gradient in Chile. In this region, the dominant vegetation types at lower elevations consist of deciduous and coniferous forests, which may have limited the establishment of non-native plants further away from the trail (Liedtke *et al.*, 2020).

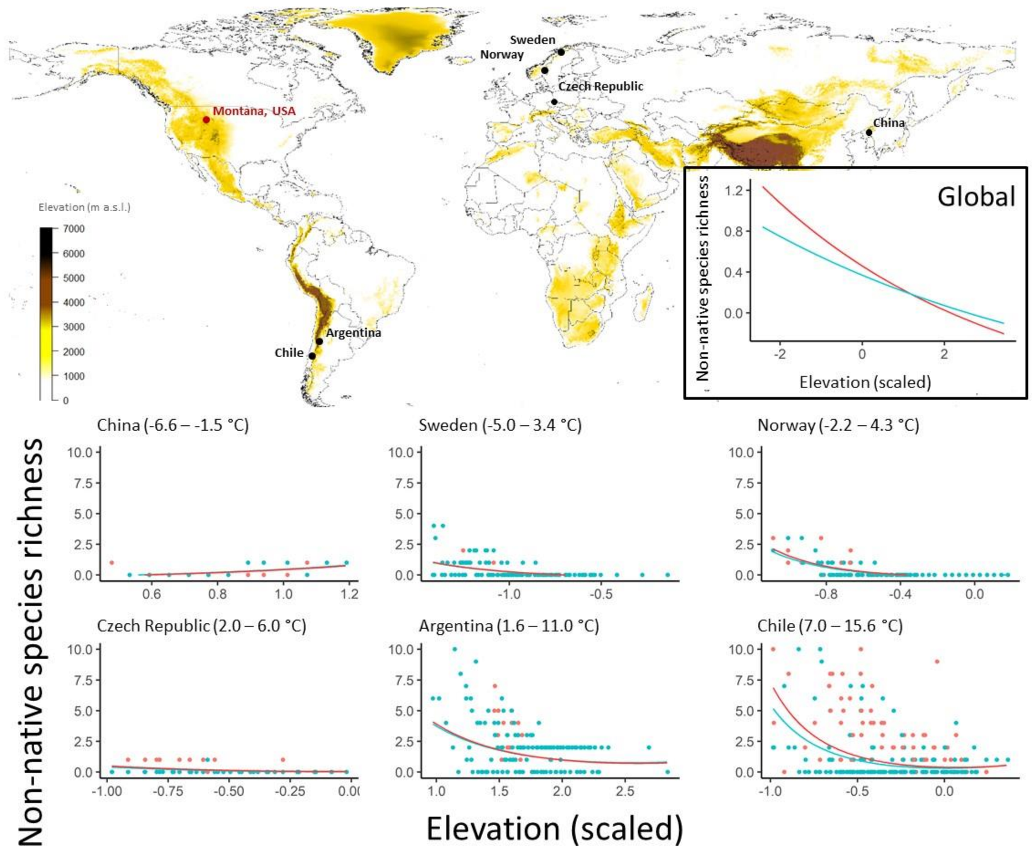

Figure 3.5. Non-native species richness along mountain trails (red lines and dots) and in the adjacent natural vegetation (turquoise lines and dots) in six mountain regions (see map; Montana, in red, did a reduced version of the survey on a selection of non-native species only and is thus not included in the analysis), data from the Mountain Invasion Research Network. The global pattern is displayed in the inset at the top. Regions are ordered by the mean annual soil temperature of their warmest plot (extracted from Lembrechts et al., 2021 at a 1 km² resolution). The range between mean annual temperature of the coldest and warmest plot in a region is given in brackets. Lines depict predictions from a linear mixed model including elevation, its quadratic effect, plot type and their two-way interactions. Note the different length of the y-axes here compared to Fig. 3.3, and thus in general much lower non-native species richness than along roads. For details on survey design, see Liedtke et al. (2020). Author's own figure.

3.4 Comparing roads and trails

While general patterns of plant invasions along mountain roads and trails are largely similar, non-native plant invasions seem to be less far advanced along mountain trails than along mountain roads. This conclusion is reinforced by the lower levels of invasion along trails than along roads in the same region (in all but northern Scandinavia, where invasion levels were low in general; Fig.3.6). However, this comparison has to be considered with caution because of the smaller plot size along trails (2 m x 10 m vs. 2 m x 50 m along roads) which directly affects the number of species recorded. Nevertheless, the much lower number of non-native species recorded in the trail survey compared to the road survey (Fig. 3.3 and Fig. 3.5) supports this conclusion.

Factors that might influence the difference in the level of invasion of roads vs. trails include the smaller disturbance effect of trails, as supported by a remarkably smaller difference in non-native richness between trailsides and adjacent vegetation than along roads (Fig. 3.6). Trails may be less susceptible to invasions due to lower propagule pressure and impact severity compared to roads (Liedtke *et al.,* 2020). Hikers, for instance, transport fewer seeds than vehicle traffic, and frequency of use of trails is often lower than that of roads (Pickering and Mount, 2010; Ansong and Pickering, 2013a). The construction and use of hiking trails creates less intensive disturbance and across a smaller area than along roads, further reducing the opportunities for trailside invasion (Ballantyne and Pickering, 2015; Liedtke *et al.*, 2020). Also, most trails are constructed without the use of imported building material such as gravel or sand (Newsome *et al.*, 2013), which excludes an invasion pathway from remote sources and profound changes in soil, nutrient and water regimes that can often be seen along roads. The smaller trail effect is reflected in the lower explained variance of the linear mixed models (Fig. 3.3, 3.5): while distance to the road explained 10.1% of observed variation in species richness (versus 8.0% by elevation), distance to the trail only explained 0.7% of variation (versus 12.3% by elevation).

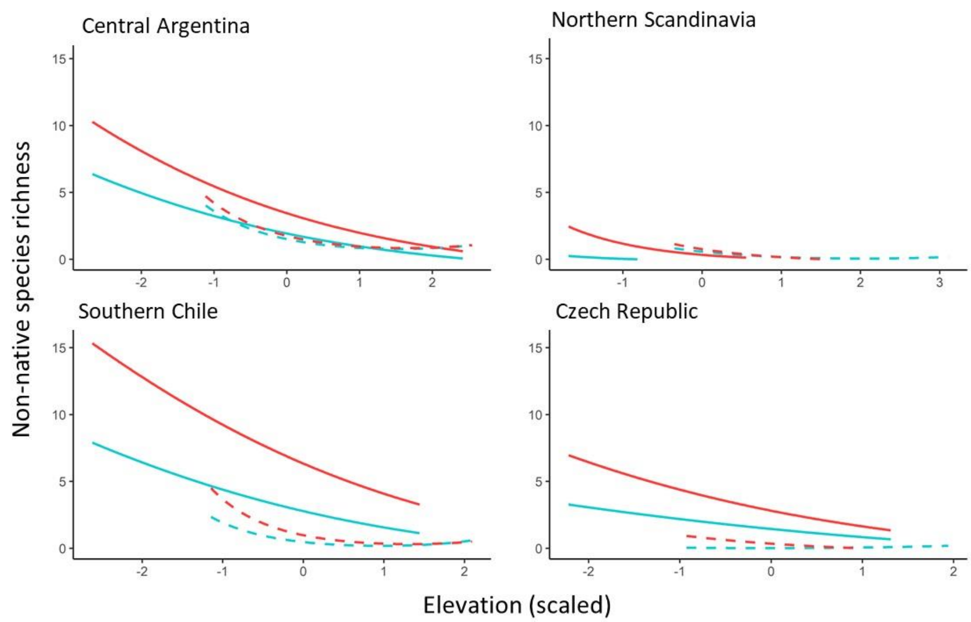

Figure 3.6. Non-native species richness along mountain roads (solid red lines) and trails (dashed red lines) and in the adjacent natural vegetation (solid and dashed turquoise lines) in four mountain regions where both roads and trails have been surveyed; data as in Fig. 3.3 and 3.5. Lines depict predictions from a linear mixed model including elevation, its quadratic effect, plot type and their two-way interactions. Author's own figure.

3.5 How to minimize plant invasions along roads and trails

Given the potential of roads and trails to promote non-native plant invasions, it is important to implement prevention and management strategies that minimize the risks. To date, management practices to prevent invasions vary enormously among regions (McDougall *et al.*, 2018), with high efforts for controlling non-native species in some mountain regions such as Canada, the United States and Australia, but very few or none in other regions such as Argentina and Chile, despite the high level of plant invasions there.

An important point to address is that although our results here show that trails have smaller effects on non-native species richness than roads, preventative actions for trails are required, particularly when they reach alpine ecosystems or cross protected areas. Although high-elevation sites have comparatively lower levels of invasion compared to lowlands (Alexander *et al.*, 2016), they are facing increasing threats due to the combined effects of climate change, recreation and tourism, and domestic livestock. Also, the nature of alpine ecosystems, characterized by open, sparse vegetation, makes it easier for hikers to wander off trails, increasing propagule pressure and disturbance across larger areas (Barros *et al.*, 2020). The latter is visible in the distribution patterns observed for the most common non-native species along mountain trails and roads (Fig. 3.4): while those species have a much higher frequency of occurrence along mountain roads than in the adjacent vegetation, their frequency in the interior vegetation next to mountain trails is similar to the one observed in trailsides.

Some of the prevention strategies to reduce propagule pressure and to mitigate road- and trailside habitat alteration include i) discouraging the use of non-native ornamental species that are known to spread into the natural vegetation from gardens and around tourism infrastructure, ii) encouraging visitors to avoid walking through verges in car parks prior to starting a hike as these areas are likely to have many non-native plants, iii) requesting hikers to always stay on the trail, iv) using vehicle wash-down facilities and boot-cleaning stations to minimize the spread of non-native plants along roads and trails and, v) require the use of weed free fodder for recreational horses and pack animals before entering protected areas (see Chapter 17, this volume).

For reducing the impacts associated with the usage and maintenance of trails and roads, their sustainable design is critical. For trails, surface hardening in popular sections and the use of local substrate for construction can substantially reduce trail widening and the impact on the adjacent vegetation and soils (Marion, 2011). For roads, sustainable practices include minimizing soil depth of roadside verges, use of infertile soils or substrates on verges to reduce seedbank accumulation by non-native species and minimizing soil surface disturbance in buffer road areas (Gelbard *et al.*, 2003). Also, restoring disturbed habitat after trail and road construction through revegetation with native plants can be effective to minimize non-native plant establishment.

References

Alexander, J. M., Chalmandrier, L., Lenoir, J., Burgess, T. I., Essl, F., *et al.* (2018). Lags in the response of mountain plant communities to climate change. *Global Change Biology,* 24(2), 563-579.

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., *et al.* (2016). Plant invasions into mountains and alpine ecosystems: current status and future challenges. *Alpine Botany*, 126(2), 89-103.

Alexander, J. M., Kueffer, C., Daehler, C. C., Edwards, P. J., Pauchard, A., *et al.* (2011). Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. *Proceedings of the National Academy of Sciences*, 108(2), 656-661.

Anderson, L. G., Rocliffe, S., Haddaway, N. R. and A.M., Dunn (2015). The role of tourism and recreation in the spread of non-native species: A systematic review and meta-analysis. *PLoS ONE*, 10(10), e0140833.

Ansong, M. and Pickering, C.M. (2013a). Are weeds hitchhiking a ride on your car? A systematic review of seed dispersal on cars. *PLoS ONE*, 8(11), e80275.

Ansong, M., and Pickering, C.M. (2013b). A global review of weeds that can germinate from horse dung. *Ecological Management and Restoration*, 14(3), 216-223.

Ansong, M and Pickering, C.M. (2014). Weed seeds on clothing: A global review. Journal of *Environmental Management*, 144, 203-211.

Ballantyne, M. and Pickering, C.M. (2015). The impacts of trail infrastructure on vegetation and soils: Current literature and future directions. *Journal of Environmental Management*, 164, 53-64.

Barni, E., Bacaro, G., Falzoi, S., Spanna, F. and Siniscalco, C. (2012). Establishing climatic constraints shaping the distribution of alien plant species along the elevation gradient in the Alps. *Plant Ecology*, 213(5), 757-767.

Barros, A., Aschero, V., Mazzolari, A., Cavieres, L. A and Pickering, C. M. (2020). Going off trails: How dispersed visitor use affects alpine vegetation. J*ournal of Environmental Management,* 267, 110546. doi:https://doi.org/10.1016/j.jenvman.2020.110546

Cavieres, L. A., Badano, E. I., Sierra-Almeida, A. and Molina-Montenegro, M. A. (2007). Microclimatic modifications of cushion plants and their consequences for seedling survival of native and non-native herbaceous species in the high Andes of central Chile. A*rctic, Antarctic, and Alpine Research*, 39(2), 229-236.

Dainese, M., Aikio, S., Hulme, P. E., Bertolli, A., Prosser, F., *et al.*. (2017). Human disturbance and upward expansion of plants in a warming climate. *Nature Climate Chan*ge, 7(8), 577-580.

Ehrenfeld, J. G. (2010). Ecosystem consequences of biological invasions. *Annual Review of Ecology, Evolution, and Systematics,* 41, 59-80.

Fekete, R., Haszonits, G., Schmidt, D., Bak, H., Vincze, O. *et al.* (2021). Rapid continental spread of a salt-tolerant plant along the European road network. *Biological Invasions*, 23(8), 2661-2674.

Fuentes-Lillo, E., Lembrechts, J. J., Cavieres, L. A., Jiménez, A., Haider, S. *et al.* (2021). Anthropogenic factors overrule local abiotic variables in determining non-native plant invasions in mountains. *Biological Invasions*, 23(12), 3671-3686.

Gelbard, J. L. and Belnap, J. (2003). Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology*, 17(2), 420-432.

Haider, S., Kueffer, C., Bruelheide, H., Seipel, T., Alexander, J. M. *et al*. (2018). Mountain roads and non-native species modify elevational patterns of plant diversity. *Global Ecology & Biogeography*, 27(6), 667-678.

Haider, S., Lembrechts, J., McDougall, K., Pauchard, A., Alexander, J. M. *et al.* (2022). Think globally, measure locally: The MIREN standardized protocol for monitoring species distributions along elevation gradients. *Ecology & Evolution,* 12, e8590. https://doi.org/10.1002/ece3.8590

IPBES (2019). Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. E. S. Brondizio, J. Settele, S. Díaz, and H. T. Ngo (editors). IPBES Secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3831673

Lembrechts, J. J., Pauchard, A., Lenoir, J., Nuñez, M. A., Geron, C. *et al*. (2016). Disturbance is the key to plant invasions in cold environments. *Proceedings of the National Academy of Sciences of the United States of America*, 113(49), 14061-14066.

Lembrechts, J. J., Van den Hoogen, J., Aalto, J., Ashcroft, M. B., De Frenne, P., *et al.* (2021). Global maps of soil temperature. *Global Change Biology,* https://doi.org/10.1111/gcb.16060

Liedtke, R., Barros, A., Essl, F., Lembrechts, J. J., Wedegärtner, R. E. M. *et al.* (2020). Hiking trails as conduits for the spread of non-native species in mountain areas. *Biological Invasions*, 22(3), 1121-1134. doi:10.1007/s10530-019-02165-9

Marini, L., Bertolli, A., Bona, E., Federici, G., Martini, F. *et al.* (2013). Beta-diversity patterns elucidate mechanisms of alien plant invasion in mountains. *Global Ecology and Biogeography*, 22(4), 450-460.

Marion, J. L., Wimpey, J. F. and Park, L. O. (2011). The science of trail surveys: Recreation ecology provides new tools for managing wilderness trails. *Park Scienc*e, 28(3), 60-65.

McDougall, K. L., Lembrechts, J., Rew, L. J., Haider, S., Cavieres, L. A, *et al.* (2018). Running off the road: roadside non-native plants invading mountain vegetation. *Biological Invasions*, 20(12), 3461-3473.

Newsome, D., Moore, S. A.and Dowling, R. K. (2013). *Natural Area Tourism Ecology*, Impacts and Management (2nd edition).Channel View Publications, Bristol, UK.

Paiaro, V., Cabido, M. and Pucheta, E. (2011). Altitudinal distribution of native and alien plant species in roadside communities from central Argentina. *Austral Ecology*, 36(2), 176-184.

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J. *et al.* (2009). Ain't no mountain high enough: Plant invasions reaching new elevations. F*rontiers in Ecology and the Environment*, 7(9), 479-486.

Petitpierre, B., Kueffer, C., Broennimann, O., Randin, C., Daehler, C., & Guisan, A. (2012). Climatic niche shifts are rare among terrestrial plant invaders. *Science*, 335(6074), 1344-1348.

Pickering, C. M.and Mount, A. (2010). Do tourists disperse weed seed? A global review of unintentional human-mediated terrestrial seed dispersal on clothing, vehicles and horses. *Journal of Sustainable Tourism*, 18(2), 239-256.

Pyšek, P., Jarošík, V., Pergl, J.and Wild, J. (2011). Colonization of high altitudes by alien plants over the last two centuries. *Proceedings of the National Academy of Sciences*, 108(2), 439-440.

Rejmanek, M.and Richardson, D. M. (1996). What attributes make some plant species more invasive? *Ecology,* 77(6), 1655-1661.

Rew, L.J, Brummer, T.J, Pollnac, F.W, Larson, C.D, Taylor, K.T. *et al.* (2018) Hitching a ride: seed accrual rates on different types of vehicles. Journal of *Environmental Management*, 206, 546-555.

Rew, L.J., McDougall, K., Alexander, J., Daehler, C, Essl, F. *et al*. (2020) Moving up and over: redistribution of plants in Alpine, Arctic and Antarctic ecosystems under global change. Arctic, Antarctic and Alpine Research, 52(1), 651-665.

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A. *et al.* (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. *Global Ecology and Biogeography*, 21(2), 236-246.

Stöckli, V., Wipf, S., Nilsson, C. and Rixen, C. (2011). Using historical plant surveys to track biodiversity on mountain summits, *Plant Ecology and Diversity*, 4(4), 415-425.

Teller, B. J., Zhan R.and Shea, K. (2016). Seed release in a changing climate: initiation of movement increases spread of an invasive species under simulated climate warming. *Diversity and Distributions*, 22(6), 708-716.

Vila, M., & Ibáñez, I. (2011). Plant invasions in the landscape. *Landscape Ecology*, 26(4), 461-472.