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The Biogeography of Invasive Plants – Projecting Range Shifts with Climate Change

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Abstract

Correlative models between species occurrences and climate (here referred to as ‘habitat suitability models’) have become increasingly popular for forecasting risk from invasive plants under current and future climate scenarios. These models have the potential to inform management and monitoring efforts by prioritizing landscapes considered at highest risk under a changing climate. However, a wide range of choices regarding climatic predictor variables, modelling approaches and even distributional data sets influences the resulting projections. The effects of these choices are seldom defined explicitly, which reduces their utility for scientists and managers alike. This chapter reviews common practices of habitat suitability modelling as they apply to invasive plants. The chapter also reviews major findings of recent projections of range shifts in invasive plants. In both cases, the aim is to explore how different choices of predictors, models and input data can influence conclusions in a habitat suitability modelling framework and develop recommendations for best practices.

Introduction

Identifying and eradicating early infestations is the most effective strategy for preventing

invasions (Moody and Mack, 1988; Strayer, 2009). Projections of potential range shifts with climate change could provide a valuable tool for informing ecosystem monitoring and targeting high-risk species (Stohlgren and Schnase, 2006). In response to this need, spatial models of species habitat under current and future climate conditions have become increasingly prevalent in recent years, with many investigators focusing on invasive plants.

Conceptually, the approach is relatively straightforward. Habitat suitability modelling uses empirical relationships between species occurrence and environmental variables (e.g. climate, topography, land use) to define habitat spatially across landscapes or regions (Fig. 15.1). Typically, this process involves collecting locations where an invasive plant is present along with spatial maps or models of the environmental conditions (Fig. 15.1a). Next, presence locations are used to specify environmental conditions in which the invasive plant can establish successfully, defining its potential habitat in environmental space (Fig. 15.1b). Finally, environmental habitat is projected back on to geographic space to determine the areas where the species could potentially establish (Fig. 15.1c). For invasive species, the potential range usually includes substantial land area that has not yet been invaded.

There are numerous synonymous or near-synonymous terms used to describe this

Fig. 15.1. Schematic representation of habitat suitability modelling. (a) Geographic occurrences of an invasive species are compared to climatic variables to calculate (b) climatic suitability for that species. (c) Climatic suitability is mapped back on to geographic space to create a model of suitable habitat, or potential invasive species range.

modelling process (Franklin, 2009). This chapter refers to it as ‘habitat suitability modelling’. Other common terms include ‘species distribution modelling’, ‘ecological niche modelling’ and ‘bioclimatic envelope modelling’. All of these models aim to predict the potential distribution of species, although the latter term typically implies that the environmental predictor variables are related to climate only.

Habitat suitability models (HSMs) do not contain any physiological information about invasive plant species, nor do they include any mechanistic information about spread or competition. They are purely empirical. As a result, they have been criticized, because correlation between climate and distribution does not necessarily imply a causal relationship between climate and distribution (Dormann, 2007). Environmental variables often have a high degree of correlation – for example, locations that have high annual precipitation also tend to be wet in the summer. So, the environmental variables that are empirically the best predictors of species distribution may not actually be the ones influencing plant growth. For a hypothetical plant, annual precipitation might provide the best fit empirically, even though summer precipitation actually affects how quickly the plant can grow and how extensive the invasion could ultimately be.

Model Limitations

How important is climate for modelling plant invasion risk? At a regional scale, climate can limit plant growth and competitive ability, defining broad boundaries on where species can establish, persist and potentially become problematic. However, at landscape and local scales, which are much more management relevant, other environmental conditions such as topography, soils and disturbance may be far more important (Pearson and Dawson, 2003). Moreover, the locations of initial introduction and dispersal ability influence how quickly non-native species are able to become an important component of regional flora (Stohlgren *et al.*, 2011; Stohlgren *et al.*, Part II, Chapter 10, this volume). All of these components play important roles in understanding invasion risk. Here, the focus is only on the approaches to modelling regional invasion risk in response to climate and climate change.

Predictor Variables

Under current climate conditions, switching one climatic predictor for another correlated predictor will have little effect on the overall projection. Studies have shown that empirical models based on current climate

have similar levels of accuracy to mechanistic models (Hijmans and Graham, 2006; Estes *et al.*, 2013). However, there is no way to measure accuracy under future climate conditions. And, if physiologically important variables are neglected in HSMs in favour of correlated but physiologically unimportant variables, this choice could have dramatic effects on projections of future range under climate change. To follow our hypothetical example above, if our plant's growth is influenced primarily by summer precipitation, then a model of future habitat based on annual average precipitation could skew projections of range shifts with climate change. Skewed projections are particularly likely if seasonal climate predictors respond differently; for example, if summer precipitation decreases but annual precipitation remains the same.

The best way to work around this problem is to select climatic predictor variables that are known to be physiologically important to the target species. This information can sometimes be found from experimental and observational data. In the absence of physiological information, climatic variables that provide the best empirical fit can be compared to expert knowledge of the species or ecosystem in order to select appropriate predictors (Bradley *et al.*, 2010a). In all cases, models of current and future distribution should be treated as hypotheses (due not only to climatic predictor uncertainty but also to uncertainty in climate projections and changes to species–environment interactions). If the climatic predictors are later found to represent physiology poorly, then any projections of future habitat should be treated with caution.

Invasive Species Equilibrium

One underlying assumption of HSMs is that the target species is at equilibrium (or near equilibrium) with current environmental conditions (Guisan and Zimmermann, 2000). If a species is not in equilibrium, then distribution points will be missing from locations where climate is actually suitable (where the species could establish if it were

introduced) and spatial models will under-predict potential establishment. Invasive species, by definition, are not at equilibrium (Václavík and Meentemeyer, 2012). Their continued spread through landscapes and regions fails the equilibrium assumption.

However, for the purposes of modelling, equilibrium does not require that the species occur everywhere that it could potentially establish *geographically*. Rather, equilibrium requires that the species encompass the *environmental* space where it could potentially establish. Environmental equilibrium is achieved much earlier than geographic equilibrium. For example, Welk (2004) showed that predictions of HSMs for purple loosestrife (*Lythrum salicaria*) invasion in North America stopped changing after distribution points from the first 120–150 years of invasion were added. Occurrence points from the past 50 years of invasion did not change the overall prediction, even though they doubled the available data. Thus, it is a reasonable assumption that long-established and/or widely introduced invasive species approximate climatic equilibrium and are appropriate for use in habitat suitability modelling. If a species is in the early stages of invasion and distribution points are only available in a localized area, then it is safe to assume that habitat models will underestimate its potential range vastly. The amount of time that it takes a species to approximate climatic equilibrium likely varies with the introduction and dispersal mechanism. For example, plants introduced as ornamentals may be widely distributed early on and approximate equilibrium sooner.

Modelling Considerations

Choice of model

A variety of different methodological approaches have been applied to habitat suitability modelling. These methods can be broken down roughly into presence/absence, presence/pseudo-absence and presence-only models. Presence/absence models require distribution not only for the location of

species presence but also for locations of species absence. Absence data for invasive plants can be problematic, because of the previously mentioned concerns with equilibrium. An absence might mean that the species cannot establish at that location. But, it could also mean that the species can establish, but has not yet been introduced. As a result, presence/absence models are typically used only at landscape or local scales to ask questions such as how local disturbance influences invasion (e.g. Bradley and Mustard, 2006). At regional scales, the likelihood of a 'false' absence skewing model results is much greater and there are few data sets that include both presence and absence. Hence, presence/absence models are rarely used to predict climatic suitability or to forecast species range shifts in response to climate change.

Presence/pseudo-absence models use species occurrences and a set of pseudo-absence points chosen at random from the available environmental space, or 'background'. Suitable climate conditions are identified by comparing the occurrences to the available climate conditions. If more occurrence points are found than expected relative to the available climate space, then those climate conditions are considered more suitable for invasive plant establishment. Many examples of presence/pseudo-absence models have been developed (Elith *et al.*, 2006), but the most widely used approach currently is based on a maximum entropy (or MaxEnt) approach developed by Phillips *et al.* (2006).

There has been considerable debate within the modelling community on how to define 'background' from which to sample pseudo-absence points (VanDerWal *et al.*, 2009a; Lobo *et al.*, 2010; Acevedo *et al.*, 2012). If defined too broadly, the model for current climate conditions might be very precise, but the predictor variables and probability relationships will be minimally informative. For example, a broad background of North America relative to an invasion in the Sonoran Desert is more likely to describe the unique climatic conditions of that desert (relative to North America) rather than the species itself. Too broad a

background will underpredict future range with climate change and may bias the climatic predictor variables if they are not defined physiologically (Acevedo *et al.*, 2012). If defined too narrowly, pseudo-absence points are more likely to be placed in locations that are actually climatically suitable, thereby biasing the projection. To date, there is no standard approach for defining background in order to minimize bias in presence/pseudo-absence models. For invasive plants, which are not in equilibrium geographically, the likelihood of creating biased models when selecting pseudo-absence points is high (Lobo *et al.*, 2010).

Presence-only models do not assume any information about absence locations or background environmental space (see Tsoar *et al.*, 2007, for examples). One example of a presence-only model is Mahalanobis Distance, which is a multivariate technique that defines perpendicular major and minor axes within the data and calculates distance from the data centroid relative to the covariance of axes lengths (Farber and Kadmon, 2003). This model measures only distance from the presence data; absences are unnecessary. Presence-only models will almost always predict a larger area of suitable habitat than presence/pseudo-absence models, because they have no direct mechanism for excluding some subsets of environmental space. Jiménez-Valverde *et al.* (2008) note that this larger predicted area has often been interpreted as an indication of overprediction, and therefore poor model performance. However, accuracy assessments are all performed relative to current species distribution, and may be unreliable measures of the accuracy of potential future range. It is equally probable that presence/pseudo-absence models underpredict future range due to biased definition of the background extents (Jiménez-Valverde *et al.*, 2008). A presence-only approach to modelling future habitat suitability under climate change is the least likely to underpredict potential range or be biased in unknown ways.

Although presence-only models are less prone to underprediction (false negatives),

presence/pseudo-absence models may be less prone to overprediction (false positives). A balance between minimizing both of these errors may ultimately be most useful for guiding regional monitoring and management. Hence, an ensemble approach (Araujo and New, 2007) that combines the projections of multiple models and climate projections could identify priority management areas.

No-analogue climate

The Earth's surface is not currently experiencing all the possible combinations of climate conditions that could exist. In some locations, rising temperatures and altered precipitation will change local climate to conditions that historically have not been experienced. These 'no-analogue' climate conditions (Williams and Jackson, 2007) may pose a problem for habitat modelling. It is possible to imagine climate conditions that do not currently exist on Earth, but in which a plant could still establish. Maybe the conditions are slightly warmer than current temperatures, or maybe they reflect an annual monsoon arriving a month later. In either case, our lack of invasive plant establishment data does not signify that the species cannot tolerate the environmental conditions, just that the location does not exist.

When projecting into these no-analogue climate conditions, models must extrapolate beyond the available data. The more complex the model, the greater the chances it will produce a problematic relationship beyond the current environmental conditions on which the model is trained. This problem is more likely in presence/absence or presence/pseudo-absence models, and provides further rationale for including presence-only models when modelling range shifts with climate change. Some presence/pseudo-absence models have developed methods for dealing with no-analogue conditions. For example, 'clamping' in MaxEnt (Phillips *et al.*, 2006) sets the value of any location outside the training range as equal to the edge of the training range. However,

as a general rule, there is much higher uncertainty in projecting range shifts into climate conditions with no current analogue (Elith *et al.*, 2010). Identifying and highlighting no-analogue climate conditions (Williams and Jackson, 2007) within habitat projections would help to convey uncertainty in risk assessments.

Range shifts with climate change

Although rising levels of CO₂ increase the growth of invasive plants relative to native plants (Blumenthal and Kray, Part I, Chapter 5, this volume), higher temperatures and altered precipitation do not necessarily provide a similar advantage. At the cold edge of invasive plant ranges, warming temperatures might enable plants to expand. For example, kudzu's (*Pueraria lobata*) range is likely limited at its northern edge in the USA by its intolerance of frost. With higher temperatures, this range margin is likely to move northwards (Bradley *et al.*, 2010b). However, at the warm edge, higher temperatures could increase evapotranspiration and reduce invasive plant competitiveness. Observations of native species have shown a consistent shift in species distribution towards the colder edge and away from the warmer edge with warming temperatures (Parmesan and Yohe, 2003; Root *et al.*, 2003). Similar range shifts are likely for invasive plants (Bradley *et al.*, 2009).

A number of projections of invasive plant range shifts with climate change have shown that both expanded and contracted range is likely (Fig. 15.2). In North America, of five problematic invasive plants, two showed primarily range expansion, two showed primarily range contraction and one showed roughly equal expansion and contraction (Bradley *et al.*, 2009). In South Africa, 30 grass species showed primarily contraction of potential range due to climate warming, with up to 50% loss of potential range by mid-century (Parker-Allie *et al.*, 2009). In Australia, three hawkweed species showed a loss of potential range of 20% by 2030 (Beaumont *et al.*, 2009). In North America, projections of range shifts by 2035 for 12

Fig. 15.2. Future climate conditions are likely to lead to substantial habitat loss for leafy spurge (*Euphorbia esula*) in the USA by the end of the century (adapted from Bradley *et al.*, 2009). Black areas on the map are currently climatically suitable, but lose suitability under climate change.

grasses and forbs showed more contraction of potential land area than expansion (Holcombe *et al.*, 2010). In Australia, models for 72 weeds of national significance projected that range will contract for the vast majority of species, with losses averaging 40% of potential range by mid-century (O'Donnell *et al.*, 2012). Losses of potential range suggest that establishment of these species might not be quite so widespread in the future.

These studies have focused primarily on shifts in potential range. However, loss of habitat within the currently invaded range is also likely. For example, of the three species with reduced potential range in North America, 30% of currently invaded areas were likely to become climatically unsuitable by the end of the century (Bradley *et al.*, 2009). This loss of habitat suggests that climate change may cause some invasive plants to become less competitive in, or even retreat from, areas that they have already invaded. Invasive species retreat highlights the intriguing possibility of widespread

restoration opportunities due to climate change (Bradley *et al.*, 2009). However, the term 'restoration' is poor for this type of opportunity, because altered climate conditions make it counterproductive to restore the native vegetation that existed prior to invasion. Rather, a type of transformative restoration may be needed to revegetate landscapes with non-invasive, regionally native species that can survive the novel climate conditions and will serve the desired function of the ecosystem (e.g. soil stability, carbon storage, animal habitat) (Bradley and Wilcove, 2009). The development of ecological goals and acceptable risks for assisted migration are needed if these opportunities are to be realized (Harris *et al.*, 2006; McLachlan *et al.*, 2007; Richardson *et al.*, 2009).

Although a number of studies now suggest that range contractions will be just as important, if not more so, than range expansion for invasive plants, the total sample size still remains small. There are thousands of non-native species that have

become invasive, but the potential ranges of only a handful have been modelled. For the majority of invasive plants that have not been modelled, potential range shifts are likely to mimic those observed in native species. Warmer temperatures are causing an upward elevational and a poleward directional shift in distributions, and a similar response is likely for invasive plants. For regional managers attempting to catalogue future risk, species that have invaded neighbouring regions with slightly warmer climates likely pose the greatest risk under climate change.

Vulnerable ecosystems

The pool of invasive species with climate change projections is small relative to all global invaders and tends to be focused on the most problematic examples. This small sample size limits the potential for identifying regions or biomes that might be more susceptible under future climate conditions. To date, only one study in Australia has attempted to assess whether some biomes have higher risk than others. O'Donnell *et al.* (2012) modelled 72 weeds of national significance in Australia under current and future climate conditions. They projected two main invasion 'hotspots', one in the south-east and one in the south-west part of the country. Both hotspots were primarily temperate ecosystems. However, this finding might reflect the unique geography of Australia rather than temperate ecosystems as higher risk. Temperate ecosystems are the poleward destination of all tropical invasive plants in Australia, and the country's central desert reduces the available range. Interestingly, the authors note that the invasion hotspots also occur in areas with intensive land use, so disturbance is likely to create more opportunities for the establishment of novel species in these particular areas.

Identifying the most vulnerable ecoregions in the near term may be linked more closely to trade, particularly imports and the availability of ornamental plants, than to climatic suitability. The horticulture

industry is the primary source of non-native plant introductions (Reichard and White, 2001; Mack and Erneberg, 2002), and trade is on the rise (Hulme, 2009; Bradley *et al.*, 2012). Ecoregions with higher rates of trade (Vila and Pujadas, 2001) and/or wider distribution of aliens versus natives (Stohlgren *et al.*, 2011) ultimately may see the most dramatic shifts in invasive species in response to climate change.

Invasion Risk: Establishment Versus Impact

Invasive plants are unique in that they have multiple different ranges: the range in which they are native and all of the locations where they have been introduced and successfully established. Often, the native range and the invaded ranges encompass different climatic space (Broennimann *et al.*, 2007), which greatly expands the potential geographic locations of invasive species establishment. Models based on the invaded range alone are likely to underestimate the potential for establishment, and therefore underestimate the potential range under climate change.

However, using all possible distribution data in a suitability model assumes that establishment is the most important stage of invasion risk to predict. For native species, trying to figure out where a plant could establish and persist under climate change is important for informing conservation (Thomas *et al.*, 2004; Kremen *et al.*, 2008), restoration and even assisted migration (Richardson *et al.*, 2009). But, for invasive species, the management applications of understanding potential establishment are much less direct. Many invasive plants have wide climatic tolerances (Pysek and Richardson, 2007), and could establish under a variety of conditions. For example, cheatgrass (*Bromus tectorum*) is established in all 50 states and most Canadian provinces (Fig. 15.3) (USDA-NRCS, 2012). But, it is abundant and has a high impact only in western states, where its early season growth helps it outcompete native species for limited water resources (Mack, 1981; Knapp, 1996) and where its fine fuel biomass

Fig. 15.3. Comparison of establishment range to abundance/impact range for cheatgrass (*Bromus tectorum*) in North America. State-/province-level establishment information is available on the USDA PLANTS website (USDA-NRCS, 2012). Point locations of high abundance come from USGS GAP analysis data for the north-west and south-west and from California.

increases fire frequency (D'Antonio and Vitousek, 1992; Brooks *et al.*, 2004; Balch *et al.*, 2013) (Fig. 15.3). These types of impacts are not a concern in eastern states, even though the species is well established.

The climatic and geographical space where an invasive species has an impact is a smaller subset of its total establishment. This space has been termed 'damage niche' (McDonald *et al.*, 2009; DiTommaso *et al.*, Part III, Chapter 16, this volume) or 'impact niche' (Bradley, 2013). Modelling the current and future geographical extents of the impact niche is much more management relevant than establishment or range alone (Hulme, 2006; McDonald *et al.*, 2009). Yet, the two invasion stages of establishment and impact (Lockwood *et al.*, 2007) are often confusingly lumped together in model projections under the single term 'invasion risk'.

If the goal of a model is to identify the current and potential range of species establishment, then all available distribution data should be used (including data from the native range and any other invaded ranges).

Not using all available establishment data will result in an underestimate of potential establishment. But, what if the goal is a more management-relevant assessment of the current and future geographical extents of impact? The choice of distribution data in this case is not straightforward. Very few distribution data sets contain any sort of measure of impact, particularly because impact itself is notoriously difficult to quantify, changes through time as species interact and could be measured for a range of different native competitors and/or ecosystem processes. An easier-to-measure proxy for impact in the case of invasive plants is abundance. Some plant species have meaningful impacts at low abundance, but high abundance always leads to greater impacts (Parker *et al.*, 1999). Unfortunately, even abundance or cover is rarely available in distribution data sets, particularly at the coarse (>1 km) spatial resolutions typically used to model climatic habitat.

One thing is clear, distribution data, even if limited to the invaded range, are a poor

proxy for impact. As a result, models of suitability based on distribution data will overestimate impact (Bradley, 2013). This is partially due to the establishment niche being much larger than the impact niche. But, this phenomenon is also a function of invasive plant distribution data sets themselves. Herbarium collections, often available online from sources such as the Global Biodiversity Information Facility (<http://www.gbif.org/>), tend to be focused on collecting individuals where the species is rare. Hence, invasive plant herbarium records are more likely to be located in sites with low abundance, and therefore low impact. Indeed, herbarium records for problematic invasive plants in the western USA were located in regions where managers considered the species to be absent nearly 60% of the time (Bradley, 2013).

Invasive plant distribution data sets collected by managers and compiled regionally, such as the Early Detection and Distribution Mapping System (<http://EDDMaps.org/>; Wallace and Barger, Part III, Chapter 13, this volume) or the invasive plant atlas of New England (<http://www.eddmaps.org/ipane/>), tend to be focused on

early detection and rapid response. Because control is more effective for early infestations (Moody and Mack, 1988), treatment tends to be focused on areas with low abundance. Hence, distribution data compiled regionally also tend to be skewed towards low abundance (Marvin *et al.*, 2009; Bradley, 2013). As a result, distribution data available in the invaded range are likely to be a poor proxy for the geographic and climatic conditions that define the impact or damage niche.

Establishment Models Do Not Predict Impact

HSMs create a surface of suitability values that are related to the probability that a species could establish at a given location (Fig. 15.4a). MaxEnt, for example, produces a 'gain' value ranging from zero to one, where high numbers indicate higher probability of establishment. It seems intuitive that higher probability of establishment should also be related to higher abundance, and therefore impact. If true, distribution data alone could be used to

Fig. 15.4. Habitat suitability scores based on distribution data are linked poorly to abundance. (a) Probability of establishment for an invasive plant in the western USA based on MaxEnt gain values (Phillips *et al.*, 2006). White is unsuitable, lighter shades have higher suitability. (b) Typical relationship between suitability scores and species abundance.

predict impact by identifying locations with the highest suitability for establishment.

Relationships between modelled suitability based on distribution data and abundance have been tested for a number of species, including birds (VanDerWal *et al.*, 2009b), mammals (Tórres *et al.*, 2012), arthropods (Jiménez-Valverde *et al.*, 2009) and plants (Pearce and Ferrier, 2001; Nielsen *et al.*, 2005; Estes *et al.*, 2013). Rather than finding a strong relationship between suitability for establishment and measured abundance, these studies have found low or insignificant correlations. In several cases, the relationship is represented best as a step function, where low suitability corresponds to absence but high suitability corresponds to any value of abundance from low to high (Fig. 15.4b).

How Can We Predict Impact?

If distribution data are a poor proxy for impact niche, and suitability for establishment is a poor predictor of impact, is there any way to predict impact niche under current and future climate conditions? One positive finding is that models appear to be able to predict abundance (a proxy for impact) using continuous abundance data. Kulhanek *et al.* (2011) showed that observed abundance of invasive carp in Minnesota lakes (USA) could be used to predict the abundance of carp in South Dakota lakes. Unfortunately, continuous abundance data are extremely rare, particularly at the regional scales needed to link abundance to climate. A few regional abundance data sets have been collected in the USA, including the forest inventory and analysis (FIA) data set (<http://www.fia.fs.fed.us/>) used by the US Forest Service to inventory forest resources and the national gap analysis program (GAP) data sets used to validate state and regional land cover classifications. However, these data are focused on native species and are likely to overlook all but the most problematic invasive plants.

One innovative approach to modelling the impact or damage niche is to create a subset of distribution data based on

locations known to have a high impact. McDonald *et al.* (2009) modelled current and future potential damage to crop species by identifying the states with high economic losses to two invasive species (Bridges, 1992), velvetleaf (*Abutilon theophrasti*) and Johnsongrass (*Sorghum halepense*), and using those state locations as distribution data for suitability models. This approach could be applied more widely to invasive plants based on expert knowledge of where the species has the highest impact, with the results less likely to overestimate impact than distribution data alone.

Finally, although suitability models based on distribution data are correlated poorly to abundance (Fig. 15.3), a recent study suggests that suitability models based on high-abundance locations are well correlated with abundance. Estes *et al.* (2013) showed that suitability values derived from a MaxEnt (Phillips *et al.*, 2006) model based on locations of high-yield maize in South Africa were correlated linearly ($R^2 = 0.6$) to continuous measurements of maize yield. This finding suggests that models based on point locations of known high abundance, such as the ones created by McDonald *et al.* (2009), could do better than just modelling impact/non-impact by predicting levels of potential abundance. The study is based on a single plant species and needs wider testing, but the approach of modelling damage or impact niche is very promising for invasive plant risk assessments.

Conclusions

Although habitat suitability modelling is conceptually straightforward, the case of forecasting plant invasions due to climate change presents a unique set of challenges. While models of establishment risk under current climate are likely robust, choices of modelling approach, distribution data and climatic predictors can have substantial effects on projections of range shifts with climate change. The limitations and likely biases of model and data choices are communicated poorly to managers, and better efforts towards transparency are needed.

The sample size of predictive models for invasive plants remains small relative to the total pool of global invaders. However, models have consistently shown both potential for expansion and likely contraction of invasive plant range. It is currently unknown whether the impact niche is projected to follow similar expansion and contraction patterns as the total range. Likely invasive plant expansion suggests that managers should be including new species in monitoring efforts, particularly the problematic species from warmer-climate neighbours. Invasive plant contraction due to climate change may open a window of unique opportunity for the targeted restoration of high-priority invaded ecosystems.

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