The dispersal potential of endangered plants versus non-native garden escapees

Ingmar R. Staude[1,2]

1. *Institute of Biology, Leipzig University, Leipzig, Germany*
2. *German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Leipzig, Germany*

Correspondence: ingmar.staude@uni-leipzig.de

Running head: *Dispersal: Endangered vs. Non-native Plants*

Abstract

Amidst global challenges like climate change, plant dispersal corridors are essential. In human-dominated landscapes, urban and rural green spaces are key dispersal avenues. Non-native plants are known to benefit from these, yet the potential benefits for endangered plants remain unclear. To address this question, I compared dispersal traits of endangered native plants with those of non-endangered and non-native species in Germany. I also compared endangered species amenable to gardening with non-native garden escapees to make my analysis pertinent to understanding the potential role of gardens in aiding native plant dispersal. I analyzed several traits including seed weight, terminal velocity, dispersal distance, germination rate, dispersal mode, seed structures, and seedbank type. Overall, dispersal traits between native and non-native plants showed minor, but in some cases statistically clear differences. Endangered plants were more often wind-dispersed and had more frequently seed appendages that might encourage a wider range of dispersal vectors. In contrast, non-native plants leaned more towards non-assisted local dispersal with slightly heavier seeds and more persistent seed banks. The other traits were largely consistent across groups. Given the minor differences in dispersal traits, green corridors may provide no lesser dispersal avenue for endangered native species. Gardens that have historically aided the dispersal of many non-native plants, could support the dispersal of endangered native plants instead.

Implications for practice

1. Traditional plant dispersal mechanisms, like herbivore-mediated pathways linking semi-natural habitats, have largely given way to human-driven dispersal connecting urban and rural areas in Central Europe.
2. Given the resulting bias towards dispersing synanthropic species, an integration of endangered species into urban and rural green spaces could help counteract this dispersal bias.
3. The results of this study indicate endangered plants do not have a lesser dispersal ability than non-native garden escapee species.
4. This indicates their potential to benefit from modern dispersal pathways, such as gardens and green corridors.

Introduction

There are increasing calls and efforts to establish corridors aiding the migration of organisms in response to global change factors, such as climate change (European Commission, 2020; Brodie et al., 2021; https://naturaconnect.eu/). While research on the efficacy of these corridors has traditionally centered on animals (Hobbs, 1992), there is an expanding interest in exploring their potential advantages for plants as well (Auffret et al., 2017; Plue et al., 2022). Despite their demonstrated capacity to enhance native species diversity (Damschen et al., 2019), evidence regarding the effectiveness of corridors for endangered species remains scarce (Kirchner et al., 2003). Endangered species may not only decline due to extrinsic factors, such as changes in land use, eutrophication, and natural successional dynamics (Godefroid et al., 2014), they may possess intrinsic factors, such as inferior dispersal and establishment, independently contributing to their population declines (Poschlod & Bonn, 1998; Young & Clarke, 2000; Ozinga et al., 2009; Holz et al., 2022). This could potentially make green corridors less effective to help these species disperse. Investigating how dispersal traits differ between species experiencing population increases and those that decline could provide insights into the potential of corridors to facilitate endangered species dispersal.

One critical type of green green corridor in our human-dominated landscape are private and public green spaces (Tallamy, 2007; Maclagan et al., 2018; Planchuelo, Kowarik, & Von der Lippe, 2020; Segar et al., 2022; Lepczyk et al., 2023). Recently, conservation gardening has emerged as a participatory approach to conservation, proposing that green spaces could serve as refuges for endangered native plants (Segar et al., 2022; Munschek et al., 2023). Additionally, meta-community theory predicts that increasing the abundance and distribution of declining plants, as can be achieved via large-scale cultivation in urban and rural green spaces, enhances the likelihood of successful dispersal and sustaining populations (Leibold & Chase, 2017; Thompson et al., 2020). While some evidence suggests that cultivating native species may positively impact population trends and Red List status, the available data is limited, particularly for endangered native plants (Segar et al., 2022; Staude, 2023). That is, cultivating endangered native plants remains scarce, resulting in a paucity of data regarding cultivation's effects on these species' population trends. Conversely, we better understand the role of gardens in the dispersal of non-native species. Many native species have become naturalized through horticulture (Reichard & White, 2001; Van Kleunen et al., 2018). Thus, contrasting the dispersal ecology of species known to benefit from such corridors may be particularly revealing in order to understand the dispersal potential of endangered native plants.

In a green space setting, certain traits may be especially beneficial for plant dispersal. In these contexts, dispersal by humans and wind—and their interplay—are likely to be particularly important (Hodkinson & Thompson, 1997; Bullock et al., 2018; Maclagan et al., 2018; Planchuelo, Kowarik, & Von der Lippe, 2020; Van Helden et al., 2020). Seeds that are smaller and lighter with reduced fall velocities are likely to be wind-dispersed over greater distances (Jongejans & Telenius, 2001). An inverse correlation between seed weight and fecundity (Moles & Westoby, 2004) may further amplify dispersal prospects (Jakobsson & Eriksson, 2003). While nearly all plants might experience epizoochory (Poschlod & Bonn, 1998; Couvreur et al., 2004), small seeds and those with specific seed structures like stickiness, hooks or barbs may enhance attachment, promoting dispersal by humans (Ridley, 1930; Wichmann et al., 2009; Auffret & Cousins, 2013). Structures such as wings or hairs can boost secondary dispersal, especially with wind from passing vehicles (Lippe & Kowarik, 2008; Lippe et al., 2013; Auffret, 2011; Rogers et al., 2019). Seeds with prolonged viability can persist in the soil for extended periods and can be dispersed through human activities such as garden throw-outs (Gioria et al., 2021; Sonkoly et al., 2022). Post-dispersal, successful germination plays a vital role. Thus, traits like seed weight, aerodynamics, dispersal mode and distance, seed structures, seedbank type, and germination rate are likely to greatly influence dispersal success of species from green spaces within a human-dominated landscape.

In this study, I investigate differences in the above dispersal traits among endangered, non-endangered native, and non-native species within the German flora. By contrasting non-native garden escapees with endangered species amenable to conservation gardening, I aim to enhance the practical implications of the findings for urban and rural green spaces. Ultimately, this research seeks to illuminate the potential of corridors in aiding the dispersal of endangered plant species.

Methods

Red List status, garden escapees and conservation gardening species. I used the 2018 German national Red List (RL) on vascular plants (Metzing et al., 2018) for species endangerment and native status. RL categories include: 0 = Extinct or lost, 1 = Critically endangered, 2 = Endangered, 3 = Vulnerable, G = Endangered - unknown extent, R = Rare, V = Near threatened, * = Not endangered, nb = Not assessed. The RL details native status as follows: “I” indicates indigenous, “N” indicates naturalized non-native, “U” indicates transient, and “F” indicates doubtful. I distinguished between endangered native species (“I” and either 0, 1, 2, 3, or G), non-endangered native species (“I” and either R, V, *, or nb), and naturalized non-native species (“N”). These categories correspond to three population trends: endangered species are declining, non-endangered native species are stable, non-native species are increasing. I only included taxa at the species level (indicated by the column “Arten”). I further inferred non-native species that escaped gardens (garden escapees) from the overlap of established non-native species in the RL, and species cultivated in gardens that naturalize in Germany as of Rothmaler Volume 5, titled "Krautige Nutz- und Zierpflanzen" (Herbaceous crop and ornamental plants; Jäger et al., 2016), which I digitized in previous work (Staude, 2023). Species amenable to conservation gardening are based on declining native species in Germany (categories 0, 1, 2, 3, G, R and V) and their inclusion in a comprehensive gardening gardening platform, natura-db.de (Munschek et al., 2023). All datasets were taxonomically harmonized with rWCVP and rWCVPdata (Brown et al., 2023). In total, there were 1054 endangered native species, 2141 non-endangered native species, 375 naturalized non-native species, 152 non-native garden escapees, and 458 conservation gardening species.

Dispersal traits. I used data from three trait databases: 1) Lososová et al., 2023, who compiled an extensive dataset of seed-related information, including seed mass, seed dispersal distance classes, and the predominant dispersal modes for most European vascular plants; 2) LEDA Database (Kleyer et al., 2008), which provides data on seed terminal velocity, seed structure and seedbank type for the Northwest European flora; and 3) Seed Information Database (SER, INSR, RBGK, 2023) which contains germination rates derived from measurements on seed collections held in Royal Botanic Garden Kew’s Millennium Seed Bank. I retrieved SID data via TRY (Kattge et al., 2020). All trait datasets were taxonomically harmonized with rWCVP and rWCVPdata (Brown et al., 2023).

Seed mass. Seed mass [mg] is a key trait that can influence a plant's ability to disperse and establish beyond garden boundaries. Lighter seeds are often better adapted for wind and long-distance dispersal and thus colonization of new areas (Westoby et al., 1997), whereas heavier seeds have a superior chance of establishing as seedlings due to a better provision of nutrients (but see Moles & Westoby, 2004). Seed mass data were available for 73% of non-native species, 62% of both endangered and non-endangered native species. Given that seed mass can vary by ten orders of magnitude across plants (Harper, 1977), I log10-transformed seed mass for subsequent analyses.

Terminal velocity. Terminal velocity [m/s] is the maximum speed a seed can reach while falling due to gravity in air without accelerating further, i.e., the rate of fall when the effects of gravity are balanced by air resistance. Seeds with a high terminal velocity tend to fall closer to the parent plant, which means they are less likely to be dispersed by wind over long distances. If species had more than one measurement, I calculated the mean terminal velocity. Terminal velocity data were available for 17%, 20% and 33% of non-native, endangered native, and non-endangered native species, respectively.

Dispersal distance. Dispersal distance is a strong indicator of dispersal ability. Lososová et al., 2023, expanding on the approach of Vittoz & Engler, 2007, classified plants into six dispersal distance classes based on a species' main dispersal mode, life form, plant height, dispersal-related diaspore features and habitat preferences. For example, species in dispersal distance class 1 have an upper distance limit of 1 m, and are small (<30cm) herbs or dwarf shrubs with a local, non-specific and unassisted dispersal, whereas species in dispersal distance class 6 are dispersed by animals and have an upper distance limit of 1500 m. Data were available for 86%, 77% and 76% of non-native, endangered native, and non-endangered native species, respectively.

Germination rate. Germination rate [%] measures the percentage of seeds from a given sample that successfully develop into seedlings under specific conditions in Petri dishes. A high germination rate increases the likelihood for plants to establish. These germination rates however, do not reflect field emergence rates (Hitchmough, 2017). Outside the sterile lab conditions under which germination rates are assessed, field emergence is almost always much lower than germination rate. The specific conditions under which germination rates are assessed in the Seed Information Database can vary within and between species. Conditions vary in pre-sow treatment (e.g., seed scarified, soaked in smoke solution), growing medium (e.g., 1% agar with varying amounts of potassium and gibberellic acid) and temperature and photoperiod conditions. I calculated species mean germination rate across conditions. Data were available for 20%, 24% and 33% of non-native, endangered native, and non-endangered native species, respectively.

Dispersal mode. The predominant dispersal mode includes nonspecific local dispersal, anemochory (dispersal by wind), hydrochory (dispersal by water) and zoochory (dispersal by animals). Zoochory is further distinguished between epizoochory (external dispersal on the surface of animals), endozoochory (dispersal via ingestion by animals), dyszoochory (dispersal by animals that lose seeds before ingestion), and myrmecochory (dispersal by ants). Dispersal modes like anemochory, hydrochory and zoochory are often associated with long-distance dispersal (i.e., seeds can be carried by wind, water or animals to distant locations), whereas more local unspecific and unassisted dispersal (e.g., by barochory) may have more limited dispersal kernels. Data were available for 89%, 81% and 71% of non-native, native endangered and native non-endangered species, respectively. Due to the low proportion of species dispersed by diszoochory (0.5%), I omitted this category from subsequent analyses.

Seed structure. Seed structures include six main morphological types: nutrient-rich structures, balloon structures, elongated appendages, flat appendages, other specializations and no appendages. Nutrient rich structures include, for example, elaiosomes, arils and pulp. Balloon structures bind air to the diaspore and include, for example, glumes (Poaceae) or urticles (Carex). Flat appendages are structures with considerably less height than width and length, for example wings as in *Acer spec.* Elongated appendages are prominent structures that make the seed look longer, for example, long pappuses as in *Centaurea spec.* or hooked structures as in *Bidens spec.* Note that in order to classify all appendages a diaspore has, any combination of the seed structure categories is possible. Data were available for 39%, 40% and 42% of non-native, native endangered and native non-endangered species, respectively.

Seedbank type. Seedbank type is a categorical variable with three categories: transient (persists in soil less than one year), short-term persistent (persists at least one year but less than five years) and long-term persistent (persists at least five years). A longer seed longevity allows species to wait for favorable conditions to germinate and establish in new habitats. This may increase their chances of successful colonization and contributes to population persistence (Stöcklin & Fischer, 1999; Reed et al., 2002). If species had multiple seedbank types, I selected the most frequent type. Data were available for 25% of both non-native and endangered native species, and for 31% of non-endangered native species.

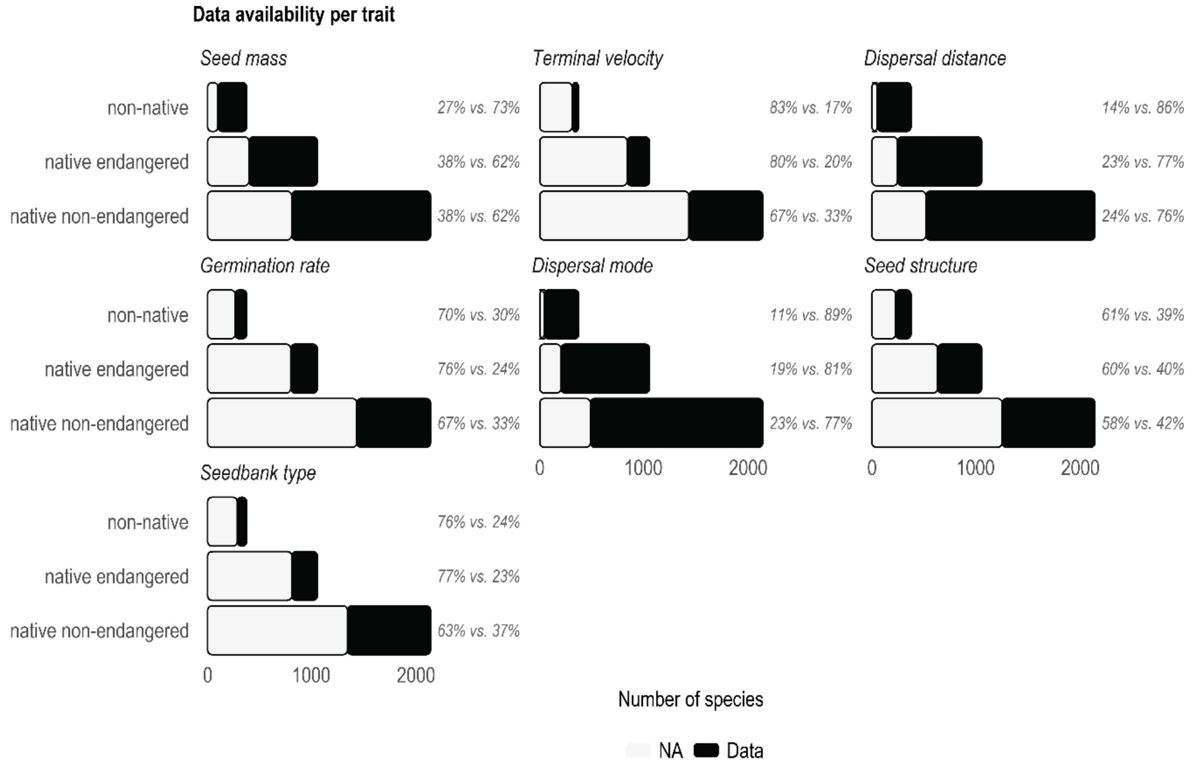

Fig. 1: **Data availability for examined dispersal traits**. Numerous species lacked data, yet there is no consistent pattern of less data for endangered native species.

Data analyses. For the continuous dispersal traits, seed mass, terminal velocity, dispersal distance, and germination rate, I employed linear models, using species type (native endangered, native non-endangered, or non-native species) as the predictor variable. In separate analyses, I also performed linear models for non-native garden escapees and species amenable to conservation gardening. For post hoc comparisons between species types, I used the emmeans package (Lenth et al., 2020). For the categorical dispersal traits, dispersal mode, seed structures, and seedbank type, I employed Chi-square tests. These tests helped determine whether there were statistically clear (Dushoff et al., 2019) differences in the proportion of species falling into specific categories across different species types. For example, I assessed whether the proportion of plants characterized by non-assisted, local dispersal (a category within the dispersal mode variable) differed among non-native, endangered, and non-endangered native species. All R code for data carpentry, analysis and visualization are available on GitHub at https://github.com/istaude/dispersal-endangered.git.

Results

I found only minor, yet in a few cases statistically clear differences in examined dispersal traits among non-native, endangered, and non-endangered native species.

Seed mass. On average, dispersules of non-native species were heavier, weighing 1.59 mg, compared to 0.61 mg for endangered species and 0.94 mg for non-endangered native species (Fig. 2a). Although the pairwise differences may appear small (given that seed mass varied by six orders of magnitude in the data), they were statistically clear (non-native vs. endangered, p-value < 0.001; non-native vs. non-endangered, p-value < 0.001; endangered vs. non-endangered, p-value < 0.001).

Terminal velocity. The differences in terminal velocity were even smaller, with non-native, endangered, and non-endangered native species' dispersules accelerating on average to 2.57 m/s, 2.19 m/s, and 2.36 m/s, respectively (Fig. 2b). None of the pairwise differences were statistically clear (non-native vs. endangered, p-value = 0.074; non-native vs. non-endangered, p-value = 0.371; endangered vs. non-endangered, p-value = 0.19).

Dispersal distance. Non-native species, on average, had a dispersal class of 2.81, while endangered species exhibited a value of 3.01, and non-endangered species showed 3.11 (Fig. 2c). Statistically, there was a clear difference in dispersal class between non-endangered and non-native species (p-value = 0.017), but not between endangered and non-native species (p-value = 0.206) or between endangered and non-endangered species (p-value = 0.41).

Germination rate. Non-native species displayed an average germination rate of 91.5%, which was not clearly higher than that of endangered species (90%, p-value = 0.548) or non-endangered native species (90.7%, p-value = 0.801; Fig. 2d). Moreover, there was no clear difference in germination rates between endangered and non-endangered native species (p-value = 0.739).

Dispersal mode. There was limited evidence to suggest that both non-endangered and non-native species diverged in the same way from endangered species or that they derived advantages from a particular mode of dispersal that might be considered superior (Fig. 2e). For instance, non-native species statistically favored local, unassisted, and non-specific dispersal to a greater extent (61%) compared to both endangered (51%; $X^2 = 9.7$, p-value = 0.0018) and non-endangered (51%; $X^2 = 9.2$, p-value = 0.0024) species. In contrast, endangered species exhibited the highest proportion of wind-dispersed species (26%), with a statistically clear difference compared to non-native (14%; $X^2 = 16.7$, p-value < 0.0001) and non-endangered (20%; $X^2 = 10.6$, p-value = 0.001) species. The only dispersal mode in which endangered species did not benefit to a greater extent was endozoochory (5%). Both non-native (9%; $X^2 = 5.4$, p-value = 0.02) and non-endangered (13%; $X^2 = 37.6$, p-value < 0.001) species were more frequently dispersed through endozoochory (for detailed pairwise comparisons for each dispersal mode category, see Supplementary Table 1.)

Seed structures. Non-native species were most often with no appendage, and both non-endangered and endangered species had across all types of appendages a clearly higher proportion of species with a given appendage (Fig. 2f; Supplementary Table 2). There was no statistically clear difference in the proportion of species with a given appendage between non-endangered and endangered species (Supplementary Table 2), except for nutrient-containing appendages which were proportionally more frequent in non-endangered species ($X^2 = 12.2$, p-value < 0.001).

Seedbank type. For seedbank type, our analysis did not reveal any clear evidence suggesting that the seed longevity of endangered species was consistently inferior to that of both non-endangered and non-native species (see Fig. 2g and Supplementary Table 3). Interestingly, non-endangered native species exhibited a higher prevalence of transient seedbanks compared to non-native species ($X^2 = 4.0$, p-value = 0.026). However, this distinction was less evident when compared to endangered native species. Notably, endangered species tended to have a higher proportion of long-term persistent seeds in comparison to non-endangered species ($X^2 = 3.9$, p-value = 0.049).

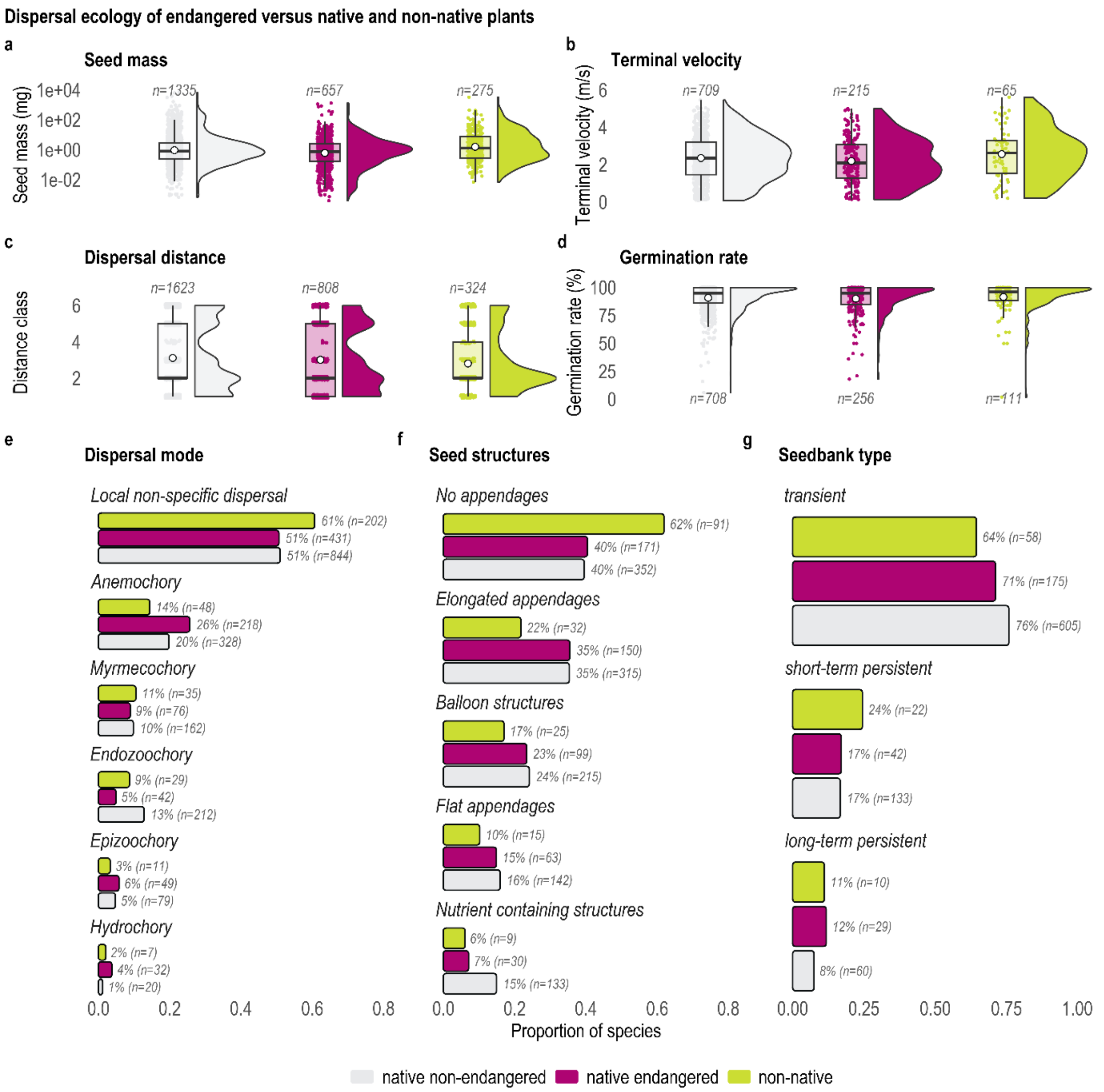

Fig. 2: **Comparison of dispersal traits between endangered native (red), non-endangered native (gray) and non-native plants (green).** a) Seed mass (log10-transformed, mg), b) terminal velocity (m/s), c) dispersal distance (ordinal), d) germination rate (%). Boxplots, jittered data, and density curves are shown; white points indicate means; 'n' signifies the number of species for which data were available. e) Dispersal mode, f) seed structures, and g) seedbank type. Bar plots illustrate species proportion in each category with adjacent percentages. E.g., 61% (or 202 species) of non-native plants exhibit local non-specific dispersal. Totals may exceed 100% for seed structures due to multiple structures per species.

The differences in examined dispersal traits between species for conservation gardening and non-native garden escapee plants were similarly minor, and largely displayed the same patterns. Conservation gardening species did not clearly differ from escapee plants in seed mass (1.47 mg vs 1.65 mg, p-value = 0.6), or terminal velocity (2.49 m/s vs 2.62 m/s, p-value = 0.612), but had a clearly higher dispersal distance (2.86 vs 2.45, p-value = 0.012). Germination rates were comparable (90.6% vs 88.5%, p-value = 0.35). Wind and endozoochory were tended to be more prevalent dispersal modes for conservation gardening species (X^2 = 3.2, p-value = 0.076 and X^2 = 4.8, p-value = 0.029)., while non-native escapees predominantly relied on unassisted dispersal (X^2 = 3.8, p-value = 0.052). This tendency is echoed in seed structures: a majority of non-native escapees (75%) lacked appendages, whereas conservation gardening species often possessed them (X^2 = 31.3, p-value < 0.001). As for seedbank longevity, non-native escapees tended to more frequently exhibit short-term persistence (20% vs 11%, p), while more conservation gardening species showcased transient seedbanks, but none of these differences were statistically clear (Supplementary Table X).

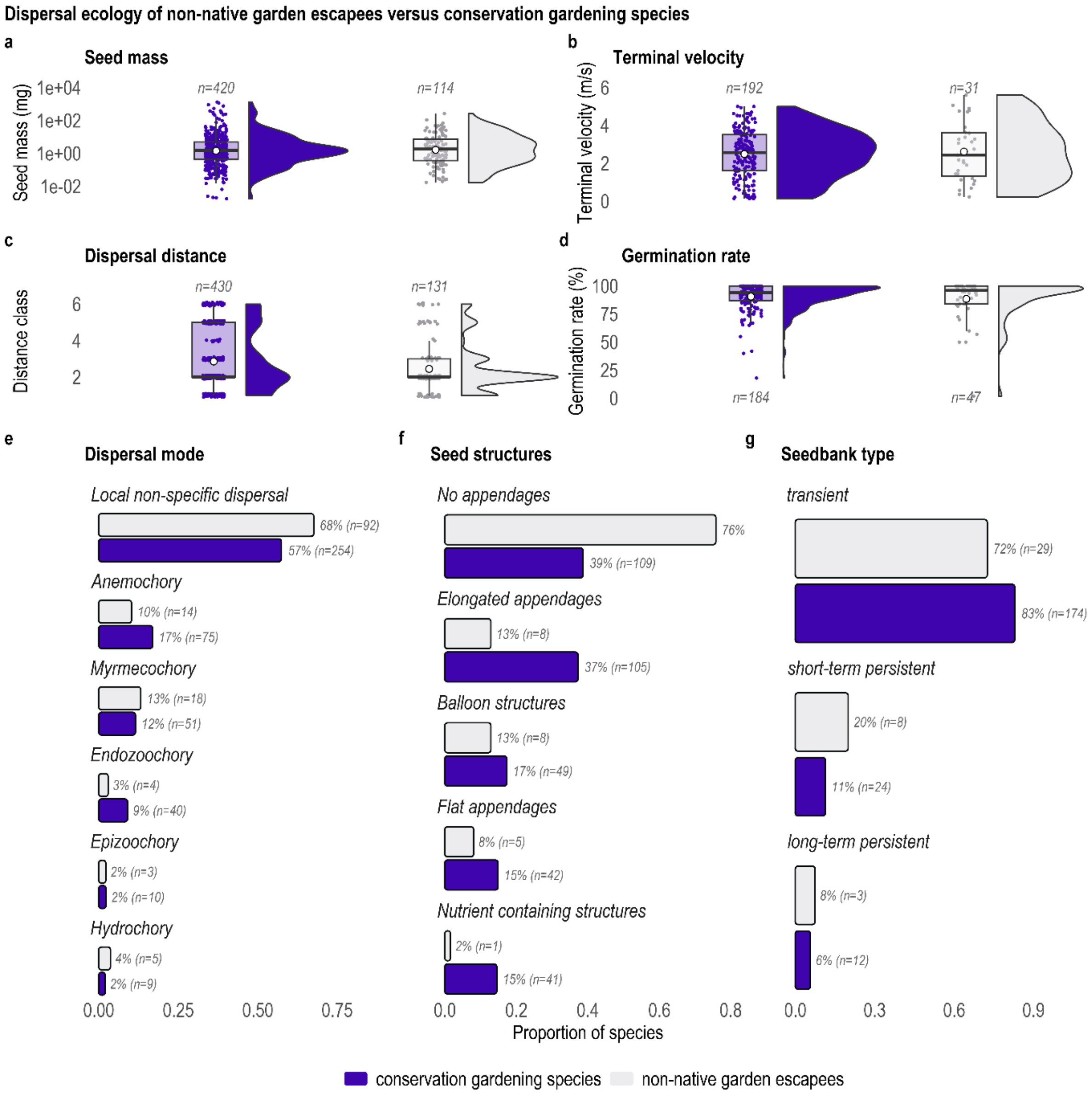

Fig. 3: **Comparison of dispersal traits between endangered native species amenable to gardening (conservation gardening species; blue), and non-native plants grown in gardens that are known to naturalize (non-native garden escapees; gray).** a) Seed mass (log10-transformed, mg), b) terminal velocity

(m/s), c) dispersal distance (ordinal), d) germination rate (%). Boxplots, jittered data, and density curves are shown; white points mark means; 'n' signifies the number of species for which data were available. e) Dispersal mode, f) seed structures, and g) seedbank type. Bar plots illustrate species proportion in each category with adjacent percentages. E.g., 68% (or 92 species) of non-native garden escapees exhibit local non-specific dispersal. Totals may exceed 100% for seed structures due to multiple structures per species.

Discussion

Given gardens' established role in dispersing non-native species (Reichard & White, 2001; Van Kleunen et al., 2018), I investigated if endangered native plants could similarly benefit. My analysis of multiple dispersal traits, including seed weight, terminal velocity, dispersal distance, germination rate, dispersal mode, seed structures, and seedbank type, revealed only subtle but in a few cases statistically clear differences. Endangered plants, both as a whole and those suitable for conservation gardening, were more often wind-dispersed and featured seed appendages that should enable them to harness a wider range of dispersal vectors. In contrast, non-native plants and garden escapees leaned more towards non-assisted local dispersal. Non-native plants further carried slightly heavier seeds and maintained a more persistent seed bank, which may aid in establishment, although germination rates were similar across groups. These findings indicate that, when viewed through the lens of dispersal ecology, green corridors, comprising gardens and other green spaces, have the potential to offer endangered native plants no lesser dispersal pathway as they currently do for non-native species.

This study suggests that endangered native plants tend to produce lighter seeds with reduced fall velocities, leading to marginally increased dispersal distances than non-native species (Fig. 2a-c and Fig. 3a-c). The more frequent presence of appendages should further help endangered plants to better capitalize on wind (via flat or elongated appendages, such as wings or pappusses), water (via balloon structures) or animal-mediated (via nutrient containing and elongated appendages, such as arils or hooks) primary and secondary dispersal than non-native plants (Fig. 2f and Fig. 3f). The higher rate of wind-dispersed species (Fig. 2d and Fig. 3d) may be particularly beneficial in urban to rural green space contexts, fostering an increased likelihood of establishing in novel urban habitats (Planchuelo, Kowarik, & Von der Lippe, 2020). Intriguingly, few native plants predominantly rely on animal dispersal (Fig. 2d and Fig. 3d). Only 5% of all endangered plants use endozoochory, where this may in fact be a notably effective mode in wildlife-friendly green spaces that are often teeming with animal dispersers (Hansen et al., 2020; Lerman et al., 2021). Nonetheless, more endangered species amenable to gardening may benefit from endozoochory compared to non-native escapee plants (Fig. 3e). Endangered plants' more transient seed banks (Fig. 2g and Fig. 3g) could imply a higher risk of reproductive failure in unpredictable, and especially early-successional environments, as are prevalent in human dominated landscapes (Larson & Funk, 2016; Gioria et al., 2021). But this trait is also common in non-native plants (64%) and garden escapees (72%), making it unlikely that this alone would facilitate much greater dispersal success. While germination rates are comparable, it does not predict field emergence (Hitchmough, 2017), for which no data were analyzed. Nonetheless, these dispersal traits suggest that the average native plant, endangered or not, possesses a dispersal ability not too different from non-native species.

Might these subtle differences between increasing (non-native; Eichenberg et al., 2021) species with declining (endangered) ones indicate that dispersal is not a primary factor in these differing population trends? The answer is more complex. Over centuries, the central European landscape and its dispersal processes have undergone profound changes, reviewed for example by Poschlod & Bonn, 1998. Given that many plants can utilize epizoochory (Poschlod & Bonn, 1998; Couvreur et al., 2004), seed dispersal by animals is a key pathway that has undergone tremendous changes. Livestock management in central Europe largely shifted from transhumance or local herding to stable or paddock systems (Hornberger, 1959; Erdmann, 1983). Additionally, the number of livestock decreased, for example, sheep numbers in Germany plummeted from 25 million in 1870 to 5 million by 1907 (Lahrkamp, 1928). Since these herbivores inhabited and linked grasslands, some authors posit that this disruption in meta-communities could be responsible for the decline in grassland species (Krauss, 1977; Ruxton & Schaefer, 2012) (many of which are in fact endangered; Staude et al., 2023). Recent research suggests that many plant species may struggle to adapt to their shifting climate niches, potentially risking extinction, as a result of defaunation (Fricke et al., 2022). Hence it is important to understand how dispersal processes are changing and that the conservation of endangered species necessitates a holistic perspective on past and present dispersal mechanisms (Ruxton & Schaefer, 2012).

Although some of the key original dispersal processes are no longer in place, there are new ones that are simulating or replacing them (Poschlod & Bonn, 1998; Auffret, 2011; Bullock et al., 2018). Modern landscapes are dominated by human-centric infrastructures—roads, railways, and gardens—that shape plant dispersal patterns (Poschlod & Bonn, 1998; Wichmann et al., 2009). These man-made pathways primarily connect human habitats, sidelining natural landscapes like semi-natural grasslands. Consequently, we tend to disperse species that thrive in these human-modified habitats (Kalusová et al., 2017; Segar et al., 2022). Given our tendency to enhance habitat fertility, coupled with frequent disturbances, means we favor the dispersal of fast-growing, nitrophilous species (Hodkinson & Thompson, 1997; Auffret & Cousins, 2013). This includes many ornamental plants favored in our gardens

(Van Kleunen et al., 2018). Together this creates a dispersal bias. To leverage the dispersal avenues in an anthropocentric landscape for biodiversity conservation, we must integrate endangered species into our urban and rural matrix, so that we start dispersing these species as well (Segar et al., 2022). Studies show, for example, that endangered species, such as orchids can thrive along roadsides, and in the neo-habitats humans create (Fekete et al., 2017; Rewicz et al., 2017; Fekete et al., 2020). Our analysis indicates that, based on dispersal ecological differences alone, there is no inherent reason why endangered species should not benefit from this type of dispersal.

Clearly this study does not provide direct and definitive evidence that green corridors aid endangered plants. Experiments are necessary to understand these dynamics, where genetic studies could illuminate the extent to which green corridors, such as urban and rural green spaces, facilitate gene flow among fragmented populations (Plue et al., 2022). Furthermore, databases are necessary that track the field emergence of endangered species, which will also be vital for conservation gardening and the design of seed mixtures. While there is evidence suggesting humans spread endangered species, research often primarily concentrates on non-native species (Auffret & Cousins, 2013). To harness human-mediated dispersal for biodiversity conservation, we need to gain a better understanding of how humans disperse endangered plants. This includes, for example, testing which and how many of these species are dispersed via footwear or clothing when they are cultivated in gardens. Furthermore, understanding where these plants successfully establish is vital. Identifying new rural and urban habitats that are conducive, and determining ways to aid this establishment process, is critical (Planchuelo, Kowarik, & von der Lippe, 2020; Planchuelo, Kowarik, & Von der Lippe, 2020). Numerous questions about the effectiveness of green corridors for endangered plants remain. However, this study suggests that associating endangered species with our green spaces, as advocated by conservation gardening, could enhance at least one critical process underlying the population persistence of declining native species: dispersal.

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