**Journal name**

*Ecology*

**Manuscript type**

Article

**Title**

Elevated temperatures shift flower head height distributions and seed dispersal patterns in two invasive thistle species

**Authors**

Trevor Drees1, Katriona Shea1

**Author affiliations**

1Department of Biology and IGDP in Ecology, The Pennsylvania State University

208 Mueller Laboratory, University Park, Pennsylvania 16802 USA

**Corresponding author**

Trevor Drees (thd5066@psu.edu)

**Data accessibility**

All data and code are on GitHub at <https://github.com/TrevorHD/FlowerHeightShifts> and will be made public upon acceptance for publication.

**Key words**

*Carduus acanthoides*; *Carduus nutans*; climate change; dispersal kernel; invasive species; weed; wind dispersal

**Abstract**

Climate change may alter how organisms disperse, with implications for models of population spread and species management. Wind dispersed plants have emerged as a useful study system for investigating effects of climate change on dispersal, but while many previous studies have successfully modeled wind dispersal, these models often assume that propagules are released from a single point on an individual. This simplifying assumption, while useful, has the potential to over- or under-estimate dispersal. Here, we investigate the effects of climate change on dispersal distances and spread rates, examining how these quantities shift when accounting for all sources of seed release on a plant. Using the wind-dispersed invasive thistles *Carduus nutans* and *Carduus acanthoides*, we quantify temperature-driven shifts in the entire distribution of flower head heights using a passive warming field experiment, and project the effects of these flower head height distribution shifts on dispersal using the Wald analytical long distance (WALD) dispersal model. We also compare dispersal distances and the resulting spread rates for both warmed and ambient temperature plants, considering the entire distribution of flower head heights versus assuming only point-source seed release at the maximum height.

An approximately 0.6 °C increase in ambient temperature increased *C. nutans* mean and maximum flower head heights by 14.1 cm (15.0%) and 14.0 cm (13.2%), respectively; larger mean and maximum flower head height increases of 21.2 cm (26.6%) and 31.8 cm (36.7%) were observed in *C. acanthoides*. Seeds from warmed individuals were more likely to exceed any given dispersal distance than those from their unwarmed counterparts; warmed *C. nutans* and *C. acanthoides* seeds were on average 1.36 and 1.71 times as likely, respectively, to travel 10 m or more, with this disparity becoming higher at longer dispersal distances. For *C. nutans*, increased temperatures boosted spread rates by approximately \_\_%, while using a maximum-height point source rather than the distribution of heights increased invasion speeds by approximately \_\_% and \_\_% for warmed and unwarmed populations, respectively. This has implications for models of population spread, as the simplifying assumption of using a maximum-height point source may overestimate spread rates.

**Introduction**

For sessile organisms, the short window of time in which seed or propagule dispersal occurs is typically the only part of their life cycle where movement is possible, with the remainder of that life cycle characterized by immobility. These sessile organisms, including fungi, plants, and marine organisms such as corals and sponges, all rely heavily upon their environment for dispersal. Here, environmental conditions play a key role in dispersal outcomes (Richmond 1990, Willocquet and Clerjeau 1998, Ozinga *et al*. 2004), with variation in individual characteristics also affecting dispersal patterns (Nanninga and Berumen 2014, Schupp *et al*. 2019, Snell *et al*. 2019). Dispersal patterns of these sessile organisms are a crucial factor in determining the spatial distribution of the next generation of individuals (Nathan and Muller-Landau 2000). Such spatial distributions may shift as a result of climate change, which may alter dispersal patterns by affecting factors that directly or indirectly control dispersal; examples include structure or fecundity of the maternal organism from which propagules come, the number of propagules produced, the shape and size of the propagules themselves, and the efficacy of dispersal vectors such as wind or water (Johnson *et al*. 2019, Snell *et al*. 2019).

In recent years, wind-dispersed plants have emerged as a study system for investigating how climate change may affect dispersal of sessile organisms, with experiments and modeling helping to illuminate possible shifts in dispersal patterns. For example, Kuparinen *et al*. (2009) use micrometeorological data to show that increased air temperatures can lead to higher rates of long-distance dispersal for a variety of plant types in a southern Finland boreal forest; Zhang *et al*. (2011) use a combination of field experiments and models to examine how increases in temperature affect reproduction and dispersal distances in the invasive thistle *Carduus nutans*, and the resulting increased rates of population spread in the northeastern United States; and Bullock *et al*. (2012) model how predicted changes in wind speed later this century would affect the spread of various wind-dispersed flora in the British Isles. Such combinations of models and experiments illustrate that climate change can impact wind-driven dispersal by affecting factors central to this process such as seed release height, wind speed, and air turbulence.

Currently, the majority of dispersal studies on wind-dispersed plants model the process as occurring from a single height, often the maximum observed seed release height; that is, when estimating dispersal distances, all seeds are assumed to be released from a single point, usually at or near the top of the plant. However, wind-dispersed plants usually have multiple reproductive structures and thus multiple points of seed release that are not necessarily the same height above the ground. Given that seed release height is an important factor in modeling dispersal distances (e.g., Katul *et al*. 2005, Kuparinen *et al*. 2006, Nathan *et al*. 2011) because it will determine how long a seed is suspended in the air and thus how far it is carried by wind, appropriately characterizing seed release heights is necessary for accurately quantifying dispersal. By accounting for the distribution of seed release heights within individuals instead of using a point source at the maximum height, more representative dispersal estimates can be constructed. However, the need for using the distribution of seed release heights in dispersal models may or may not be necessary depending on plant morphology and variation in these seed release heights (Cousens and Rawlinson 2001). On one hand, for plants with a single small inflorescence or many small inflorescences at approximately the same height, estimation of seed release from a single point source may be appropriate; on the other hand, for plants where inflorescence height varies dramatically, using the distribution of flower heights may be more appropriate.

Here, we combine field experiments and dispersal models to assess effects of climate change on dispersal when accounting for multiple release heights within individuals where seed release heights vary extensively. We focus our investigation on climate change in the form of increases in mean temperature, and seek to examine effects of increased growing temperature on dispersal and invasion rates while accounting for the entire distribution of seed release heights rather than a point source at the maximum. Here, we address three questions of interest. First, how does increased temperature affect the distribution of flower head heights (and thus seed release heights)? Second, how does increased growing temperature affect dispersal distances and invasion rates over the distribution of flower head heights? And third, are there differences in dispersal distances and invasion rates when using the distribution of seed release heights rather than a point source at the maximum height?

**Methods and materials**

***Study******species***

*Carduus nutans* L. (“musk thistle” or “nodding thistle”) and *Carduus acanthoides* L. (“plumeless thistle”) are closely related invasive thistles in the Asteraceae family. Both species germinate in autumn or spring and bolt in the early summer (Zhang *et al*. 2012), reproduce exclusively by seed, and have monocarpic perennial life cycles that have been demonstrated (for *C. nutans*) to shift from biennial towards annual under warming conditions (Keller and Shea 2021). While sharing similar life histories, the two species display significant morphological differences in dispersal-related characteristics such as number of flower heads, flower head size, seed count per flower head, and distribution of flower heads across an individual (Desrochers *et al*. 1988). These thistles have high reproductive potential and are a considerable agricultural pest since they thrive in pastures, are unpalatable to most grazers, and decrease pasture productivity (Trumble and Kok 1982). In addition to pastures, these thistles occur in other highly disturbed areas such as drainages and roadsides. Both species can be found across the U.S. (Allen and Shea 2006), and are listed as noxious weeds in several states (Skinner *et al.* 2000).

Wind serves as the primary dispersal vector in both *C. nutans* and *C. acanthoides*. Seeds of both species display a prominent pappus that, when remaining attached to the achene, decreases falling velocity and makes it possible for seeds to be carried long distances. For example, Skarpaas and Shea (2007) detected seeds traveling away from their parent plants at distances up to 96 m for *C. nutans* and 16 m for *C. acanthoides*, though longer distances are likely possible in extreme wind events. The achenes and pappi of both species differ in size; *C. nutans* seedsare on average larger than those of *C. acanthoides* (Skarpaas *et al*. 2011), and there is between- and within-species variation in plume loading and plume density that generates variation in seed terminal velocity. Here, we focus on wind-driven primary dispersal that may be affected by an additional potential source of variation: seed release height.

***Experimental design***

Experiments measuring the effects of warming on the distribution of *C. nutans* and *C. acanthoides* flower head heights were conducted at the Russell E. Larson Agricultural Research Farm in Rock Springs, Pennsylvania. The field site was previously a rocky-soiled pasture at the base of a mountain ridge dominated by deciduous forest. To simulate the disturbed habitats in which these thistles are often found, aboveground vegetation at the site was killed using an offset disk, and the soil surface was leveled using a roller harrow before any planting occurred.

After being started in a greenhouse for approximately one month, thistle rosettes for each species were planted in groups of four, each in a 2m x 2m plot. To simulate increased ambient temperatures, 100 *C. nutans* and 48 *C. acanthoides* were randomly chosen to be passively warmed using fiberglass open-top chambers (OTCs) shortly after being planted in the field. A total of 272 *C. nutans* and 136 *C. acanthoides* were planted in 17 blocks, with each block consisting of 16 *C. nutans* and 8 *C. acanthoides*. The OTCs used in this experiment simulate an approximately 0.6 °C increase in temperature while not affecting soil moisture or snow depth (Zhang *et al*. 2011) and are built to the specifications listed in the International Tundra Experiment Manual (Molau and Mølgaard 1996). All OTCs were held into the ground with rebar and remained in place for the remainder of the thistle life cycle. Over the course of the experiment, plot vegetation was trimmed to prevent confounding with different vegetation interactions while simulating growth after invasion into newly disturbed ground; however, vegetation was not trimmed in winter due to snow cover.

As flowers began to set seed, mesh pollen bags were used to keep seeds from escaping into the environment while still allowing the flowers access to air, water, and sunlight. Once any particular individual either a) desiccated, b) collapsed under its own weight, or c) stopped producing new flower buds, the heights of all flower heads on the individual were measured before cutting the plant down. In instances where pollen bags caused flower heads to droop, the erect height of the flower head was measured. All flower head height measurements were taken over the course of three weeks, starting in mid-July and terminating in early August.

***Dispersal and spread models***

Dispersal was modeled using the Wald analytical long-distance dispersal (WALD) model (Katul *et al*. 2005). This mechanistic model, based in fluid dynamics, predicts the distribution of propagule dispersal distances by wind and has been shown to be a suitable approximation of empirically determined kernels for wind-dispersed plants (Katul *et al*. 2005, Skarpaas and Shea 2007, Bullock *et al*. 2012). The WALD kernel is an inverse Gaussian distribution of the form

|  |  |  |
| --- | --- | --- |
|  |  | [1] |

where denotes a given dispersal distance. The location parameter and scale parameter are functions of a fixed seed release height , mean wind speed , a fixed seed terminal velocity , and turbulent flow parameter .

To correct wind speed measurements for use at any seed release height rather than at measurement height, we used the same procedure as Skarpaas and Shea (2007) and Bullock *et al*. (2012) and integrate wind speed over the logarithmic wind profile

|  |  |  |
| --- | --- | --- |
|  |  | [2] |

where is the friction velocity, is the height above the ground, d is the zero-plane displacement, is the roughness length, and is the von Karman constant. Methods for calculating zero-plane displacement and roughness length can be found in Raupach (1994); suitable approximations of these values for grasslands can be found in Wiernga (1993). Calculations for and were performed using the methods in Skarpaas and Shea (2007).

While the WALD model can use fixed or mean values of wind speed and terminal velocity, as is often done when calculating and in Equation 1, failure to account for variation in these parameters may over- or under- estimate dispersal. To better account for effects of this variation, we integrate over these parameters using the same methods as Skarpaas and Shea (2007), who have applied this technique to both *C. nutans* and *C. acanthoides*. This yields the kernel

|  |  |  |
| --- | --- | --- |
|  |  | [3] |

where and are the probability density functions for seed terminal velocity and wind speed, and is the dispersal kernel in Equation 1 using variable wind speeds and terminal velocities rather than fixed values. Wind speed data were obtained from a local weather station, and the distribution of terminal velocities from seed drop experiments in a laboratory setting. To incorporate the entire distribution of flower head heights, we build upon this by integrating across the distribution of heights from the field experiment such that

|  |  |  |
| --- | --- | --- |
|  |  | [4] |

so the new dispersal kernel now also accounts for the different flower head heights from which seeds can be released rather than just using a fixed release height (Skarpaas *et al*. 2011). Note that here, represents the distribution of all flower head heights, not just the maxima. Dispersal was then simulated for the various combinations of species, warmed/unwarmed treatment, and maximum/distributed flower heights so that dispersal kernels between these various combinations could be compared.

To examine the effects of shifts in flower head height and dispersal kernels on rates of invasion, we modeled population spread as a moving wave by combining these shifts with a four-stage demographic model similar to that used in previous studies (Zhang *et al*. 2011, Teller *et al*. 2016). This model estimates reproductive output for each stage and then uses the moment-generating function of the simulated dispersal kernels to estimate wave speeds. Due to limited demographic information on *C. acanthoides*, spread rates were only estimated for *C. nutans*. The methods used to model these spread rates are described in greater detail in Appendix S1.

***Statistical Analyses***

All modeling and statistical analyses were performed in R version 4.0.3 (R Development Core Team, 2020). For each thistle species, effects of warming treatment on mean flower head height were assessed using a linear mixed-effects model with the **lmer** function from the package **lme4** version 1.1-26 (Bates *et al*. 2020). Within each model, warming treatment or lack thereof was treated as a fixed effect, initial rosette diameter shortly after planting as a covariate, and treatment block as a random effect. Response variables and rosette diameter covariates for both models were plot-averaged, as has been done for previous analyses in this system where multiple individuals are located within a single experimental unit (Zhang *et al*. 2011). Backward stepwise selection was subsequently applied after models were fit, eliminating fixed effects until Akaike’s information criterion (AIC) was minimized. A combination of Shapiro-Wilk tests and quantile-quantile (Q-Q) plots was used to assess normality of data and model residuals, ensuring model assumptions were not violated; Kolmogorov-Smirnov tests were used to assess the significance of differences between flower head height distributions for warmed and unwarmed groups, as well as differences between dispersal kernels for warmed and unwarmed groups.

**Results**

***Flower heights***

In both species, individuals that received warming treatments had taller flower heads on average; a 14.1 3.9 cm (15.0%) increase in mean flower head height was observed in *C. nutans* (d.f. = 55, *t* = 3.613, *n* = 57, *p* < 0.001), while a 21.2 4.6 cm (26.6%) increase in mean flower head height was observed in *C. acanthoides* (d.f. = 20, *t* = 4.592, *n* = 24, *p* < 0.001). The resulting increases in mean flower head height were associated with rightward shifts in the overall distributions of flower head height (Figure 1; see Appendix S2, Figure S1 for an alternative representation); this resulted in significant differences between the warmed and unwarmed flower head height distributions for *C. nutans* (*D* = 0.272, *n*1 = 310, *n*2 = 1094, *p* < 0.001) and *C. acanthoides* (*D* = 0.461, *n*1 = 588, *n*2 = 931, *p* < 0.001). Individuals that received warming treatments also generally displayed greater maximum flower head heights; a 14.0 4.4 cm (13.2%) increase in mean maximum flower head height was observed in *C. nutans* (d.f. = 54, *t* = 3.186, *n* = 57, *p* = 0.002), while a 31.8 6.1 cm (36.7%) increase mean maximum flower head height was observed in *C. acanthoides* (d.f. = 19, *t* = 5.206, *n* = 24, *p* < 0.001). Plus/minus terms on the increases reported above indicate one standard error on the mean.

***Dispersal: warmed vs. unwarmed***

Shifts in the distributions of flower head height also resulted in different dispersal kernels for warmed and unwarmed individuals of each species (Figure 2). For ten million simulated dispersal events per combination of species and warmed/unwarmed treatment, dispersal kernels for warmed individuals displayed notably lower peaks and fatter tails compared those of unwarmed individuals, with markedly different dispersal kernels in both *C. nutans* (*D* = 0.068, *p* < 0.001) and *C. acanthoides* (*D* = 0.116, *p* < 0.001). The projected mean *C. nutans* dispersal distance increased 21.3% from 3.05 m to 3.70 m, while the mean *C. acanthoides* dispersal distance increased 37.8% from 2.46 m to 3.39 m (Appendix S2, Table S1).

The frequency of longer-distance dispersal events was also affected by the warming-induced shift in flower head height distribution. The mean 95th and 99th percentile dispersal distances for *C. nutans* were 10.53 m and 25.15 m respectively for unwarmed individuals, with a shift to 12.76 and 30.37 m respectively for warmed individuals; for *C. acanthoides*, the mean 95th and 99th percentile dispersal distances were 8.48 m and 19.97 m respectively for unwarmed individuals, with a shift to 11.69 m and 27.58 m for warmed individuals (Appendix S2, Table S1). Warming also increased the probability that a seed would exceed a given distance, with this effect becoming more pronounced at higher dispersal distances (Figure 3); seeds from warmed *C. nutans* and *C. acanthoides* were on average 1.36 and 1.71 times as likely to travel 10 m or more as their respective unwarmed counterparts, and on average 1.66 and 2.44 times as likely to travel 50 m or more as their respective unwarmed counterparts. Uncertainty in these relative frequencies of dispersal events also increases and becomes quite large at high dispersal distances, which echoes the fact that long-distance dispersal events are relatively rare and difficult to capture not only empirically, but in dispersal simulations as well.

***Dispersal: incorporating maximum height vs. height distribution***

Using the maximum flower head height as a point source instead of the distribution of flower head heights in the WALD dispersal model resulted in a significantly different dispersal kernel (Figure 4). For ten million simulated dispersal events per combination of species, warmed/unwarmed treatment, and maximum/distributed height, Kolmogorov-Smirnov tests indicate this was evident for warmed *C. nutans* (*D* = 0.030, *p* < 0.001), unwarmed *C. nutans* (*D* = 0.090, *p* < 0.001), warmed *C. acanthoides* (*D* = 0.058, *p* < 0.001), and unwarmed *C. acanthoides* (*D* = 0.025, *p* < 0.001). Effects on the shape of the dispersal kernel when using the maximum flower head height instead of the distribution of flower head heights were similar to the effects of the warming treatment, as lower peaks and fatter tails were present when using the maximum flower head height. However, though Kolmogorov-Smirnov tests indicate that dispersal kernels constructed using the height distribution and dispersal kernels constructed using only the maximum height as a point source are distinct, visual inspection of the kernels in Figure 4 suggest that the change in kernel shape is not as obvious as that observed when comparing warmed and unwarmed individuals in Figure 2.

The frequency of longer-distance dispersal events was also affected when using the maximum flower head height as a point source instead of the flower head height distribution. Using maximum flower head height increased mean 95th and 99th percentile dispersal distances across the four combinations of species and warming treatment (Appendix S2, Table S2), though not by much compared to the differences between warmed and unwarmed treatment groups when using the entire flower head height distribution. The probability of seeds exceeding a given dispersal distance is higher when using the maximum flower head height than when using the entire distribution of flower head heights (Figure 5) at lower distances, but the uncertainty surrounding this trend increases dramatically at higher distances; here, the bootstrap intervals sometimes even envelop the null risk ratio of 1 associated with no significant difference.

***Spread rates***

Warming-induced shifts in the distributions of *C. nutans* flower head height resulted in increased median spread rates, with an approximately \_\_% increase from \_\_ m/yr to \_\_ m/yr. When also accounting for other demographic changes caused by increased growing temperatures in addition to the height increases, we observed an even larger increase in median spread rate of approximately \_\_% from \_\_ m/yr to \_\_ m/yr. Using mean maximum flower head height as a point source resulted in a median spread rate of \_\_ m/yr, or \_\_% higher than the \_\_ m/yr estimated when using the distribution of flower head heights. This difference was higher when accounting for other demographic changes caused by increased growing temperatures in addition to the height increases, with a \_\_% increase from \_\_ m/yr to \_\_ m/yr. See Appendix S1, Table S1 for additional details. Note that we report median invasion rates here so we can make direct comparisons to previous studies that report results with that same metric.

**Discussion**

Our results demonstrate that increases in temperature, which are expected in the wake of climate change across much of the geographical range of *C. nutans* and *C. acanthoides*, can dramatically increase mean flower head height and significantly shift the distribution of flower head heights in these two non-native, invasive thistles. Changes in flower head heights and distributions for *C.* *acanthoides* have not been previously studied; here, we show that even mild increases in average ambient temperatures have striking effects on *C.* *acanthoides* plant height (which was increased by 31.8 cm) and flower head distributions. For *C. nutans*, our results are similar to those from a prior study conducted by Zhang *et al*. (2011).

These changes in flower head height and distribution have important ramifications for dispersal. Such changes in flower head height shift dispersal kernels and increase seed dispersal distances, especially along the right tail of the kernels. Seeds from warmed thistles had a higher probability of exceeding a given dispersal distance than those from their unwarmed counterparts; warmed *C. nutans* and *C. acanthoides* seeds were on average 1.66 and 2.44 times as likely, respectively, to travel 50 m or more. This disparity is stronger at longer threshold dispersal distances; in a patchy landscape such differences can differentiate success or failure to reach suitable habitat. However, using the maximum height rather than height distribution when modeling dispersal kernels may overestimate dispersal at higher dispersal distances, though the resulting differences in dispersal outcomes are smaller than the differences generated by warming; for example, seeds from warmed and unwarmed *C. acanthoides* were on average 1.57 and 1.38 times as likely, respectively, to travel 50 m or more when using the maximum flower head height rather than the distribution of heights, while seeds were 2.44 times as likely to travel 50 m or more if they dispersed from warmed rather than unwarmed *C. acanthoides*.

Warming-induced height increases are only one response that may affect dispersal patterns in wind-dispersed plants under climate change: shifts in air turbulence, wind speeds, and frequency of extreme events may also alter dispersal patterns (Kuparinen *et al*. 2009, Soons *et al*. 2004, Bullock *et al*. 2012). How these changes interact and play out will critically determine distributions of invasive species in future climates. Dispersal kernels like the ones in this study are often used to estimate the rate at which species spread, and are often included in integrodifference and integral projection models of population spread (e.g., Kot *et al*. 1996, Neubert and Caswell 2000, Ellner and Rees 2006, Jongejans et al. 2011). These models can be highly sensitive to the right tails of dispersal kernels used in the model framework; population movement here can often be driven by a small handful of long-distance dispersal events (Kot *et al*. 1996, Clark *et al*. 1998, 2001). Because propagules dispersed long distances can escape density-dependent mortality (Janzen 1970, Connell 1971) from sources such as predation (Blundell and Peart 1998, Norghauer *et al*. 2010) or infection by pathogens (Augspurger 1983, Augspurger and Kelly 1984), or because they can simply find more suitable habitat, they may experience increased fitness and make significant contributions to population growth and spread. If warming due to climate change increases the likelihood or magnitude of dispersal events in the right tail of the kernel as we have shown, it may have strong effects on spread rates, even if increased dispersal distances are somewhat modest. The information used to construct dispersal kernels can affect estimates of spread rates too, and modeling dispersal using maximum flower head height rather than the distribution of flower head heights may overestimate spread rates since models using the maximum height will assume that propagules are released from higher above the ground than they actually are; propagules will then spend more time in the air and thus be carried further by wind, leading to an overrepresentation of longer dispersal distances. Sentence about table S3, and how using the distribution of maxima rather than mean maximum as a point source may get us closer to the truth without much more effort.

Given the sensitivity of population spread models to long distance dispersal events, a better understanding of how to quantify long distance dispersal is necessary for more accurate measurements of population spread; this improved accuracy is especially important since climate change and increased temperatures may, as our study demonstrates, make long-distance dispersal more common and necessitate new management strategies for invasive species like *C. nutans* and *C. acanthoides*. However, quantifying long distance dispersal events can be quite challenging and comes with a large degree of uncertainty (Cain *et al*. 2000, Nathan *et al*. 2003, Nathan 2006). This is quite evident in Figures 3 and 5; as dispersal distance increases, the uncertainty greatly increases because long-distance dispersal events are inherently rare. Even with the approximately ten million dispersal events simulated in each figure panel, extremely rare long distance dispersal events may not even show up in the dispersal simulations; this same problem occurs in real dispersal as well, as propagule trapping or tracking may fail to detect such long-distance dispersal. The elusive nature of these long-distance dispersal events can be problematic for estimating how quickly species spread; if models of population spread are often heavily influenced by these long-distance dispersal events, yet we struggle to adequately capture them, then uncertainty in the frequency of such events can adversely affect our ability to measure spread rates (Skarpaas and Shea 2007).

In addition to the implications for models of population spread, our findings have potential implications for invasive species management as well. For invaders like *C. nutans* and *C. acanthoides,* climate change poses a significant challenge to managementby possibly altering how they grow, reproduce, and spread (Mainka and Howard 2010, Ziska *et al.* 2011). As a result, current invasive species management strategies may no longer match the phenology of their target species and become less effective or even entirely unsuitable in future climates (Hellmann *et al.* 2008, Keller and Shea 2021). Concerns regarding invasive species control in future climates have become widespread among natural resource managers; a recent survey by Beaury *et* *al.* (2020) reports that many natural resource managers worry about how climate change will affect invasive species management, and 65% of managers have incorporated climate change into their management plans. One particular challenge for managers of invasive species lies within how climate change affects dispersal, as dispersal governs where and how quickly invasive species populations spread. Knowledge of how climate change may alter movement and dispersal patterns would be useful in helping invasive species managers improve predictions of spread speeds and make more informed management decisions (Caplat *et al*. 2013), ultimately saving time and money in the process.

Our findings also contribute to the body of literature addressing individual variation within populations and how it determines dispersal (Schupp *et al*. 2019). While we account for variation in terminal velocity and wind speed when modeling dispersal, we specifically focus on variation in flower head height and demonstrate that accounting for this variation yields a different dispersal kernel than assuming dispersal from the maximum head height, though this difference appears to be less stark as that from warming effects. Neglecting the distribution of propagule release heights in favor of the maximum height will overestimate dispersal distances, likely resulting in inflated rates of projected population spread. Depending on the study system and magnitude of this inflation, this may or may not matter for management purposes; if the amount of effort necessary for data collection is high, minor overestimates in population spread resulting from assuming the maximum might be worth foregoing the need to gather data on the entire distribution of propagule release heights.

Overall, accurately quantifying dispersal and how it is affected by environmental factors is crucial for understanding how organisms move across a landscape, as the dispersal process drives this movement. This is especially true for sessile organisms where environmental factors play a critical role in governing dispersal patterns, with these environmental factors subject to the effects of climate change. Here, we demonstrate that the common simplifying assumption of representing seed release as a point source from a single flower head at the maximum height can result in markedly different dispersal patterns than when seed release is represented as a distribution of point sources within an individual; for species like *C. nutans* and *C. acanthoides* that have flower heads at a wide variety of heights, the latter of these representations better matches these species’ morphology and may thus produce more accurate estimates of dispersal, likely resulting in more accurate estimates of species spread. We also show that even minor increases in growing temperature can drastically increase seed release heights, with this change in morphology resulting in greater dispersal distances. Developing a better understanding of how climate change affects these dispersal patterns will be important for better modeling and managing the spread of invasive plant species, as well as the movement of other sessile species, and for better predicting how their populations will shift over the coming decades.

**Acknowledgements**

The authors would like to thank J. Darlington, M. Dwyer, A. Kirt, and other personnel at the Russell E. Larson Agricultural Research Center for logistical support, and E. Howerton, E. Johnson, and J. Keller for assisting in setup of the experiment. O. Skarpaas provided advice on dispersal models. E. Hanks, L. Leites, and C. Lowry helped provide feedback on the manuscript. K. Marchetto and E. Jongejans kindly provided *C. nutans* and *C. acanthoides* terminal velocity data for use in wind dispersal models, and B. Teller provided code for invasion speeds. T.H.D. was supported by the Alumni Professor in the Biological Sciences endowment to K.S.

**Author Contributions**

T.H.D and K.S. designed the field experiment. T.H.D. conducted the field experiment and associated data and modeling analyses, and led writing of the initial manuscript. T.H.D. and K.S. made contributions to subsequent versions of the manuscript.

**References**

Allen, M.R., & Shea, K. (2006). Spatial segregation of congeneric invaders in central Pennsylvania, USA. *Biological Invasions*, 8(3), 509-521.

Augspurger, C. K. (1983). Seed dispersal of the tropical tree, *Platypodium elegans*, and the escape of its seedlings from fungal pathogens. *The Journal of Ecology*, 71(3), 759-771.

Augspurger, C.K. & Kelly, C.K. (1984). Pathogen mortality of tropical tree seedlings: experimental studies of the effects of dispersal distance, seedling density, and light conditions. *Oecologia*, 61(2), 211-217.

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2020). Package ‘lme4’, version 1.1-26. https://cran.r-project.org/web/packages/lme4/index.html

Beaury, E.M., Fusco, E.J., Jackson, M.R., Laginhas, B.B., Morelli, T.L., Allen, J.M., Pasquarella, V.J., & Bradley, B.A. (2020). Incorporating climate change into invasive species management: insights from managers. *Biological Invasions*, 22(2), 233-252.

Blundell, A.G. & Peart, D.R. (1998). Distance-dependence in herbivory and foliar condition for juvenile *Shorea* trees in Bornean dipterocarp rain forest. *Oecologia*, 117(1-2), 151-160.

Bullock, J.M., White, S.M., Prudhomme, C., Tansey, C., Perea, R., & Hooftman, D.A. (2012). Modeling spread of British wind‐dispersed plants under future wind speeds in a changing climate. *Journal of Ecology*, 100(1), 104-115.

Cain, M.L., Milligan, B.G., & Strand, A.E. (2000). Long‐distance seed dispersal in plant populations. *American Journal of Botany*, 87(9), 1217-1227.

Caplat, P., Cheptou, P.O., Diez, J., Guisan, A., Larson, B.M., Macdougall, A.S., Peltzer, D.A., Richardson, D.M., Shea, K., van Kleunen, M., Zhang, R., & Buckley, Y. M. (2013). Movement, impacts and management of plant distributions in response to climate change: insights from invasions. *Oikos*, 122(9), 1265-1274.

Clark, J.S., Fastie, C., Hurtt, G., Jackson, S.T., Johnson, C., King, G.A., Lewis, M., Lynch, J., Pacala, S., Prentice, C., & Schupp, E.W. (1998). Reid's paradox of rapid plant migration: dispersal theory and interpretation of paleoecological records. *BioScience*, 48(1), 13-24.

Clark, J.S., Lewis, M., & Horvath, L. (2001). Invasion by extremes: population spread with variation in dispersal and reproduction. *The American Naturalist*, 157(5), 537-554.

Connell, J.H. (1971). On the role of natural enemies in preventing competitive exclusion in some marine animals and in rain forest trees. *Dynamics of Populations*, 298-312.

Cousens, R.D. & Rawlinson, A.A. (2001). When will plant morphology affect the shape of a seed dispersal “kernel”?. *Journal of Theoretical Biology*, 211(3), 229-238.

Desrochers, A.M., Bain, J.F., & Warwick, S.I. (1988). The Biology of Canadian Weeds.: 89. *Carduus nutans* L. and *Carduus acanthoides* L. *Canadian Journal of Plant Science*, 68(4), 1053-1068.

Ellner, S.P. & Rees, M. (2006). Integral projection models for species with complex demography. *The American Naturalist*, 167(3), 410-428.

Hellmann, J.J., Byers, J.E., Bierwagen, B.G., & Dukes, J. S. (2008). Five potential consequences of climate change for invasive species. *Conservation Biology*, 22(3), 534-543.

Janzen, D.H. (1970). Herbivores and the number of tree species in tropical forests. *The American Naturalist*, 104(940), 501-528.

Johnson, J. S., Cantrell, R. S., Cosner, C., Hartig, F., Hastings, A., Rogers, H. S., Schupp, E. W., Shea, K., Teller, B. J., Yu, X. Zurell, D., & Pufal, G. (2019). Rapid changes in seed dispersal traits may modify plant responses to global change. *AoB Plants*, 11(3), plz020.

Jongejans, E., Shea, K., Skarpaas, O., Kelly, D., & Ellner, S.P. (2011). Importance of individual and environmental variation for invasive species spread: a spatial integral projection model. *Ecology*, 92(1), 86-97.

Katul, G.G., Porporato, A., Nathan, R., Siqueira, M., Soons, M.B., Poggi, D., Horn, H.S., & Levin, S.A. (2005). Mechanistic analytical models for long-distance seed dispersal by wind. *The American Naturalist*, 166(3), 368-381.

Keller, J.A. and Shea, K. (2021). Warming and shifting phenology accelerate an invasive plant life cycle. *Ecology*, 102(1), e03219.

Kot, M., Lewis, M.A. & van den Driessche, P. (1996). Dispersal data and the spread of invading organisms. *Ecology*, 77(7), 2027-2042.

Kuparinen, A. (2006). Mechanistic models for wind dispersal. *Trends in Plant Science*, 11(6), 296-301.

Kuparinen, A., Katul, G., Nathan, R., & Schurr, F.M. (2009). Increases in air temperature can promote wind-driven dispersal and spread of plants. *Proceedings of the Royal Society B: Biological Sciences*, 276(1670), 3081-3087.

Mainka, S.A. & Howard, G. W. (2010). Climate change and invasive species: double jeopardy. *Integrative Zoology*, 5(2), 102-111.

Molau, U. & Mølgaard, P. (1996). International Tundra Experiment Manual. Danish Polar Centre, Copenhagen.

Nanninga, G.B. & Berumen, M.L. (2014). The role of individual variation in marine larval dispersal. *Frontiers in Marine Science*, 1, 71.

Nathan, R. & Muller-Landau, H.C. (2000). Spatial patterns of seed dispersal, their determinants and consequences for recruitment. *Trends in Ecology & Evolution*, 15(7), 278-285.

Nathan, R., Perry, G., Cronin, J.T., Strand, A.E., & Cain, M.L. (2003). Methods for estimating long‐distance dispersal. *Oikos*, 103(2), 261-273.

Nathan, R. (2006). Long-distance dispersal of plants. *Science*, 313(5788), 786-788.

Nathan, R., Katul, G.G., Bohrer, G., Kuparinen, A., Soons, M.B., Thompson, S.E., Trakhtenbrot, A., & Horn, H.S. (2011). Mechanistic models of seed dispersal by wind. *Theoretical Ecology*, 4(2), 113-132.

Neubert, M.G. & Caswell, H. (2000). Demography and dispersal: calculation and sensitivity analysis of invasion speed for structured populations. *Ecology*, 81(6), 1613-1628.

Norghauer, J.M., Grogan, J., Malcolm, J.R., & Felfili, J.M. (2010). Long-distance dispersal helps germinating mahogany seedlings escape defoliation by a specialist caterpillar. *Oecologia*, 162(2), 405-412.

Ozinga, W.A., Bekker, R.M., Schaminee, J.H., & Van Groenendael, J.M. (2004). Dispersal potential in plant communities depends on environmental conditions. *Journal of Ecology,* 92(5), 767-777.

R Development Core Team (2020) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.r-project.org/

Raupach, M. R. (1994). Simplified expressions for vegetation roughness length and zero-plane displacement as functions of canopy height and area index. *Boundary-Layer Meteorology*, 71(1), 211-216.

Richmond, R.H. (1990). The effects of the El Niño/Southern Oscillation on the dispersal of corals and other marine organisms. *Elsevier Oceanography Series*, 52, 127-140.

Schupp, E.W., Zwolak, R., Jones, L.R., Snell, R.S., Beckman, N.G., Aslan, C., Cavazos, B.R., Effiom, E., Fricke, E.C., Montaño-Centellas, F., Poulsen, J., Razafindratsima, O.H., Sandor, M.E., & Shea, K. (2019). Intrinsic and extrinsic drivers of intraspecific variation in seed dispersal are diverse and pervasive. *AoB Plants*, 11(6), plz067.

Skarpaas, O. & Shea, K. (2007). Dispersal patterns, dispersal mechanisms, and invasion wave speeds for invasive thistles. *The American Naturalist*, 170(3), 421-430.

Skarpaas, O., Silverman, E. J., Jongejans, E., & Shea, K. (2011). Are the best dispersers the best colonizers? Seed mass, dispersal and establishment in Carduus thistles. *Evolutionary Ecology*, 25(1), 155-169.

Skinner, K., Smith, L., & Rice, P. (2000). Using noxious weed lists to prioritize targets for developing weed management strategies. *Weed Science*, 48(5), 640-644.

Snell, R.S., Beckman, N.G., Fricke, E., Loiselle, B.A., Carvalho, C.S., Jones, L.R., Lichti, N.I., Lustenhouwer, N., Schreiber, S.J., Strickland, C. and Sullivan, L.L. (2019). Consequences of intraspecific variation in seed dispersal for plant demography, communities, evolution and global change. *AoB Plants*, 11(4), p.plz016.

Soons, M.B., Nathan, R., & Katul, G.G. (2004). Human effects on long‐distance wind dispersal and colonization by grassland plants. *Ecology*, 85(11), 3069-3079.

Teller, B.J., Zhang, R., & 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.

Trumble, J.T. & Kok, L.T. (1982). Integrated pest management techniques in thistle suppression in pastures of North America. *Weed Research*, 22(6), 345-359.

Willocquet, L. & Clerjeau, M. (1998). An analysis of the effects of environmental factors on conidial dispersal of *Uncinula necator* (grape powdery mildew) in vineyards. *Plant Pathology*, 47(3), 227-233.

Wiernga, J. (1993). Representative roughness parameters for homogeneous terrain. *Boundary-Layer Meteorology*, 63(4), 323-363.

Zhang, R., Jongejans, E., & Shea, K. (2011). Warming increases the spread of an invasive thistle. *PLoS One*, 6(6), e21725.

Zhang, R., Post, E., & Shea, K. (2012). Warming leads to divergent responses but similarly improved performance of two invasive thistles. *Population Ecology*, 54(4), 583-589.

Ziska, L.H., Blumenthal, D.M., Runion, G.B., Hunt, E.R., & Diaz-Soltero, H. (2011). Invasive species and climate change: an agronomic perspective. *Climatic Change*, *105*(1-2), 13-42.

**Figure Captions**

**Figure 1.** Distribution of observed flower head heights for *C. nutans* and *C. acanthoides* under the control and warming treatments. Solid lines indicate mean probability density for a given height, and error bands indicate a 95% bootstrap interval.

**Figure 2.** Dispersal kernels for *C. nutans* and *C. acanthoides* for the control and warming treatments, both using the observed distribution of flower head heights. Solid lines indicate mean probability density for a given distance, and error bands indicate a 95% bootstrap interval.

**Figure 3.** Relative risk of a seed exceeding a given distance when originating from a warmed maternal plant for *C. nutans* and *C. acanthoides*, as measured by the ratio of complementary cumulative distribution functions for the warmed and unwarmed groups. Solid lines indicate the mean relative risk for a given distance, while the dotted lines indicate a relative risk of 1. Error bands indicate a 95% bootstrap interval.

**Figure 4.** Dispersal kernels for warmed and unwarmed *C. nutans*, as well as for warmed and unwarmed *C. acanthoides*. Solid lines indicate mean dispersal kernels generated using the entire observed distribution of flower heights, while the dotted lines indicate mean dispersal kernels generated using only the maximum flower height. Error bands indicate a 95% bootstrap interval.

**Figure 5.** Relative risk of a seed exceeding a given distance when modeling dispersal using the maximum flower head height rather than the flower head height distribution for warmed and unwarmed *C. nutans*, as well as for warmed and unwarmed *C. acanthoides*. Solid lines indicate the mean relative risk for a given distance, while the dotted line indicate a relative risk of 1. Error bands indicate a 95% bootstrap interval.

**Chart, line chart

Description automatically generated**

**Figure 1**



**Figure 2**



**Figure 3**

**Figure 4**



**Figure 5**