

The Hitchhiker's Guide to Invading an Ecosystem: Effects of Recreation on Non-Native Plant Dispersal

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Author contributions

A.B., L.B., and D.K. participated in the literature review and experimental design. A.B. and L.B. carried out field data collection. A.B. and L.B. completed data preparation, analysis, visualization, and data management. L.B. and D.K. drafted the introduction, and A.B. and D.K. wrote the methods section. A.B. prepared figures, plotted hypotheses and predictions, and wrote the results section, and L.B. wrote the discussion section. D.K. drafted the limitations of the study and conclusion. All reviewed and edited the document prior to submission.

Abstract

Anthropogenic activity influences ecosystems in diverse and often context-specific ways. As human use of protected areas expands, especially in areas near urban developments, the introduction rates of non-native species are continuously accelerating – increasing the risk of ecological disruption through potential invasions. Here, we examine the effects of human recreation on non-native plant dispersal, testing the hypothesis that human recreation facilitates the establishment of non-native species into protected areas by acting as an agent of dispersal. We laid out 250m transects along two trails – one high-use and one low-use – in Pacific Spirit Park, Vancouver, BC, a large urban park with extensive recreational use, and recorded non-native species richness, abundance, and composition every 25m from the trailhead. Using Bayesian generalized linear models and non-metric multidimensional scaling (NMDS) ordination, we examined the effects of trail use and distance from trailhead on non-native species richness, abundance, and composition. We found that non-native richness and abundance **decreased with distance from trailhead**, but surprisingly, also found that trail use had a **strong negative effect** on non-native richness and abundance, possibly due to high trampling preventing seedling establishment in the high-use trail. Our findings underscore the complex nature of human-mediated ecosystem change and the context-dependent role of recreation in non-native species introductions.

1. Introduction

The proliferation of non-native species poses a significant threat to biodiversity and ecological integrity worldwide. Non-native species are “species that are not naturally present in a native assemblage, but have been moved beyond the limits of their normal geographic ranges by human actions” (Blackburn et al., 2011). Human activity has facilitated the introduction of non-native species to new ecosystems, leading to many substantive and unexplored impacts on these ecosystems (Jeschke et al., 2014). Much of this facilitation has arisen from globalization and the global trade networks, which assist species dispersal beyond their normal ranges (Chapman et al., 2017).

Non-native species can impact ecosystem function and biodiversity in complex and context-dependent ways, making their effects hard to predict. Non-native species interact positively and negatively with the ecological function and biodiversity of the ecosystems they inhabit (Jeschke et al., 2014). Many hypothetical mechanisms have been reported to explain the impact that non-native species exert on ecosystems. These are based on ecosystem and species traits, community structure, niche partitioning and competitive exclusion, changing abiotic conditions, organismal influx, and synergistic effects (Ricciardi et al., 2013). Overall, these mechanisms highlight the impact that human activity has on ecological processes, driving changes in community composition.

Human activities play a major role in species introductions, yet gaps remain in our understanding of their influence on dispersal patterns. While it is known that large-scale networks such as global trade play a role in the introduction of non-native

species, the effects of local-scale anthropogenic activities are less studied in literature today (Chapman et al., 2017). Small-scale anthropogenic pathways such as nearby roads and trails may help facilitate the dispersal of non-native species into previously undisturbed local habitats, contributing to changes in community structure (Liedtke et al., 2020). The impact of human activity on dispersal down these trails, however, remains unclear.

Trails in protected areas act as both disturbances and potential pathways for species dispersal, yet the specific mechanisms by which trails facilitate dispersal are not well explored. The presence of trails alters the composition of such areas, with modified habitat conditions and physical disturbances present, while also containing heightened human and animal movement (Underwood et al., 2004). Species with specific epizoochorous dispersal mechanisms, including non-native species, are likely to be transported by humans and animals, such as dogs, through activities such as hiking and mountain biking, with trails acting as pathways into forest interiors (Pickering, 2022). Non-native plant species richness slightly increases with proximity to high-use forest trails (Liedtke et al., 2020). Despite this, specific mechanisms by which these trails facilitate the dispersal of non-native plant species remain unknown. Few studies have explored the effect of frequency of use on these trails on dispersal and establishment rates, with little knowledge of the differences between high-use and low-use trails, respectively. Understanding what role trails and human traffic play in the dispersal of non-native species is critical to understanding non-native species dynamics in these protected areas.

Given the potential dispersal-related impact that trails may have, we investigated **whether the frequency of recreational trail use by humans and dogs had an effect on the zoochorous dispersal of non-native plant species in protected areas**. We hypothesized that anthropogenic trail use facilitates the zoochorous dispersal of non-native species, and predicted that trails with high use by humans and dogs would exhibit greater richness and abundance of non-native species further from trailheads when compared to low-use trails – a *human vector* hypothesis (**Fig. 1**). Additionally, we expected that high-use trails would show greater proportions of zoochorous-dispersed non-native species farther from the trailhead when compared to low-use trails (**Fig. 2**). Last, we predicted that non-native species composition at sites as represented in ordination space would initially overlap between trails at the first few quadrats, due to edge effects and proximity to trailhead, but diverge with distance from trailhead (**Fig. 3**). Our null hypothesis was that frequency of trail use does not impact zoochorous dispersal of non-native species. Insight into this relationship will help us further understand the relationship between trail use frequency and the dispersal of non-native species, which can help inform future conservation strategies.

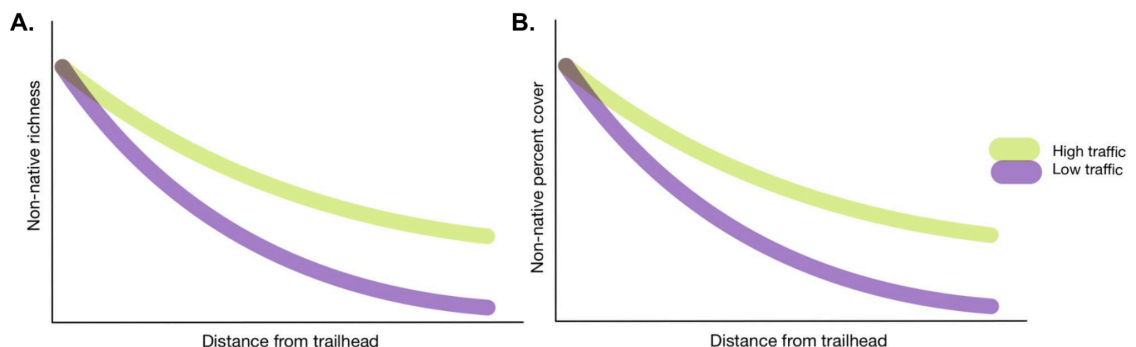


Figure 1. Non-native species richness and percent cover are predicted to decrease with distance from trailhead, with high-use trails maintaining higher non-native richness and percent cover due to greater human-mediated dispersal. Predictions for non-native richness (**left**) and percent cover (**right**) along distance from trailhead under the *human vector* hypothesis, with colour representing trail use.

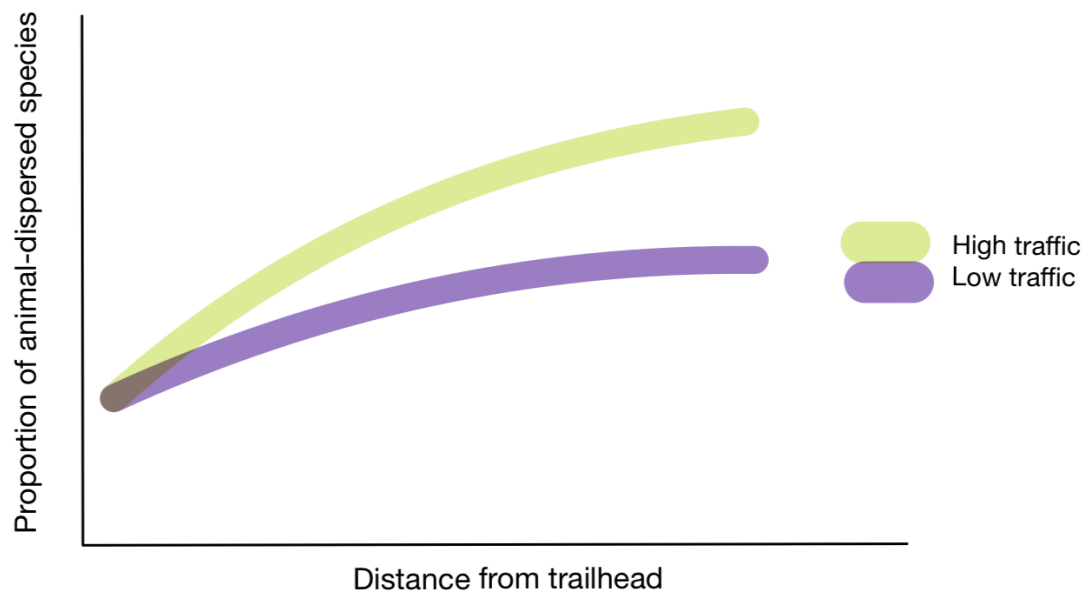


Figure 2. Proportion of animal-dispersed species within the immediate plot species pool is predicted to increase with distance from trailhead, and be higher in high-use trails, due to greater human- and animal-mediated dispersal of zoochorous dispersed species. Colour represents trail use.

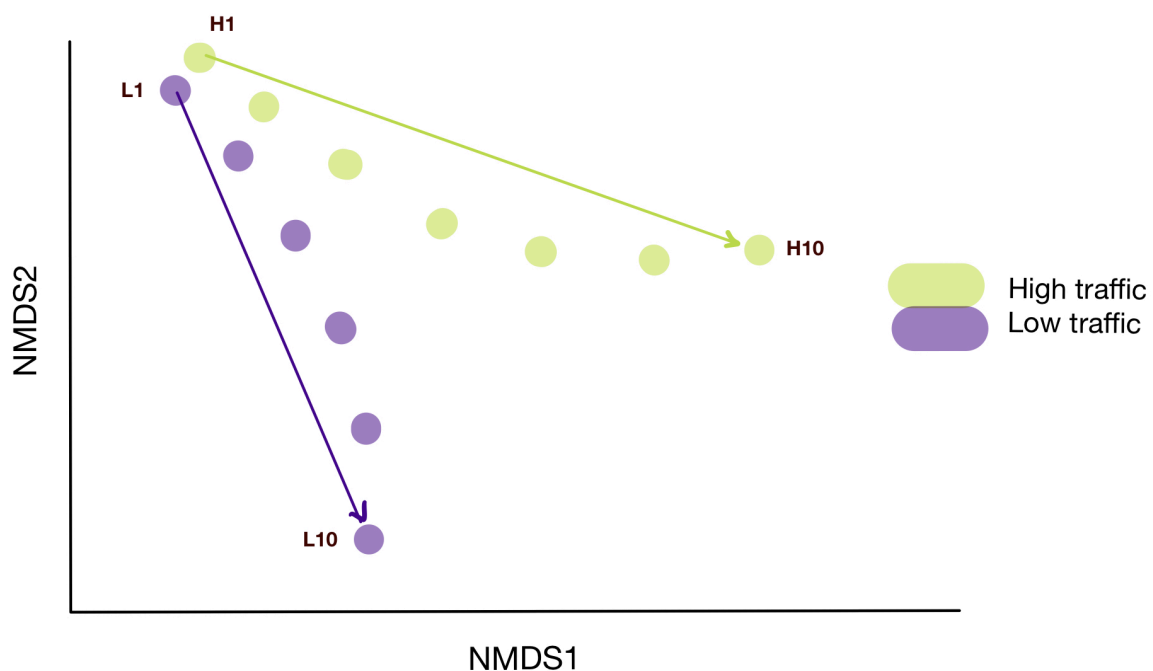


Figure 3. Non-native species composition is predicted to initially overlap across trail use due to edge effects and proximity to trailhead, but diverge with distance from trailhead due to differences in dispersal with differences in trail use. Divergence is represented with arrows and colours represent trail use.

2. Methods

Study site

Pacific Spirit Park is a temperate second-growth forested area located in Vancouver, British Columbia. It is the traditional, ancestral, and unceded territory of the hə́n̓q̓əmin̓ə́m-speaking xʷməθkʷə́yəm, who have stewarded this land since times immemorial. Despite ongoing colonial pressures, including the establishment of Pacific Spirit Park itself, xʷməθkʷə́yəm continue to assert their rights and responsibilities to this land. The 763-hectare park contains a network of trails mainly used for various recreational activities such as hiking, biking, and dog-walking. The forest contains many native tree species such as douglas fir (*Pseudotsuga menziesii*), western redcedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*) (GVRD Parks, 1991). Non-native plant species that are present in this park which were included in our study were lesser periwinkle (*Vinca minor*), creeping buttercup (*Ranunculus repens*), spanish bluebell (*Hyacinthoides hispanica*), english ivy (*Hedera helix*), himalayan *Rubus armeniacus*, english laurel (*Prunus laurocerasus*), common nipplewort (*Lapsana communis*), english holly (*Ilex aquifolium*), and cutleaf blackberry (*Rubus laciniatus*). This particular study site was chosen due to the high frequency of human activity within the park's trails, and its close proximity to developed areas.

Trail description

We selected the Top and Sasamat trails using Strava Heat Maps to visualize relative trail use (<https://www.strava.com/maps/global-heatmap>). We designated Sasamat trail as a high-use trail and Top trail as a low-use trail based on relative

estimates of activity along each trail. We identified these trails due to their close locations and shared environmental conditions, controlling for differences in environmental conditions between the two as much as possible.

Field Methods

All fieldwork was conducted on March 10th, 2025, under overcast weather with light rain. We set up 250-meter long transects with 5 x 5 meter quadrats located at 25 meter intervals (N = 10 plots per trail), for each trail (N = 2 trails, **Fig. 4**). All quadrats were placed facing west, and distance from transect perpendicularly was constant at 1.5 metre. For each of the quadrats, percent cover of 12 non-native species listed above, non-native species richness, dominant canopy species, canopy cover, and proportion of bare ground was recorded. The percent cover for each of respective non-native species was visually estimated and averaged between two observers.



Figure 4. Map of sampled transects and quadrat locations (*left*) along Sasamat trail (high traffic) and Top trail (low traffic) in Pacific Spirit Park, BC. Strava Heat Maps (*right*) showed higher relative trail use along Sasamat trail than Top trail.

Data and code availability

All data analyzed in this study was collected and analyzed by authors using R statistical software version 4.4.3 (2025-02-28) (R Core Team, 2025), primarily using packages *brms* (v2.22.0), *lme4* (v1.1-37), *vegan* (v2.6.8) and functions from the *tidyverse* (Wickham et al., 2019). Collected data is publicly available at https://github.com/abeauche/BIOL406-2025-Final-Project/tree/main/data_cleaned and all code is available at <https://github.com/abeauche/BIOL406-2025-Final-Project/>.

Data analysis

We fit two Bayesian generalized linear regression models to evaluate the relationship between (1) non-native species richness and (2) non-native percent cover with distance from the trailhead. The model for species richness was fit with a Poisson distribution after testing for overdispersion and run with 4 chains and 2000 iterations, with a warmup of 1000 iterations, and the model for percent cover was fit with a Zero-Inflated Beta distribution to account for the presence of zeros in the dataset and run with 4 chains and 4000 iterations, with a warmup of 2000 iterations. Both models were specified with weakly informative priors centered on zero and thinning of 1. We first attempted running the model with a random slope term for traffic, but the models encountered too many divergent transitions and we removed the term. Convergence diagnostics (\hat{R} = 1.0) confirmed good mixing for both. We determined that effect sizes with 95% credible intervals excluding 0 were strongly supported by our data.

We then completed a literature review to identify key dispersal syndromes of the non-native species we identified in the field. Species were annotated for any of animal dispersed, wind dispersed, water dispersed, and sprawling, with possibility for multiple

dispersal syndromes. Species without any reported literature syndromes were marked as “unspecified,” with syndromes reported in **Table B3**. We identified all non-native species reported in a plot, and estimated the proportion of species documented to be animal-dispersed to some extent for each plot in relation to the total number of non-native species observed at each plot. Data were visualized and modelled using a generalized linear model with binomial family predicting number of animal-dispersed species within total non-native species count according to distance from trailhead and traffic, with tests for assumption of non-overdispersed data.

Last, we completed an NMDS ordination of non-native vegetation community structure at each site using the *vegan* package in R and Bray-Curtis distance. Stress for ordination with $k = 2$ dimensions was 0.075 so it was decided to carry out the ordination along two axes. We decided not to complete further statistical analysis on this due to limited data. Tables with all model outputs are reported in the Appendix.

3. Results

Non-native species richness decreased with distance (**Fig. 5A**; slope = -0.01, CI = -0.01, 0.00) and was significantly lower with high trail use (**Fig. 5A**; effect = -1.36, CI = -2.11, -0.62), with credible intervals narrowly excluding zero, indicating strong support in the magnitude and direction of the effect sizes (**Table B1**). The zero-inflation component of percent cover – representing presence-absence – decreased with distance (**Fig. 5B**; slope = -0.04, CI = -0.10, -0.01) and high trail use (**Fig. 5B**; effect = -9.98, CI = -19.49, -3.81), indicating strong support for a negative effect of the two on the likelihood of presence of non-native cover at each plot, with credible intervals again excluding zero

(**Table B2**). However, there was no detected effect of either distance (slope = 0.00, CI = -0.01, 0.01) or trail use (effect = 0.23, CI = -1.44, 1.79) in the beta (continuous) component of this model, reflecting more uncertainty about the relationship with percent cover. The proportion of animal-dispersed non-native species was not different between high and low traffic trails (**Fig. 6**; estimate = 1.170 ± 1.002 , $z = -1.168$, $p = 0.243$), and did not change with distance from the trailhead (**Fig. 6**; estimate = 0.00361 ± 0.00526 , $z = 0.686$, $p = 0.493$), with model outputs reported in **Table B4**. Last, qualitative assessment showed non-native species assemblages clustered differently in NMDS ordination space along high- and low-use trails, and diverged with distance from the trailhead (**Fig. 7**).

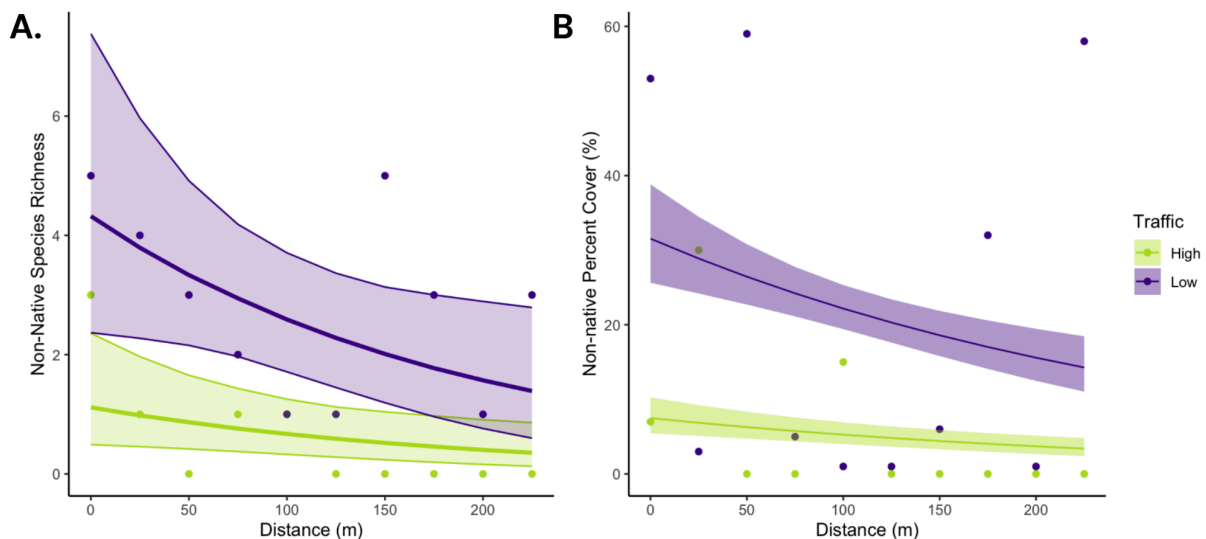


Figure 5. Non-native richness and percent cover decreased with distance from trailhead and increased trail use, as modelled using Bayesian generalized linear models with (1) Poisson and (2) Zero-Inflated Beta families. Panel (A) shows predicted values for non-native richness along distance from trailhead (slope = -0.01, CI = -0.01, 0.00), with effect of trail traffic (effect = -1.36, CI = -2.11, -0.62) represented by colour, with 95% credible intervals plotted as the shaded area (**Table B1**). **Panel (B)** shows predicted values for non-native percent cover along distance from trailhead, with effect of trail traffic represented by colour, as represented using a general linear regression model, with 95% confidence intervals plotted as the shaded area. This relationship was evaluated using a Bayesian general linear model with a Zero-Inflated Beta family. While distance from trailhead (slope = -0.04, CI = -0.10, -0.01) and high trail use (effect = -9.98, CI = -19.49, -3.81) had a negative effect on the likelihood of occurrence of non-native cover at a plot in the zero-inflation component, no effect was detected in the

Beta (continuous) component (**Table B2**). Data were collected from 10 5×5 m plots along each of Top trail (low-use) and Sasamat trail (high-use) in Pacific Spirit Park, BC, on March 10, 2025 (N = 2 trails).

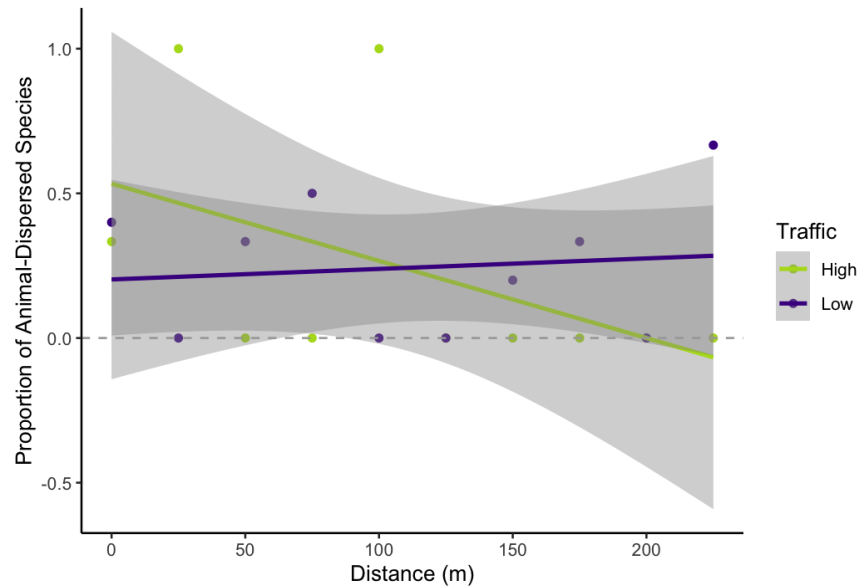


Figure 6. Proportion of zoochorous-dispersed non-native species in a plot-level species pool showed no trends as distance from trailheads increased (estimate = 1.170 ± 1.002 , $z = -1.168$, $p = 0.243$), **or with increased trail use** (estimate = 0.00361 ± 0.00526 , $z = 0.686$, $p = 0.493$). Proportion was calculated as the number of non-native species with reported animal dispersal from the total number of species observed at a plot. This relationship was modelled using a generalized linear model with binomial family, with outputs reported in **Table B4**. Data were collected from 10 5×5 m plots along each of Top trail (low-use) and Sasamat trail (high-use) in Pacific Spirit Park, BC, on March 10, 2025 (N = 2 trails), and identified dispersal syndromes are reported in **Table B3**.

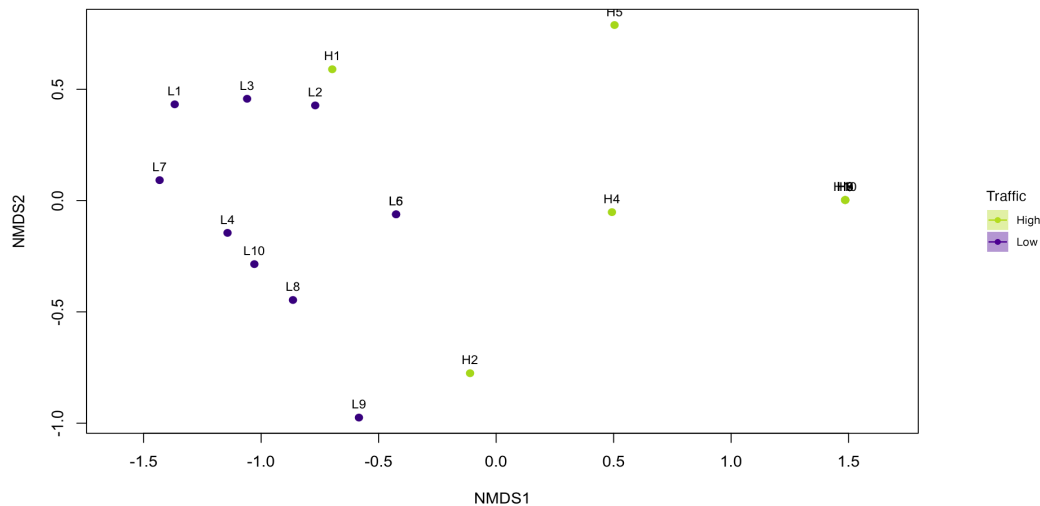


Figure 7. Non-native species assemblages clustered differently in NMDS ordination space along high- and low-use trails, and diverged with distance from the trailhead. NMDS ordination was carried out with Bray-Curtis dissimilarity and $k = 2$. Points are labelled as L (low) or H (high) for trail use, with a number representing their sequential order from trailhead. Data were collected from 10 5×5 m plots along each of Top trail (low-use) and Sasamat trail (high-use) in Pacific Spirit Park, BC, on March 10, 2025 (N = 2 trails).

4. Discussion

This study tested the hypothesis that human recreation along trails facilitates the dispersal of non-native species. This hypothesis was generally supported by our findings, which, along with a study conducted by Liedtke et al. (2020), found a gradual drop-off of non-native species richness and abundance the further along a trail, consistent with a dispersal kernel (**Fig. 5**). The high- and low-use trails also clustered differently in NMDS, which indicated a divergence in community structure between high- and low-use trails, as expected under our hypothesis (**Fig. 7**). However, the other predictions that arose from our hypothesis were not supported by our models.

Non-native species richness was expected to be significantly lower in low-use trails as compared to high-use ones, simply because there were less potential human vectors using human trails (Underwood et al., 2004). Contrary to our expectations, we found that non-native species richness was actually higher on low-use trails. This likely resulted from increased disturbance in the form of trampling beside high-use trails, which may have prevented the establishment of non-native species that show species-specific responses to trampling (Chardon et al., 2023). This trampling may have been human-caused, or it could be a result of the many off-leash dogs we observed on the trail while we were sampling. With how species-specific each response to trampling is, the available niche in a highly trampled area is likely to narrow as species from the full regional pool are filtered out, which reduces richness (Pescott & Stewart, 2014).

Non-native species percent cover was also expected to be lower in low-use areas, yet also showed the opposite relationship in our models. Once again, trampling was likely the driver that reduced percent cover, as it disturbed available substrate for

plants to establish in, and caused direct tissue damage to any plants that would have established (Liddle, 1975). Quadrats along the high-use trail were observed to be quite bare, and many showed disturbed, muddy substrate from dog and human traffic – which is likely reflected in the effect of trail use on likelihood of presence of any non-native cover at a site, and may have contributed to the low number of sites with continuous values for non-native percent cover along the high-use trail.

No significant relationship was found between the proportion of zoochorous non-native plants and the distance of their presence along each trail. We expected this proportion to increase further along a trail, as the distance into the forest that can't be covered by human-mediated dispersal would filter out non-zoochorous non-native species. We expected this slope to be steeper for high-use trails, due to the increased human-mediated zoochorous dispersal associated with higher traffic. The lack of these relationships in our data indicates that if any effect of human-mediated dispersal on species composition and functional diversity exists, it's a small one. Pickering (2022) and Johnson et al. (2020) found that the traits of seeds themselves influences how well they adhere to human vectors, which indicates that one would expect the proportion of seeds with epizoochorous traits to increase if human vectors were playing a significant role. The fact that we missed this pattern likely indicates that we were limited by the relatively small distance we examined and the two trails we looked at.

Last, visual assessment of NMDS clustering showed support for differences in non-native species assemblage between the two trails and with distance from trailhead, suggesting potential directions for further research. In the case of this study, this difference in clustering is likely driven by the sites along the high-use trail consistently

being entirely bare ground, with minimal non-native species present. Nevertheless, the directionality of points along a distance from trailhead gradient supported that sites closer to each other were more similar in non-native assemblage than sites more distant from each other, which could provide a valuable direction for future research.

Future research and limitations

A key limitation of this study was the limited number of trails sampled. Because we were only able to sample two trails, it is likely that our results were biased by differences in invasive species management between the trails or differences in trail maintenance, and thus that our findings primarily reflect these and not a broader impact of trail use on non-native species dynamics. Additionally, identifying and measuring percent cover for all species in each quadrat and not only non-native species would allow for a more exhaustive understanding of the role of dispersal syndrome in shaping patterns of species distributions along recreational trails in protected area.

Future research should focus on implementing a broader gradient of trail use frequency, which would provide more insight into the relationship present. Increasing the number of trails used in the study and using a randomized block design may provide a broader gradient of traffic. Additionally, accounting for potentially confounding variables such as differences in management practices and history of disturbances would help generalize the specific impact that trail use frequency may have on non-native dispersal. Finally, future studies should consider temporal variation; repeated experiments across time periods are needed to account for possible phenological differences in plant species. Addressing the limitations of our study will provide a

greater understanding of the specific mechanisms that drive non-native species dispersal in protected areas.

5. Conclusion

Our study provided us with an improved understanding of how anthropogenic use can impact non-native species dispersal. However, it is important to note that the active management of these species can make the study of these dynamics a bit challenging in situ. With the active management along with the limitations mentioned previously, there remains a gap in our knowledge regarding the specific impact that anthropogenic use has on plant communities and plant composition. As recreation increases, identifying these mechanisms through future studies is crucial towards improving our overall understanding of ecological processes, as well as for conservation and management practices, to ensure that biodiversity is maintained in these protected areas.

References

- Blackburn, T. M., Pyšek, P., Bacher, S., Carlton, J. T., Duncan, R. P., Jarošík, V., Wilson, J. R. U., & Richardson, D. M. (2011). A proposed unified framework for biological invasions. *Trends in Ecology & Evolution*, 26(7), 333–339.
<https://doi.org/10.1016/j.tree.2011.03.023>
- Chapman, D., Purse, B. V., Roy, H. E., & Bullock, J. M. (2017). Global trade networks determine the distribution of invasive non-native species. *Global Ecology and Biogeography*, 26(7/8), 907–917.
- Chardon, N. I., Stone, P., Hilbert, C., MacLachlan, T., Ragsdale, B., Zhao, A., Goodwin, K., Collins, C. G., Hewitt, N., & Elphinstone, C. (2023). Species-Specific Responses to Human Trampling Indicate Alpine Plant Size Is More Sensitive than Reproduction to Disturbance. *Plants*, 12(17), 3040. <https://doi.org/10.3390/plants12173040>
- GVRD Parks. (1991). *Pacific Spirit Regional Park management plan*.
<https://metrovanancouver.org/services/regional-parks/Documents/pacific-spirit-regional-park-management-plan.pdf>
- Jeschke, J. M., Bacher, S., Blackburn, T. M., Dick, J. T. A., Essl, F., Evans, T., Gaertner, M., Hulme, P. E., Kühn, I., Mrugała, A., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D. M., Sendek, A., Vilà, M., Winter, M., & Kumschick, S. (2014). Defining the Impact of Non-Native Species. *Conservation Biology*, 28(5), 1188–1194.
- Johnson, M. K. A., Johnson, O. P. J., Johnson, R. A., & Johnson, M. T. J. (2020). The role of spines in anthropogenic seed dispersal on the Galápagos Islands. *Ecology and Evolution*, 10(3), 1639–1647. <https://doi.org/10.1002/ece3.6020>
- Liddle, M. J. (1975). A selective review of the ecological effects of human trampling on natural ecosystems. *Biological Conservation*, 7(1), 17–36.
[https://doi.org/10.1016/0006-3207\(75\)90028-2](https://doi.org/10.1016/0006-3207(75)90028-2)

- Liedtke, R., Barros, A., Essl, F., Lembrechts, J. J., Wedegärtner, R. E. M., Pauchard, A., & Dullinger, S. (2020). Hiking trails as conduits for the spread of non-native species in mountain areas. *Biological Invasions*, 22(3), 1121–1134.
<https://doi.org/10.1007/s10530-019-02165-9>
- Pescott, O. L., & Stewart, G. B. (2014). Assessing the impact of human trampling on vegetation: A systematic review and meta-analysis of experimental evidence. *PeerJ*, 2, e360.
<https://doi.org/10.7717/peerj.360>
- Pickering, C. (2022). Mountain bike riding and hiking can contribute to the dispersal of weed seeds. *Journal of Environmental Management*, 319, 115693.
<https://doi.org/10.1016/j.jenvman.2022.115693>
- Ricciardi, A., Hoopes, M. F., Marchetti, M. P., & Lockwood, J. L. (2013). Progress toward understanding the ecological impacts of nonnative species. *Ecological Monographs*, 83(3), 263–282.
- Underwood, E. C., Klinger, R., & Moore, P. E. (2004). Predicting Patterns of Non-Native Plant Invasions in Yosemite National Park, California, USA. *Diversity and Distributions*, 10(5/6), 447–459.

Appendix

Table B1. Model outputs for a Bayesian generalized linear model with Poisson family predicting non-native richness along distance from trailhead, with a term for trail traffic. 95% credible intervals for both the distance and traffic term excluded 0. The model was specified with weakly informative priors and run with 4 chains and 2000 iterations, with a warmup of 1000 iterations, and thinning by 1. Model output table produced using *knitr::kable()*.

Parameter	Estimate	Est.Error	l.95..CI	u.95..CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.10	0.41	-0.71	0.86	1	2185	2724
distance_m	-0.01	0.00	-0.01	0.00	1	3443	2714
TrafficLow	1.36	0.38	0.62	2.11	1	1684	2086

Table B2. Model outputs for a Bayesian generalized linear model with Zero-Inflated Beta family predicting non-native percent cover along distance from trailhead, with a term for trail traffic. All parameter estimates in the Beta component had 95% credible intervals overlapping zero, indicating considerable uncertainty in the direction and magnitude of these effects. In contrast, zero-inflated parameters for both distance and traffic had credible intervals that excluded zero, suggesting stronger evidence for their effects on the probability of excess zeros (complete absence of non-native cover). The model was specified with weakly informative priors and run with 4 chains and 4000 iterations, with a warmup of 2000 iterations, and thinning by 1. Model output table produced using *knitr::kable()*.

Parameter	Estimate	Est.Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	-1.06	0.72	-2.49	0.34	1.00	4032	2949
zi_Intercept	-1.99	1.72	-5.94	1.01	1.00	2508	1741
distance_m	0.00	0.01	-0.01	0.01	1.00	2958	3033
TrafficLow	-0.23	0.83	-1.79	1.44	1.00	2459	2825
zi_distance_m	0.04	0.02	0.01	0.10	1.00	1669	1185
zi_TrafficLow	-9.98	4.09	-19.49	-3.81	1.01	1034	1492
phi	2.69	1.09	1.07	5.31	1.00	2839	2571

Table B3. Non-native species dispersal syndromes as reported in literature. Species are labelled based on data collection label, and dispersal syndrome is listed as column name. A 1 in a column indicates that the species has been reported to have the corresponding dispersal syndrome. Table produced using *knitr::kable()*.

species	animal	wind	water	sprawling	unspecified
spanish_bluebell_cover	0	1	1	0	0
rumex_cover	0	0	0	0	1
creeping_buttercup_cover	1	1	1	1	0
lesser_periwinkle_cover	1	0	0	1	0
english_ivy_cover	1	0	0	1	0
grass_cover	0	0	0	0	1
himalayan_blackberry_cover	1	0	0	1	0
english_laurel_cover	1	0	0	1	0
nipplewort_cover	0	0	0	0	1
english_holly_cover	1	0	0	0	0
cutleaf_blackberry_cover	1	0	0	1	0

Table B4. Model outputs for a generalized linear model with binomial family predicting number of animal-dispersed species within total non-native species count per plot along distance from trailhead, with a term for trail traffic. Model output table produced using *knitr::kable()*.

Term	Estimate	Std. Error	z value	Pr(>)
(Intercept)	-0.120	0.837	-0.143	0.886
distance_m	0.00361	0.00526	0.686	0.493
TrafficLow	-1.170	1.002	-1.168	0.243

- **Null deviance:** 13.023 on 13 df
- **Residual deviance:** 11.562 on 11 df
- **AIC:** 29.346
- **Fisher Scoring iterations:** 4