

Plant removal across an elevational gradient marginally reduces rates, substantially reduces variation in mineralization

KENNA E. REWCASTLE ^{1,2,3,8} JEREMIAH A. HENNING ^{3,4} QUENTIN D. READ ^{3,5} REBECCA E. IRWIN,^{3,6}
 NATHAN J. SANDERS ^{3,7} AND AIMÉE T. CLASSEN ^{3,7}

¹*Rubenstein School of Environment and Natural Resources, University of Vermont, 81 Carrigan Dr., Burlington, Vermont 05405 USA*

²*Gund Institute for Environment, University of Vermont, 210 Colchester Ave., Burlington, Vermont 05405 USA*

³*Rocky Mountain Biological Laboratory, P.O. Box 519, Crested Butte, Colorado 81224 USA*

⁴*Department of Biology, University of South Alabama, 5871 USA Dr. N, Mobile, Alabama 36688 USA*

⁵*National Socio-Environmental Synthesis Center (SESYNC), 1 Park Pl., Annapolis, Maryland 21401 USA*

⁶*Department of Applied Ecology, North Carolina State University, Campus Box 7617, Raleigh, North Carolina 27695 USA*

⁷*Department of Ecology and Evolutionary Biology, University of Michigan, 1105 North University Ave., Ann Arbor, Michigan 48109 USA*

Citation: Rewcastle, K. E., J. A. Henning, Q. D. Read, R. E. Irwin, N. J. Sanders, and A. T. Classen. 2022. Plant removal across an elevational gradient marginally reduces rates, substantially reduces variation in mineralization. *Ecology* 103(1):e03546. 10.1002/ecy.3546

Abstract. The loss of aboveground plant diversity alters belowground ecosystem function; yet, the mechanisms underpinning this relationship and the degree to which plant community structure and climate mediate the effects of plant species loss remain unclear. Here, we explored how plant species loss through experimental removal shaped belowground function in ecosystems characterized by different climatic regimes and edaphic properties. We measured plant community composition as well as potential carbon (C) and nitrogen (N) mineralization and microbial extracellular enzyme activity in soils collected from four unique plant removal experiments located along an elevational gradient in Colorado, USA. We found that, regardless of the identity of the removed species or the climate at each site, plant removal decreased the absolute variation in potential N mineralization rates and marginally reduced the magnitude of N mineralization rates. While plant species removal also marginally reduced C mineralization rates, C mineralization, unlike N mineralization, displayed sensitivity to the climatic and edaphic differences among sites, where C mineralization was greatest at the high elevation site that receives the most precipitation annually and contains the largest soil total C pool. Plant removal had little impact on soil enzyme activity. Removal effects were not contingent on the amount of biomass removed annually, and shifts in mineralization rates occurred despite only marginal shifts in plant community structure following plant species removal. Our results present a surprisingly simple and consistent pattern of belowground response to the loss of dominant plant species across an elevational gradient with different climatic and edaphic properties, suggesting a common response of belowground ecosystem function to plant species loss regardless of which plant species are lost or the broader climatic context.

Key words: biodiversity loss; carbon mineralization; elevational gradient; nitrogen mineralization; plant removal; plant–soil linkages.

INTRODUCTION

Loss of plant species from communities will impact the structure and function of ecosystems as interactions among plant species, evolutionary dynamics such as competition and facilitation that shape plant communities, and plant–soil linkages shift in response to diminished biodiversity (Pugnaire et al. 2019). Ecologists often explore the impact of plant species loss on ecosystem function by building correlations between ecosystem

functions and aspects of plant community structure such as dominant plant functional type, functional traits, species relative abundances, and the diversity of the remaining plant community (Symstad et al. 1998, Lyons and Schwartz 2001, Gilman et al. 2010, Ockendon 2014). However, the properties of communities or ecosystems that determine ecosystem sensitivity to changes in plant communities or to abiotic changes remain unclear (Klanderud 2005, Gilman et al. 2010, Wardle et al. 2011, Adler et al. 2012). Furthermore, we know little about the sensitivity of belowground soil processes to changes in species interactions relative to aboveground ecosystem functions (Zak et al. 2003). Despite ample research defining the relationship between species richness and ecosystem function in experimental and observational frameworks

Manuscript received 30 April 2021; accepted 27 May 2021; final version received 14 September 2021. Corresponding Editor: Pamela H. Templer.

⁸E-mail: Kenna.Rewcastle@uvm.edu

(Tilman and Wedin 1996, Grace 2016), we also know that the loss of individual species from a plant community can have a different impact on the biodiversity–ecosystem function relationship than is captured by studies that assess this relationship in the direction of increasing diversity across a biodiversity gradient (Naeem et al. 1995, Fox and Kerr 2012, Kardol and Fanin 2018). Plant removal experiments, especially when conducted across an environmental gradient, can be used as a tool to address many of these problems that arise when attempting to quantify the impact of losing a plant species on ecosystem function (Sundqvist et al. 2013).

Some plant species have a disproportionately large impact on soil processes due to their dominance (Smith and Knapp 2003, Avolio et al. 2019), the uniqueness of their traits in a community, or their longevity on the landscape, i.e., long-lived perennial species. The loss of these influential plant species from a community should have a greater impact on ecosystem processes than the loss of a plant species with a less pronounced presence in the community (Tilman et al. 1997, Chapin 1998, McLaren and Turkington 2010). In short, the identity of the plant species that is lost from an ecosystem should matter when predicting the impact of species loss on belowground processes (Wardle et al. 1999, Johnson and Phoenix 2008). In a 2003 review, Díaz et al. proposed three mechanisms as pathways by which plant species removal impacts ecosystem function as the loss of a single plant species shifts species interactions: (1) Ecosystems could respond to the loss of the specific functional role filled by the removed species, (2) changes to ecosystem function could be a response to the reassembly of the remaining community members following the loss of a species, and (3) changes to ecosystem function could be a response to the disturbance of the removal treatment itself through loss of aboveground biomass. By incorporating treatments that account for each of these mechanisms of plant species removal impact, removal experiments are uniquely positioned as an experimental framework to discern the pathways by which removing a single plant species can impact belowground ecosystem function.

Environmental conditions such as growing season temperature and precipitation modulate the relationship between above- and belowground ecosystem components, meaning that the loss of the same plant species could have a starkly different impact on belowground processes in ecosystems characterized by different climates (Klanderud and Totland 2005, Brooker 2006, Bardgett et al. 2013). Changes in climate as well as the associated environmental stress alter both the direction and magnitude of plant–soil feedbacks (van der Putten et al. 2016, Baert et al. 2018). The nature of this feedback effect is highly variable and depends on the context of both present environmental conditions and the legacy of historical climate (Kaisermann et al. 2017). The soil microbial communities involved in carbon (C) and nitrogen (N) mineralization, the nutrient recycling processes

in terrestrial ecosystems, may also be limited by different abiotic factors than is the aboveground community. Thus, global change drivers may decouple above- and belowground linkages as plant and microbial communities respond differently to environmental change (Wardle et al. 2013, Classen et al. 2015).

Taken together, the effect of losing a plant species from a community on belowground ecosystem processes likely depends on the functional identity of the plant species removed, climatic context, and the degree to which plant community structure shifts following the species loss (Díaz et al. 2003, McLaren and Turkington 2010, Wardle et al. 2011, Pugnaire et al. 2019). However, the influence of the combination of factors described above on the relationship between plant species removal and belowground ecosystem processes has yet to be empirically tested using field experiments. Therefore, to understand the biotic and abiotic variables that shape the impact of species loss on potential C and N mineralization, we sampled a series of plant removal experiments established at different sites across an elevational gradient. We predicted that the effect of removing a dominant plant species would vary across the environmental gradient, with the greatest decrease in belowground mineralization rates following plant removal occurring at the lowest elevational site where moisture limitation, i.e., environmental stress, is the highest (García-Palacios et al. 2018, Pugnaire et al. 2019). We also hypothesized that the impact of plant species removal on belowground processes would scale with the amount of biomass removed from each plot (Díaz et al. 2003, McLaren and Turkington 2010). Our results lend insight to how the loss of species from communities might alter important nutrient cycles in climatically different ecosystems through the restructuring of plant communities and ecosystem function.

MATERIALS AND METHODS

To investigate the impact of removing a plant species on C and N mineralization, we sampled four existing, independent plant removal experiments located at four locations along an elevational gradient near the Rocky Mountain Biological Laboratory, Gothic, Colorado, USA. Each of the four experiments removed different focal plant species, and the experiments have been running from 5 to 17 years. A dominant plant species was removed at each site along the elevational gradient, but, because there is nearly complete turnover of the plant community across the elevational gradient, the identity of the removed species was different at each elevation, confounding the functional type of the removed species as well as environmental conditions across the elevational gradient.

At the low elevation site, located at 2,740 m above sea level (asl; 38.71° N, 106.82° W), we removed the dominant plant species *Wyethia* × *magna* (Asteraceae), a perennial forb that is a stable hybrid of *Wyethia amplexicaulis* and *W. arizonica* (Weaver 1915), by clipping

aboveground biomass to the soil surface annually for 5 yr prior to sampling. Plots at this site were 2.0 × 2.0 m in area ($n = 8$). Soils at the low elevation site were classified as Mayoworth loam. At the mid-low elevation site, located at 2,890 m asl (38.95° N, 106.98° W), we maintained a 17-yr removal of *Linaria vulgaris* (Plantaginaceae), a perennial, invasive forb with an extensive horizontal root system, in 2.0 × 2.0 m plots ($n = 6$) (Wilke and Irwin 2010). Plants were removed from the mid-low elevation site by gently pulling on the base of the plant to remove the aboveground biomass and a small portion of the *L. vulgaris* root mat. Soils at the mid-low elevation site were classified as Tine sandy loam and Bassel sandy loam, depending on the slope of each plot. At the mid-high elevation site, we removed *Festuca thurberi* (Poaceae), a perennial grass that forms shallow but dense root systems, by clipping to the soil surface annually for 7 yr prior to sampling. Removal treatments at the mid-high elevation site (2,904 m asl, 38.94° N, 106.99° W) were applied in 1.5 × 1.5 m plots ($n = 4$; Read et al. 2018, Henning et al. 2019), with soils at this site classified as Leaps silty clay loam. In the *W. magna* and *F. thurberi* removals at the mid-high and low elevation sites, a small amount of glyphosate herbicide was applied to the remaining base of the stems after the aboveground biomass was removed to attempt to kill belowground biomass. Herbicide was only applied to stem bases at these sites for the first 2 yr of the removal treatments, and because of the meticulous way in which it was applied to the base of the remaining stems following removal using a paintbrush, we are confident that herbicide application had negligible effects on the fitness of the other plant species. At the high elevation site (3,460 m asl, 38.99° N, 107.06° W), we removed the dominant species *Juncus drummondii* (Juncaceae), a perennial rush that grows in thick bunches, by clipping aboveground biomass to the soil surface annually for

5 yr before sampling. Plots at this site covered an area of 2.0 × 2.0 m ($n = 8$). Soils at the high elevation site were classified as Moran-Rubble land complex, characterized by extremely gravelly loam/sandy loam. Roots of the removed plants species were left intact at all sites that used clipping to remove aboveground biomass in removal treatments (low, mid-high, and high elevation sites) in order to minimize disturbance caused by the removal treatment for the remaining plant species and the soil community. While *L. vulgaris* individuals were removed by tugging gently on the base of the stem, removing some root biomass in the process, the majority of the root runners that extend horizontally just beneath the soil surface remained in the soil.

In addition to the control (i.e., no plant biomass removed from plots with the focal species naturally present) and individual species removal treatments present at all sites, the mid-low and mid-high elevation sites had an additional random biomass removal treatment where biomass from random plant species was removed to reflect the amount of biomass that was removed annually from the treatments in which a specific plant species was removed. This treatment was intended to isolate the disturbance effect that plant removal has on the remaining plant community through loss of aboveground biomass. Finally, a natural control treatment, i.e., plots where the focal plant species was naturally absent, existed at the mid-low elevation site. Table 1 summarizes site-level experimental design, climate data for the 2018 growing season, relative abundance of the focal species at each site, and the average amount of biomass removed annually from removal treatment plots at each site.

During the peak of the 2018 growing season, when aboveground biomass was greatest at each site, we conducted plant community surveys by visually assessing the percentage of the plot area covered by each plant species. Rare species were recorded as covering <1%. At

TABLE 1. Focal species removed, experimental design, and climate variables for each of the four field experiments included in this study.

Site	Length of experiment (yr)	Plant species removed	Replication (n)	Elevation (m above sea level)	Growing season mean			Mean		
					Temperature (°C)	Precipitation (mm)	Soil moisture (VWC, %)	Relative abundance of focal species (%)	Biomass removal rate (dry g/yr)	Soil total C (g C/kg dry soil)
Low elevation	5	<i>Wyethia × magna</i>	8	2,740	15.6	61.4	3.61	19.9	173.02	111.6
Mid-low elevation	17	<i>Linaria vulgaris</i>	6	2,890	12.1	98.3	9.67	19.8	–	109.4
Mid-high elevation	7	<i>Festuca thurberi</i>	4	2,904	–	–	–	29.5	29.51	104.2
High elevation	5	<i>Juncus drummondii</i>	8	3,460	12.2	127.6	7.71	19.1	52.49	153.5

Notes: Both mid-elevation sites are located in close proximity to one another and therefore share climate data. Collection of growing season climate data for all sites began on 1 June 2018, and ended on 31 August 2018, with the exception of the high-elevation field site where the weather station was installed on 27 June at the beginning of the growing season. The relative abundance of the focal species targeted in removal treatments at each site was calculated by dividing the percentage of the plot area covered by the focal species in control plots by the total area covered by all plant species in each control plot. Data quantifying the amount of biomass removed annually from removal treatments at the mid-low elevation site were not collected. Climate data for the mid-elevation sites are made available through the Rocky Mountain Biological Laboratory (RMBL, see Acknowledgments).

the low, mid-high, and high elevation sites, data were collected annually on the amount of plant biomass removed from each plot that received a removal treatment by collecting removed biomass in paper bags and drying the plant biomass for 48 h at 60°C. We calculated the average amount of biomass removed annually from each plot by pooling the total amount of biomass removed over the entire length of the experiment at each site, and dividing this cumulative number by the length of each experiment in years (Table 1). On 23–25 July 2018, we took two separate soil cores (<15 cm deep), one for the potential mineralization incubation and one for extracellular enzyme assays, from random locations within each plot. Soil samples were refrigerated and transported to the University of Vermont, Burlington, Vermont, USA for processing. We sieved each sample to remove rocks and large plant material (>2 mm) and measured gravimetric water content (g H₂O/g dry soil) by drying 10 g of field-moist soil in an oven at 105°C for 48 h (Jarrell et al. 1999). We measured soil organic matter content using the loss on ignition method (SOM-LOI) by combusting oven-dried soil samples (105°C) at 550°C for 6 h, and then measuring SOM-LOI (g C/kg dry soil) by quantifying the mass difference between oven-dried and combusted soil samples (Hoogsteen et al. 2015).

To measure the effect of plant removal on potential C and N mineralization in soils under standardized environmental conditions, we conducted a 30-d laboratory incubation of soils exposed to ideal moisture and temperature conditions (30% volumetric water content, 20°C) (Robertson et al. 1999). Thirty percent volumetric water content was the maximum soil moisture content recorded during the growing season at these four sites in the two years preceding this study, so this moisture threshold was chosen to represent ideal but realistic conditions under which we could investigate the full capacity of the microbial community in sampled soils to mineralize C and N. We divided field-moist soil samples from each plot into paired 10-g subsamples. One subsample was extracted immediately using a 2.0 mol/L KCl solution to measure extractable NH₄⁺ and NO₃⁻. We incubated the second subsample in the dark for 30 d in 1-L clear, glass jars fitted with rubber septa in the metal lid. We measured the CO₂ evolved in the headspace of the jars via direct injection by using a syringe to sample the air in each sealed jar and injecting 7 mL of air into a LI-COR 7810 trace gas analyzer (LI-COR Instruments, Lincoln, Nebraska, USA) at six time points (days 1, 2, 4, 8, 16, and 30 of the incubation) to track potential C mineralization (μg C/g dry soil⁻¹·d⁻¹). At the completion of the incubation, we extracted the incubated soil subsample with 2 mol/L KCl to again measure the concentration of extractable NO₃⁻ and NH₄⁺. We measured the concentration of NO₃⁻ and NH₄⁺ in the extractions colorimetrically (Doane and Horwath 2003) using a Synergy HT microplate fluorimeter/spectrophotometer (Synergy HT, Biotek, Winooski, Vermont, USA). We then calculated total N

mineralization rates by subtracting the sum of NO₃⁻ and NH₄⁺ (i.e., inorganic N) in the initial subsample from the sum of extractable inorganic N in the final subsample and dividing the amount of inorganic N produced during the reaction by the length of the incubation in days (mg N·kg dry soil⁻¹·d⁻¹).

In addition to a laboratory incubation of soils to measure N and C mineralization rates, we measured the activities of six different soil extracellular enzymes to understand how plant removal impacts potential microbial activity. Using the protocol established by Saiya-Cork and Sinsabaugh (2002), we assayed the carbon degrading enzymes α-glucosidase (AG), β-glucosidase (BG), cellobiohydrolase (CBH), and β-Xylosidase (XYL), nitrogen-acquiring enzyme β-N-acetylglucosaminidase (NAG), and phosphorus-acquiring enzyme acid phosphatase (PHOS). All assays were performed by incubating enzymes in a soil slurry prepared with a buffered solution (pH 5.0 sodium acetate buffer) at an ideal temperature (20.0°C) with non-limiting amounts of substrate. Following incubation, potential enzyme activities were quantified using a Synergy HT microplate spectrophotometer (Biotek).

To analyze shifts in plant community structure in response to the plant removal treatments across the elevational gradient, we performed nonmetric multidimensional scaling (NMDS) on the plant community survey data in each plot using the metaMDS function in the vegan: Community Ecology Package in R package (Oksanen 2019) with distance between plant communities in plots within the same treatment calculated according to the modified Gower (altGower) method (Anderson and Ellingsen 2006). We performed a permutational multivariate analysis of variance using the adonis function in the vegan package with two thousand permutations to quantify the extent to which the removal treatment explains dissimilarity between communities. To calculate dissimilarity among plant communities at each site and with the removal treatments, we chose to use the modified Gower method (Anderson et al. 2006; altGower in the vegdist function in the vegan package). The modified Gower method for calculating dissimilarity between communities is most appropriate for our study because it explicitly weights an order-of-magnitude change in abundance equivalent to a change in species composition. This feature is important because this analysis relies on our ability to detect changes in abundance driven by our removal treatments despite high turnover in species composition across the elevational gradient that would overwhelm changes in abundance in most other dissimilarity indices that consider abundance (Anderson et al. 2006). Finally, we used the betadisper function to evaluate the homogeneity of variances in community structure across treatment groups to understand how the removal treatments, and separately, site, affect the variation in plant community structure. This test calculates the average linear distance between individual plots within a treatment group and

the within-group centroid, which represents the median community structure for that group.

To measure the impact of plant removal on potential C and N mineralization, potential soil enzyme activity, and SOM-LOI content, we performed ANOVAs using the Anova function in the car package to measure the extent to which elevation, removal treatment (dominant species removal vs. non-removal), and the interaction between these two variables explained variation in C or N mineralization (Fox and Weisberg 2019). To account for heteroscedasticity in potential N mineralization rates among removal treatment and site groups, we used a White-adjustment for corrected standard errors within the Anova function (White 1980). Additionally, to understand if the effect of species removal on mineralization rates was mediated by simply the removal of aboveground biomass, we again conducted ANOVAs to analyze variation in potential mineralization rates across removal treatments using only the mid-low and mid-high elevations where random biomass removal treatments were applied. To further understand whether potential mineralization rates were influenced by the amount of biomass removed from each plot, we analyzed linear relationships between average biomass removed annually from each plot and the residual variation in C and N mineralization after accounting for the effects of removal treatments and site. Finally, we conducted an ANOVA using only the mid-low elevation plots that included a treatment where the focal species, *L. vulgaris*, was naturally absent to analyze the similarity between mineralization rates in focal species removal plots and mineralization rates in plots where the focal species was absent. A high degree of similarity between potential mineralization rates in removal treatment plots and focal species absent plots would lend support to one of the hypotheses proposed by Díaz et al. (2003), that the impact of removing a species from a system is mediated by the loss of functions or influence uniquely attributed to that plant species. To understand how site and removal treatment impact variation among potential C and N mineralization rates, we also performed modified Levene's tests (applying an ANOVA test to the absolute deviation of each observation from the group median) using the levene.test function in the lawstat package (Brown and Forsythe 1974, Gastwirth et al. 2019). All statistical analyses were performed in RStudio (R version 3.6.3; Rstudio Team 2016).

RESULTS

Plant species removal marginally decreased the magnitude of potential N mineralization rates by 27% ($F_{1,43} = 3.32$, $P = 0.075$) and significantly decreased the variation in potential N mineralization ($P = 0.016$), regardless of plant species removed or climate at each elevation (Fig. 1A). Neither site ($F_{3,43} = 0.84$, $P = 0.481$) nor the interaction between site and removal treatment ($F_{3,43} = 0.35$, $P = 0.791$) significantly affected

N mineralization. Variation in potential N mineralization rates within a single site was relatively homogenous across the elevational gradient ($P = 0.462$). Conversely, potential C mineralization rates varied significantly across the elevational gradient ($F_{3,42} = 5.31$, $P = 0.003$), with notably higher C mineralization rates at the high elevation site confirmed by a Tukey HSD test (Fig. 1B). Plant species removal tended to decrease potential C mineralization rates by 9% ($F_{1,42} = 3.22$, $P = 0.08$), whereas the interaction of site and plant removal treatment had no discernible impact on potential C mineralization rates ($F_{3,42} = 0.57$, $P = 0.637$). Rates of C mineralization over the course of the incubation showed consistent patterns across all treatments and sites, with CO₂ efflux peaking during the first days of the incubation and stabilizing at a lower mineralization rate after the first week of the incubation (Appendix S1: Fig. S1). SOM content, measured via LOI, reflected the same pattern as potential C mineralization rates, where SOM-LOI was greatest at the high elevation site ($F_{3,42} = 3.93$, $P = 0.015$) with no detectable response to removal treatments ($F_{1,42} = 1.00$, $P = 0.322$) nor an interaction between removal treatments and site ($F_{3,42} = 0.52$, $P = 0.669$) (Appendix S1: Fig. S2). We did not find an effect of site or removal treatment on the variation in potential C mineralization rates within a treatment group at each site (site $P = 0.349$, removal treatment $P = 0.112$).

While the activity of all soil extracellular enzymes differed significantly across the elevational gradient (AG $P = 0.002$, BG $P < 0.001$, CBH $P = 0.007$, XYL $P = 0.010$, NAG $P < 0.001$, PHOS $P < 0.001$), we did not detect a response of potential soil enzyme activity to the plant removal treatments (AG $P = 0.232$, BG $P = 0.113$, CBH $P = 0.241$, NAG $P = 0.533$, PHOS $P = 0.220$), with the exception of XYL activity that decreased marginally with plant removal ($F_{1,36} = 3.65$, $P = 0.064$; Appendix S1: Fig. S3). Activity of NAG, a nitrogen degrading enzyme, and XYL, a carbon degrading enzyme involved in the breakdown of hemicellulose, both showed significant differences in response to the interaction between site and removal treatment (NAG $F_{3,41} = 3.55$, $P = 0.023$; XYL $F_{3,36} = 4.33$, $P = 0.010$).

Plant species removal at all sites had only a marginal impact on plant community structure ($F_{1,43} = 1.81$, $P = 0.060$), but community structure shifted significantly across the elevational gradient, largely driven by species turnover ($F_{3,43} = 7.76$, $P < 0.001$; Fig. 2). We did not detect a significant interactive effect between plant removal treatment and site on plant community structure ($F_{3,43} = 0.054$, $P = 0.125$). Overall, site, removal treatment, and the interaction of these two parameters explained 40% of the variation in plant community structure across all plots. Furthermore, neither site nor removal treatment altered the dispersion or within-group variation in plant community structure (site $F_{3,47} = 1.23$, $P = 0.310$; removal treatment $F_{1,49} = 0.10$, $P = 0.750$; Appendix S1: Fig. S4).

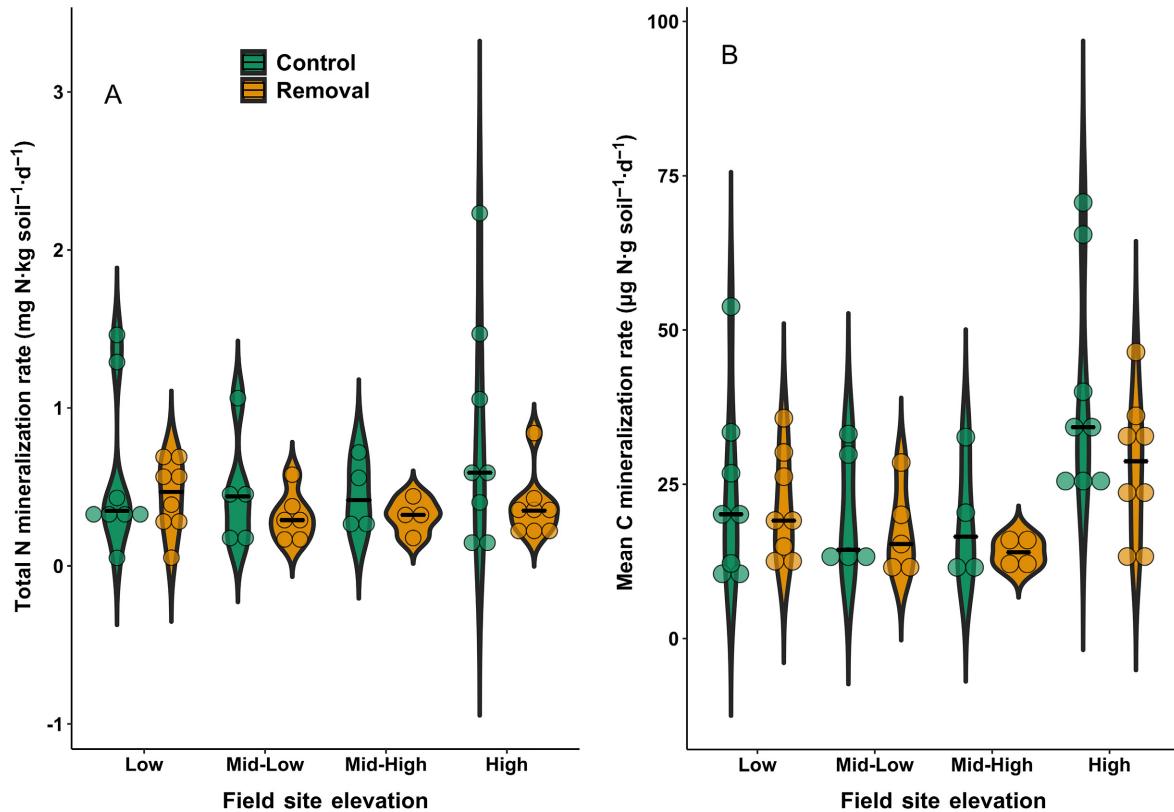


FIG. 1. (A) Plant removal decreased the variation in and, marginally, the magnitude of N mineralization rates. Site and the interaction between elevation and removal treatment had no discernible effect. (B) C mineralization rates varied significantly among sites and were marginally impacted by plant species removal while the interaction between site and species removal treatment did not significantly impact C mineralization rates. Crossbars indicate group medians.

Our comparison of plant community structure and potential soil mineralization rates in random biomass removal plots vs. focal plant species removal plots verified that the effect of removal treatments does not stem from biomass removal alone. However, we could not detect a distinct effect of plant removal on plant community structure or potential mineralization rates in data from this subset of field sites. When analyzing the plant community structure at the two sites that included random biomass removal treatments (mid-high and mid-low elevation sites), we found only a marginally significant difference in plant community dissimilarity across removal treatments ($F_{2,23} = 1.45$, $P = 0.083$) and did not find a significant change in the distribution of variation in plant community structure in response to either of the removal treatments ($F_{2,26} = 0.135$, $P = 0.875$). Likewise, potential C mineralization ($F_{2,22} = 0.68$, $P = 0.519$) and potential N mineralization ($F_{2,23} = 0.98$, $P = 0.389$) were seemingly unaffected by removal treatments, and mineralization rates in random biomass removal plots were indistinguishable from the rates in control plots (C mineralization $P = 0.750$, N mineralization $P = 0.667$) and species removal plots (C mineralization $P = 0.928$, N mineralization $P = 0.912$). Linear regressions

correlating the amount of biomass removed annually with the residual variation in N and C mineralization rates after accounting for the effects of site and removal treatment further supported our finding that the amount of aboveground biomass removed was not significantly correlated with belowground mineralization rates in these meadow ecosystems (C mineralization $P = 0.588$, N mineralization $P = 0.455$). These results indicate that the mechanism driving changes in belowground ecosystem function in response to plant removal treatments stems beyond loss of aboveground biomass.

By isolating the mid-low elevation site “natural removal control” plots where the focal species, *L. vulgaris*, was naturally absent from plots, our results showed that none of our removal treatments had a discernible effect on any plant community properties or ecosystem function rates as measured above (community dissimilarity $F_{3,18} = 0.86$, $P = 0.735$; community dispersion $F_{3,18} = 0.48$, $P = 0.703$; potential C mineralization $F_{3,17} = 1.41$, $P = 0.275$; potential N mineralization $F_{3,18} = 0.83$, $P = 0.494$). Furthermore, potential C mineralization and N mineralization in the plots where *L. vulgaris* was naturally absent were indistinguishable from potential mineralization rates in control plots (C

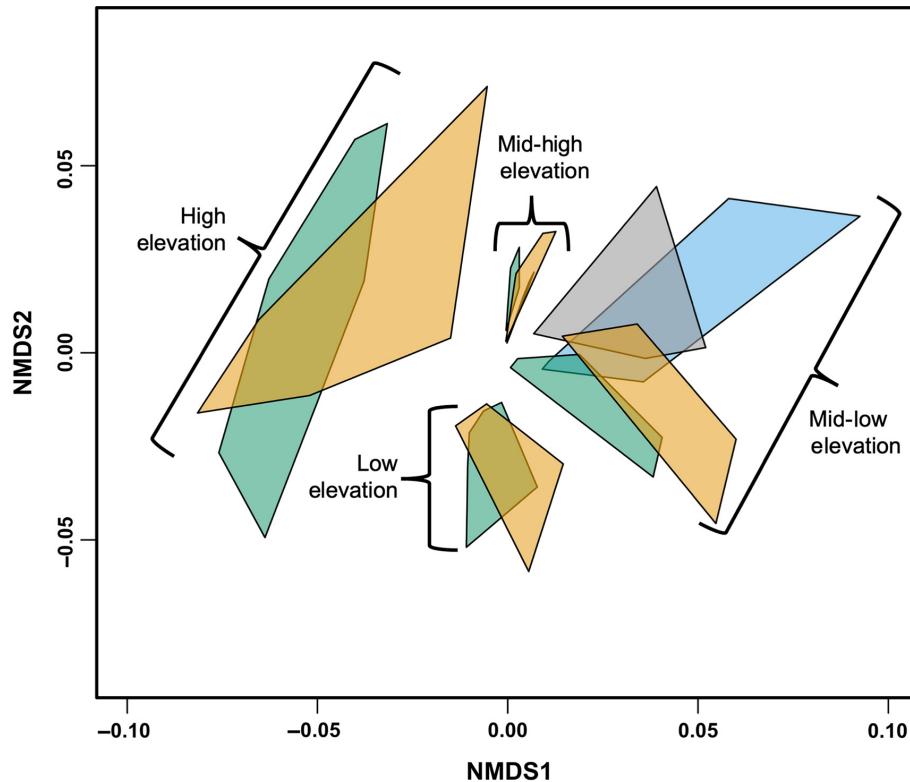


FIG. 2. A nonmetric multidimensional scaling (NMDS) ordination of plant community structure in response to plant removal treatments at four different sites along an elevational gradient. Polygons encompass communities in all plots within a specific removal treatment at each site. Teal polygons indicate control treatments, gold polygons encompass communities that received the plant removal treatment, blue polygons indicate the random biomass removal treatment, and the gray polygon at the mid-low elevation site indicates a natural removal control treatment where the focal plant species, *Linaria vulgaris*, naturally did not occur. Community structure varied by site but only varied marginally with removal treatments and was not significantly impacted by an interaction between removal treatment and site.

mineralization $P = 0.906$, N mineralization $P = 0.993$) and *L. vulgaris* removal plots (C mineralization $P = 0.9997$, N mineralization $P = 0.905$), indicating that the effect of removing the dominant species cannot be attributed to the loss of the specific influence of this species on belowground processes.

DISCUSSION

By incubating soils sampled from plant species removal experiments spread across an elevational gradient, our results show that species removal consistently decreases the variation in potential N mineralization rates, with the magnitude of N mineralization rates in plant removal plots being marginally lower. These results were consistent across the entire elevational gradient, with no differences in potential N mineralization rates among sites, a finding that stands in contrast to our hypothesis that the effect of dominant plant species removal would be mediated by variation in climate and environmental stress across the elevational gradient. While potential C mineralization rates also decreased

subtly with removal treatments, we found significantly higher mineralization rates at the high elevation site where soil moisture across the growing season is relatively higher. This result likely reflects SOM content across the elevational gradient, where potential C mineralization rates were highest at the high elevation site where SOM content was significantly greater than at all other sites (Appendix S1: Fig. S2). We also found that variation in soil potential N mineralization rates was significantly lower in the plant species removal treatment, indicating convergence in potential N mineralization with the loss of a plant species regardless of its identity. Interestingly, we observed these changes in belowground processes in response to plant species removal despite little to no change in overall plant species composition and potential extracellular enzyme activity with the removal of a focal species in each ecosystem.

Why might there be an interactive effect of elevation and removal on XYL and NAG enzyme activities? We suspect that shifts in NAG activity in response to removal might be sensitive to the changes in edaphic properties like soil total C and soil moisture that vary

across the elevational gradient. The NAG activity patterns revealed here likely mirror our N mineralization results because this enzyme is a key agent in the nitrogen mineralization process. Additionally, the interactive effects of site and removal treatment might be linked to the functional identity of the plant species removed at each site. XYL enzymes hydrolyze specific linkages in β -(1,4)-xylose compounds found in the cell walls of plant species. Because the cell walls of monocot species substitute a different xylan compound as the structural backbones of cell walls, the biomass of dicot species contains more of the β -(1,4)-xylose XYL substrate (Hatfield and Rancour 2017). Therefore, removal of a monocot (as at the high and mid-high elevational sites) vs. the removal of a dicot (mid-low and low elevational sites) could have contrasting effects on the synthesis and activity of extracellular enzymes like XYL that aid in the decomposition of lignocellulosic biomass.

Previous work in these meadow ecosystems revealed fairly strong resistance to change in both above- and belowground communities following *F. thurberi* removal, where plant community composition, community-weighted plant functional traits, and fungal colonization of plant roots showed no clear response to removal of the dominant species or nitrogen fertilization (Read et al. 2018, Henning et al. 2019). Alpine ecosystems experience frequent disturbances at a variety of scales, from burrowing mammals to avalanches and landslides, possibly conditioning these ecosystems and the plant and soil communities that inhabit montane meadows to a relatively large degree of disturbance, resulting in resilience to more minor disturbances like plant species removal. These results also suggest that plant species loss impacts belowground processes by some other mechanism beyond the loss of the specific influence of the removed plant species on belowground nutrient cycling.

By comparing the response of potential N mineralization to removal of four distinct species along an elevational gradient, we can discern between the mechanisms by which plant removal might impact ecosystem function proposed by Díaz et al. (2003). Because removal decreased potential N mineralization at all field sites, regardless of the functional identity of the plant species removed, our results suggest that this ecosystem response is not due to the loss of the influence of a particular species, eliminating the first mechanism as a plausible explanation. When considering the “natural removal control” plots at the mid-low elevation, we find that any response of N mineralization rates to the removal of this species was not due to the loss of a specific function performed by the removed species, *L. vulgaris*, because N mineralization rates did not differ between plots where *L. vulgaris* was present and plots where *L. vulgaris* was naturally absent. The results from our analysis of shifts in the structure of the plant communities in each experiment in response to removals (Fig. 1A) showed that the removal treatment had only a marginal effect on plant community structure, making it

unlikely that community re-assembly following removal could be driving the shift in ecosystem function found here. Finally, potential N mineralization rates did not differ significantly between control plots and random biomass removal plots, indicating that disturbance is also unlikely to be the dominant driver of shifts in potential N mineralization rates in response to removal. Our experimental design therefore allows us to speculatively rule out all three of the mechanisms that drive removal effects as proposed by Díaz et al. (2003), so we turn to other more indirect mechanisms that might drive variation in mineralization rates following species removal.

Plant removal may have several indirect impacts on belowground microbial processes that could explain our results. Removals across all experiments were conducted by either clipping aboveground biomass of the removed species to ground level, or by gently pulling on aboveground biomass to break shoots from the extensive network of belowground roots, as in the *L. vulgaris* removal. This method of removal leaves much of the belowground biomass of the removed plant behind, potentially constituting a significant pool of N that is immobilized in plant root litter. Several studies estimate that root mean residence time is approximately four years for fine roots in ecosystems characterized by a -1.8 – 2.7°C range in mean annual temperature (Gill and Jackson 2000, Leifeld et al. 2015), and root turnover in grasslands globally is limited by precipitation (Wang et al. 2019). Estimates of fine root decomposition in arid grasslands with precipitation regimes similar to the climate of this study system indicate that as much as 60% of root biomass remains after 4–10 yr of decomposition, immobilizing 60% of the N content of root litter at the onset of decomposition (Parton 2007). Delay in root decomposition following the onset of plant removal could lead to a short-term decrease in the size of the soil N pool that is available to microbes and plants until the roots of removed plants are decomposed. Our data lend some support to this explanation when N mineralization rates are analyzed within the context of each individual experiment across the elevational gradient. Results from the mid-low elevation site, where *L. vulgaris* has been actively removed for 17 yr, show no effect of removal, or any of other treatments, on potential N mineralization rates, possibly indicating that the legacy of the removal treatment has faded as roots have decomposed following removal to release N immobilized in belowground *L. vulgaris* root litter. In contrast, the effect of species removal on soil N mineralization rates remains across all three younger experiments. Immobilization of N in belowground root litter might therefore be a mechanism by which loss of species locally could reduce ecosystem N cycling in the short term.

Species removal likely affects ecosystem function more generally by reducing ecosystem resilience to climatic extremes (Tilman and Downing 1994). This response to species removals may be a belowground manifestation of

an effect that Kardol et al. (2018) found in a long-term plant removal experiment where plant species loss led to greater temporal variability in aboveground plant biomass. Perhaps plant species loss leads to similar variability belowground where properties such as soil microbial community composition, microbial biomass, or microbial activity and carbon use efficiency in communities that have lost plant species are especially vulnerable to climatic extremes, driving marked shifts in belowground processes that are not seen in more diverse and resilient communities. Heightened temporal variability of above- and/or belowground ecosystem components in response to species loss may also negatively impact ecosystem resilience (Oliver 2015). As 2018 was an especially dry summer in our study area, loss of the ability of plant and microbial communities to function in spite of climatic events like a severe drought may explain lower mineralization rates in plant removal plots.

In conclusion, we found that removing the dominant plant species consistently reduces the variation in soil N mineralization rates, while marginally decreasing the magnitude of C and N mineralization rates, in alpine meadows. We find these changes to belowground function in response to plant removal despite only subtle shifts in aboveground plant community structure and no clear changes in extracellular enzyme activity in response to removal. While our results are limited in their ability to pinpoint a clear mechanism by which species loss affects potential soil mineralization rates, the results do offer an unexpectedly simple pattern describing the overall effect of species loss on soil N and C cycling that holds across an elevational gradient. Moreover, our study offers insight into how loss of aboveground plant species might indirectly impact belowground processes with implications for ecosystem function in a future world characterized by global change.

ACKNOWLEDGMENTS

We would like to thank Pamela Templer and all reviewers of our manuscript for their thoughtful comments and valuable feedback. This study was partially supported by a Semper Ardens grant from the Carlsbergfondet and by a Catalyst Grant from the Gund Institute for Environment at the University of Vermont. Installation, maintenance and publication of data from the RMBL weather stations has been supported by NSF award (MRI-0821369) and DOE funding as well as ongoing operational support from RMBL, the Upper Gunnison River Water Conservancy District and the LBNL Science Focus Area 2.0 project. This research is also based upon work supported by the National Science Foundation Graduate Research Fellowship Program under Grant No. NSF-16-588 awarded to K. Rewcastle. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the National Science Foundation. Author contributions: A. Classen, N. Sanders, J. Henning, Q. Read, and R. Irwin designed the field experiments that were sampled as part of this study. All authors contributed to maintaining the plant removal treatments and together collected annual estimates of biomass removed and plant community composition across all field sites. K. Rewcastle collected field samples, conducted laboratory analyses, and analyzed all

data with support from A. Classen, N. Sanders, J. Henning, Q. Read, and R. Irwin. K. Rewcastle was primarily responsible for writing the manuscript with contributions and revisions from all coauthors.

LITERATURE CITED

- Adler, P. B., H. J. Dalglish, and S. P. Ellner. 2012. Forecasting plant community impacts of climate variability and change: when do competitive interactions matter? *Journal of Ecology* 100:478–487.
- Anderson, M. J., K. E. Ellingsen, and B. H. McArdle. 2006. Multivariate dispersion as a measure of beta diversity. *Ecology Letters* 9:683–693.
- Avolio, M. L., E. J. Forrester, C. C. Chang, K. J. La Pierre, K. T. Burghardt, and M. D. Smith. 2019. Demystifying dominant species. *New Phytologist* 223:1106–1126.
- Baert, J. M., N. Eisenhauer, C. R. Janssen, and F. D. Laender. 2018. Biodiversity effects on ecosystem functioning respond unimodally to environmental stress. *Ecology Letters* 21:1191–1199.
- Bardgett, R. D., P. Manning, E. Morriën, and F. T. De Vries. 2013. Hierarchical responses of plant–soil interactions to climate change: consequences for the global carbon cycle. *Journal of Ecology* 101:334–343.
- Brooker, R. W. 2006. Plant–plant interactions and environmental change. *New Phytologist* 171:271–284.
- Brown, M. B., and A. B. Forsythe. 1974. Robust tests for the equality of variances. *Journal of the American Statistical Association* 69:364–367.
- Chapin III, F. S. 1998. Ecosystem consequences of changing biodiversity. *BioScience* 48:45–52.
- Classen, A. T., M. K. Sundqvist, J. A. Henning, G. S. Newman, J. A. M. Moore, M. A. Cregger, L. C. Moorhead, and C. M. Patterson. 2015. Direct and indirect effects of climate change on soil microbial and soil microbial–plant interactions: What lies ahead? *Ecosphere* 6:art130.
- Diaz, S., A. J. Symstad, F. Stuart Chapin, D. A. Wardle, and L. F. Huenneke. 2003. Functional diversity revealed by removal experiments. *Trends in Ecology & Evolution* 18:140–146.
- Doane, T. A., and W. R. Horwath. 2003. Spectrophotometric determination of nitrate with a single reagent. *Analytical Letters* 36:2713–2722.
- Fox, J. W., and B. Kerr. 2012. Analyzing the effects of species gain and loss on ecosystem function using the extended Price equation partition. *Oikos* 121:290–298.
- Fox, J., and S. Weisberg. 2019. An R companion to applied regression. Sage, Thousand Oaks, California, USA.
- García-Palacios, P., N. Gross, J. Gaitán, and F. T. Maestre. 2018. Climate mediates the biodiversity–ecosystem stability relationship globally. *Proceedings of the National Academy of Sciences USA* 115:8400–8405.
- Gastwirth, J. L., Y. R. Gel, W. L. Wallace Hui, V. Lyubchich, W. Miao, and Noguchi, K. 2019. lawstat: tools for biostatistics, public policy, and law. R package version 3.3. <https://cran.r-project.org/web/packages/lawstat/index.html>
- Gill, R. A., and R. B. Jackson. 2000. Global patterns of root turnover for terrestrial ecosystems. *New Phytologist* 147:13–31.
- Gilman, S. E., M. C. Urban, J. Tewksbury, G. W. Gilchrist, and R. D. Holt. 2010. A framework for community interactions under climate change. *Trends in Ecology & Evolution* 25:325–331.
- Grace, J. B., et al. 2016. Integrative modelling reveals mechanisms linking productivity and plant species richness. *Nature* 529:390–393.
- Hatfield, R. D., D. M. Rancour, and J. M. Marita. 2017. Grass cell walls: a story of cross-linking. *Frontiers in Plant Science* 7:2056.

- Henning, J. A., Q. D. Read, N. J. Sanders, and A. T. Classen. 2019. Fungal colonization of plant roots is resistant to nitrogen addition and resilient to dominant species losses. *Ecosphere* 10:e02640.
- Hoogsteen, M. J. J., E. A. Lantinga, E. J. Bakker, J. C. J. Groot, and P. A. Tittone. 2015. Estimating soil organic carbon through loss on ignition: effects of ignition conditions and structural water loss. *European Journal of Soil Science* 66:320–328.
- Jarrell, W. M., D. E. Armstrong, D. F. Grigal, E. F. Kelly, H. C. Monger, and D. A. Wedin. 1999. Soil Water and Temperature Status. Pages 55–73 in G. P. Robertson, D. C. Coleman, C. S. Bledsoe, and P. Sollins, editors. *Standard Soil Methods for Long-Term Ecological Research, Long-Term Ecological Research Network Series*. Oxford University Press, Oxford, New York, USA.
- Johnson, D., G. K. Phoenix, and J. P. Grime. 2008. Plant community composition, not diversity, regulates soil respiration in grasslands. *Biology Letters* 4:345–348.
- Kaisermann, A., F. T. de Vries, R. I. Griffiths, and R. D. Bardgett. 2017. Legacy effects of drought on plant–soil feedbacks and plant–plant interactions. *New Phytologist* 215:1413–1424.
- Kardol, P., N. Fanin, and D. A. Wardle. 2018. Long-term effects of species loss on community properties across contrasting ecosystems. *Nature* 557:710–713.
- Klanderud, K. 2005. Climate change effects on species interactions in an alpine plant community. *Journal of Ecology* 93:127–137.
- Klanderud, K., and Ø. Totland. 2005. Simulated climate change altered dominance hierarchies and diversity of an alpine biodiversity hotspot. *Ecology* 86:2047–2054.
- Leifeld, J., S. Meyer, K. Budge, M. T. Sebastia, M. Zimmermann, and J. Führer. 2015. Turnover of grassland roots in mountain ecosystems revealed by their radiocarbon signature: role of temperature and management. *PLoS ONE* 10:e0119184.
- Lyons, K. G., and M. W. Schwartz. 2001. Rare species loss alters ecosystem function—invasion resistance. *Ecology Letters* 4:358–365.
- McLaren, J. R., and R. Turkington. 2010. Ecosystem properties determined by plant functional group identity. *Journal of Ecology* 98:459–469.
- Naeem, S., L. J. Thompson, S. P. Lawler, J. H. Lawton, and R. M. Woodfin. 1995. Empirical evidence that declining species diversity may alter the performance of terrestrial ecosystems. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences* 347:249–262.
- Ockendon, N., et al. 2014. Mechanisms underpinning climatic impacts on natural populations: altered species interactions are more important than direct effects. *Global Change Biology* 20:2221–2229.
- Oksanen, J., et al. 2019. Vegan: community ecology package. <https://cran.r-project.org/web/packages/vegan/index.html>
- Oliver, T. H., et al. 2015. Biodiversity and resilience of ecosystem functions. *Trends in Ecology & Evolution* 30:673–684.
- Parton, W., et al. 2007. Global-scale similarities in nitrogen release patterns during long-term decomposition. *Science* 315:361–364.
- Pugnaire, F. I., J. A. Morillo, J. Peñuelas, P. B. Reich, R. D. Bardgett, A. Gaxiola, D. A. Wardle, and W. H. van der Putten. 2019. Climate change effects on plant–soil feedbacks and consequences for biodiversity and functioning of terrestrial ecosystems. *Science Advances* 5:eaz1834.
- Read, Q. D., J. A. Henning, A. T. Classen, and N. J. Sanders. 2018. Aboveground resilience to species loss but belowground resistance to nitrogen addition in a montane plant community. *Journal of Plant Ecology* 11:351–363.
- Rewcastle, K. E., J. A. Henning, Q. D. Read, R. E. Irwin, N. J. Sanders, and A. T. Classen. 2021. Effects of plant removal on mineralization rates at the Rocky Mountain Biological Laboratory, Gunnison County, Colorado: 2018 ver 1. Environmental Data Initiative. <https://doi.org/10.6073/pasta/11a8123a58bb45d76c61fbb1f5b88d7>
- Robertson, G. P., D. Wedin, P. M. Groffman, J. M. Blair, E. A. Holland, K. J. Nadelhoffer, and D. Harris. 1999. Soil carbon and nitrogen availability: nitrogen mineralization, nitrification, and soil respiration potentials. Pages 258–271 in G. P. Robertson, D. C. Coleman, C. S. Bledsoe, and P. Sollins, editors. *Standard soil methods for long-term ecological research, long-term ecological research network series*. Oxford University Press, Oxford, New York, USA.
- RStudio Team. 2016. RStudio: Integrated Development for R. RStudio, Boston, Massachusetts, USA.
- Saiya-Cork, K. R., R. L. Sinsabaugh, and D. R. Zak. 2002. The effects of long term nitrogen deposition on extracellular enzyme activity in an *Acer saccharum* forest soil. *Soil Biology and Biochemistry* 34:1309–1315.
- Smith, M. D., and A. K. Knapp. 2003. Dominant species maintain ecosystem function with non-random species loss. *Ecology Letters* 6:509–517.
- Sundqvist, M. K., N. J. Sanders, and D. A. Wardle. 2013. Community and ecosystem responses to elevational gradients: Processes, mechanisms, and insights for global change. *Annual Review of Ecology, Evolution, and Systematics* 44:261–280.
- Symstad, A. J., D. Tilman, J. Willson, and J. M. H. Knops. 1998. Species loss and ecosystem functioning: effects of species identity and community composition. *Oikos* 81:389–397.
- Tilman, D., and J. A. Downing. 1994. Biodiversity and stability in grasslands. *Nature* 367:363–365.
- Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie, and E. Siemann. 1997. The influence of functional diversity and composition on ecosystem processes. *Science* 277:1300–1302.
- Tilman, D., D. Wedin, and J. Knops. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* 379:718–720.
- van der Putten, W. H., M. A. Bradford, E. P. Brinkman, T. F. J. van de Voorde, and G. F. Veen. 2016. Where, when and how plant–soil feedback matters in a changing world. *Functional Ecology* 30:1109–1121.
- Wang, J., J. Sun, Z. Yu, Y. Li, D. Tian, B. Wang, Z. Li, and S. Niu. 2019. Vegetation type controls root turnover in global grasslands. *Global Ecology and Biogeography* 28:442–455.
- Wardle, D. A., R. D. Bardgett, R. M. Callaway, and W. H. van der Putten. 2011. Terrestrial ecosystem responses to species gains and losses. *Science* 332:1273–1277.
- Wardle, D. A., K. I. Bonner, G. M. Barker, G. W. Yeates, K. S. Nicholson, R. D. Bardgett, R. N. Watson, and A. Ghani. 1999. Plant removals in perennial grassland: vegetation dynamics, decomposers, soil biodiversity, and ecosystem properties. *Ecological Monographs* 69:535–568.
- Wardle, D. A., M. J. Gundale, A. Jäderlund, and M.-C. Nilsson. 2013. Decoupled long-term effects of nutrient enrichment on aboveground and belowground properties in subalpine tundra. *Ecology* 94:904–919.
- Weaver, J. E. 1915. A study of the root-systems of prairie plants of Southeastern Washington. *Plant World* 18:227–248.
- White, H. 1980. A heteroskedasticity-consistent covariance matrix estimator and a direct test for heteroskedasticity. *Econometrica* 48:817–838.
- Wilke, B. J., and R. E. Irwin. 2010. Variation in the phenology and abundance of flowering by native and exotic plants in subalpine meadows. *Biological Invasions* 12:2363–2372.
- Zak, D. R., W. E. Holmes, D. C. White, A. D. Peacock, and D. Tilman. 2003. Plant diversity, soil microbial communities, and ecosystem function: Are there any links? *Ecology* 84:2042–2050.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of this article at <http://onlinelibrary.wiley.com/doi/10.1002/ecy.3546/suppinfo>

OPEN RESEARCH

All data (Rewcastle et al. 2021) have been made publicly available through the Environmental Data Initiative (EDI): <https://doi.org/10.6073/pasta/11a8123a58cbb45d76c61fbb1f5b88d7>