2015

**A framework for quantifying the magnitude and variability of community responses to global change drivers**

Meghan L. Avolio
Kimberly J. La Pierre
Gregory R. Houseman
Sally E. Koerner
Emily Grman

*See next page for additional authors*

Follow this and additional works at: https://scholarworks.wm.edu/vimsarticles

Part of the Marine Biology Commons

**Recommended Citation**


This Article is brought to you for free and open access by the Virginia Institute of Marine Science at W&M ScholarWorks. It has been accepted for inclusion in VIMS Articles by an authorized administrator of W&M ScholarWorks. For more information, please contact scholarworks@wm.edu.
A framework for quantifying the magnitude and variability of community responses to global change drivers

MEGHAN L. AVOLIO,1,2† KIMBERLY J. LA PIERRE,3 GREGORY R. HOUSEMAN,4 SALLY E. KOERNER,5 EMILY GRMAN,6 FOREST ISBELL,7 DAVID SAMUEL JOHNSON,8,9 AND KEVIN R. WILCOX5

1Department of Biology, University of Utah, Salt Lake City, Utah 84112 USA
2National Socio-Environmental Synthesis Center, Annapolis, Maryland 21401 USA
3Department of Integrative Biology, University of California, Berkeley, Berkeley, California 94720 USA
4Department of Biological Sciences, Wichita State University, Wichita, Kansas 67026 USA
5Department of Biology and Graduate Degree Program in Ecology, Colorado State University, Fort Collins, Colorado 80523 USA
6Department of Biology, Eastern Michigan University, Ypsilanti, Michigan 48197 USA
7Department of Ecology, Evolution and Behavior, University of Minnesota, Saint Paul, Minnesota 55108 USA
8Department of Biological Sciences, Virginia Institute of Marine Science, Gloucester Point, Virginia 23062 USA
9Marine Biological Laboratory, Woods Hole, Massachusetts 02543 USA


Abstract. A major challenge in global change ecology is to predict the trajectory and magnitude of community change in response to global change drivers (GCDs). Here, we present a new framework that not only increases the predictive power of individual studies, but also allows for synthesis across GCD studies and ecosystems. First, we suggest that by quantifying community dissimilarity of replicates both among and within treatments, we can infer both the magnitude and predictability of community change, respectively. Second, we demonstrate the utility of integrating rank abundance curves with measures of community dissimilarity to understand the species-level dynamics driving community changes and propose a series of testable hypotheses linking changes in rank abundance curves with shifts in community dissimilarity. Finally, we review six case studies that demonstrate how our new conceptual framework can be applied. Overall, we present a new framework for holistically predicting community responses to GCDs that has broad applicability in this era of unprecedented global change and novel environmental conditions.

Key words: beta diversity; community dissimilarity; convergence; divergence; multivariate analysis; non-metric multidimensional scaling; rank abundance curve; species composition.

Received 27 May 2015; revised 12 June 2015; accepted 23 June 2015; published 21 December 2015. Corresponding Editor: D. P. C. Peters.

Copyright: © 2015 Avolio et al. This is an open-access article distributed under the terms of the Creative Commons Attribution License, which permits unrestricted use, distribution, and reproduction in any medium, provided the original author and source are credited. http://creativecommons.org/licenses/by/3.0/

† E-mail: mavolio@sesync.org

INTRODUCTION

Ecologists are currently tasked with predicting how communities will respond to a wide variety of global changes including chronic resource alterations (Smith et al. 2009) and shifting biodiversity (Chapin et al. 2000). As such, the field of experimental global change ecology is rapidly expanding, with many experiments aiming to understand how different global change...
Drivers (GCDs) will impact communities (see Box 1 for definitions of italicized words). However, many of these studies focus on univariate responses only, such as summary measures of community diversity (e.g., richness). As noted by Collins et al. (2008), these univariate approaches can obfuscate complex community changes. Instead, a multivariate approach that considers community composition in its entirety can be more informative, if it can be done in a generalizable manner. Indeed, changes in community composition can be systematically studied using distance-based dissimilarity metrics, recently popularized as useful for measures of beta diversity (Anderson et al. 2006, 2011). Dissimilarity metrics enable the study of both the magnitude and the predictability of community responses by comparing experimental replicates both among treatments as well as within treatments. Community dissimilarity of replicates among treatments is informative for understanding the magnitude of community change corresponding with a change in environmental conditions, while community dissimilarity of replicates within treatments can indicate the predictability of community change. For example, Inouye and Tilman (1988, 1995) found that replicate tallgrass prairie plant communities receiving high-level nitrogen additions diverged from one another, while those receiving less nitrogen converged. Likewise, Houseman and colleagues (2008) found patterns of convergence among replicate grassland plant communities with disturbance and divergence with fertilization treatment. Finally, Koerner and colleagues (unpublished manuscript) found decreasing levels of community divergence among replicates with higher levels of nitrogen and phosphorus additions. Thus, one might infer from these three studies that depending on the degree of fertilization, communities may become more heterogeneous making it more difficult to predict community responses. Studying broad patterns enables generalizations and comparison across studies, as well as enhancing our ability to identify the mechanisms that lead to patterns of convergence or divergence, as characterized in previous succession and community assembly studies (Clements 1936, Leps and Rejmanek 1991, Fukami 2010, Weiher et al. 2011).

Understanding patterns of community change over time (Clements 1916, Gleason 1926) and identifying mechanisms of change (Connell and

---

**Box 1 Glossary of terms**

**global change driver (GCD):** A human-driven exogenous perturbation that changes abiotic (e.g., resources, temperature, pollution) or biotic (e.g., invasive species) conditions typically occurring as widespread chronic presses. Global change drivers can also change the magnitude or frequency of episodic natural disturbance regimes (fire suppression, hurricanes).

**community composition:** The identity and abundance of all species in a given space and time.

**among treatments:** Comparing the community composition of treatment and control plots. This concept is similar to turnover beta diversity (see Vellend 2001, Anderson et al. 2011), however, instead of comparing how species composition of plots changes along a gradient, it is mean dissimilarity of the treatment plots from the control plots.

**within treatments:** Comparing the community composition of plots within a treatment. This is variation beta diversity (see Vellend 2001, Anderson et al. 2011).

**convergence:** Communities become more similar to one another in composition through time.

**divergence:** Communities become less similar to one another in composition through time.

**parallel change:** Communities that change compositionally through time, but do not converge or diverge.

**rank abundance curves (RAC):** A display of species abundance distributions that presents each species’ abundance on the vertical axis and its rank in abundance (ordered from highest to lowest abundance) on the horizontal axis.
Slatyer 1977) was an early and consistent focus of ecological research. Prach and Walker (2011) suggest re-visiting the successional literature to increase our understanding of how GCDs alter communities. GCDs are environmental perturbations that typically result in chronic resource alterations, but do not clear space (Smith et al. 2009) to initiate primary or secondary successional dynamics. However, the processes, mechanisms, and terminology of GCD induced community change are the same as those used to describe succession. As with succession, the list of potential mechanisms for global change driven community alterations is extensive (Connell and Slatyer 1977, Walker and Chapin 1987, Pickett and McDonnell 1989). While ultimately it is desirable to know the mechanisms by which a community changed, mechanisms can be case specific and prevent generalizations. Across the plethora of global change experiments, it might be more informative to first determine whether there are clear patterns of community change in response to GCDs. After patterns of community responses have been identified, specific mechanisms may more easily be elucidated.

Here, we present a new framework for studying GCD caused community changes by simultaneously using multivariate community dissimilarity metrics and rank abundance curves (Fig. 1). We propose that comparing dissimilarity of replicates both among and within treatments will give insight into both the magnitude and the predictability or repeatability of community change. One drawback of such an approach is that it is difficult to understand what aspect of the community was altered using dissimilarity metrics alone. To overcome this potential limitation, we demonstrate how integrating rank abundance curves (RACs) with community dissimilarity metrics can be used to identify concomitant changes in community structure (Fig. 1). Towards this end, we hypothesize how species changes reflected in RACs might change community dissimilarity metrics both among and within treatments, and present examples of community studies that use multivariate methods to highlight this new framework. Lastly, we suggest future directions for this framework and provide testable hypotheses for the field.

**Detecting the Magnitude and Predictability of Community Change Using Dissimilarly Metrics**

There are two important measures of dissimilarity among replicates (Houseman et al. 2008; Figs. 1 and 2): (1) the mean dissimilarity of replicates among treatments (i.e., do control and treated replicates have similar communities?) and (2) the mean dissimilarity of replicates within treatments (i.e., do control or treated replicates have different community variability?). The first measure, mean dissimilarity of replicates among treatments, is the separation between treatment centroids in multivariate community space and is similar to the concept of turnover beta diversity (see Box 1; Vellend 2001, Anderson et al. 2011). The second, mean dissimilarity of replicates within treatments, is the dispersion of replicates around the treatment centroid in multivariate community space (Clarke 1993) and is the same as variation beta diversity (see Box 1; Vellend 2001, Anderson et al. 2011). Both measures of variability are affected by the species present and their abundances (Clarke 1993).

Using multivariate measures to determine changes in the composition of communities is a way to quantify beta diversity (Legendre et al. 2005, Anderson 2006, Bacaro et al. 2012, Legendre and De Cáceres 2013). More traditionally, beta diversity has been calculated directly from alpha and gamma diversity (Whittaker 1960, Tuomisto 2010a). Beta diversity is highly influenced by local alpha diversity and regional gamma diversity (Anderson et al. 2011, Chase et al. 2011, Kraft et al. 2011). However, because our framework is for an experimental examination of GCDs on communities, we are less concerned with how changes in alpha or gamma diversity will affect beta diversity. We consider gamma diversity to be the total species pool for the experiment and thus static, and we consider alpha diversity to be the number of species in a treatment and therefore changeable over time. While studies typically find that GCDs reduce alpha diversity (Chalcraft et al. 2008, Powell et al. 2013), these reductions are often not drastic. For example, across 40 grassland sites, three years of NPK additions resulted on average a loss of two species (Borer et al. 2014), and such minor losses are commonly found (see Zavaleta et al. 2003,
Avolio et al. 2014). However, if within an experiment there are very large changes in alpha diversity, null models may be a useful tool to understand the true changes in beta diversity that are not influenced by gamma or alpha diversity (Chase et al. 2011, Chase and Myers 2011, Kraft et al. 2011). In this paper, because we are interested in both aspects of beta diversity simultaneously (turnover and variation), we are moving away from the tortured beta diversity language and simply discuss multivariate patterns below.

We envision eight ways that individual replicates can differ from one another in their dissimilarity both among and within treatments and demonstrate these patterns in a hypothetical multivariate community space (Bray and Curtis 1957) for easy visualization (Fig. 2). While investigating community changes using dissimilarity metrics and visualizing differences is a method commonly used in community ecology (see Houseman et al. 2008), here we suggest that looking at changes in dissimilarity of replicates among treatments and decreased overlap in multivariate community space. “Δ Mean” indicates high dissimilarity of replicates among treatments and increased overlap in multivariate community space. “Δ Variance” indicates similar dissimilarity of replicates within treatments and similar dispersion of replicates around the centroid in multivariate community space. Lastly, “Δ Variance” indicates changes in the dissimilarity of replicates within treatments and a different degree of dispersion in multivariate community space.
replicates (convergence; Fig. 2C). These four variance responses could also be accompanied by a change in mean (Fig. 2D–F). Regardless of the pattern in variability among treated replicates, both stochastic and deterministic processes can cause either community convergence or divergence (Chase and Myers 2011). Thus, it is important to note that the observed pattern among replicates does not necessarily indicate the nature of the mechanism that brought about the community change, but it does provide important quantification of the pattern that may be generalizable across studies.

By investigating the variability of the community response among replicates (Fraterrigo and Rusak 2008, Murphy and Romanuk 2012), patterns can be identified that give insight into the repeatability and predictability of community change. An observed pattern of parallel change or convergence among replicates would suggest that all treated replicates respond the same way and community changes may be predictable. Conversely, if all replicates diverged from one another in their community composition, then the community response may be difficult or impossible to predict.

**MEASURING AND ANALYZING PATTERNS OF COMMUNITY DISSIMILARITY**

There have been many influential papers on multivariate community analyses. Here we briefly review the steps necessary for such analyses, from creating a dissimilarity matrix to performing statistical tests. The first step is to carefully select an appropriate dissimilarity metric and create a dissimilarity matrix of all replicates to one another (Anderson et al. 2006, Anderson et al. 2011). We suggest using abundance data and an associated dissimilarity metric (e.g., Bray-Curtis dissimilarity) over presence/absence data for three reasons. First, abundance data also allows for the subsequent examination of rank abundance curves (see below for details). Second, abundance data is more sensitive to detecting deterministic changes in community composition compared with presence/absence data (Segre et al. 2014). Third, calculating dissimilarity from presence/absence data is highly dependent differences in on alpha diversity (Chase et al. 2011). Next, it is helpful to visualize differences in community composition of replicates via non-metric multidimensional scaling (NMDS) or
Fig. 3. There are four ways the rank abundance curves (RAC) could change in response to the community changes identified in Fig. 2. Panels A–H should be compared with the control plot on top of the Fig. First, species reordering occurs when all the same species remain in the community but their order changes (A, B). Both a gain (C, D) and a loss of species (E, F) from the community occurs when species immigrate into or go extinct from the community. Note that both of these changes necessitate species re-ordering unless they happen in the tail of the RAC. Lastly, there can be a change in evenness only (G, H). Note that the scale of the x-axis changes. Depending on the nature and degree of the change to the RAC, these four basic changes a community can lead to changes in community dissimilarity both among and within treatments (see Table 1).
principal coordinates analysis (PCO or PCoA; Legendre and Legendre 2012). Both analyses will produce a plot showing symbols representing replicates appearing close together if their local communities are similar (based on the dissimilarity metric chosen) or far apart if their communities are dissimilar (see Fig. 4 for examples). After the patterns have been discerned, it is desirable to test whether differences among treatments in both their mean (among treatment differences) and dispersion of replicates (within treatment differences) are statistically significant. To test for differences of replicates among treatments (difference of centroids), permutational multivariate analysis of variance (PERMANOVA) is perhaps the most popular method. Anderson and Walsh (2013) recently demonstrated that with balanced experimental designs, PERMANOVA is relatively insensitive to changes in variance among groups. There are a multitude of different approaches to test for differences in dispersion of replicates within treatments (Tuomisto 2010b, Anderson et al. 2011); however, many do not allow statistical tests because they produce only one estimate of variation (Anderson 2006). Two approaches that do allow statistical testing are permutational tests for homogeneity of variance (Anderson 2001, 2006) or general linear model-based approaches (Wang et al. 2012). Because of their widespread use and known capabilities for detecting patterns of convergence and divergence, overall we advocate use of PERMANOVA to test for changes in mean among treatments and permutational tests for homogeneity of variance to test for changes in dispersion.

**INTEGRATING RANK ABUNDANCE CURVES WITH COMMUNITY DISSIMILARITY METRICS**

We argue that measuring community dissimilarity among replicates in response to GCDs offers insight into the magnitude and predictability of community change. However, it is possible to lose valuable information when using community dissimilarity metrics alone (Collins et al. 2008), because it is unclear how the community changed. One way to understand these community-level changes is to examine changes in specific species that drive the community response (e.g., Houseman et al. 2008), but this approach has limited application across systems. A second more generalizable approach is to pair community dissimilarity analyses with examination of changes in the community’s species abundance distribution, which describe the relative abundances of all constituent species within a community. Species abundance distributions are useful as visualization tools to help ecologists understand how communities are structured and are relatively easy to create using commonly collected species abundance data (McGill et al. 2007, Ulrich et al. 2010). Rank abundance curves (RACs; e.g., Fig. 3), are a type of species abundance distribution first introduced in the mid-20th century (MacArthur 1957, Whittaker 1965). Here, we suggest using RACs to isolate specific aspects of community change in response to GCDs (McGill et al. 2007, Collins et al. 2008; Fig. 1). Once the patterns of change in the RAC have been identified and classified (see below; Fig. 1), mechanisms responsible for that change can be more easily explored. Further, by identifying changes in RAC, the findings from studies are more generalizable, facilitating comparisons across systems, GCDs, and experiments.

There are four ways that GCDs can modify RACs (Fig. 3) within a replicate following a perturbation: (1) RACs can be altered due to species re-ordering; that is, the rank of species can change because one or more species have either decreased or increased in abundance. Additionally, RACs can be modified by the (2) loss and/or (3) gain of species. While re-ordering alone can occur without a gain or loss of species, it is important to recognize that the loss or gain of any species that is not the lowest rank must co-occur with species reordering. Lastly, RACs can be changed by (4) shifts in species evenness; that is, the shape of the curve can become steeper (greater dominance) or flatter (greater evenness). We hypothesize that for each of the eight possible patterns of community change (Fig. 2), there are a limited number of ways the RAC can change, and present a table of testable hypotheses integrating changes in community dissimilarity and RACs (Table 1), by comparing a control with a GCD treatment. For example, we hypothesize that the re-ordering, gain or loss of only rare species in the treated replicates will change the variability among replicates within treatments but not the mean (Table 1), while for common
species, all three processes could result in shifts in mean and changes in dispersion (Table 1). Many of these scenarios are not mutually exclusive and all four modifications to the RAC can occur simultaneously.

Overall, there is great interest in determining the mechanisms underlying community responses to GCDs. However, without a clear understanding of how communities shift in response to GCDs (e.g., loss of dominance, turnover of functional types), these mechanisms will likely be difficult to determine. Key to understanding patterns of community responses to global change is correctly measuring the species pool. Indeed, recent studies have shown that patterns of diversity responses to GCDs can be altered by the area sampled alone (Chase and Knight 2013, Powell et al. 2013). As suggested by Chase and Knight (2013) collecting abundance data is potentially key to obtaining unbiased effect sizes of GCDs on communities, and may make researchers less prone to draw conclusions from probabilistic sampling effects. Further, RACs necessitate collecting abundance data. Additionally, with a thorough understanding of the patterns of change in RACs in response to GCDs, future experiments can be designed to test potential mechanisms underlying any observed response, either deterministic or stochastic.

**CASE STUDIES**

We present six examples of the changes in multivariate community space illustrating the application of our conceptual framework. For each case study, we describe observed changes in community dissimilarity mean and variance among replicates between control and treatments and graph the non-metric multi-dimensional scaling plots for each case study comparing the control plots to those altered by a treatment (Fig. 4). Where possible, we also detail how changes in the RACs led to the observed changes. Lastly, we explore how understanding these changes in RACs can lead to hypothesized mechanisms determining community responses. Where possible we use examples where the treatment is a GCD, however as GCD studies examining multivariate community shifts are relatively rare, this was not possible for all scenarios.

**No change in mean, no change in variance (Fig. 4A)**

We hypothesized that no change in mean or variance would occur when there is no change in the RAC (Table 1). Johnson and Fleeger (2009) examined the responses of salt marsh benthic annelids to nutrient additions and large predator removals in two pairs of tidal creeks in Plum Island Estuary, Massachusetts, USA. They found no consistent changes in the abundance of any species and also observed no change in the community composition among treatments, as measured by shifts in NMDS plots and analyzed using ANOSIM.
We hypothesized that no change in mean with an increase in variance should be observed with species reordering, loss, or gain, but only when these changes occur for rare species, or with changes in species evenness (Table 1). Terlizzi et al. (2009) examined changes in the composition of marine benthic mollusk communities that were either located adjacent to or far from gas drilling platforms in the Ionian Sea off of the coast of Italy. They observed no change in mean (as measured by PERMANOVA), but an increase in variance (as measured by PERMDISP, a permutational test for homogeneity of variance) of mollusk communities located adjacent to the gas platforms. This increase in variance was likely caused by greater environmental heterogeneity due to fouling from the drilling platforms onto the sea floor. The increased environmental heterogeneity caused species gains or losses from individual sampling locations (Bevilacqua, personal communication).

**No change in mean, decrease in variance (Fig. 4C)**

We hypothesized that no change in mean with a decrease in variance should be observed with species reordering, loss, or gain, but only when these changes occur for rare species, or with changes in species evenness (Table 1). Houseman et al. (2008) examined changes in species composition in response to a one-time removal of biomass following solarization and herbicide application in grasslands in southwest Michigan, USA. In response to the one-time disturbance, there was no change in the centroid in multivariate community space (as measured by PERMANOVA) with a decrease in the variance (as measured by PERMDISP). This pattern appeared to have been driven by a decrease in species evenness in the disturbed plots.
Change in mean, no change in variance (Fig. 4D)

We hypothesized that a change in mean with no change in variance should be observed with species reordering, loss, or gain, but only when these processes occur to the same extent in all replicates (Table 1). Sitzia et al. (2012) examined changes in the richness and composition of tree species in two Silver Fir forests in Italy that were either abandoned or lightly managed after a common history of heavy exploitation. Sitzia et al. (2012) observed changes in species number and a similar reordering of species with abandonment as compared to plots that were lightly managed, which corresponded with a change in the centroid in multivariate community space (as measured by PERMANOVA) but no change in the variance (as measured by PERMDISP).

Change in mean, increase in variance (Fig. 4E)

We hypothesized that community divergence accompanied by a change in the mean in multivariate community space can be observed with species reordering when a different species becomes dominant in each plot or by differential species gain or loss, which must necessarily be accompanied by species reordering (Table 1). Cooling et al. (2012) examined the effects of the invasive Argentine ant on native ant communities across New Zealand. In ant communities that never had Argentine ants, the native ant community showed little variability between sites (Cooling et al. 2012). However, when Argentine ants invaded an area, the native ant community drastically altered the mean in multivariate community space and increased the variability, as measured by PERMDISP. Cooling et al. (2012) found a decrease in ant species richness with Argentine ant invasions, indicating species loss. The species that were lost due to Argentine ant invasion varied among the plots, resulting in differential species loss and specific species reordering for each plot.

Change in mean, decrease in variance (Fig. 4F)

We hypothesized that community convergence accompanied by a change in the mean in multivariate community space can be observed with species reordering when the same species becomes dominant in each plot or by similar species gains or losses, which must necessarily be accompanied by species reordering (Table 1).

Wolkovich (2010) examined changes in the composition of scrubland arthropod communities in response to invasive grass litter additions in southern California, USA, which resulted in a shift in the centroid in multivariate community space (as measured by PERMANOVA) and a decrease in the variance (as measured by PERMDISP). With this last example, we show how changes in RACs link to changes in community dissimilarity (Fig. 5). The shifts in multivariate space were driven by the systematic loss of less common morphospecies, gain of new morphospecies, and increased abundances of the same morphospecies (i.e., reordering) across all treatment replicates (Fig. 5).

Conclusions and Future Directions

We are in a new age of experimental ecology, where a multitude of studies are being conducted around the world to understand how GCDs are altering communities and ecosystem function. The number of meta-analyses synthesizing the effects of precipitation manipulations (Wu et al. 2010), nutrient additions (Suding et al. 2005, Elser et al. 2007), temperature changes (Rustad et al. 2001, Wu et al. 2010), invasion (Liao et al. 2008), and carbon dioxide increases (Curtis and Wang 1998) on communities are indicative of the plethora of experiments. Many studies focus on measures of ecosystem function, and we suggest that community composition of replicates should also always be recorded, as it is relatively inexpensive and can be very informative. We contend that future community analyses in GCD experiments should assess multivariate patterns of replicates among and within treatments. Our framework presents a generalizable approach to studying the magnitude, predictability and trajectory of community change in response to global change by integrating measures of community dissimilarity with rank abundance curves. Additionally, enhancing our understanding of the predictability of community responses to GCDs might lead to a more predictive framework for ecosystem responses to GCDs, because ecosystem responses are typically driven by changes in community structure (Smith et al. 2009, Isbell et al. 2013, Avolio et al. 2014).

Because many global change studies collect both community and ecosystem data, we can begin to
synthesize across GCD studies and link patterns of community convergence or divergence with alterations in ecosystem functioning using our framework.

Many global change studies only manipulate a single GCD, and seldom compare observed community changes to different GCDs (e.g., comparing N addition to drought). However, such comparisons are informative (Winfree et al. 2009, Murphy and Romanuk 2012, Tilman et al. 2012, Murphy and Romanuk 2014) and experiments are increasingly studying multiple global change drivers in single experiments (Shaw et al. 2002). Future studies should focus on what

Fig. 5. RAC changes for the eight control and treated replicates in the Wolkovich (2010) paper. Each graph is a replicate. The top eight gray line-graphs are the control replicates, and the bottom eight black line-graphs are the treated replicates. Each morphospecies was numbered (39 total species) and its relative abundance was plotted according to its rank in each replicate. A blue number is a morphospecies that was only present in the control replicates, and red number is morphospecies that was only found in the treated replicates.
conditions result in divergence, convergence or parallel change of communities. For instance, does alteration of only one GCD commonly result in patterns of convergence while simultaneous alteration of two or more GCDs commonly results in patterns of divergence? Do particular GCDs (e.g., nitrogen or drought) lead to convergence, divergence or parallel change? Are ecosystems that have greater gamma diversity more prone to change compared with species poor communities? Our framework will facilitate more comparisons and greater synthesis across studies and drivers, because we are less focused on the mechanisms, which can be specific to study systems, organism type, and GCD manipulated. Instead we are interested in patterns of community change, and the generality of these patterns. Ultimately, our new framework will advance the field of community ecology global change studies in two ways: (1) by helping ecologist identify the trajectory, magnitude and predictability of community change and (2) providing a framework to synthesize community responses across ecosystems and GCDs.

ACKNOWLEDGMENTS

We thank LTER Network Office for funding our working group and the National Socio-Environmental Synthesis Center for additional funding. We also thank K. Gross, M. Power, M. Smith, T. Fukami, K. Suding, W. Bowman, E. Gasarch and D. Milchunas for helping develop the initial ideas. Lastly, we are grateful to S. Collins and two anonymous reviewers for constructive and thoughtful comments on our manuscript.

LITERATURE CITED


response to chronic resource alterations induced by 
Suding, K., S. L. Collins, L. Gough, C. M. Clark, E. E. 
Cleland, K. L. Gross, D. G. Milchunas, and S. 
mechanisms explain diversity loss due to N 
fertilization. Proceedings of the National Academy 
of Sciences USA 102:4387–4392.
Terlizzi, A., M. J. Anderson, S. Bevilacqua, S. Fraschet-
higher-level taxa reflect heterogeneity in species 
composition? Diversity and Distributions 15:450– 
458.
Tilman, D., P. B. Reich, and F. Isbell. 2012. Biodiversity 
impacts ecosystem productivity as much as re-
sources, disturbance, or herbivory. Proceedings of 
the National Academy of Sciences USA 109:10394– 
10397.
Tuomisto, H. 2010a. A diversity of beta diversities: 
straightening up a concept gone awry. Part 1. 
Defining beta diversity as a function of alpha and 
gamma diversity. Ecography 33:2–22.
Tuomisto, H. 2010b. A diversity of beta diversities: 
straightening up a concept gone awry. Part 2. 
Quantifying beta diversity and related phenomena. 
Ecography 33:23–45.
Ulrich, W., M. Ollik, and K. I. Ugland. 2010. A meta-
analysis of species–abundance distributions. Oikos 
119:1149–1155.
Vellend, M. 2001. Do commonly used indices of β-
diversity measure species turnover? Journal of 
Walker, L. R., and F. S. Chapin. 1987. Interactions 
among processes controlling successional change. 
Oikos 50:131–135.
2012. mvabund: an R package for model-based 
analysis of multivariate abundance data. Methods 
Weih, E., D. Freund, T. Bunton, A. Stefanski, T. Lee, 
and S. Bentivenga. 2011. Advances, challenges and 
a developing synthesis of ecological community 
assembly theory. Philosophical Transactions of the 
Whittaker, R. H. 1960. Vegetation of the Siskiyou 
Mountains, Oregon and California. Ecological 
Monographs 30:279–338.
Whittaker, R. H. 1965. Dominance and diversity in 
land plant communities: numerical relations of 
species express the importance of competition in 
Winfree, R., R. Aguilar, D. P. Vázquez, G. LeBuhn, and 
M. A. Aizen. 2009. A meta-analysis of bees’ 
responses to anthropogenic disturbance. Ecology 
90:2068–2076.
Wolkovich, E. M. 2010. Nonnative grass litter enhances 
grazing arthropod assemblages by increasing 
Wu, Z., P. Dijkstra, G. W. Koch, J. Peñuelas, and B. A. 
Hungate. 2010. Responses of terrestrial ecosystems 
to temperature and precipitation change: a meta-
analysis of experimental manipulation. Global 
Change Biology 17:927–942.
Zavaleta, E. S., R. Shaw, N. R. Chirriello, B. D. Thomas, 
Grassland responses to three years of elevated 
temperature, CO₂, and N deposition. Ecological 