

# Compensatory dynamics are rare in natural ecological communities

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In population ecology, there has been a fundamental controversy about the relative importance of competition-driven (density-dependent) population regulation vs. abiotic influences such as temperature and precipitation. The same issue arises at the community level; are population sizes driven primarily by changes in the abundances of cooccurring competitors (i.e., compensatory dynamics), or do most species have a common response to environmental factors? Competitive interactions have had a central place in ecological theory, dating back to Gleason, Volterra, Hutchinson and MacArthur, and, more recently, Hubbell's influential unified neutral theory of biodiversity and biogeography. If competitive interactions are important in driving year-to-year fluctuations in abundance, then changes in the abundance of one species should generally be accompanied by compensatory changes in the abundances of others. Thus, one necessary consequence of strong compensatory forces is that, on average, species within communities will covary negatively. Here we use measures of community covariance to assess the prevalence of negative covariance in 41 natural communities comprising different taxa at a range of spatial scales. We found that species in natural communities tended to covary positively rather than negatively, the opposite of what would be expected if compensatory dynamics were important. These findings suggest that abiotic factors such as temperature and precipitation are more important than competitive interactions in driving year-to-year fluctuations in species abundance within communities.

biological interactions | community dynamics | negative covariance | neutral models | zero-sum

A foundational controversy in ecology has centered on the long-term stability of population and community abundance, sometimes called “the balance of nature” (1). Darwin's famous “struggle for existence” on the “entangled bank” poetically expressed Thomas Malthus' principal idea that species' capacity to reproduce greatly exceeds their resources (2). Hence, fierce competition should structure the species and assemblages we see today. Similarly, papers in the 1920s–1950s presented the view that population abundances fluctuate much less than their intrinsic rates of increase would allow (3–5). This observation suggested to early ecologists that populations were regulated by density-dependent factors, and that competition was the most plausible underlying mechanism. In contrast, other authors emphasized abiotic environmental factors as the primary drivers of population fluctuations, often largely in the absence of competition (6–9). Recurring debates about the relative impor-

tance of biotic regulation vs. abiotic forcing have been dubbed “ecology's 12-year cycle” (1, 10).

The same set of issues applies at the community level. Diamond (11), Tilman (12, 13), and Wisheu and Keddy (14), among others, have presented models of plant community structure based on the relative competitive abilities of community members. More recently, Hubbell's (15) unified neutral theory of biodiversity and biogeography similarly, “. . . rests on a key first principle, namely that the interspecific dynamics of ecological communities is a stochastic zero-sum game” (16). That is, the total number of individuals in a community is constant or at least only stochastically varying.

Yet, Cooper (1) points out that arguments about the balance of nature, “attempt to settle questions about what kinds of ecological factors are most important, as determinants of demographic behavior and/or community structure, from a largely *a priori* perspective, with at best a smattering of empirical cases sprinkled in for good measure.” For example, in Hubbell's book (15), support for the zero-sum assumption comes in the form of: (i) an empirical linear relationship between the size of the sampling unit (SU) and the number of individual trees found on a 50-hectare plot on Barro Colorado Island, Panama; and (ii) logical arguments based on finite resources. The empirical relationship is unconvincing because, whereas a positive area–abundance relationship is necessary in a world where biological communities are saturated, it is certainly not sufficient. Hubbell's (15) logical arguments about communities resemble Lack's arguments about individual populations: “Limiting resource availability per unit area will ultimately impose a finite limit on the density of competing organisms within a given ecological community in a defined space”. This argument works only if communities are assumed always to be at or near carrying capacity. Hubbell ends this discussion with the statement of a general principle, that “large landscapes are essentially always

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Abbreviation: SU, sampling unit.

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biotically saturated with individuals" (15). This assertion, in our view, is an open empirical question.

Operationally, theories that postulate strong competitive interactions (such as centrifugal organization and resource partitioning) and neutral model zero-sum community dynamics both imply that communities will show compensatory dynamics. This hypothesis makes a strong prediction. By the strictest definition, it predicts that (as Hubbell states; ref. 15), "any increase in one species must be accompanied by a matching decrease in the collective number of all other species in the community". A less-stringent definition that allows for limited stochastic variation implies that the covariance among population abundances in a community must, on average, be negative.

To test for negative covariance among species, we calculated a community-level measure of covariance (i.e., the sum of all pair-wise covariances). The variance of a sum can be expressed as the sum of all possible variances and covariances (17), providing an analytical tool for detecting compensatory dynamics within communities (17–19). That is, the variance of community abundance can be expressed as the sum of all of the species variances plus all of the pair-wise species covariances. For example, in a two-species community comprising species  $x$  and  $y$ ,  $\sigma^2(x + y) = \sigma^2(x) + \sigma^2(y) + 2\sigma(x, y)$ , where  $\sigma^2(x)$  = the temporal variance in abundance of species  $x$ ,  $\sigma^2(y)$  = the temporal variance in abundance of species  $y$ ,  $\sigma(x, y)$  = the temporal covariance in abundance of species  $x$  and  $y$ , and  $\sigma^2(x + y)$  = the temporal variance in the combined abundance of species  $x$  and  $y$ . More generally, in an  $n$  species community,  $\sigma^2(\sum_{i=1}^n x_i) = \sum_{i=1}^n \sigma^2(x_i) + 2 \sum_{i < j} \sigma(x_i, x_j)$ . Thus, we can compute the total covariance for a community as  $\sum_{i=1}^n \sum_{j < i} \sigma^2(x_j, x_i) = (\sigma^2(\sum_{i=1}^n x_i) - \sum_{i=1}^n \sigma^2(x_i))/2$ . In a community with zero-sum dynamics, the variance of the sum is zero. Because the sum of the variances is a positive number, these simple equations show that the sum of the covariances must be negative. We used this method to calculate community covariance for each of the communities in each of 41 data sets (Table 1).

## Results and Discussion

We estimated the distribution of community covariances in different SU for 41 different plant and animal data sets; 36 of 41 data sets showed >50% positive covariances (Fig. 1). Thirty-one of those had more positive covariances than would be expected by chance alone (assuming zero covariance). Only 3 of 41 data sets had fewer positive covariances than would be expected by chance ( $\alpha = 0.10$ ), all of which were plant communities: trees from the Hubbard Brook Bird Area plots at the quadrat scale ( $25 \times 10$  m), Cedar Creek plants at the transect scale, and Sonoran herbaceous plants at the line scale. Indeed, even the tree community on Barro Colorado Island (BCI) on which Hubbell based his theoretical concepts showed strong positive covariance over six sampling periods from 1981 to 2005 (mean = 232,868; coefficient of variation = 0.052; covariance = 124,105,931) (note: the BCI data are not part of the 41 data sets analyzed here, because we used only data sets that had multiple sites). Our results demonstrate that positive covariance among species is far more common in nature than negative covariance (Fig. 1).

However, the distribution of community covariances is scale-dependent. Nine data sets analyzed here have data for at least two spatial scales and, in eight of nine cases, the smaller spatial scales have an equal or larger proportion of negative covariances (although still <50% in most cases) (Fig. 2), which implies that the factors causing negative covariance between species may be more important at small than large scales.

Why did we observe so few negatively covarying species dynamics? Hubbell (15) suggested three cases in which the zero-sum assumption might not hold: (i) aggregation of taxa that are not at the same trophic level, (ii) a severe disturbance regime

that maintains community abundance at levels below carrying capacity, and (iii) spatial variability in productivity (1). We have constrained our analyses to communities of species of similar trophic status, so inappropriate aggregation is unlikely to be the explanation for the ubiquity of positive covariances. In addition, for two of the data sets (i.e., Rothamsted moths and Wisconsin Lake fish), we were able to analyze subsets of the communities likely to have more similar resource requirements (e.g., planktivorous fish rather than all fish). We would expect that more trophically similar communities would show more negative covariances than less trophically similar communities if a lack of trophic similarity was a cause of the positive covariances, but we found little evidence that this prediction was true (Rothamsted moths, all moths had 105 sites with positive covariance and 0 with negative covariance; Noctuid moths, 105:0; Geometrid moths, 103:2; Ennominaea moths, 95:8; Wisconsin fish, all fish 3:2; zooplanktivores, 3:2; benthivores, 4:1; and piscivores, 2:3).

The second and third explanations are both subsets of a more general possibility consistent with our results, i.e., that the abundances of large sets of species in a community vary in response to a common set of environmental drivers. It would not be surprising that many coexisting species respond similarly to disturbance, temporal variability in productivity, and fluctuations in climatic conditions. Fischer *et al.* (20), for example, showed that compensatory dynamics are limited in communities where species respond in similar ways to changing environmental conditions. Synchronized fluctuations in abundance over time would be reflected in generally positive covariance among abundances of species in communities. Positive covariance suggests that the proposition that "large landscapes are essentially always biotically saturated with individuals" (15) is not true, or at least that the carrying capacity for large landscapes varies dramatically over time, primarily in response to environmental drivers (21).

More broadly, our results show that competition is unlikely to be the primary factor responsible for observed variability in community abundance over time. This does not preclude pairwise negative covariances between individual pairs of species, nor does it rule out differing competitive responses to the primary factors that drive community abundance. Rather, our results suggest that the signature of competition on temporal variability in community abundance is weak compared with the signature of other, probably abiotic, forcing variables.

The results, from many natural communities and across multiple scales, suggest that community-level negative covariance in abundance is generally rare. Our results have important implications for ecological hypotheses that emphasize competition as the primary driver of community dynamics, such as centrifugal (14) and resource partitioning (13) theories, because our results suggest that the primary driver of community dynamics is abiotic environmental forcing, not competition. Our results do not rule out the possibility of "ghosts of competition past" (22, 23), i.e., communities structured to minimize competition, but they strongly suggest that fluctuations in abundance are not driven primarily by competition. An additional caveat is that we have examined communities over relatively short temporal scales. It may be that disturbance events and environmental variability drive community dynamics over short time periods, whereas over longer time periods, competition would lead to more negative covariances.

Similarly, our results also impact the unified neutral theory of biodiversity and biogeography (15). Its assumption of zero-sum community dynamics is apparently not commonly found in nature. That said, neutral model theory and our analyses have focused on the abundance of organisms, but there is theoretical and empirical evidence suggesting that compensatory dynamics are more likely to be seen in variables such as biomass and energy utilization than in abundance (18, 24, 25). It is possible that

**Table 1. Data sets used in covariance analyses**

Data set	Abbreviation	Taxa	Sampling period	Time interval, yr (times sampled)	Scale	Sampling units (mean CV of community abundance)
Harvard Forest, Massachusetts, Lyford Mapped Tree Plot	HLF	Trees	1969–2001	5–17 (4)	920 m <sup>2</sup>	32 (0.212)
Harvard Hurricane Recovery Plot	HHRP	Trees	1937–1991	3–30 (5)	250–1,000 m <sup>2</sup>	14 (0.132)
Harvard Pisgah Forest Plots	HPF	Trees	1984–2001	5–6 (4)	400 m <sup>2</sup>	14 (0.909)
Hubbard Brook, New Hampshire, Bird Area	HBBAQ	Trees	1991–2003	2 (7)	250 m <sup>2</sup>	234 (0.067)
Hubbard Brook, New Hampshire, Watershed 6	HBW6	Trees	1991–2003	2 (7)	10,000 m <sup>2</sup>	4 (0.010)
Hubbard Brook, New Hampshire, Watershed 6	HBW6	Trees	1965–2002	5–12 (7)	625 m <sup>2</sup>	208 (0.362)
Sonora	SPL	Herbs	1949–1992	1–14 (21)	0.09 m <sup>2</sup>	288 (0.777)
	SPP	Herbs	1949–1992	1–14 (21)	1,200-m transect	24 (0.602)
	SPPa	Herbs	1949–1992	1–14 (21)	Pasture (various sizes)	8 (0.523)
Portal, New Mexico	PWAS	Winter annuals	1989–2002	1 (14)	0.25 m <sup>2</sup>	384 (1.782)
	PWAP	Winter annuals	1989–2002	1 (14)	2,500 m <sup>2</sup>	24 (1.226)
	PSAS	Summer annuals	1989–2002	1 (14)	0.25 m <sup>2</sup>	384 (1.715)
	PSAP	Summer annuals	1989–2002	1 (14)	2,500 m <sup>2</sup>	24 (1.839)
Big Bend, Texas	BBPT	All plants	1955–1996	6–15 (5)	1.85 m <sup>2</sup>	51 (1.049)
	BBPS	All plants	1955–1996	6–15 (5)	5.5 m <sup>2</sup>	17 (1.118)
	BBPeT	Perennials	1955–1996	6–15 (5)	1.85 m <sup>2</sup>	51 (1.054)
	BBPeS	Perennials	1955–1996	6–15 (5)	5.5 m <sup>2</sup>	17 (1.123)
Cedar Creek, Minnesota	CCF	All plants	1988–1998	1 (11)	Field (various sizes)	14 (0.367)
	CCT	All plants	1988–1998	1 (11)	12 m <sup>2</sup>	52 (0.438)
Jornada, New Mexico	JPQ	All plants	1989–2002	1 (14)	1 m <sup>2</sup>	734 (0.524)
	JPS	All plants	1989–2002	1 (14)	6,400 m <sup>2</sup>	15 (0.380)
	JPZ	All plants	1989–2002	1 (14)	Vegetation zones (various sizes)	5 (0.339)
Rothamsted, United Kingdom	RM	Moths	1965–2003	1 (10–39)	Light-trap	105 (1.264)
	RE	Ennomiids	1965–2003	1 (10–39)	Light-trap	103 (0.460)
	RG	Geometrids	1965–2003	1 (10–39)	Light-trap	105 (0.403)
	RN	Noctuids	1965–2003	1 (10–39)	Light-trap	105 (0.406)
Wisconsin LTER Lakes	WF	Fish (all trophic levels)	1981–2001	1 (21)	Whole-lake	5 (1.015)
	WB	Benthivore fish	1981–2001	1 (21)	Whole-lake	5 (0.883)
	WP	Piscivore fish	1981–2001	1 (21)	Whole-lake	5 (0.553)
	WZ	Zooplanktivore fish	1981–2001	1 (21)	Whole-lake	5 (1.269)
Konza LTER, Kansas	KMSL	Small mammals (spring)	1982–1997	1 (16)	300-m trapline (20 traps)	14 (0.917)
	KMSW	Small mammals (spring)	1982–1997	1 (16)	Watershed	7 (0.840)
	KMAL	Small mammals (autumn)	1982–1997	1 (16)	300-m trapline (20 traps)	12 (0.714)
	KMAW	Small mammals (autumn)	1982–1997	1 (16)	Watershed	6 (0.663)
Konza LTER, Kansas	KG	Grasshoppers	1982–1991 and 1996–2003	1 (7–19)	Watershed	10 (1.250)
Sevilleta, New Mexico	SSpR	Rodents (spring)	1989–2003	1 (15)	444 trap grid	6 (0.798)
	SSR	Rodents (summer)	1989–2003	1 (15)	444 trap grid	6 (0.777)
United Kingdom Butterfly Monitoring Scheme	BMS	Butterflies	1976–2002	1 (10–27)	2,000–4,000 m	58 (0.539)
Cedar Creek, Minnesota	CCG	Grasshoppers	1989–1998	1 (10)	Field (various sizes)	20 (0.697)
Jornada, New Mexico	JM	Small mammals	1989–1994	1 (6)	1,000-m transect	5 (1.362)
Jornada, New Mexico	JR	Reptiles	1989–1994	1 (6)	1,000-m transect	5 (0.721)

LTER, National Science Foundation Long-Term Ecological Research; CV, coefficient of variation.

zero-sum assumptions about biomass and energy use may be more consistent with patterns seen in nature than zero-sum abundance assumptions.

The neutral model of molecular evolution has demonstrated that neutral theories can avoid the zero-sum assumption (26), but it has done so by incorporating the concept of “effective population size”





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