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# Productivity Is a Poor Predictor of Plant Species Richness

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For more than 30 years, the relationship between net primary productivity and species richness has generated intense debate in ecology about the processes regulating local diversity. The original view, which is still widely accepted, holds that the relationship is hump-shaped, with richness first rising and then declining with increasing productivity. Although recent meta-analyses questioned the generality of hump-shaped patterns, these syntheses have been criticized for failing to account for methodological differences among studies. We addressed such concerns by conducting standardized sampling in 48 herbaceous-dominated plant communities on five continents. We found no clear relationship between productivity and fine-scale (meters<sup>-2</sup>) richness within sites, within regions, or across the globe. Ecologists should focus on fresh, mechanistic approaches to understanding the multivariate links between productivity and richness.

For more than three decades, ecologists have debated the role of primary productivity in regulating plant species richness at fine spatial scales (1, 2). Although some studies have advocated multivariate approaches (3–5), much of the debate remains focused on evidence for a single, general relationship between productivity and richness. This classic productivity–richness relationship (PRR) is hump-shaped, with richness increasing at low to intermediate levels of productivity and decreasing at

high productivity (6). The mechanisms invoked to explain the decreasing phase of the PRR in terrestrial plant communities have attracted the greatest controversy and include disturbance (3, 7), competitive exclusion mediated by shifts in the identity or heterogeneity of limiting resources (8–10), and evolutionary history and dispersal limitation (11).

However, the theoretical justification for a hump-shaped PRR has been challenged (12), and the empirical evidence is mixed. For ex-

ample, recent meta-analytical syntheses concluded that evidence for a single, canonical pattern was weak (13–15). A large percentage of studies exhibited negative, U-shaped, or nonsignificant PRRs in addition to unimodal and positive linear patterns, and the frequency of these various patterns depended on taxon and spatial scale. Subsequent critiques of the meta-analyses argued that the apparent lack of generality in PRRs might simply reflect methodological inconsistencies among the field studies (16, 17). First, PRR studies vary widely in their choice of both the grain (the area of the sampling unit) and extent (the area over which sampling units are spread) (15, 18). Because of the strong effects of area and heterogeneity on richness, such differences in scale confound cross-study comparisons (19). Second, many of the studies included in PRR meta-analyses did not measure primary production directly but used weakly related surrogates such as latitude, temperature, or altitude (14).

We assessed the generality of the PRR and addressed previous methodological inconsistencies by conducting standardized, observational sampling in 48 herbaceous-dominated plant communities on five continents (Fig. 1 and table S1) (20). We sampled plant species richness in standard 1-m<sup>2</sup> quadrats located in blocks of 10 plots, holding grain constant and minimizing differences in extent across sites. In addition, we used the same protocol at all sites for estimating aboveground net primary production (ANPP) as peak-growing-season live biomass, an effective measure of ANPP in herbaceous vegetation (21), especially when consumption by herbivores is low (fig. S1).

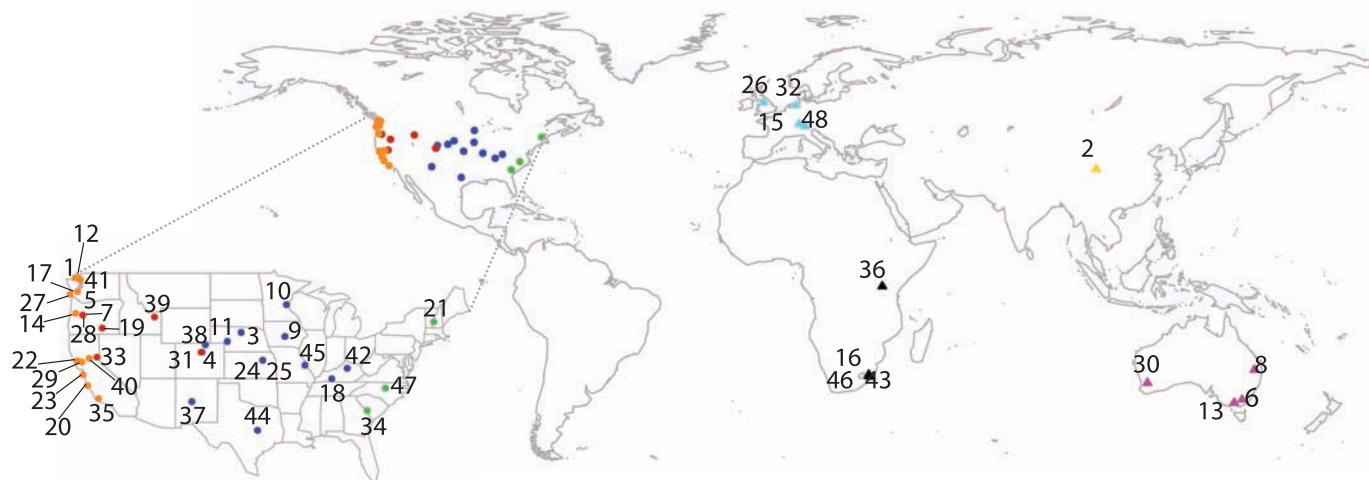
Previous work indicated that the form of the PRR might vary with the spatial extent of sampling. Although significant PRRs have been observed at spatial extents ranging from individual

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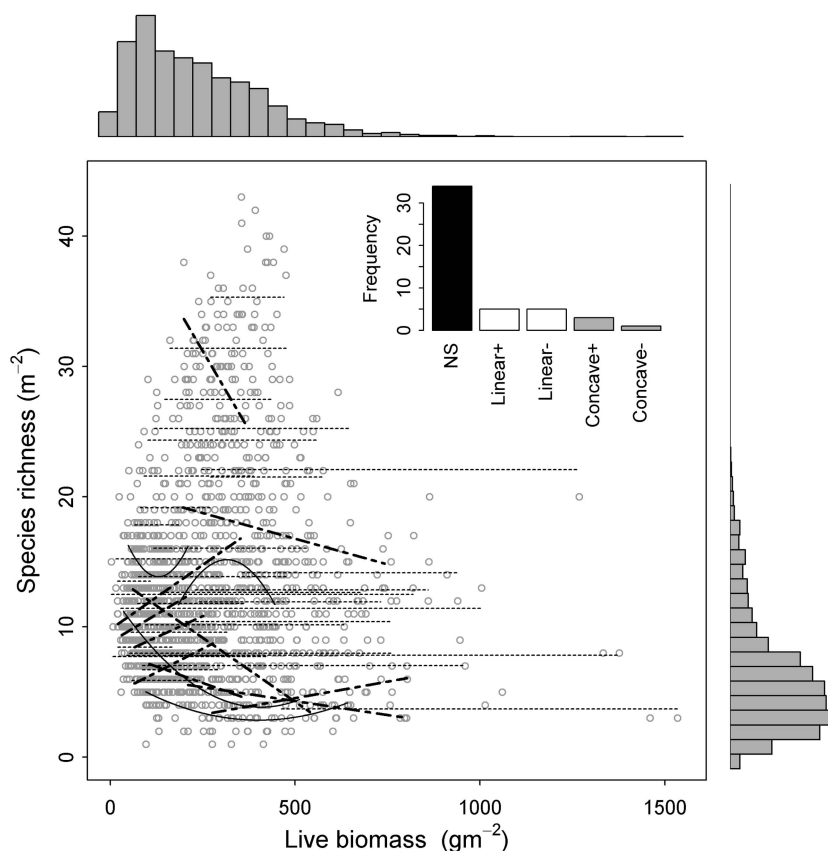
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**Fig. 1.** Locations of the 48 Nutrient Network sites that provided data for this study. Numbers correspond to the “code” column in table S1. Colors and symbols represent the distinct biogeographic regions also shown in Fig. 3 (see Fig. 3 for key).



**Fig. 2.** Within-site relationships between productivity, measured as peak live biomass (dry weight) and species richness. The inset shows the frequencies of relationships that were nonsignificant (NS, thin dashed lines), positive or negative linear (thick dashed lines), and concave-up (+) or down (–) (solid curves). Statistical results and separate figures for each of the 48 sites are available in table S2 and fig. S1, respectively. The marginal histograms show the frequency of species richness and peak live biomass across all sites.

plots located within one community to means of sites spread across continents, the hump-shaped pattern has emerged most frequently in studies that cross community boundaries (14, 22). PRRs described within communities may be weaker

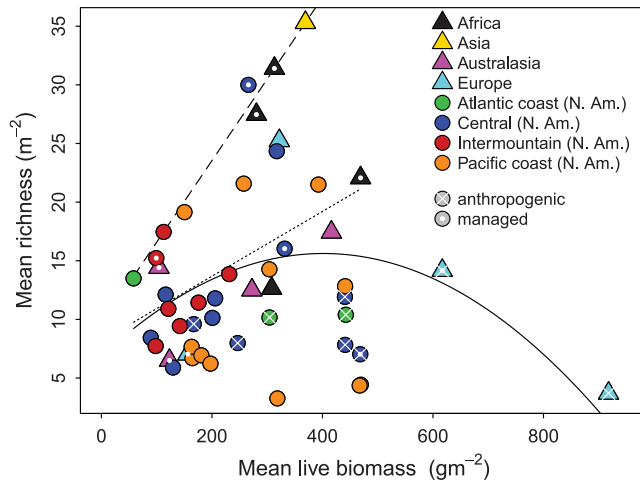
because of the potential for limited variation in productivity among sampling plots as well as measurement error on individual samples (22) and because mechanisms involving variation in species pools and dispersal are excluded. We

tested for scale-dependence by characterizing the shape of the PRR at three spatial extents: (i) The within-site extent compares richness and productivity sampled in individual plots; (ii) the regional extent compares site-level averages for 1-m<sup>2</sup> richness and productivity among sites occurring within a biogeographic province; and (iii) the global extent compares site-level averages for richness and productivity among all sites.

The 48 within-site PRRs took all possible shapes in parametric regressions of species richness on productivity (Fig. 2, fig. S2, and tables S2 and S3). The most common relationship was nonsignificant (34 sites), 5 sites had a positive linear pattern, 5 sites had a negative linear pattern, 3 sites were concave-up (U-shape), and 1 site was concave-down (the classical hump shape). Repeating this analysis with quasipoisson regression (20) gave similar results (34 nonsignificant, 5 positive linear, 6 negative linear, 2 concave-up, and 1 concave-down). We did not find factors that explained the variation in the shape of the within-site PRRs. For example, if unproductive sites had positive linear PRRs and highly productive sites had negative PRRs, then there should be a correlation between site-level productivity and the slope of the within-site linear relationship (18). We found no such pattern (correlation coefficient  $r = 0.07$ ,  $df = 46$ ,  $P = 0.62$ ), nor were sites that spanned larger ranges in productivity more likely to show significant PRRs. Specifically, the probability of finding a non-null PRR was unrelated to the range of ANPP within a site (logistic regression  $P = 0.20$ ).

We tested the regional relationship between site-level-average species richness (meters<sup>-2</sup>) and average biomass production in the three biogeographic provinces of North America in which we had more than four sites (Fig. 3). For the 11 Pacific coast sites, located west of the Cascade/Sierra Mountain ranges and dominated by non-native species (along with one salt

**Fig. 3.** Global relationship between mean productivity, measured as peak live biomass (dry weight), and mean species richness (meters<sup>-2</sup>) at each site. White dots indicate managed sites (burned regularly or grazed by domestic livestock) and crosses indicate sites of anthropogenic origin (pastures, old fields, and restored prairies). The solid curve shows the quadratic relationship between productivity and richness with all sites included; the dotted line shows the linear relationship that remains when the anthropogenic sites are removed; and the dashed line shows the 0.95 quantile regression with all sites included. N. Am., North America.



marsh), there was no significant quadratic ( $t = -1.0$ ,  $P = 0.33$ ) or linear ( $t = -0.27$ ,  $P = 0.79$ ) effect of productivity on richness. Removing the highly productive salt marsh site did not change this result. Results for the seven Intermountain West sites located between the Cascade/Sierra and Rocky Mountains were similar: Neither the quadratic ( $t = 0.52$ ,  $P = 0.63$ ) nor linear ( $t = 0.14$ ,  $P = 0.89$ ) effects of productivity were significant, and removing the one site grazed by domestic livestock did not change this result. For the 13 Central Region grassland sites east of the Rockies and west of the Appalachian Mountains, we did find evidence of a hump shape, with a significant quadratic effect of productivity on richness ( $t = -2.35$ ,  $P = 0.041$ ). However, when we removed five sites of anthropogenic origin (restored prairies, pastures, or old fields), the quadratic term was no longer significant ( $t = -0.177$ ,  $P = 0.87$ ), whereas the linear term was significant ( $t = 2.5$ ,  $P = 0.046$ ).

At the global extent (Fig. 3), the quadratic effect of productivity on richness was significant ( $t = -2.39$ ,  $P = 0.021$ ). However, this hump-shaped model, which ignored uncertainty in estimates of site means, explained little variation in average species richness (coefficient of determination  $R^2 = 0.11$ ). Furthermore, the pattern was sensitive to land-use history. When we removed nine sites of anthropogenic origin and the one salt marsh, the quadratic effect was no longer significant ( $t = -1.36$ ,  $P = 0.18$ ), but a positive linear effect was significant ( $t = 2.61$ ,  $P = 0.013$ ).

An alternative hypothesis states that productivity sets the upper limit on richness, with stochastic forces such as disturbance causing deviations below this limit (3, 23). We tested for a hump-shaped constraint on maximum richness by conducting quantile regressions on our data at within-site and global extents (we did not have sufficient data to address the regional extent). At the within-site extent, results for the 0.95

quantile regressions were similar to our standard regression analysis, with 39 nonsignificant tests, 2 positive linear, 5 negative linear, 1 concave-up, and 1 concave-down pattern (fig. S2). The use of lower quantiles (0.7, 0.8, 0.9) generated fewer significant PRRs. At the global extent (Fig. 3), the quadratic effect was not significant ( $t = -1.63$ ,  $P = 0.11$ ); instead, a positive linear trend emerged ( $t = 2.19$ ,  $P = 0.034$ ). Testing the relationship between mean productivity at a site and maximum richness observed at that site (fig. S3) produced a similar nonsignificant quadratic effect ( $t = -1.50$ ,  $P = 0.14$ ) and marginally significant linear effect ( $t = 2.01$ ,  $P = 0.051$ ).

Overall, we found no consistent, general relationship between productivity and richness of herbaceous-dominated plant communities at the local, regional, or global extent. When we used both standard and quantile regressions, nonsignificant relationships were most common. Although linear or hump-shaped patterns occurred in particular cases, no strong correlates explained these idiosyncrasies. Furthermore, consideration of land-use history and management changed the form of the regional and global scale relationships. Despite using consistent and appropriate data-collection methods, our results show even less support for a general PRR than did previous synthesis efforts based on meta-analysis (13–15), indicating that inadequate or noncomparable data are not the explanation for the lack of a general PRR.

If theory provided a strong prediction for the form of the PRR, then deviations from the expected pattern would be informative. However, ecologists have proposed many competing models that predict every form of the PRR (12). Furthermore, recent work has emphasized that productivity does not have a direct, mechanistic effect on fine-scale species richness, but rather a complex set of interactions links the two variables (5, 24). For example, productivity and

richness each respond to the supply rate as well as the stoichiometry of resources (25–27), with variation in these factors leading to different forms of the PRR. In addition, richness may respond more strongly to disturbance, habitat heterogeneity, and biogeographic and assembly history (3, 11, 28–30) than to productivity. Finally, richness is not simply a function of productivity but it may feed back to influence productivity (31). The weak and variable PRRs we found are consistent with these hypotheses.

Rather than investing continued effort in attempting to identify a general PRR, ecologists should focus on more sophisticated approaches already available for investigating the complex, multivariate processes that regulate both productivity and richness (5, 25, 26). Coordinated, global networks represent a research approach that will be invaluable not only for addressing longstanding debates about the generality of empirical patterns but also for testing the underlying mechanisms.

#### References and Notes

1. J. P. Grime, *Nature* **242**, 344 (1973).
2. M. M. Al-Mufti, C. L. Sydes, S. B. Furness, J. P. Grime, S. R. Band, *J. Ecol.* **65**, 759 (1977).
3. M. Huston, *Am. Nat.* **113**, 81 (1979).
4. M. A. Huston, *Biological Diversity: The Coexistence of Species on Changing Landscapes* (Cambridge Univ. Press, Cambridge, 1994).
5. J. B. Grace *et al.*, *Ecol. Lett.* **10**, 680 (2007).
6. M. L. Rosenzweig, Z. Abramsky, in *Species Diversity in Ecological Communities* (Univ. of Chicago Press, Chicago, 1993), pp. 52–65.
7. M. Kondoh, *Proc. Biol. Sci.* **268**, 269 (2001).
8. D. Tilman, *Resource Competition and Community Structure* (Princeton Univ. Press, Princeton, NJ, 1982).
9. D. E. Goldberg, T. E. Miller, *Ecology* **71**, 213 (1990).
10. D. Tilman, S. Pacala, in *Species Diversity in Ecological Communities* (Univ. of Chicago Press, Chicago, 1993), pp. 13–25.
11. M. Zobel, M. Pärtel, *Glob. Ecol. Biogeogr.* **17**, 679 (2008).
12. P. A. Abrams, *Ecology* **76**, 2019 (1995).
13. R. Waide *et al.*, *Annu. Rev. Ecol. Syst.* **30**, 257 (1999).
14. G. Mittelbach *et al.*, *Ecology* **82**, 2381 (2001).
15. L. N. Gillman, S. D. Wright, *Ecology* **87**, 1234 (2006).
16. R. J. Whittaker, E. Heegaard, *Ecology* **84**, 3384 (2003).
17. R. J. Whittaker, *Ecology* **91**, 2522 (2010).
18. S. M. Scheiner *et al.*, *Evol. Ecol. Res.* **2**, 791 (2000).
19. N. Gotelli, R. Colwell, *Ecol. Lett.* **4**, 379 (2001).
20. Materials and methods are available as supporting material on Science Online.
21. M. Oesterheld, S. J. McNaughton, in *Methods in Ecosystem Science*, O. E. Sala, R. B. Jackson, H. A. Mooney, R. W. Howarth, Eds. (Springer, New York, 2000), pp. 151–157.
22. J. Grace, H. Jutila, *Oikos* **85**, 398 (1999).
23. M. Huston, *Oikos* **86**, 393 (1999).
24. W. Ma *et al.*, *Glob. Ecol. Biogeogr.* **19**, 233 (2010).
25. B. J. Cardinale, D. M. Bennett, C. E. Nelson, K. Gross, *Ecology* **90**, 1227 (2009).
26. B. J. Cardinale, H. Hillebrand, W. S. Harpole, K. Gross, R. Ptacnik, *Ecol. Lett.* **12**, 475 (2009).
27. W. S. Harpole, D. Tilman, *Nature* **446**, 791 (2007).
28. J. M. Chase, M. A. Leibold, *Nature* **416**, 427 (2002).
29. T. Fukami, P. J. Morin, *Nature* **424**, 423 (2003).
30. J. M. Chase, *Science* **328**, 1388 (2010); 10.1126/science.1187820.
31. D. Hooper *et al.*, *Ecol. Monogr.* **75**, 3 (2005).

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#### Supporting Online Material

www.sciencemag.org/cgi/content/full/333/6050/1750/DC1  
Materials and Methods

Figs. S1 to S3  
Tables S1 to S3  
References (32–34)

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# African Wild Ungulates Compete with or Facilitate Cattle Depending on Season

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Savannas worldwide are vital for both socioeconomic and biodiversity values. In these ecosystems, management decisions are based on the perception that wildlife and livestock compete for food, yet there are virtually no experimental data to support this assumption. We examined the effects of wild African ungulates on cattle performance, food intake, and diet quality. Wild ungulates depressed cattle food intake and performance during the dry season (competition) but enhanced cattle diet quality and performance during the wet season (facilitation). These results extend our understanding of the context-dependent–competition-facilitation balance, in general, and are critical for better understanding and managing wildlife-livestock coexistence in human-occupied savanna landscapes.

Savannas cover ~20% of the global land surface and occur more extensively in Africa than in any other continent (1). These ecosystems vitally support large proportions of the world's human, livestock, and wildlife populations (1). In savannas worldwide—and especially in the ungulate-rich African savannas (2)—domestic and wild herbivores commonly share food and other resources. Such sharing of habitat by guilds of herbivores can result in varied interactions ranging from negative (competition) to positive (facilitation) (3).

In savanna rangelands worldwide, management decisions are based on the supposition that wild fauna and domestic stock compete for forage resources, but there are little experimental data to support this assumption. For competition to occur, a shared resource must be in short supply, and its joint exploitation by two or more herbivore species must lead to reduced performance (such as survivorship, fecundity, or weight gain) of at least one species (3). Although changes in several factors—including food availability, quality, and intake—can alter herbivore performance, a change in one or more of these factors without an effect on performance of the species involved is not in itself evidence of competition (3).

The food habits of domestic and wild ungulates—and dietary overlap between these herbivore guilds—have been studied widely (4–7). In addition, the effects of wildlife on livestock food habits and foraging patterns have been documented (8, 9). However, the critical assessment of whether or not wild ungulates alter livestock

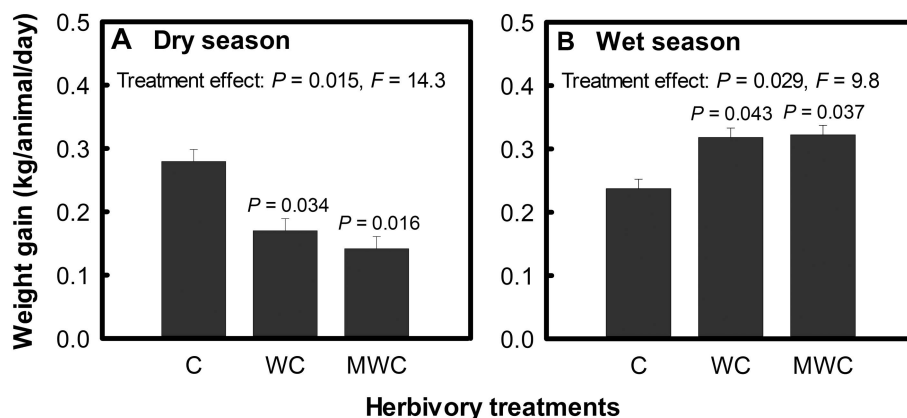
performance has rarely been carried out, and never in a tropical savanna biome. Yet, such an appraisal is urgently needed to guide management efforts toward enhancing wildlife-livestock coexistence in human-occupied landscapes, especially in the African savanna biome, which hosts the last remnants of an intact large herbivore fauna.

We used a controlled replicated experiment to assess whether or not medium-sized wild ungulates (>20 kg; plains zebra *Equus burchelli*, Grevy's zebra *E. grevyi*, African buffalo *Syncerus caffer*, eland *Tragelaphus oryx*, hartebeest *Acelaphus buselaphus*, oryx *Oryx gazella*, and Grant's gazelle *Gazella granti*) and megaherbivores (African elephant *Loxodonta africana* and giraffe *Giraffa camelopardalis*) compete with cattle in a savanna ecosystem in Kenya. Specifically, we hypothesized that if these ungulates compete with cattle, food availability and quality should decrease in the shared foraging areas, resulting in reductions in food intake, diet quality, and most importantly,

weight gain of cattle. Additionally, we hypothesized that these effects would reduce after experimental exclusion of megaherbivores, especially elephants, because of their documented seasonal resource overlap with cattle (10). Last, we expected greater competitive effects during the dry season, when food is less abundant.

We compared cattle weight gain, organic matter food intake (OMI), diet selection, dietary digestible organic matter (DOM), crude protein (CP), DOM/CP ratio, and herbage cover in treatment plots that cattle accessed exclusively and those they shared with wild ungulates, excluding or including megaherbivores, during wet and dry seasons (11). Consistent with our hypothesis, cattle experienced depressed weight gain when they shared foraging areas with wild herbivores during the dry season (Fig. 1A), providing evidence of competition. In contrast, this pattern was reversed in the wet season, with increased cattle performance in the shared treatments (Fig. 1B), demonstrating a surprising facilitative interaction that was nearly great enough to overcome the preceding season's competition.

Competition was associated with depressed food intake in the shared treatments (Table 1), which corresponded with reductions in cover and selection by cattle of *Pennisetum stramineum* (Fig. 2, A to C), suggesting that wildlife and cattle competed for this grass. For all other major herbaceous species, cover was not significantly different among herbivore treatments (table S1). Relative bites on *Themeda triandra* increased in the treatment accessible to all three guilds of herbivores during wet season, but no other major plant species showed treatment effects on either relative bites or selection by cattle (tables S2 and



**Fig. 1.** Weight gain of cattle within treatment plots they accessed exclusively (C) and those they shared with wild herbivores, with megaherbivores absent (WC) or present (MWC). (A) During dry season. (B) During wet season. Error bars are SEM ( $n = 3$  experimental blocks). The  $P$  values over the WC and MWC treatments are for comparisons with treatment C (Tukey's post hoc test).

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animal viruses, parasites, and other pathogens to their cellular hosts, binding of bacteria to plants and of pollen to the plant stigma, binding of amphibian and marine sperm to eggs, and sexual agglutination in yeast. All of these events are considered to be mediated by glycans (6).

The results reported by Pang *et al.* raise a number of questions. The functional analyses were carried out with SLeX from total human ZP, not with SLeX from individually purified ZP glycoproteins. It is thus unclear whether sperm bind to all human ZP glycoproteins (ZP1 to ZP4) containing SLeX or whether binding is restricted to one or more of them. In this context, the finding that SLeX that is covalently linked to BSA is orders of magnitude more effective than SLeX alone as an inhibitor of sperm binding to eggs suggests that the oligosaccharide's orientation may be affected by the polypeptide to which it is linked (10, 12). Indeed, the binding of mouse sperm to egg ZP oligosaccharides is influenced by the polypeptide to which they are

linked (13). Because Pang *et al.* conclude that the effectiveness of SLeX in sperm binding depends on the presence of a terminal sialic acid, it is uncertain whether the monosaccharide's negative charge is responsible for binding. It is likely that SLeX does not inhibit binding of mouse sperm to eggs because of the negative charge introduced by sialic acid (10). Whether SLeX can block the binding of other species of mammalian sperm to homologous eggs also remains unanswered; this would bear on the issue of species specificity during mammalian fertilization.

Another issue concerns the nature of the egg-binding proteins on human sperm that recognize and bind to SLeX. Perhaps derivatives of SLeX could be used as effective probes to tag the proteins. Even in the well-studied case of the binding of mouse sperm to the egg's ZP, the nature of the egg-binding proteins remains contentious (14). Despite these lingering issues, the study by Pang *et al.* should stimulate considerable interest in the molecular basis of sperm-egg interaction in

humans and may ultimately lead to development of new contraceptives.

#### References

1. R. Yanagimachi, in *Physiology of Reproduction*, E. Knobil, J. D. Neill, Eds. (Raven, New York, 1994), vol. 1, pp. 189–318.
2. H. M. Florman, T. Ducibella, in *Physiology of Reproduction*, J. D. Neill, Ed. (Academic Press, New York, 2006), vol. 1, pp. 55–112.
3. P.-C. Pang *et al.*, *Science* **333**, 1761 (2011); 10.1126/science.1207438.
4. A. Varki *et al.*, *Essentials of Glycobiology* (Cold Spring Harbor Laboratory, Cold Spring Harbor, New York, ed. 2, 2008).
5. P. M. Wassarman, *J. Biol. Chem.* **283**, 24285 (2008).
6. L. Jovine, C. C. Darie, E. S. Litscher, P. M. Wassarman, *Annu. Rev. Biochem.* **74**, 83 (2005).
7. M. Fukuda *et al.*, *J. Biol. Chem.* **260**, 12957 (1985).
8. H. M. Florman, P. M. Wassarman, *Cell* **41**, 313 (1985).
9. E. S. Litscher *et al.*, *Biochemistry* **34**, 4662 (1995).
10. C. L. Kerr, W. F. Hanna, J. H. Shaper, W. W. Wright, *Biol. Reprod.* **71**, 770 (2004).
11. D. S. Johnston *et al.*, *J. Biol. Chem.* **273**, 1888 (1998).
12. J. P. Carver, S. W. Michnick, A. Imberty, D. A. Cumming, *Ciba Found. Symp.* **145**, 6, discussion 18 (1989).
13. L. Han *et al.*, *Cell* **143**, 404 (2010).
14. B. D. Shur, *Int. J. Dev. Biol.* **52**, 703 (2008).

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## ECOLOGY

# Biodiversity and Productivity

Michael R. Willig

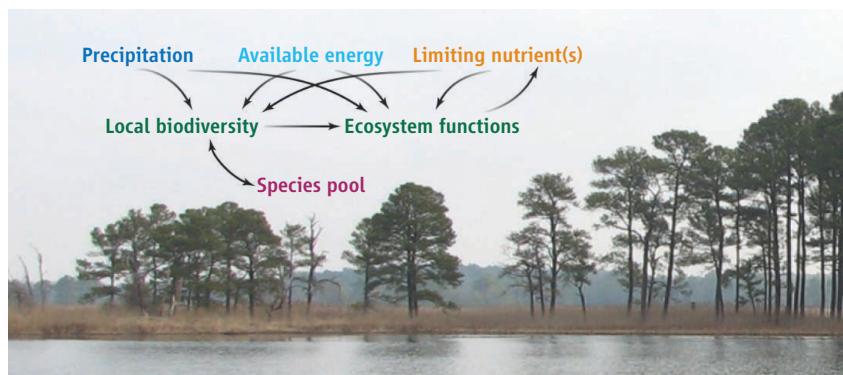
Researchers predict that human activities—especially landscape modification and climate change—will have a considerable impact on the distribution and abundance of species at local, regional, and global scales in the 21st century (1, 2). This is a concern for a number of reasons, including the potential loss of goods and services that biodiversity provides to people (3, 4). Exactly how biodiversity relates to ecosystem function and productivity, however, has been a widely studied and highly controversial issue over the past few decades. Initially, for example, many researchers believed that the relationship between species richness and net primary

productivity (often expressed as the number of grams of carbon produced within a square meter of an ecosystem over a year) could be visualized as a hump-shaped or modal curve (5), with richness first rising and then declining with increasing productivity. However, subsequent theoretical and empirical research, including meta-analyses, seriously diminished acceptance of the modal pattern

The relationship between species richness and ecosystem productivity appears to be very complex.

as a canonical relationship (6–11). On page 1750 of this issue, Adler *et al.* (12) carry the critique a step further. In a multiscale assessment of 48 plant communities on five continents, they demonstrate that the modal productivity-diversity pattern is quite rare in nature, rather than the dominant relationship. Their findings suggest that ecological understanding may advance more rapidly if investigators focus on exploring a range of topics that are germane to the productivity-diversity relationship in a changing world, rather than continue the search for a dominant pattern.

One topic that merits attention is developing a better understanding of the concepts underlying gradients in species richness, over both space and time. In general, biodiversity gradients can appear (13, 14) if (i) one or more limiting resources differ in



**Complex relationship.** The relationship between biodiversity (e.g., species richness) and associated ecosystem functions (e.g., net annual primary productivity) is governed by a suite of abiotic and climatic characteristics, as well as biotic feedback. To fully understand the underlying mechanistic bases for the biodiversity-productivity relationship and to predict how it might respond to climatic change and land use change, an effective synoptic network must minimally estimate these characteristics at multiple sites and scales, and must do so over the long term.

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time or space; (ii) more individuals lead to more species, given a uniform environment of fixed area; (iii) the variance of an environmental characteristic increases with its mean within an area of fixed size or time of fixed duration; and (iv) nonmonotonic relationships require trade-offs in organismal, population, or species characteristics with respect to the environmental gradient (e.g., competitive ability versus stress tolerance or competitive versus colonizing ability). By identifying critical trade-offs, researchers can identify contexts, including both times and places, in which modal patterns may be most likely to occur in natural settings and distinguish them from places and times in which monotonic patterns may be expected. Alternatively, the absence of a modal pattern suggests the absence of defining trade-offs. Finally, for a modal pattern to emerge, the trade-offs must be strong and pervasive. If the biota is large, comprising many species of different physiology or life history, a single dominant trade-off may not be in operation, reducing the likelihood of detecting a modal pattern.

A second issue is that biodiversity has multiple dimensions; species richness is but one of its many attributes. Areas that are ripe for investigation include the way in which productivity varies with other components of the taxonomic dimension of biodiversity, such as species evenness, diversity, or rarity (15), or the way in which other dimensions of biodiversity (e.g., functional, phylogenetic, genetic, or trait) vary with productivity (16, 17). Such comparisons may be useful in identifying causal mechanisms affecting empirical patterns from both ecological and evolutionary perspectives.

A final topic is the role of multicausality in a complex world. For example, although variation in available energy may mold patterns of species richness (and other attributes of biodiversity), variation in species richness (and other attributes of biodiversity) may in turn mold patterns of plant productivity. Each of these attributes may also respond to other driving factors, both environmental (e.g., energy, temperature, and precipitation) and evolutionary (e.g., size and composition of species pools). It should not be surprising that the relationship between biodiversity and productivity is complex, scale dependent, and context specific in nature.

Understanding the consequences of changes in land use and species richness at multiple scales may be critical for long-term sustainability, because these changes will affect the relationship between biodiversity and ecosystem functions. In this regard, Adler *et al.*

are correct in arguing for the establishment of large-scale environmental networks or global biodiversity observatories. To address global patterns and overarching conceptual issues, such as the relationship between assemblage structure and ecosystem function, networks must be coordinated and synoptic in nature (18, 19), measuring similar characteristics in similar ways at a variety of spatial scales. They also must be supported by cyberinfrastructure and connected to other networks and evolving databases, such as GenBank ([www.ncbi.nlm.nih.gov](http://www.ncbi.nlm.nih.gov)), TreeBASE (a phylogenetic data source at [www.treebase.org](http://www.treebase.org)), or TraitNet (a repository for trait characteristics at <http://traitnet.ecoinformatics.org>). The answers to some of the greatest challenges facing society may depend on sustained support of biodiversity observatories that are designed to address the relationships between the multiple dimensions of biodiversity and a suite of ecosystem functions that provide critical services of value to humans.

#### References and Notes

1. C. D. Thomas *et al.*, *Nature* **427**, 145 (2004).
2. Intergovernmental Panel on Climate Change, *Climate Change and Biodiversity* (IPCC, New York, 2002).

10.1126/science.1212453

3. Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Biodiversity Synthesis* (World Resources Institute, Washington, DC, 2005).
4. R. Haines-Young, *Land Use Policy* **26**, S178 (2009).
5. M. L. Rosenzweig, Z. Abramsky, in *Species Diversity in Ecological Communities*, R. E. Ricklefs, D. Schluter, Eds. (University of Chicago Press, Chicago, 1993), pp. 13–25.
6. G. G. Mittelbach *et al.*, *Ecology* **82**, 2381 (2001).
7. J. B. Grace *et al.*, *Ecol. Lett.* **10**, 680 (2007).
8. L. N. Gillman, S. D. Wright, *Ecology* **87**, 1234 (2006).
9. R. B. Waide *et al.*, *Annu. Rev. Ecol. Syst.* **30**, 257 (1999).
10. R. J. Whittaker, E. Heegaard, *Ecology* **84**, 3384 (2003).
11. R. J. Whittaker, *Ecology* **91**, 2522 (2010).
12. P. B. Adler *et al.*, *Science* **333**, 1750 (2011).
13. G. A. Fox *et al.*, in *Theory of Ecology*, S. M. Scheiner, M. R. Willig, Eds. (University of Chicago Press, Chicago, 2011), pp. 283–307.
14. S. M. Scheiner, M. R. Willig, *Am. Nat.* **166**, 458 (2005).
15. D. R. Chalcraft, B. J. Wilsey, C. Bowles, M. R. Willig, *Biodivers. Conserv.* **18**, 91 (2009).
16. M. W. Cadotte, J. Cavender-Bares, D. Tilman, T. H. Oakley, *PLoS ONE* **4**, e5695 (2009).
17. D. F. B. Flynn *et al.*, *Ecology* **92**, 1573 (2011).
18. S. J. Andelman, M. R. Willig, *Science* **305**, 1565 (2004).
19. S. J. Andelman, M. Bowles, R. Willig, B. Waide, *Bioscience* **54**, 240 (2004).
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## ECOLOGY

# Coexisting with Cattle

Johan T. du Toit

In East Africa, large wild herbivores both compete with and benefit cattle.

Many large plant-eating mammals have evolved to live in multispecies assemblages, with species competing for food and other resources. Through domestication and animal husbandry, however, humans have enabled a few species of livestock, such as cattle, to dominate such assemblages. One standard practice in livestock production on rangelands, espoused by commercial ranchers and subsistence pastoralists alike, is the eradication of large, indigenous herbivores that are believed to compete with livestock for food. These eradication efforts have increasingly problematic implications for biodiversity conservation (1). So it is timely that on page 1753 of this issue, Odadi *et al.* (2) report on a relatively simple experiment that tested the assumption that cattle and wildlife compete for food. Their study, conducted in an East African savanna

renowned for its large herbivore diversity, revealed that cattle do compete with herbivores such as zebras and gazelles during the dry season, when food quantity is low. In contrast, during the wet season, when food quantity is high, grazing by wildlife benefits cattle by improving the quality of forage. The findings highlight ecological processes that promote coexistence among large herbivores in grasslands and savannas, and hence could be useful for conservation.

Large herbivores (>5 kg) generally belong to either a grazing guild (eating mostly grass) or a browsing guild (eating mostly foliage on trees and shrubs); a few are “mixed feeders” that alternate in response to seasonal changes in food plants (3). This grazer-browser dichotomy is a key factor promoting resource partitioning, with coexisting herbivores feeding on different plants or plant parts in the same area (4). In addition, coexisting species within each guild often differ in body size and/or digestive

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