



Detecting Insect Pollinator Declines on Regional and Global Scales

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Abstract: *Recently there has been considerable concern about declines in bee communities in agricultural and natural habitats. The value of pollination to agriculture, provided primarily by bees, is >\$200 billion/year worldwide, and in natural ecosystems it is thought to be even greater. However, no monitoring program exists to accurately detect declines in abundance of insect pollinators; thus, it is difficult to quantify the status of bee communities or estimate the extent of declines. We used data from 11 multiyear studies of bee communities to devise a program to monitor pollinators at regional, national, or international scales. In these studies, 7 different methods for sampling bees were used and bees were sampled on 3 different continents. We estimated that a monitoring program with 200–250 sampling locations each sampled twice over 5 years would provide sufficient power to detect small (2–5%) annual declines in the number of species and in total abundance and would cost U.S.\$2,000,000. To detect declines as small as 1% annually over the same period would require >300 sampling locations. Given the role of pollinators in food security and ecosystem function, we recommend establishment of integrated regional and international monitoring programs to detect changes in pollinator communities.*

Keywords: Apiformes, Apoidea, bees, monitoring, power analysis

Detección de Declinaciones de Insectos Polinizadores a Escalas Regional y Global

Resumen: *Recientemente ha incrementado la preocupación sobre las declinaciones de las comunidades abejas en hábitats agrícolas y naturales. El valor de la polinización para la agricultura, proporcionado principalmente por abejas, es >\$200 billones/año en todo el mundo, y se piensa que es aun mayor en ecosistemas naturales. Sin embargo, no existe un programa de monitoreo para detectar, con precisión, declinaciones en la abundancia de insectos polinizadores. Utilizamos datos de 11 estudios multianuales de comunidades de abejas para diseñar un programa de monitoreo de polinizadores a escalas regional,*

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nacional o internacional. En estos estudios, se emplearon 7 métodos diferentes para muestrear abejas y muestrearon abejas en 3 continentes. Estimamos que un programa de monitoreo, con 200–250 localidades muestreadas dos veces a lo largo de 5 años, tendría suficiente poder para detectar declinaciones anuales pequeñas (2–5%) en el número de especies y en la abundancia total y tendría un costo de U.S.\$2,000,000. La detección de declinaciones tan pequeñas como 1% anual en el mismo período requeriría de >300 localidades de muestreo. Considerando el papel de los polinizadores en la seguridad alimentaria y en el funcionamiento del ecosistema, recomendamos el establecimiento de programas de monitoreo regionales e internacionales para detectar cambios en las comunidades de polinizadores.

Palabras Clave: Abejas, análisis de poder, Apiformes, Apoidea, monitoreo

Introduction

Thirty-five percent of global food supply is increased by or depends on animal pollinators (Klein et al. 2007; Ollerton et al. 2011). Consequently, production will decrease if pollinators become extinct or substantially less abundant. Gallai et al. (2009) estimated recently that the global economic value of crops pollinated by insects is \$190.5 billion/year, which is 9.5% of the value of food produced for human consumption in 2005. Crops that depend on animal pollination are of high value and have an average value \$1948/t, whereas other crops such as rice, wheat, or corn, which are wind pollinated, average \$181/t (Gallai et al. 2009). Pollinators allow individuals with small farms to diversify into higher-value horticultural crops. This diversification of income stream helps alleviate poverty (Krishna 2007). Animal pollination maintains or increases yields in agricultural and horticultural crops and therefore affects food production and security, diet quality, and farmer livelihoods. Besides pollinating crops, animals pollinate approximately three-fourths of all flowering plant species (Ashman et al. 2004; Aguilar et al. 2006; Ollerton et al. 2011). The value of those plants' contributions to ecosystem function (e.g., water and habitat conservation, timber production) likely equals or exceeds their contribution to agricultural production (USDA 2007).

Most crops rely on a small number of managed pollinators, primarily the honey bee (*Apis mellifera*) (Aebi et al. 2012). Managed pollinators are brought into the crop system when needed and are sometimes less effective pollinators of crops than wild pollinators (Westerkamp 1991; Klein et al. 2007; Gallai et al. 2009). However, there are no regional or international programs that monitor the status and trends of pollinators, particularly native bees. Since 2006, a 30–40% loss of commercial honey bee colonies in the United States has been attributed in part to colony collapse disorder, a phenomenon in which worker bees abruptly disappear (Cox-Foster et al. 2007; Stokstad 2007a, 2007b). Similarly, in central Europe there has been a 25% loss of honey-bee colonies since 1985 and a 54% loss in the United Kingdom (Potts et al. 2010). Indirect evidence of declines of some native pollinators has been reported at point locations in Europe, Asia, North America, South America, Africa, and

Australia. These reports were based primarily on post hoc analyses or resurveys of study sites not originally intended to yield data that would unequivocally detect a change or trend (e.g., Rasmont & Mersch 1988; National Research Council 2007; Goulson et al. 2008). The existing data collected in these long-term studies, due to the inherent variation in pollinator abundance and detectability, seem to remain within statistical boundaries that define stasis.

Attempts to detect declines in the abundances of other animal species suggest that interannual variability in abundance is so high that for most species it might take as much as 20 years of monitoring to detect a mean decline of 2%/year. As a result, populations may be reduced by almost 50% before evidence for a decline could be detected (e.g., Gibbs et al. 1998; Hatch 2003; Maxwell & Jennings 2005). To effectively detect a temporal trend in bee assemblages, a monitoring program should be simple, repeatable, inexpensive, and, most importantly, have the ability to quickly detect declines if they are occurring (i.e., have adequate statistical power over a short period). This latter point has not been emphasized in the most thorough studies on pollinator trends published thus far (Roubik 2001; Roubik & Villanueva-Gutiérrez 2009). Monitoring protocols for systems with high variances in the monitored attribute require large numbers of sampling locations to detect a trend of a given magnitude, which may drive costs beyond the means of funding agencies.

We evaluated whether a monitoring program could be designed that would detect regional, national, or global changes in bee assemblages within 5 years in a cost-effective fashion. We compared estimates of annual variability of abundance and species richness of bee assemblages derived from data collected via 7 techniques commonly used to sample bees. We used computer simulations of 2 assemblage-level attributes (species richness and total abundance) to estimate statistical power for a range sampling-location numbers. To compare the cost of monitoring among different numbers of sampling locations, sampling techniques, and intervals between sampling, we estimated the component costs and the overall cost of a spatially extensive pollinator-monitoring program.

Table 1. The cost of detecting 1% (5% cumulative), 2% annual (8% cumulative), 5% annual (14% cumulative), and 7% annual (25% cumulative) declines in the total abundance of bees over 5 years with different amounts of statistical power for an $\alpha = 0.05$ and a coefficient of variation between years = ~ 31.22 if abundance is stable.

Sites	Estimated cost (U.S.\$)	Power to detect declines of			
		1%	2%	5%	7%
25	390,315	0.13	0.27	0.69	0.79
50	616,555	0.16	0.40	0.89	0.95*
75	842,794	0.27	0.57	0.98*	1*
100	1,069,034	0.34	0.69	1*	1*
200	1,973,993	0.50	0.93*	1*	1*
300	2,878,953	0.56	0.97*	1*	1*

*Power values above 0.90 (our recommended minimum power for monitoring).

Methods

Estimating Annual Variation

To estimate annual sampling variation associated with different methods of sampling bees, we searched in Google Scholar for articles published from 1945 through December 2009 using the following terms: *pollinator** OR *bee* OR *bees* OR *Apoidea* OR *pollinat**. In addition, we requested unpublished data from individuals who sampled bee communities for at least 3 sequential years. (Supporting Information). We found 8 published and 3 unpublished studies that applied 7 different sampling techniques (pan traps, Moericke traps, visual counts of the number of animals, malaise traps, hand netting, funnel traps, and baits). These studies were conducted in North America, South America, and Europe from the northern temperate zone to the tropics. From each data set, we estimated the mean annual coefficient of variation (CV) for species richness and total abundance. Because each study included multiple sites, we estimated CVs by computing the mean and SD of annual estimates of each response variable at each site and then averaging values of each variable among sites. A final estimate of the CV for each sampling technique, and its SE, was obtained by averaging among studies. We compared the CVs for each sampling method with a 2-tailed Kruskal-Wallis test in SAS (version 9.1) (SAS Institute, Cary, North Carolina).

Power Analyses

We assumed that in analyses of monitoring data for detection of trends researchers compared values of species richness and abundance at the beginning and end of a fixed period at each sampled site and that sites were selected independently and at random from within the region of interest. We used this within-subjects design because variation due to initial differences in species richness and abundance between subjects or to differ-

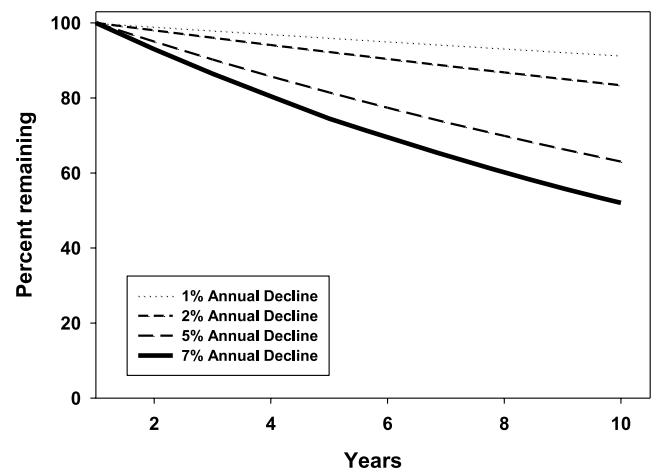


Figure 1. Reduction in species richness or total abundance over time of bees for various percent annual declines in species richness and abundance.

ences in the level of responsiveness of subjects to factors affecting species richness and abundance does not affect the estimate of the trends within sites (Winer et al. 1991). This is especially important when sites are heterogeneous and potentially span the globe. The difference between the variable at the beginning and end of each fixed period was analyzed with a paired *t* test.

To determine the sample size required to detect changes in species richness and abundance of bees, we conducted a simulation in which bee communities had annual declines in species richness and abundance of 1%, 2%, 5%, or 7% (Fig. 1). We focused on declines because this is primarily the trend managers wish to detect. Such a range of declines is similar to those found in long-term surveys of other groups of animals (e.g., North American breeding bird survey [Sauer et al. 2008]).

We developed a model of decrease in the assemblage-level metrics of species richness and total bee abundance. Our model is not mechanistic; rather, it is simply a model of the decline of an attribute in a statistical population. We began with a simple random-walk model with a given trend in the population size or number of species to simulate the dynamics of a metric of a single bee assemblage (similar to Pollard et al. 1987).

$$x_{t+1} = (1 - \delta_t) \times x_t + e_{np}, \quad (1)$$

where t is year zero, x_t is the initial number of species collected, x_{t+1} is the number of species or number of animals collected in successive years, δ is the normally distributed effect size expressed as a proportional decline, and e_{np} is a normally distributed term that represents nonprocess error. We set $\delta_t = N(\mu_\delta, \sigma_{ep})$, where μ_δ is the targeted effect size and σ_{ep} is the process error estimated from the observed CV of the trends among the studies we examined in which pan traps were used. We set $e_{np} = N[0, b(x_t)]$, where b is a constant multiplier

that expresses the detrended SD of the annual variability as a proportion of the previous year's value. Hence, the CV of annual variability is held constant, but SD is directly related to the prior year's value. If $\mu_\delta = 0$, the value of the variable is stable over time.

We then simulated assemblages independently at different sites to represent potential samples from an underlying statistical population of sites about which one would like to make an inference. Because the distribution of declines from our simulated assemblages was not symmetrical (i.e., it shifted slightly toward a greater proportion of declines than increases), we used Monte Carlo simulation to empirically estimate the $(1-\alpha)$ quantile of the distribution of assemblage declines under the null hypothesis that $\delta = 0$. From these quantiles, we calculated the power of the estimate under the alternative hypothesis $\delta > 0$ by numerical simulation.

We specified an initial value of the community metric of interest drawn from a Poisson distribution with $\mu =$ the mean species richness or the total abundance of bees detected in field studies that sampled bees with pan traps (i.e., 56.3 bee species and 375.1 total bees). We used values of the parameter h to achieve the detrended mean annual CV for stable species richness or abundance that we estimated from field data from pan traps (i.e., CV = 25.34 and $b = 0.127$ for species richness, CV = 31.22 and $b = 0.155$ for total abundance) (Fig. 2). We used the method outlined in Gibbs et al. (1998) to calculate the detrended CVs. For our estimate of process error, we used the CVs of the annual trends estimated from pan trap studies (i.e., 1.11 for species richness and 2.05 for total abundance).

We focused our simulations on sampling periods of 5 years, but could have performed power calculations for shorter or longer study durations. For all simulations for each site, the study period was chosen uniformly at random from all possible study periods. For example, with a 4-year gap between samples the samples could be taken in years 1 and 5, 5 and 9, 9 and 13, 13 and 17, 17 and 21, or 21 and 25. We ran simulations in which the number of sites sampled ranged from 25 to 300 in increments of 25 and with effect sizes including 1%, 2%, 5%, and 7% annual declines. We ran 1000 iterations for each combination of number of sites (δ) and ran simulations for 25 years.

We attempted to capture global variation in species richness and abundance of bee assemblages, but because we had data from only 3 continents, we may have underestimated the true level of global variation in these response variables. However, we do not believe our data were from sites that had lower variance than the global average. It is also conceivable that we overestimated the magnitude of process and nonprocess errors. Similarly, we may have overestimated continental or regional variation because the sites spanned continents and regions. To assess the effect of underestimating process and nonprocess error, we performed an additional set of simulations

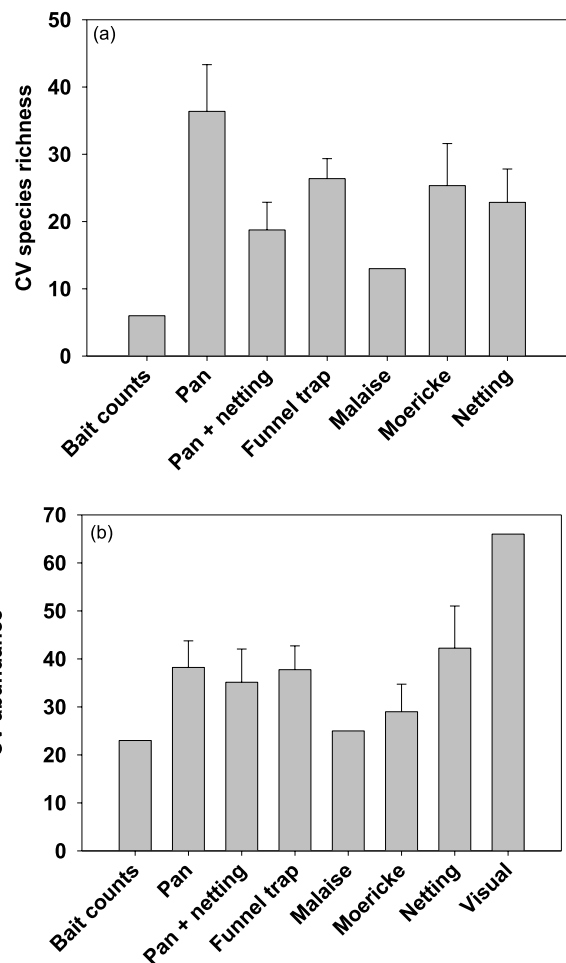


Figure 2. Mean and standard error of the coefficient (CV) of variation for bee (a) species richness and (b) total abundance with different sampling techniques (sample sizes for richness and abundance respectively: bait, 1 and 1; pan trap, 8 and 14; pan trap and netting, 8 and 8; funnel, 3 and 8; malaise, 1 and 1; Moericke [large pan] trap, 6 and 6; netting, 13 and 13; visual, no data and 1). There were no estimates of species richness available from visual-count studies.

in which we assumed our empirical estimates of both were underestimates by as much as 50%.

Evaluating Costs

We compared how study costs varied with effect size and power. On the basis of cost estimates from several spatially extensive surveys in Europe (S.G.P., ALARM, www.alarmproject.net) and the United States (T. Griswold), we estimated that if sampling was conducted with the standard protocol of 30 pan traps for each sample period (LeBuhn et al. 2003) and each site was sampled at 2-week intervals over a year (26 samples/year), each site could yield approximately 5000 bees/year (average

number of bees collected per day for a set of 30 pans ranged from 70 to 97 in the western United States [T. Griswold, unpublished data] and 15–30 in the eastern United States [S.D., unpublished data]. We present cost estimates for supplies and equipment in Supporting Information.

We evaluated how long it would take to fully process the specimens (i.e., remove them from liquid storage bags; clean, pin, and label the specimens; and enter the data about the specimen). We used data from studies in Zion and Yosemite National Parks, (U.S.A.) (T. G., unpublished) and in Europe (Klein et al. 2007) to estimate processing time, which ranged from 1.05 to 1.25 minutes/specimen. T. Griswold and H. Ikerd (U.S. Department of Agriculture Bee Lab, Logan, Utah) estimated the time needed to identify the specimens to morphospecies: 0.19 minutes/specimen. This estimate reflects how sorting is done in this lab. They use a combination of 2 novice sorters and 2 expert sorters working together. They also estimated that identifying each of the specimens to species would take an average of 0.96 minutes/specimen. The time required to sample, process, and identify bees and to enter data for a single site would be approximately 201 h (Supporting Information). Sampling 100 sites would require 20,076 h or 10.45 people working 5 days/week, 40 h/week for 48 weeks. By staggering the starting dates of sampling sites, the cost could be spread over 10 years and reduce the annual cost of sampling and identification to a little over one full-time salary.

It is difficult to estimate a budget because of inequities in world salaries and the variation in the time necessary to identify specimens. In areas where there is little taxonomic expertise or many undescribed taxa or taxa that are difficult to identify, the number of hours needed to identify bees at the level of morphospecies or species would be greater. To err on the side of higher costs, we used estimated typical labor costs for a laboratory worker in the United States (Supporting Information).

Results

Annual CVs did not differ significantly across sampling techniques for species richness or total abundance of bees (species richness, $\chi^2 = 9.47$, $p > 0.15$, $n = 7$; abundance, $\chi^2 = 5.20$, $p > 0.98$, $n = 11$).

Power increased as sample size, α , effect size, and time interval between samples increased, but power decreased as CV increased. For the modeled CVs, power appeared to be >90% to detect a 5% or even a 2% annual decline with a sampling interval of 5 years, sample sizes from 100 to 250 sites, and $\alpha = 0.05$ or 0.10 (Figs. 3 & 4). For longer sampling intervals (e.g., 10 years, 15 years), the power to detect declines increased, whereas the required sample size decreased rapidly. The modeled

CVs matched the estimates of CVs for pan traps and were within the range of CVs we estimated for field studies that applied a variety of sampling methods. Even if we underestimated the magnitude of process and nonprocess error, the study design and monitoring protocol we propose would still have sufficient power to detect 5% and potentially 2% annual declines in species richness and abundance. The simulation in which we assumed our empirical estimates of process and nonprocess error were both underestimated by as much as 50% indicated that with sample sizes ranging from 200 to 300 sites, 5 years of monitoring with $\alpha = 0.10$ would have at least 80% power.

Assuming the average hourly wage per worker on the project was U.S.\$20, resources for storage were required at all sites, and specimens would be shipped for identification, a regional or international monitoring program tracking 100 sites and sampling them twice at 5-year intervals could be fully implemented for just over U.S.\$1 million (Table 1). For 200 sites the cost was just over U.S.\$2 million (Supporting Information). The cost of collecting the bees was <15% of the total cost of monitoring.

Discussion

Our results suggest that monitoring of bees with pan traps will detect changes in total abundance and species richness. Our simulations suggest that the number of sites required to detect declines in species richness or abundance of 2% to 5% would be in the range of 100–300 sites for a national, regional, or international monitoring program. Given the low annual CV, cost, and simplicity of setting pan traps, we recommend the use of pan traps in most monitoring programs (Westphal et al. 2008). However, pan traps have been evaluated primarily in North America and Europe and collect a large proportion of the bee fauna in all forms of open land-cover types throughout the world. In areas with dense vegetation (e.g., tropical forests), few bees are caught with any standard method, and pan traps deployed on the ground are almost certainly inadequate. However, canopy-level pan traps have been used in humid tropical regions with some success (Nuttman et al. 2011).

The cost of monitoring bees is significantly less than is required for most large biological monitoring projects, such as monitoring associated with the proposed Amphibian Conservation Action Plan (U.S.\$11.5 million for 2 years) (Parra et al. 2007). Although identifying bees to species provides important data on the distribution, relative abundance, and, over long periods, the changes in individual species, it is time-consuming and only a limited number of specialists currently have the ability to correctly identify species. In some parts of the world, the bee fauna is only partially known and identification

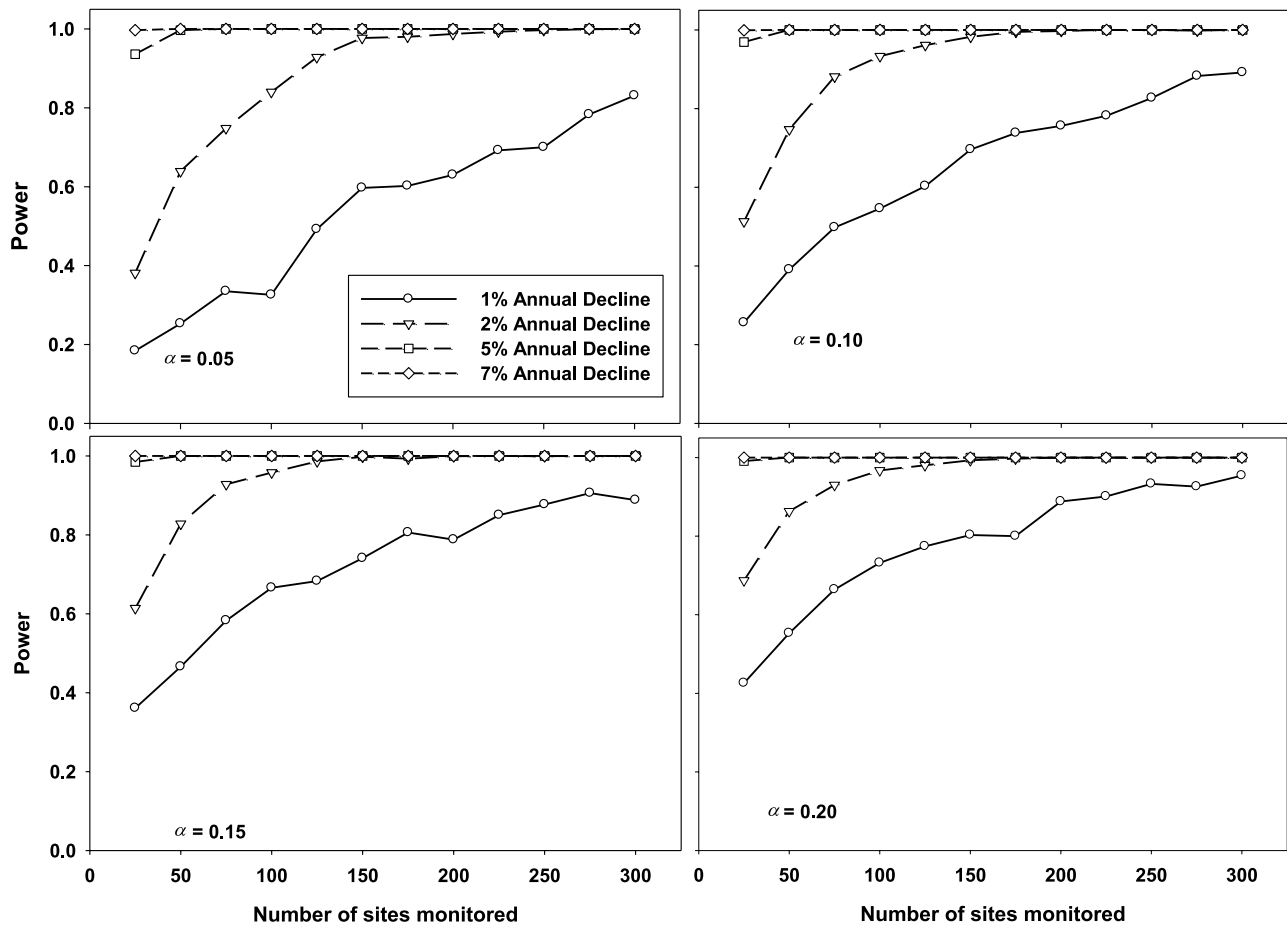


Figure 3. Results of power analysis of ability of a pan-trap study to detect a decline in species richness of bees over 5 years assuming a detrended annual coefficient of variation of 25.34 for 4 values of alpha.

to species is impossible. However, identification to genus is usually straightforward and can be done by inspecting washed and dried specimens in petri dishes, which eliminates the need to pin and label specimens. It is easy to train technicians to identify specimens to genus.

Approximately 85% of the cost of monitoring bees is in the handling, processing, shipping, storage, and identification of bees. If bees were simply counted or identified only to genus, then costs would be dramatically lower, but the power to detect change in abundance would remain high. Archiving specimens following the initial identification to genus would allow for future identification to the species level.

Our modeling results showed that the sample size required to detect decreases in species richness and abundance of bees of 2% to 5% was possible when 100–300 sites were sampled regionally or internationally. Thus, if one wished to detect pollinator declines in specific regions, then a similar number of study sites located in each region would be required. However, additional preliminary data could show that sampling variance for individ-

ual regions is less than for an international study; hence, sampling bees in these low-variance regions would require a smaller sample size within each region.

It could be argued that monitoring should be designed to have a probability of missing a true decline (type II error) that is as low as or lower than the probability of incorrectly concluding there is a decline (type I error) (Field et al. 2004). One could build into the study design a buffer in sample size to offset the chance that preliminary estimates of the CVs of measures of bee status are underestimates. Decreasing precision from $p = 0.05$ to $p = 0.10$ or $p = 0.20$ would require a smaller sample size and lower the cost of the project.

To minimize effects of short-term weather fluctuations on detecting multiyear trends in species richness and abundance, a study with a gap between surveys should sample equal proportions of sites in each year. For example with 200 sites, each year 40 sites should be sampled. This feature also spreads the cost out over time but would not necessarily simplify coordination, especially over a large area and if sites to be sampled in a given year had been selected at random.

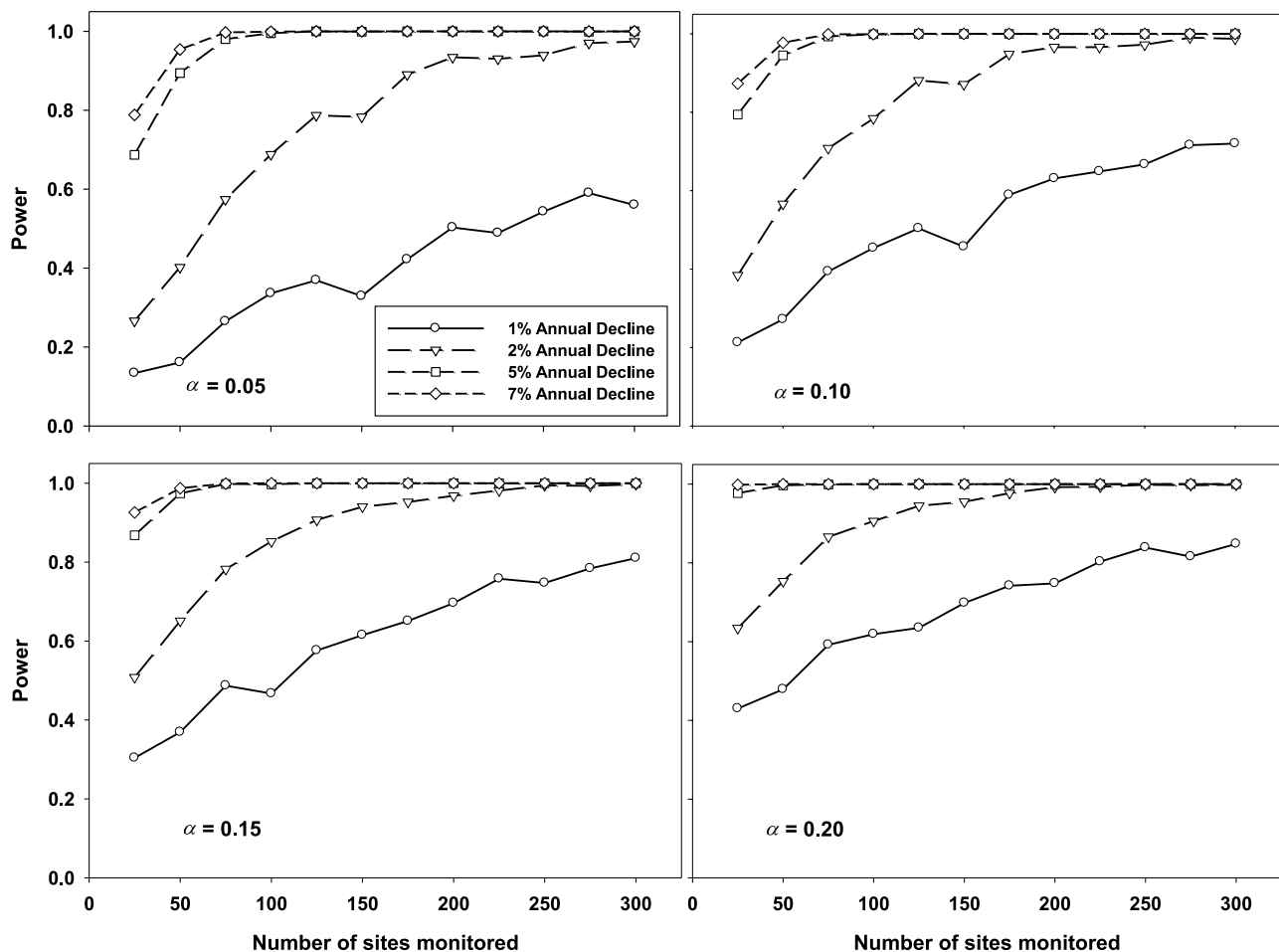


Figure 4. Results of power analysis of ability of a pan-trap study to detect a decline in total abundance of bees over 5 years assuming a detrended interannual coefficient of variation of 31.22 for 4 values of alpha.

Establishment of regional, national, or international monitoring programs would allow tracking of the status and trends of pollinators (that is all monitoring would do). The estimated cost of sustaining an international monitoring program represents a relatively small investment compared to the potential economic cost of severe pollinator losses or the increased costs of fixing a problem that was only detected once it reached crisis level (Gallai et al. 2009). Regional, national, or international monitoring programs could therefore underpin an early-warning system or document whether declines are occurring. Such programs would allow for mitigation of pollinator losses and avoid the financial and nutritional crisis that would result if there were an unforeseen and rapid collapse of pollinator communities.

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Supporting Information

Details about the studies used in the analyses (Appendix S1), estimates of cost of materials (Appendix S2) and estimates of total costs (Appendix S3) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

Literature Cited

- Aebi, A., B. E. Vaissière, D. vanEngelsdorp, K. S. Delaplane, D. W. Roubik, and P. Neumann. 2012. Back to the future: *Apis* versus non-*Apis* pollination. *Trends in Ecology & Evolution* **27**:147–148.
- Aguilar, R., L. Ashworth, L. Galetto, and M. Aizen. 2006. Plant reproductive susceptibility to habitat fragmentation: review and synthesis through a meta-analysis. *Ecology Letters* **9**:968–980.
- Ashman, T.-L., et al. 2004. Pollen limitation of plant reproduction: ecological and evolutionary causes and consequences. *Ecology* **85**:2408–2421.
- Cox-Foster, D. L., et al. 2007. A metagenomic survey of microbes in honey bee colony collapse disorder. *Science* **318**:283–286.
- Field, S., A. Tyre, N. Jonzén, J. Rhodes, and H. Possingham. 2004. Minimizing the cost of environmental management decisions by optimizing statistical thresholds. *Ecology Letters* **7**:669–675.
- Gallai, N., J.-M. Salles, J. Settele, and B. E. Vaissière. 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics* **68**:810–821.
- Gibbs, J. P., S. Droege, and P. Eagle. 1998. Monitoring populations of plants and animals. *BioScience* **48**:935–940.
- Goulson, D. G., C. Lye, and B. Darvill. 2008. Decline and conservation of bumblebees. *Annual Review of Entomology* **53**:191–208.
- Hatch, S. 2003. Statistical power for detecting trends with applications to seabird monitoring. *Biological Conservation* **111**:317–329.
- Klein, A. M., B. Vaissière, J. H. Cane, I. Steffan-Dewenter, S. A. Cunningham, C. Kremen, and T. Tscharntke. 2007. Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society of London, Series B* **274**:303–313.
- Krishna, A. 2007. For reducing poverty faster: target reasons before people. *World Development* **35**:1947–1960.
- LeBuhn, G., et al. 2003. A standardized method for monitoring bee populations—the bee inventory (BI) plot. Available from <http://online.sfsu.edu/~beeplot/pdfs/Bee%20Plot%202003.pdf> (accessed June 2010).
- Maxwell, D., and S. Jennings. 2005. Power of monitoring programmes to detect decline and recovery of rare and vulnerable fish. *Journal of Applied Ecology* **42**:25–37.
- National Research Council. 2007. Status of pollinators in North America. The National Academies Press, Washington, D.C.
- Nuttman, C. V., M. Otieno, P. K. Kwapong, R. Combey, P. G. Willmer, and S. G. Potts. 2011. Aerial pan-trapping: a method for assessing insect pollinators in tree canopies. *Journal Kansas Entomological Society* **84**:260–270.
- Ollerton, J. et al. 2011. How many flowering plants are pollinated by animals? *Oikos* **120**:321–326.
- Parra, G., et al. 2007. Systematics and conservation. Pages 45–48 in C. Gascon, J. P. Collins, R. D. Moore, D. R. Church, J. E. McKay, and J. R. Mendelson III, editors. Amphibian conservation action plan. Amphibian Specialist Group, IUCN, Gland, Switzerland.
- Pollard, E., K. H. Lakhani, and O. Rothery. 1987. The detection of density-dependence from a series of annual censuses. *Ecology* **68**:2046–2055.
- Potts, S. G., S. P. M. Roberts, R. Dean, G. Marris, M. Brown, R. Jones, and J. Settele. 2010. Declines of managed honeybees and beekeepers in Europe. *Journal of Apicultural Research* **49**:15–22.
- Rasmont, P., and P. Mersch. 1988. Première estimation de la dérive faunique chez les Bourdons de la Belgique (Hymenoptera, Apidae). *Annales de la Société Royale Zoologique de Belgique* **118**:141–147.
- Roubik, D. W. 2001. Ups and downs in pollinator populations: When is there a decline? *Conservation Ecology* **5**:2. <http://consecol.org/vol5/iss1/art2>.
- Roubik, D. W., and R. Villanueva-Gutiérrez. 2009. Invasive Africanized honey bee impact on native solitary bees: a pollen resource and trap nest analysis. *Biological Journal of the Linnean Society* **98**:152–160.
- Sauer, J. R., J. E. Hines, and J. Fallon. 2008. The North American breeding bird survey, results and analysis 1966–2007. Version 5.15.2008. U.S. Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland.
- Stokstad, E. 2007a. The case of the empty hives. *Science* **316**:970–972.
- Stokstad, E. 2007b. Puzzling decline of U.S. bees linked to virus from Australia. *Science* **317**:1304–1305.
- USDA (U.S. Department of Agriculture). 2007. Crop values 2007 summary. USDA, National Agricultural Statistics Service, Washington, D.C. Available from <http://usda01.library.cornell.edu/usda/nass/CropValuSu//2000s/2008/CropValuSu-02-14-2008.pdf> (accessed September 2012).
- Westerkamp, C. 1991. Honeybees are poor pollinators-why? *Plant Systematics and Evolution* **177**:71–75.
- Westphal, C., et al. 2008. Measuring bee biodiversity in different European habitats and biogeographical regions. *Ecological Monographs* **78**:653–671.
- Winer, B. J., D. R. Brown, and K. M. Michels. 1991. Statistical principles in experimental design. 3rd edition. McGraw Hill, Boston, Massachusetts.

