

Benefits from living together? Clades whose species use similar habitats may persist as a result of eco-evolutionary feedbacks

Andreas Prinzing¹, Wim A. Ozinga^{2,3}, Martin Brändle⁴, Pierre-Emmanuel Courty⁵, Françoise Hennion¹, Conrad Labandeira^{6,7,8}, Christian Parisod⁹, Mickael Pihain¹ and Igor V. Bartish¹⁰

¹University Rennes 1/Centre National de la Recherche Scientifique, Research Unit 'Ecobio – Ecosystèmes, Biodiversité, Evolution', Campus Beaulieu, Bâtiment 14 A, 35042 Rennes, France; ²Experimental Plant Ecology, Institute for Water and Wetland Research, Radboud University Nijmegen, Nijmegen, the Netherlands; ³Alterra, Wageningen University & Research, PO Box 47, NL-6700, AA Wageningen, the Netherlands; ⁴Department of Ecology – Animal Ecology, Faculty of Biology, Philipps-Universität Marburg, Karl-von-Frisch Strasse 8, D-35032 Marburg, Germany; ⁵Department of Biology, University of Fribourg, Chemin du Musée 10, CH-1700 Fribourg, Switzerland; ⁶Department of Paleobiology, National Museum of Natural History, Smithsonian Institution, Washington, DC 20013, USA; ⁷Department of Entomology and BEES Program, University of Maryland, College Park, MD 20742, USA; ⁸College of Life Sciences, Capital Normal University, Beijing 100048, China; ⁹Laboratory of Evolutionary Botany, Institute of Biology, University of Neuchâtel, Rue Emile-Argand 11, 2000 Neuchâtel, Switzerland; ¹⁰Institute of Botany, Academy of Sciences of Czech Republic, Zámek 1, Pruhonice 25243, Czech Republic

Author for correspondence:

Andreas Prinzing

Tel: +33 6 06 65 13 87

Email: andreas.prinzing@univ-rennes1.fr

Key words: assembly of present and fossil communities, competition, conservation biology, enemy pressure and mutualism of coexisting species, evolution and conservatism, hybridization, niche breadth.

Summary

Recent decades have seen declines of entire plant clades while other clades persist despite changing environments. We suggest that one reason why some clades persist is that species within these clades use similar habitats, because such similarity may increase the degree of co-occurrence of species within clades. Traditionally, co-occurrence among clade members has been suggested to be disadvantageous because of increased competition and enemy pressure. Here, we hypothesize that increased co-occurrence among clade members promotes mutualist exchange, niche expansion or hybridization, thereby helping species avoid population decline from environmental change. We review the literature and analyse published data for hundreds

of plant clades (genera) within a well-studied region and find major differences in the degree to which species within clades occupy similar habitats. We tentatively show that, in clades for which species occupy similar habitats, species tend to exhibit increased co-occurrence, mutualism, niche expansion, and hybridization – and rarely decline. Consistently, throughout the geological past, clades whose species occupied similar habitats often persisted through long time-spans. Overall, for many plant species, the occupation of similar habitats among fellow clade members apparently reduced their vulnerability to environmental change. Future research should identify when and how this previously unrecognized eco-evolutionary feedback operates.

I. Entire clades decline while others persist; we suggest this might reflect an eco-evolutionary feedback between habitat similarity among clade members, their co-occurrence, and their reduced vulnerability to environmental change

Environmental change appears to threaten entire clades: changing environments are considered the principal cause of species decline (McKinney, 1997), and declining species often are phylogenetically clustered (Purvis *et al.*, 2000; Sjöström & Gross, 2006; Thuiller *et al.*, 2005; Purvis, 2008 for a review, but see Davies *et al.*, 2011). Consequences of phylogenetic clustering may be the decline of entire branches of the phylogenetic tree (Vamosi & Wilson, 2008) and of phylogenetic diversities of regional biota (Eiserhardt *et al.*, 2015). Species in such declining clades might be vulnerable for two reasons. First, species within declining clades might share particular traits that make them less able to respond to environmental change (McKinney, 1997). For instance, body size (Cardillo *et al.*, 2005; Davies *et al.*, 2011), life form (Sodhi *et al.*, 2008; Davies *et al.*, 2011), cold tolerance (Eiserhardt *et al.*, 2015), and fruit type (Sjöström & Gross, 2006) have been shown to be related to species decline and to characterize declining clades. However, explained variances remain low (e.g. Sodhi *et al.*, 2008) or trait relationships are inconsistent among clades (Sjöström & Gross, 2006), and opposite relationships have been reported in different studies (reviewed in McKinney, 1997). Second, species in declining clades might be vulnerable as a consequence of clade-level traits, such as species richness or phylogenetic age. Several studies have reported that species in species-poor clades have a higher risk of going extinct (Purvis *et al.*, 2000; Sjöström & Gross, 2006). However, several other studies report the opposite relationship of species in speciose clades suffering higher extinction risk (Schwartz & Simberloff, 2001; Davies *et al.*, 2011). In addition, Wang *et al.* (2013) reported that rare, putatively vulnerable species belong to phylogenetically young clades. However, also, clade-level traits often explain only a limited portion of the variance in species rarity or decline (Wang *et al.*, 2013) and may be inconsistent among families (Sjöström & Gross, 2006; Davies *et al.*, 2011).

Here, we suggest a new perspective on how clade-level traits can affect the vulnerability of species: vulnerability of species within a clade may depend on whether these species occupy similar habitats. Species within some clades occupy surprisingly similar habitats, and species within other clades surprisingly different habitats. Emblematic examples, for instance, are Potamogetonaceae or Cactaceae, using almost exclusively submerged or dry habitats, respectively,

versus Asteraceae or Poaceae, each of which use both very dry and very wet habitats. Individual clades in which species occupy particularly similar habitats have been recognized by Pearman *et al.* (2008), Donoghue (2008) and Olalla-Tárraga *et al.* (2011) and were systematically quantified by Lavergne *et al.* (2013). Such similar habitat use among species within a clade may result from slow evolutionary divergence of occupied habitats among clade members (Wiens *et al.*, 2010; Losos, 2008; Kellermann *et al.*, 2012; see also Petitpierre *et al.*, 2012). Past evolutionary divergence of occupied habitats among species within a clade can be constrained by: (1) climatic, geographical and geological factors, such as low spatial and temporal habitat heterogeneity within a given biogeographical region in which the clade diversified (Pennington *et al.*, 2004; Crisp *et al.*, 2009); or (2) clade-specific factors, such as genetic or trait constraints (Alcantara *et al.*, 2014; Christin & Osborne, 2014; Johnson *et al.*, 2015; Siefert *et al.*, 2015), or constraints caused by clade-specific biotic partners (Wise & Rausher, 2013; Yu & Nason, 2013). Similarity in habitat use among clade members, in turn, might increase the frequency of their local co-occurrence (Prinzing *et al.*, 2016). Eventually, local co-occurrence of fellow clade members might either increase or decrease the vulnerability of species to environmental change. Vulnerability may increase as a result of competitive replacement between clade members from abiotically suitable environments or enemy exchange. By contrast, vulnerability may decrease as a result of exchange of mutualists or increased intraspecific variability. We detail these hypotheses in the following sections.

We focus on similarity among clade members in terms of habitat conditions (habitat niche; Grubb, 1977) quantified as the species' position along abiotic gradients such as soil moisture, pH, productivity, light availability and within-region gradients of temperature. In the hierarchical concept of Silvertown *et al.* (2006), the habitat use of a species corresponds to its beta niche. Two species using similar habitat conditions may co-occur in the same patch of a habitat, being spatially sufficiently proximate to interact. By contrast, larger scale macroclimatic conditions used in the literature on 'species distribution modelling' are less appropriate for our purpose as species within the same macroclimatic zone may still be spatially very far from each other and only share the same geographical range (gamma niche according to Silvertown *et al.*, 2006). Conversely, the use of those smaller scale micro-environmental conditions that ultimately control whether two species interact or not – microhabitats as determined by soil depth, phenology, and interacting predators or mutualists (alpha niche; Silvertown *et al.*, 2006) – may also be less appropriate for our

purpose. Species may use different microenvironmental conditions but nevertheless co-occur and initially interact within the same habitat patch. In fact, microenvironmental divergence or convergence among species may be a consequence rather than the cause of interactions among species co-occurring in a habitat patch.

In this study, we develop the reasoning for each step of an eco-evolutionary feedback connecting similarity of habitat use among species within clades to co-occurrence among fellow clade members and co-occurrence among clade members to their vulnerability to environmental change. We review the literature to provide evidence and, if this is unavailable, we re-analyse published accounts. We start with the implications from the ideas that seem to be mainstream in community ecology – that species suffer from co-occurring with fellow clade members, and that such biotic pressure renders species more vulnerable to changing abiotic environments. We then analyse and advocate the opposite view: that co-occurring with fellow clade members may render species less vulnerable to changing environmental conditions. We finally review the existing evidence in favour of the overall relationship between the degree to which species within clades occupy similar habitats and the risk of decline of species within these clades, both at present and in the fossil record. While this evidence remains tentative, it does permit the conclusion that, if species within a clade occupy similar habitats, this increases the chance that these species (1) locally co-occur, (2) locally help each other at least as much as they can impede each other, and (3) persist under environmental change often even more than species in a clade in which habitat use is more variable. Past evolutionary constraints on habitat use of species within a clade might hence improve the present fate of the clade members and thereby of the entire clade, and this relationship might be mediated by interactions among locally co-occurring species.

II. Definitions, and methods used to infer tentative evidence from published results

We focus throughout this review on angiosperms (flowering plants), as they currently represent the most diverse and dominant lineage of land plants. We use published data to study angiosperms from the Netherlands where uniquely detailed databases are available containing information at the species and community levels, as outlined later in this section. For the entire regional species pool, we accessed species-specific information on (1) habitat use along multiple environmental gradients (habitat positions and habitat breadths; Ozinga *et al.*, 2013; using data from the National Dutch Vegetation Database; Global Index of Vegetation-Plot Databases ID: EU-NL-001; Schaminée *et al.*, 2012), (2) interactions with mycorrhizal mutualists (Hempel *et al.*, 2013), (3) life histories (Ozinga *et al.*, 2005, using information from the LEDA trait database of Kleyer *et al.*, 2008) and their consequences for competitiveness (Grime, 2001; Klotz *et al.*, 2002), (4) phylogenetic ages (Bartish *et al.*, 2016), (5) hybrid status (Frank & Klotz, 1990; Jäger & Werner, 2005), and (6) local co-occurrence (Prinzing *et al.*, 2016, a work, again, based on the Dutch National Vegetation Database, the most complete regional vegetation database globally available, covering all habitat types across the Netherlands). Despite the unique data availability, restriction to the Netherlands has

obvious shortcomings: the Netherlands have a low proportion of entirely natural habitats, and belong to a region of the world with limited overall richness of species and supraspecific clades. Also, as a consequence of the restricted surface of the Netherlands, it is possible that outside the Netherlands a given species might co-occur with clade members that are not present in the Netherlands. However, most species in the Netherlands have similar co-occurring relatives elsewhere in Europe: in comparisons of specialist and generalist clades in the Dutch flora, Ozinga *et al.* (2013) showed that the results were not biased by geographical sampling of lineages; that is, they did not depend on whether clades belonged to lineages that were well or poorly represented in the Netherlands. We provide additional examples of between-region consistency below in this section. Overall, a study on the Netherlands appears to be a good starting point, albeit that analyses on other regions are desirable, should the data become available.

We focus here on clades at the level of genera and characterize the variation among species within genera. In the Netherlands, these genera are mostly entirely monophyletic and, if not, almost monophyletic (Durka & Michalski, 2012). As a general tendency, species within angiosperm genera tend to use similar habitats, notably in terms of pH, temperature and nitrogen (Prinzing *et al.*, 2001); to share many natural enemies (Schoonhoven *et al.*, 2005; Wardhaugh, 2014 on many phytophages feeding on only a single host genus), and to share many mutualists (Armbruster, 2012 on preference of many pollinators for particular plant genera over others); and hybridizations in angiosperms are mostly within rather than between genera (e.g. Jäger & Werner, 2005). Each of these issues will be treated in the present review. Obviously, genera lack strict biological meaning as they differ in age and richness, and so we accounted for these characters in further analyses (see below in this section for details of age estimations).

Specifically, we focused on variation in habitat use along abiotic gradients, which provides a context for the subsequent study of biotic consequences, notably of co-occurrences with and interactions among fellow clade members. Positions of species are known along gradients of light, temperature, soil moisture, pH and productivity from Ellenberg indicator values (Ellenberg *et al.*, 1992). These species-specific indicator values are expert-knowledge classifications, ranging from 1 to 9 (or 1 to 12 for moisture), for the optimal occurrence of species along environmental gradients. Evidence for the accuracy of these indicator values has been provided by several studies reporting a close correlation between average indicator values and corresponding direct measurements of environmental variables (e.g. Hill & Carey, 1997; Schaffers & Sykora, 2000; Diekmann, 2003; Ozinga *et al.*, 2013) and a very strong correlation between indicator values of the same species on different continents (Niinemets & Valladares, 2006). These values were adapted to the Netherlands and rendered more continuous using within-plot averages of indicator values for all plots in which a species was found (across 36 853 plots, as in Ozinga *et al.*, 2013), but with practically identical results to those obtained when using original indicator values. For a given species and a given gradient (e.g. pH), we then calculated the standard deviation across the local plot means of all plots in which this species occurred (as in Ozinga *et al.*, 2013). This provided an estimate of the variation of occupied

environments within that particular species along the gradient. This approach is much more differentiated and precise than many of the often-used classifications based on numbers of habitat types occupied. We multiplied the values obtained for the different gradients to identify a volume of the habitat niche occupied by a given species and used this product as an integrative measure of within-species variation in habitat use (Ozinga *et al.*, 2013). We also characterized genera by calculating, for each gradient, the environmental position of a genus as the mean value across its constituent species, and the within-genus environmental variation as the standard deviation across its constituent species. Standard deviations along different niche axes were always positively correlated (mean $r = 0.33$), and consequently we averaged for each genus the standard deviations for the different gradients. Low (mean) variation indicates that species within a genus use similar habitats.

Information on local co-occurrences was available from Prinzing *et al.* (2016). These authors quantified the frequency at which species locally encounter congeners using the Dutch National Vegetation Database (Hennekens *et al.*, 2010; Schaminée *et al.*, 2012), containing spatially explicit descriptions of species composition (presence/absence) in more than 350 000 small plots. For each species, the average number of co-occurring congeners per plot is extracted and these per-species values are averaged within genera (see Prinzing *et al.*, 2016 for detailed justification of this approach). Only genera occurring in multiple plots were considered. Interestingly, genera scoring high for co-occurrence in the Netherlands also score high in a distant region, South Africa (after partialling out differences in species richness between regions; Prinzing *et al.*, 2016).

Information on hybridization, mycorrhization and population trends came from a variety of sources. Information on hybridization was available from Frank & Klotz (1990), checked against Jäger & Werner (2005). These authors present 'successful', that is, persistently established, hybrids accepted by botanists. More ephemeral hybridization that goes unnoticed by botanists is not relevant here. Information on mycorrhization, life histories and their consequences for competitiveness (*sensu* Grime, 2001) was available, respectively, from Hempel *et al.* (2013), from Ozinga *et al.* (2005, based on information in the LEDA trait database; Kleyer *et al.*, 2008), and from Klotz *et al.* (2002). Information on population trends during the 20th century was available from Ozinga *et al.* (2009). These authors used species occurrences in the Netherlands across a 1-km² grid during 1902–1949 and during 1975–1998. Specifically, a selection of nearly 25% of the grid cells with a high sampling intensity in both periods was used, supplemented by a correction factor for temporal differences in sampling intensity (Van der Meijden *et al.*, 2000). Our definition of decline corresponds to a regional application of International Union for Conservation of Nature red-list criteria (Ozinga *et al.*, 2009). As trend data are sensitive to various sources of bias and to differences in spatial and temporal scale, we used a binary classification: species were labelled as declining if the number of grid-cell occurrences had declined by > 25%.

Genera differ in age and hence time for divergence of habitat niche among species. Consequently, we controlled our analyses of

habitat similarity within genera for differences in age. To do so, we used crown age, that is, the age of the most recent common ancestor of all species of a clade present in the study region (the Netherlands) or, in other words, the age of the earliest diversification event among all the regional species within the clade. Crown ages should more realistically reflect the time for diversification of habitat niches within genera than stem age. High crown age of genera indeed was associated with somewhat increased variation in occupied environments, notably in terms of temperature, pH and nitrogen ($P < 0.05$), with graphical inspection showing that existing relationships are linear. Genus crown ages were inferred from a dated and finely resolved phylogeny covering a total of 557 genera. Details of the analyses for reconstruction of the dated phylogeny of Dutch angiosperm genera are provided in Hermant *et al.* (2012, their Appendix E) and in Bartish *et al.* (2016, their Appendix S3). In brief, these authors began their analyses by focusing on phylogenetic relationships at the level of all families, which can be represented by sequences of the same gene, the large subunit of the ribulose-bisphosphate carboxylase (*rbcL*), and then proceeded to genera within larger families and to particularly old genera (subtrees). Several genes from chloroplast genomes (chloroplast DNA (cpDNA)) and internal transcribed spacer (ITS) regions of nuclear DNA were used in these studies for phylogenetic reconstructions within the subtrees. The genes from cpDNA were selected and retrieved from GenBank for maximal representation of the regional sample of species within the subtrees. For their dating analyses, the authors, similarly, first obtained age estimates for diversifications among all families, and then for subtrees in their sample. At the level of all families, reconstructions were based on the same phylogeny and the same gene (*rbcL*) and established a set of reference nodes for calibration of the stem nodes of the subtrees. This approach ensures that age estimates for the stem and crown nodes of genera are comparable across families, as ages of all families are derived from the same basic node age estimates. If no phylogenetic information was available for intrageneric relationships, ages of the crown nodes were simply assessed as half of ages of the stem nodes of the corresponding genera. A dated tree in Newick format based on sequence relationships of all genera and about half of native angiosperms species represented in the Netherlands is available from TreeBase (ID: S13572). The tree is highly congruent with, but often more resolved than that of Durka & Michalski (2012) for the larger region of Europe. The tree is also congruent with that of Zanne *et al.* (2014). Zanne *et al.*'s tree covers some 10–15% of the global angiosperm flora, which renders it less complete and representative of the Dutch genera than the complete tree that we used.

III. The interface between variation in habitat use within clades and the assembly of local communities: clade members occupying similar habitats tend to locally co-occur

Within a region, species in some clades occupy very similar habitats, as exemplified for the Netherlands in Supporting Information, Notes S1 (Fig. S1). How does this affect local co-occurrence among these species? If we assume that competition leads to local

replacement of species within clades (e.g. Webb *et al.*, 2002; Violle *et al.*, 2011), then we should expect clade members to never co-occur even if they have similar habitat requirements. If no such competition occurred, and if dispersal was unlimited and survival in a new habitat never possible, then we should expect local co-occurrence of clade members to depend entirely on the similarity of their habitat use: maximal co-occurrence among species in all clades of highest habitat similarity, minimal or no co-occurrence among species in all clades of lowest habitat similarity. To date, this relationship between habitat similarity among clade members and their local co-occurrence has to our knowledge only been studied by Prinzing *et al.* (2016, but see Sedio *et al.*, 2012 for a case study on a single clade, and Villalobos *et al.*, 2013 for comparing co-occurrences among clades). These authors used data on habitat use along individual environmental gradients and co-occurrences within genera of angiosperms and demonstrated that high co-occurrence among species is indeed found in genera in which species occupy similar habitats. Fig. 1 shows a more comprehensive analysis of their data accounting for multiple gradients (see Section II), confirming that local co-occurrence within genera increases with within-genus similarity of habitat use. In other words, constraints during the evolutionary past leading to low variation in habitat use among species within a clade partly control the ('microecological') assembly of communities in local habitat patches (as suggested by Gerhold *et al.*, 2015). The pattern, however, shows some scatter; the reality lies between the extreme

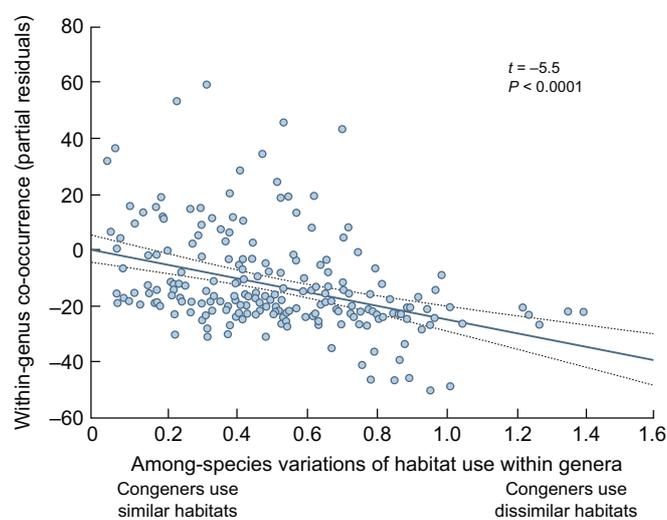


Fig. 1 Co-occurrence among species within a genus is high in those genera in which species use similar habitats, that is, in genera of low variation in habitat use (multiple regression: $df = 201$; $t = -5.5$; $P < 0.0001$). Data points are angiosperm genera studied in the Netherlands. The within-genus co-occurrence is the number of fellow congeners with which an average species will co-occur on an average plot (from Prinzing *et al.*, 2016). Within-genus co-occurrence is given as partial residuals from an analysis accounting for confounding effects of other variables increasing the chance for co-occurrence: (1) genus–species richness, (2) genus crown age (e.g. Perret *et al.*, 2007), and (3) mean intraspecific variation in habitat use (from Ozinga *et al.*, 2013; Hermant *et al.*, 2012, as explained in Section II). Here and elsewhere, we verified normality and homogeneity of residuals graphically and ensured robustness to exclusion of possible outliers. This negative relationship is phylogenetically independent: it was found in 13 out of 15 angiosperm orders.

expectations outlined above: the effect of habitat similarity among clade members on co-occurrence might be overlaid by competition, dispersal limitation (Siefert *et al.*, 2015; Renwick & Rocca, 2015) or transient survival in unsuitable habitats.

IV. The mainstream hypotheses in community ecology imply that co-occurrence with fellow clade members is detrimental, but the evidence is equivocal

Classically, community ecologists consider co-occurrence among species within clades as a disadvantage because of the increased intensity of competition and enemy pressure. Such relationships would imply a negative eco-evolutionary feedback between similarity in habitat niche within clades and the survival of species.

1. Competition pressure

Species suffering from high competition pressure might suffer more from environmental change than species facing weak competition pressure. First, local competitors can limit access to declining abiotic resources (Martinez-Vilalta *et al.*, 2012). Second, competition among native species may facilitate establishment of introduced species (Gerhold *et al.*, 2011). In both cases, competition pressure would impose increased investment in competitiveness, implying a reduced investment in tolerance of a deteriorating environment (Grime, 2001). Multiple examples of trade-offs between investment in competitiveness and in the response to harsh environments are known, such as use of carbohydrates for drought tolerance versus use of carbohydrates for growth (Alpert, 2006); or early germination to escape competitors versus late germination to escape late frost (Ross & Harper, 1972), a case of the 'ecological costs' of competitiveness (Koricheva, 2002). Such trade-offs between competitiveness and tolerance reflect to a large extent differences in the underlying functional traits that ensure these capacities (Adler *et al.*, 2014), which may incur a burden if the respective capacity is not needed (Kunstler *et al.*, 2016). The trade-off between competitiveness on the one hand and tolerance of stressful environments (or the opportunistic use of temporally favourable environments) on the other has often been considered as the major axis along which plant life histories are differentiated, albeit that the issue remains debated (Grime, 2001; Craine, 2005; Reich, 2014 for a syntheses).

Such high competition pressure may result from co-occurrence with fellow clade members, which, in turn, is promoted by similarity of habitat use (Fig. 1). Closely related species tend to be, on average, more ecologically similar than distantly related species (e.g. Burns & Strauss, 2011), and co-existing, closely related species might therefore compete strongly (Violle *et al.*, 2011). Increased competition among co-occurring related species might force such species to invest more energy in competitiveness and less in abiotic tolerance (Fig. 2).

However, co-occurrence of related species does not necessarily increase competition. Competition might be tempered as a result of character displacement among competitors (Dayan & Simberloff, 2005; see Section V.2). Moreover, recent studies investigating trait assembly under competition suggest that one possible outcome of

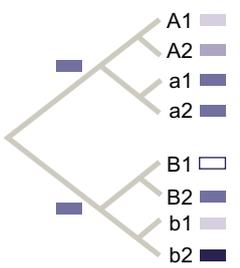
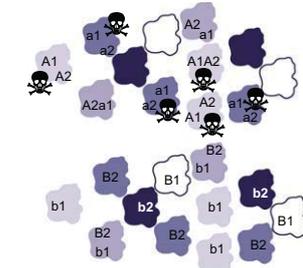
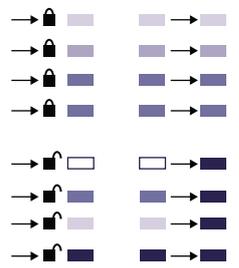
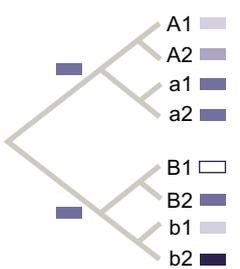
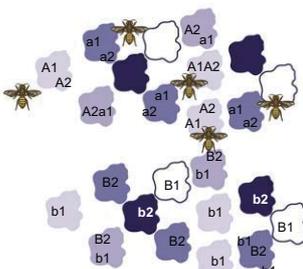
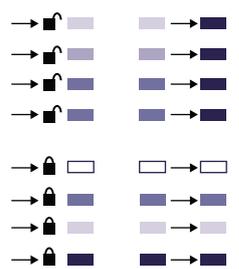
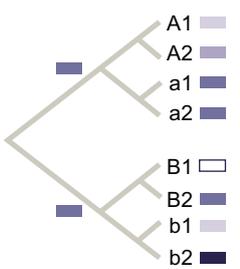
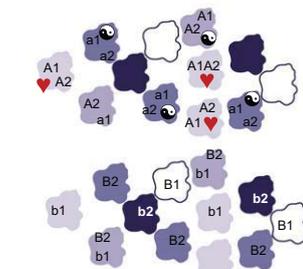
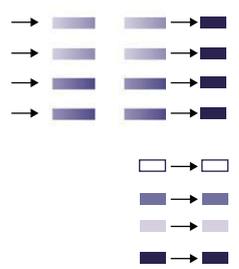
	Habitat similarity within clades	Mechanism	Redistribution under environmental change towards 	Expected population trend
<p>Implications from mainstream hypotheses: More vulnerable – species in clade of similar habitat use often co-occur in the same patches, compete and exchange enemies () reducing flexibility ( vs )</p>				
<p>Our hypothesis: Less vulnerable – species in clade of similar habitat use co-occur and exchange mutualists () increasing flexibility ( vs )</p>				
<p>Our hypothesis (cont.): Less vulnerable – species in clade of similar habitat use co-occur, hybridize () or displace characters () increasing intraspecific variation (shading)</p>				

Fig. 2 Scheme summarizing the hypotheses regarding the consequences of similarity in habitat use among clade members: increases in vulnerability of species to current environmental change (more vulnerable; ‘Implications from mainstream hypotheses’) and decreases in vulnerability (less vulnerable; ‘Our hypothesis’). Species in the upper of the two clades occupy similar habitats, in contrast to species in the lower clade. Shades of blue correspond to the environments used; for example, different moisture conditions.

competition is increased trait similarity and not trait dissimilarity (Scheffer & Van Nes, 2006; Mayfield & Levine, 2010; Vergnon *et al.*, 2012; Tobias *et al.*, 2014). In that case, similar, co-occurring, closely related species would not suffer from increased competition intensity, but profit from decreased competition asymmetry. Such decreased competition asymmetry makes the process of competitive exclusion slow enough to be offset by the many equalizing mechanisms that help prevent competitive exclusion. So there are two contrasting windows of opportunity for coexistence: being sufficiently different or being sufficiently similar (Scheffer & Van Nes, 2006). Finally, similar species may facilitate each other, such as through shared effects on microclimates. Piston *et al.* (2015), for example, showed that the effects of cushion plants on closely related species changed from negative to positive as environmental conditions became more severe (see also Cavieres *et al.*, 2016). Hence, it is not clear whether or when co-occurring related species suffer more from competition than do other species, or whether and when such competition prevents these related species from responding to environmental change (Table 1).

To tentatively explore the evidence for increased competition pressure on fellow clade members using similar habitats, we used the same genera as above (Fig. 1). We tested whether increased habitat similarity among relatives increased the investment of plants in competitiveness (inferred from the classification by Grime (2001), as explained in Notes S2 and Fig. S2). We analysed the relationship for genera in which habitat similarity indeed corresponded to a high co-occurrence, and possibly high competition, that is, genera with small unsigned residuals in the above relationship of co-occurrence versus habitat similarity (‘small’ being defined as the lowest quartile). We also analysed this relationship in genera for which co-occurrence corresponds minimally to habitat similarity, indicated by an unsigned residual co-occurrence in the highest quartile. An increase of competitiveness for genera whose species occupy similar habitats was not found in either of the two groups. Indeed, the relationship of habitat variation to competitiveness was positive for both groups of genera (Notes S2; Fig. S2). In a separate analysis, we treated residual co-occurrence as a continuous variable and found the interaction

Table 1 Several questions that need to be resolved to understand whether species within clades using similar habitats facilitates or impedes response to present environmental change

Mainstream hypotheses imply that co-occurrence with fellow clade members is detrimental and increases vulnerability to environmental change.

Competition

When is competition in nature particularly intense among fellow clade members using similar habitats?

Does competition with fellow clade members using similar habitats limit the distribution of species? Does this limitation increase vulnerability to environmental change?

Do fellow clade members using similar habitats invest more in competitiveness? Does this limit investments in responses to environmental change?

Natural enemies

Do fellow clade members using similar habitats suffer more from natural enemies than other species? Does this increase vulnerability to environmental change?

Do fellow clade members using similar habitats invest more in defence against natural enemies? Does this limit alternative investments in responses to environmental change? Which types of defence are particularly costly?

We hypothesize that co-occurrence with fellow clade members is often beneficial and reduces vulnerability to environmental change.

Mutualism

Do fellow clade members using similar habitats profit particularly greatly from mutualists?

Do fellow members of host clades using similar habitats compete for mutualists? Conversely, do the mutualists compete with fellow clade members for hosts?

Do the costs of the competitive interactions outweigh the benefits for the mutualist?

Character displacement

Is there local character displacement among clade members if they use similar habitats?

Does intra-specific variation of characters increase the flexibility to environmental change?

Hybridization

Do fellow clade members using similar habitats exhibit higher rates of hybridization, and can this be explained by high intra-clade co-occurrence?

Do hybrids better tolerate environmental change?

term ‘habitat similarity × residual co-occurrence’ to be nonsignificant ($t = -1.29$; $P > 0.2$). Overall, habitat similarity and co-occurrence among fellow clade members does not appear to impose higher investment in traits that confer competitiveness.

2. Enemy pressure

The response to environmental change may be particularly difficult for organisms that suffer elevated pressure from natural enemies. For instance, Siemsen *et al.* (2009) have demonstrated that mustards, which invest strongly in defence against natural enemies, have reduced capacity to respond to increasingly xeric abiotic environments. Below, we contend that such pressure from natural enemies might be higher in clades whose species use similar habitats than for species in other clades.

Closely related species belonging to the same clade are more likely than distantly related species to share or exchange enemies and diseases, including phytophagous insects, fungal pathogens

and several invertebrate and vertebrate diseases (Daszak *et al.*, 2000; Brändle & Brandl, 2006; Watanabe *et al.*, 2014). The sharing of enemies among related host species appears to result from similarities in morphology, physiology, phenology, habitat use and range distribution among closely related hosts (Brändle & Brandl, 2006). As explained above, related species using similar habitats tend to have a higher degree of co-occurrence. A host surrounded by closely related neighbours consequently may face an increased risk of infection by its neighbours’ enemies and diseases (Daszak *et al.*, 2000; Gossner *et al.*, 2009; Vialatte *et al.*, 2010), increasing the damage caused by these enemies (Yguel *et al.*, 2011; Parker *et al.*, 2015) but also the enemy pressure suffered by the enemies themselves (Yguel *et al.*, 2014a). Such pressure from enemies may reduce the tolerance of the host to stresses such as environmental change (Siemsen *et al.*, 2009) by triggering defences, which often may be costly (Strauss *et al.*, 2006). Enemy pressure may also cause a reduction of host population size, loss of genetic diversity (Dhondt *et al.*, 2006; Breed *et al.*, 2009) and, finally, local extinction of host genotypes or species (McCallum & Dobson, 1995). Reduction in size of and diversity within populations may reduce their tolerance to environmental change.

Defence against enemies does not, however, automatically reduce the capacity to respond to changes in the habitat environment. The degree to which this happens may strongly depend on the type of defences (Fürstenberg-Hägg *et al.*, 2013) and the type of costs that these defences trigger within the specific environment studied (Koricheva, 2002). First, enemy defence may be of different types, and some types of defence may be more costly than others. Costs may be high in ‘quantitative’ defences requiring extensive tissues or large amounts of chemical compounds to reduce the edibility and digestibility of plant tissues, compared with ‘qualitative’ defences requiring only small amounts of toxic compounds (Price, 1995). Moreover, ‘constitutive’, permanently established defences (Brennan & Weinbaum, 2001a,b) may be more costly in terms of resource allocation than ‘induced’ defences produced only upon need (Karban *et al.*, 1997; Kessler & Baldwin, 2001; Zavala *et al.*, 2004), although opposite relationships exist for ecological costs (Shudo & Iwasa, 2001; Cipollini *et al.*, 2003). Defences that are costly – quantitative or constitutive defences – might hence constrain the capacity to respond to environmental change more than low-cost defences. However, some quantitative defences have been reported also to increase resistance against abiotic stress, such as tannins or waxes, increasing resistance against both herbivory and desiccation (Jetter *et al.*, 2000; Brennan & Weinbaum, 2001a,b).

Second, enemy defence may have different types of costs, some of which may be more constraining under habitat change, others less so (Koricheva, 2002). These costs may be expressed in terms of energy allocation, as shown for mustards (Siemsen *et al.*, 2009). Costs may also be expressed as the loss of opportunities, such as diapause during high enemy pressure at the cost of missing the opportunity to take up nutrients and overgrow competitors (Baldwin & Hamilton, 2000). Costs may, finally, be ecological (Gassman & Hare, 2005; Van Velzen & Etienne, 2015), such as early budburst decreasing the pressure by late-season herbivores but increasing the risk

from frost events. The latter type of cost, for instance, may become less important under an increasingly warm climate. Overall, whether or not defence against enemies incurs a cost in terms of resistance to environmental change is highly context dependent. In particular cases, enemy pressure might even facilitate the response to changes in the abiotic environment.

While each of the individual mechanisms we invoked has been documented, we do not yet know their overall consequences (Table 1). Do species within a clade that use similar habitats currently suffer increased enemy pressure or alternatively invest more in defence against enemies? Does this trade-off handicap a species' response to environmental change? Currently, we are lacking the data needed to explore these questions. Meaningful analyses will require investigations characterizing the impact of enemies on plant hosts across an entire region involving large groups of hosts, as well as assessment of host investment in various modes of enemy defence. We admit that the above comparisons were among hosts that co-occur with similar, closely related as opposed to less similar and distantly related heterospecific hosts. Another scenario would be that all co-occurring plant hosts are conspecifics and hence phenetically very similar, which probably would incur even greater enemy pressure.

V. We suggest that co-occurrence with fellow clade members is often beneficial and we present evidence

Contrary to what is implied by mainstream ecology, we suggest that co-occurrence among species within clades may also be advantageous and hence decrease their vulnerability to environmental change. We see two ways in which this may happen. First, fellow clade members using similar habitats may exchange mutualists, thus increasing their tolerance to environmental change. Second, fellow clade members using similar habitats compete and hybridize, thus increasing variation of the habitat niche within species and thereby the flexibility to environmental change. In the following sections, we outline the possible mechanisms and the existing evidence.

1. Mutualist exchange

Similarity in habitat use among species within clades brings these species together within local patches of a particular habitat type (Fig. 1; Cavender-Bares *et al.*, 2009; Prinzing *et al.*, 2016). Co-occurring clade members might interact *positively* by sharing common mutualists and symbionts (Sargent *et al.*, 2011 for pollinators). Symbionts may help hosts to better tolerate harsh and changing environments. We will explore below how a symbiosis may be favoured if each of the partners shares similar habitat preferences with its fellow clade members, and how the symbiosis might, in turn, favour the hosts' response to environmental change (Fig. 2). We will do so, as an example, for a mutualism of particular importance: the mycorrhizal symbiosis. In terrestrial ecosystems, > 80% of plant species live in symbiosis with mutualistic fungi and form mycorrhizas (Smith & Read, 2008). Mycorrhizal fungi provide soil nutrients to the plant, and in return

the plant delivers carbohydrates to mycorrhizal fungi. Finally, mycorrhizal fungi have been shown to increase tolerance of their hosts to environmental changes (Courty *et al.*, 2010).

Given phylogenetic signal in habitat use, in mycorrhizal partnerships and in associated traits, related plants are likely to co-occur with related mycorrhizal species (Peay *et al.*, 2010; Anacker *et al.*, 2014). Co-occurrence might increase the probability that the same mycorrhizal fungal species are locally shared between related host species within a common mycorrhizal network (CMN). Such sets of interacting host and host-specific mycorrhizal species often will be nested within larger networks involving the same hosts interacting with nonspecific mycorrhizas. However, in some mycorrhizal interactions, such sets of specific host/fungal species may form distinct modules in the interaction web, rendering these modules potentially more important for hosts (Van der Heijden *et al.*, 2015). Such CMNs facilitate resource exchange between conspecific or nonconspecific neighbouring plants (Selosse *et al.*, 2006; Walder *et al.*, 2015) and seedling recruitment (Teste & Simard, 2008). In addition, CMNs might increase plant competitiveness (Van der Heijden, 2002), and thereby promote selection for single plant species (Wilkinson, 1998). This selection would contribute to the maintenance of low-diversity plant communities (McGuire, 2007), and in particular of closely related species, that is, species within clades using similar habitats. Within such a CMN, plants may invest more carbon in their fungal partners and become their major source of nutrients (Kiers *et al.*, 2011). Consequently, one may hypothesize that, among multiple interacting plant species, CMNs promote and maintain a set of closely related mycorrhizal species and of closely related plant host species, that is, co-occurrence within clades. These sets of closely related species co-occur within the same patches of the same habitat niche and exchange of mutualists may hence contribute to conservation of the habitat niches of these species.

From the arguments outlined above for a specific mutualism, we hypothesize that co-occurring, closely related members of a clade of host plants might benefit from shared mutualists. Such shared mutualists may supply hosts with nutrients and are genetically highly variable, thereby potentially increasing the hosts' tolerance to environmental changes (Johnson *et al.*, 2013). As a tentative test for our hypothesis, we explored whether the effect of intra-genus co-occurrence on decline (as described in Section II) depends on the degree to which the genera use mycorrhizas (taken from Hempel *et al.*, 2013; details in Fig. 3). We found a significant negative interaction: intra-generic co-occurrence reduces the proportions of declining species, but only in genera that have a high degree of mycorrhizal symbiosis and hence a strong potential to locally interact with congeners via shared CMNs (Fig. 3). This pattern is consistent with CMNs among co-occurring congeners decreasing their vulnerability to environmental change. However, whether such increased support of a co-occurring clade member by CMNs actually happens remains to be tested in future studies (Table 1). In fact, these host plants that support mycorrhizas may also suffer from multiple risks. Mycorrhizal symbionts, for instance, may be functionally redundant and hence provide only a limited range of services to their hosts (Rineau & Courty, 2011;

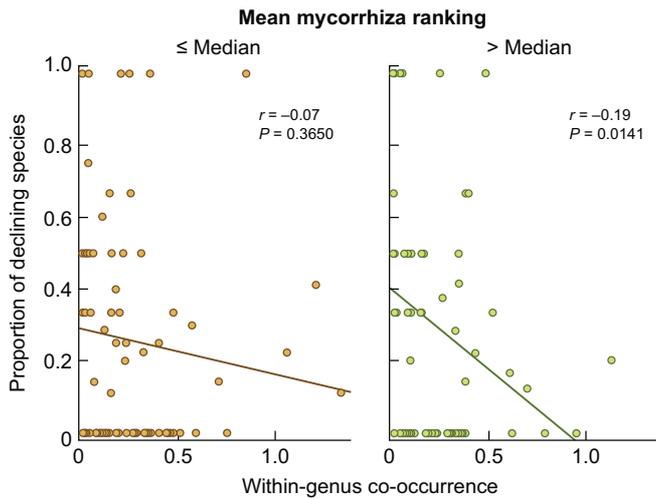


Fig. 3 A high degree of intra-genus co-occurrence (as in Fig. 1) corresponds to a low proportion of declining species within a genus, provided that its species are colonized by mycorrhizas (above median mycorrhization rank, right graph, versus left graph). Decline or nondecline of species during the 20th century was recorded (from Ozinga *et al.*, 2009). Mycorrhization was extracted for each species from Hempel *et al.* (2013) as 0, 1 or 0.5 (only 'nonmycorrhized', only 'mycorrhized', or both) and averaged within genera (median of averages = 0.8). An analysis treating the mycorrhization of species within genera as a continuous variable yields a significant interaction term 'co-occurrence × mycorrhization' ($t = 1.996$; $P = 0.046$; three extreme outliers excluded).

Courty *et al.*, 2016). Host plants may compete for these services provided by mycorrhizas (Walder *et al.*, 2012, 2015), which may also compete among themselves (Maherali & Klironomos, 2007; Yguel *et al.*, 2014b). Consequently, plant–mycorrhizal interactions may sometimes destabilize entire communities (Bever *et al.*, 2010). Finally, mycorrhizas themselves may be susceptible to environmental change (Courty *et al.*, 2010). In the future, quantification of the benefits versus risks of co-occurring, related hosts exchanging mutualists will be important – not only for plants and mycorrhizas, but for all types of mutualist interactions. Should the benefits predominate, this would provide an improved understanding of how the sharing of habitats among clade members may ultimately lead to better support of host individuals through mutualists, facilitating responses to environmental change.

2. Competition and hybridization increasing within-species variation in habitat use

Here we argue that similarity in habitat use among species within clades contributes to their local co-occurrence, which, in turn, can trigger habitat expansion through character displacement or hybridization. Similar habitat niches resulting in habitat-niche expansion is paradoxical, but might nevertheless be true and explain the lack of perfect niche convergence among related species (Losos, 2008), or the existence of a phylogenetic signal in habitat-niche position but not in niche breadth (Brändle *et al.*, 2002). A species in which the habitat niche remains flexible as a consequence of character displacement or hybridization might be more adaptable to environmental change (Fig. 2).

Character displacement among competitors As explained above (Fig. 1), similar habitat use among clade members within a clade is the necessary prerequisite for their local co-occurrence. Co-occurrence with clade members may trigger not only ecological but also evolutionary interactions. One of these evolutionary interactions is character and niche shifts within each of the co-occurring clade members, resulting in character and niche displacement among species to reduce competition (Dayan & Simberloff, 2005; Stuart & Losos, 2013; Fig. 2). Such character and niche displacement may operate rapidly, within decades or less (Dayan & Simberloff, 2005). Indeed, we find evidence for character displacement in communities composed of phylogenetically closely related species (Prinzing *et al.*, 2008; Gerhold *et al.*, 2011). These local character and niche shifts within species in response to co-occurring fellow clade members will increase the within-species variation in habitat niches and in characters among localities. Increased variation between populations within species may increase the chance that some populations are able to cope with environmental change (Yoshimura & Jansen, 1996; Le Gac *et al.*, 2012). Therefore, we hypothesize that similar habitat use among species within a clade may increase microevolutionary variation in characters and niches within the species of that clade (Silvertown *et al.*, 2006). Such a result might increase the capacity of species to respond to niche-related changes (Fig. 2).

We tentatively tested whether congeneric species occupying similar habitats may increase their intraspecific habitat-niche variation through local interference. We used the published data presented in Section II. We studied genera in which similar habitat use among species corresponds to local co-occurrence and those in which it does not (i.e. low and high, respectively, unsigned residual co-occurrences). For the former, we expect within-species variation in habitat niche to increase with increasingly similar habitat use among species. We indeed found that within-species variation in the habitat niche of genera was influenced by a positive interaction term 'among-species similarity of habitat niche × residual co-occurrence' ($t = 2.5$; $P = 0.014$; Fig. 4): habitat niches within species are particularly variable in those genera in which species occupy similar habitats and locally co-occur. This effect might be attributable to character displacement among relatives within genera. However, note that, with the data at hand, we cannot exclude the possibility that increased within-species variation might also be the cause, rather than the consequence, of co-occurrence. Causalities among the co-occurrence of clade members, their niche variability and their response to environmental change require further investigation (Table 1).

Hybridization The occupation of similar habitats by close relatives within a clade increases the probability of local co-occurrence of these species within patches of the same habitat (Fig. 1). Co-occurrence of relatives increases the likelihood of hybridization (Cavender-Bares *et al.*, 2009), and hybridization may, in turn, often increase evolutionary innovation within species (Aguilee *et al.*, 2012; Abbott *et al.*, 2013; Fig. 2). Hybridization might trigger niche innovation as hybrids often show transgressive traits, exhibiting extremes compared with their parents as a result of

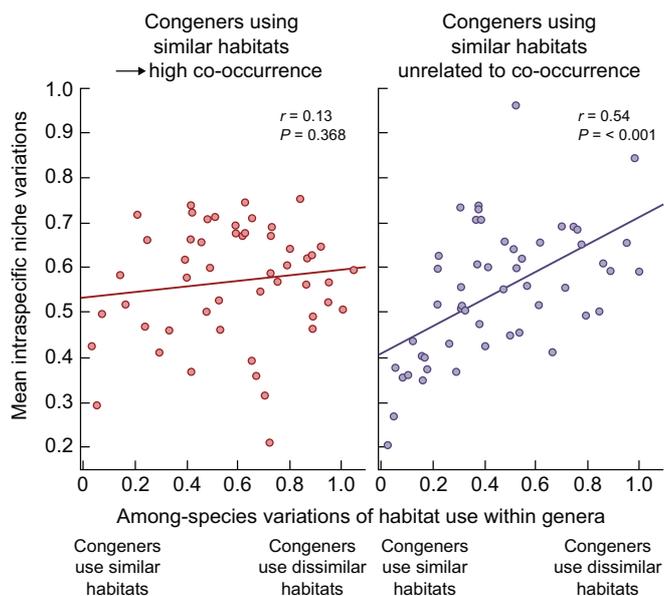


Fig. 4 Genera in which species use similar habitats show low within-species habitat variation (from Ozinga *et al.*, 2013), except if high habitat similarity corresponds to high within-genus co-occurrence (right and left graphs: highest and lowest quartiles of unsigned residual co-occurrences, respectively). The co-occurrence among congeners that occupy similar habitats consequently leads to a *relative* increase of within-species habitat variation. An analysis including genus crown age and species richness as covariables and treating residual co-occurrence as a continuous variable yields a significant interaction term 'habitat similarity \times residual co-occurrence' ($t = 2.5$; $P = 0.014$).

either segregation of parental alleles in hybrids (Rieseberg & Willis, 2007), or of drastic genome reorganization modulating gene expression (Hill & Kotanen, 2011; Doyle *et al.*, 2008). For instance, the 150-yr-old hybridization event between two *Spartina* species that co-occurred in patches of a shared, salt-marsh habitat niche has triggered structural and epigenetic changes in the newly formed allopolyploid *Spartina anglica* (Parisod *et al.*, 2009). These changes are associated with enhanced phenotypic plasticity and increased invasiveness of the plant species within its salt-marsh niche, despite substantial ongoing anthropogenic modification (Ainouche *et al.*, 2009). Finally, hybrids might avoid competition with their parent species by establishing themselves in a new niche, although more research is needed (Glennon *et al.*, 2014). Overall, similar habitat use among related species might accelerate microevolutionary variation in niches within species resulting from increased rates of hybridization. This microevolutionary variation may facilitate the response of extant species to changing environments. The response may happen very rapidly, on time scales of current global change, as in the example of *Spartina* above.

Future research needs to identify when the positive effects of hybridization on evolutionary innovation dominate over the possible negative effects (Table 1). Such negative effects include gene flow between incipient species resulting in genetic homogenization and eventually disappearance of limits between parent species; hybrid species replacing parent species; hybrids suffering from the merging of incompatible parental genomes, or genomes adapted to distinct environments (Burke & Arnold, 2001),

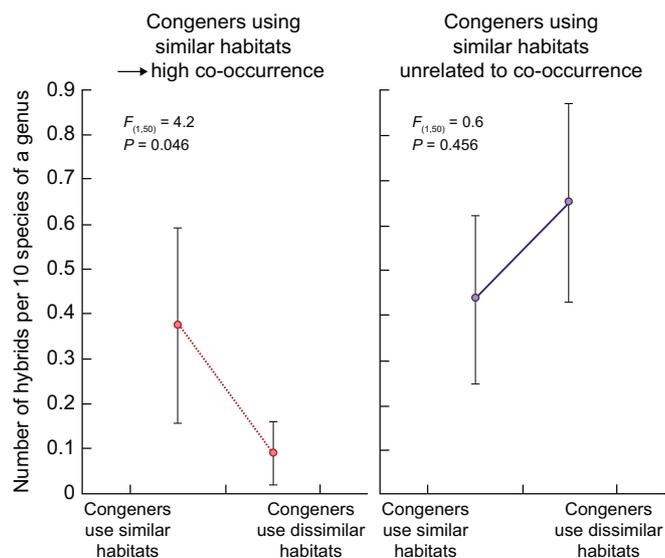


Fig. 5 The relative number of hybrids is high for genera in which species use similar habitats (i.e. below-median habitat variation), provided that using similar habitats corresponds to high intra-generic co-occurrence (left and right graphs: unsigned residual co-occurrence in lower quartile and in upper quartile, respectively). Hybrids are taken from Frank & Klotz (1990) and Jäger & Werner (2005); the similarity in habitat use and co-occurrence are as in Fig. 1 and explained in Section II. Clade members using similar habitats consequently increase the probability of hybridization. Note that analyses using an interaction term and a continuous gradient of niche variation are impossible because of numerous zero values in the dependent variable.

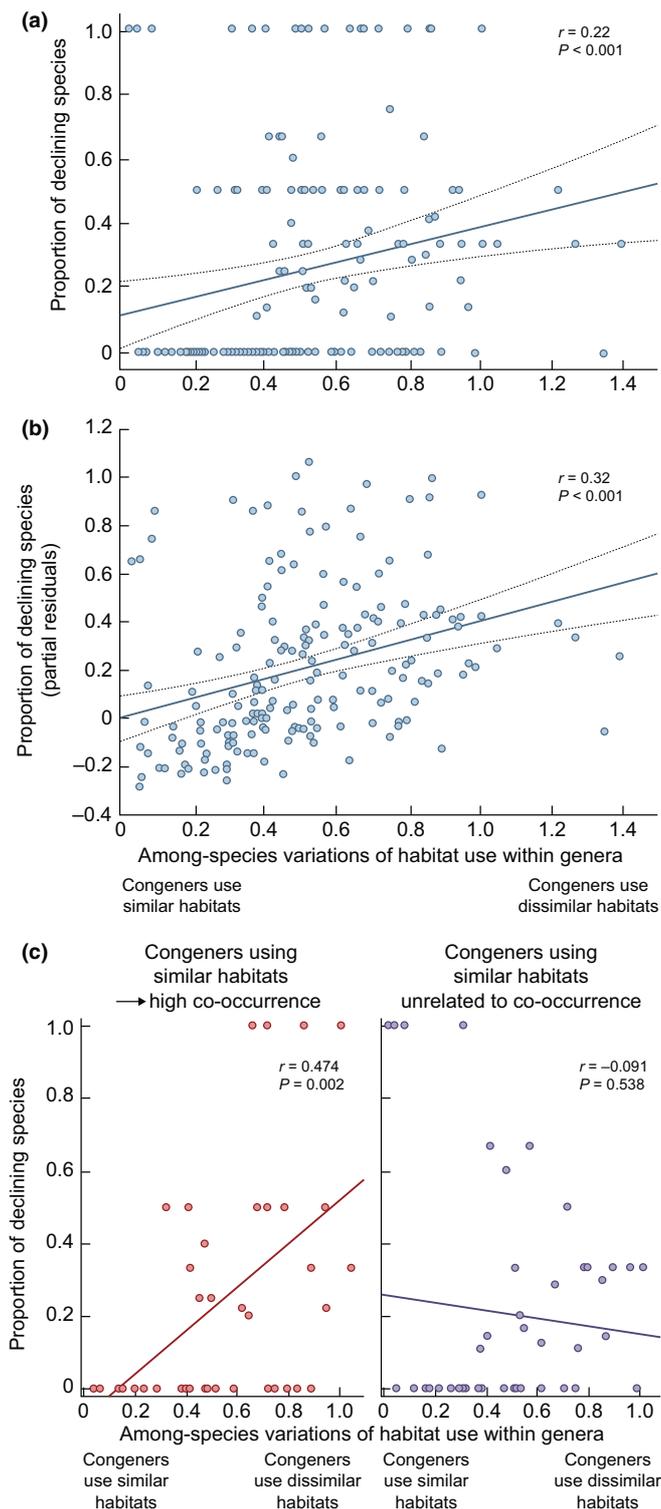
generally leading to hybrid depression. The particular conditions under which hybridization triggers rather than suppresses evolutionary innovation have, to our knowledge, previously not been tested.

As a first step to understanding the relationship between habitat similarity within clades and hybridization, we tested whether occupation of similar habitats among congeneric species corresponds to an increased rate of hybridization, through an increased rate of co-occurrence (this evaluation used published data, as explained in Section II). Indeed, we found hybridization to be higher in those genera whose species use similar habitat types, provided that this similarity in habitat use among congeners corresponds to an elevated co-occurrence, as demonstrated by low unsigned residual co-occurrences (Fig. 5, consistent with the observations of Prinzing *et al.*, 2016).

VI. There is tentative evidence for an eco-evolutionary feedback between habitat similarity among clade members, their co-occurrence, and their reduced vulnerability to environmental change

1. Relationship to present-day decline

Species in clades of high similarity in habitat use and high co-occurrence rarely decline We are not aware of any study that has related habitat similarity within extant clades to the tendency of their constituent species to decline under ongoing environmental change. We hence compared within-genus similarity in habitat use (Section II) to the per-genus proportion of species that declined



during the 20th century (from Ozinga *et al.*, 2009; Section II). We found that genera composed of species occupying similar habitats tend to have a smaller percentage of declining species, even after accounting for the evolutionary age or species richness of the genera (Fig. 6a,b). This lack of decline in species that share similar habitats with congeners might be attributable to the benefits discussed in Section V. Each of these benefits results from increased local co-

Fig. 6 Genera in which species use similar habitats show a low proportion of declining species during the 20th century (from Ozinga *et al.*, 2009).

(a) Simple relationship ($r = 0.22$; $P < 0.001$); (b) partial residuals from a multiple regression analysis accounting for multiple covariables that might influence within-genus variation of habitats or fates of species (species richness, phylogenetic age, and mean within-species variation of habitats) ($r = 0.32$; $P < 0.001$). Note that this relationship is phylogenetically independent: it was found in 14 out of 15 orders. (c) The above relationship is particularly strong for genera in which similar habitat use corresponds to high intra-generic co-occurrence but disappears if similar habitat use does not correspond to co-occurrence (left and right graphs: unsigned residual co-occurrence in lower and higher quartiles, respectively). An analysis including genus crown age and species richness as covariables and treating residual co-occurrence as a continuous variable yields a significant interaction term 'similarity in habitat use \times residual co-occurrence' ($t = 2.48$; $P = 0.014$). The term remains significant ($t = 2.14$; $P = 0.034$) after including mean niche positions as covariables, which are variables classically used to explain species decline.

occurrence among congeners sharing similar habitats. Indeed, this is the trend that we found: that similar habitat use among species within genera corresponds to a low proportion of declining species, provided that similar habitat use in itself corresponds to high local co-occurrence (i.e. unsigned residual co-occurrences are low): the interaction term *habitat use similarity* \times *residual co-occurrence* is significantly positive ($t = 2.48$; $P = 0.014$; Fig. 6c). Tentatively, this indicates that, for angiosperms of the Netherlands, the use of similar habitats by congeners might reduce the risk of population decline through increasing the rate of co-occurrence. The mechanisms involved might include increased intra-specific niche variation, exchange of mutualists, an increased rate of hybridization, or other mechanisms. This conclusion is consistent with our above analyses, which generally confirmed the suggestion that use of similar habitats among congeners decreases vulnerability to environmental change.

In some clades, slow niche evolution in the past correlates to present decline, but the relationship generally is weak Little variation in habitat niches among relatives may, among other causes, result from constrained niche evolution in the past. It would hence be interesting to understand whether lineages that have undergone slow niche evolution in the past are favoured or handicapped today. We are aware of only a single study that systematically links the fate of species to the degree to which these species have retained their ancestors' niches. Lavergne *et al.* (2013) quantified the rate of past niche evolution in families of plants and birds, inferring niche from life history, climate distribution and trophic position. The authors then related rates of past niche evolution to present-day declines during 1970–1990 and during 1990–2000. Overall, families whose species evolved slowly along two of three niche axes tended to decline more strongly during 1970–1990 than families of faster niche evolution. However, this pattern had a major unexplained variation: some families had very slow niche evolution during the past and nevertheless their species are not declining today. Moreover, for the third niche axis, and for all three niche axes versus declines from 1990 to 2000, there was no relationship between the rate of past niche evolution and decline. Overall, there appear to be numerous cases where slow niche

evolution does not render species more vulnerable to environmental change.

2. Relationship to past lifespan in the fossil record

Paleoecological studies permit the documentation of similarity in habitat niches among relatives at long time intervals. Sometimes, such studies permit the tracking of evolution of habitat niches from ancestors to descendants (Notes S3).

Clades of narrow abiotic ranges sometimes had short lifespans in the fossil record ('fossil lifespan' from here on) The range of habitats occupied by entire clades has only rarely been systematically compared to the lifespans of these clades, and only for animals. Liow (2007) found that ostracode genera whose species each occupied a narrow bathymetric range show decreased lifespans, but only in two out of nine data sets. Also from the plant fossil record, several clades are known whose species occupied similar habitats, usually inferred from similar, specialized functional traits. Some of these clades suffered from environmental change such as specialized families in New Zealand during profound Miocene environmental changes (Lee *et al.*, 2001; Conran *et al.*, 2014), or specialized Antarctic clades under ice-cap engulfment of mountains 4–2 million yr ago (Francis & Hill, 1996). Other clades, however, apparently did not suffer from major changes, such as 12 plant lineages in the western North America chaparral from 48 to 11.5 million yr ago that successfully survived within changing chaparral environments (Ackerly, 2004). Overall, habitat similarity within clades does not necessarily decrease clade longevity, although systematic quantitative overviews such as that of Liow (2007) are rare.

Clades of narrow host niches could have very long fossil lifespans We are aware of no example of habitat use by plants directly observed in the fossil record during the evolutionary history of a plant clade. There are, however, observations on habitat use by herbivores. For an insect herbivore, a habitat roughly corresponds to a host plant species and the types of tissues consumed on that host plant. In one example, species of several moth genera were mining foliar tissue of the oak *Quercus agrifolia* for >7 million yr (Opler, 1973, 1974; Fig. S3A). In another example, the component community of arthropod herbivores and detritivores occupying the marattialean tree fern *Psaronius chasei*, from the Euramerican Late Pennsylvanian (Labandeira & Phillips, 2002), included both detritivore and herbivore lineages persisting for up to 45 million yr (D'Rozario *et al.*, 2011) as well as herbivore lineages entering and exiting the *Psaronius* host plant (Fig. S3B). Both case studies indicate that some clades of insect herbivores were successfully specialized on, and phylogenetically conserved, the same 'habitat type' (a tissue of a plant host) for several millions of years, despite major spatiotemporal shifts and profound changes in the ambient palaeoenvironment that made other clades disappear.

VII. Conclusions and future directions

In conclusion, clade members occupying similar habitats do not necessarily suffer more from environmental change than clade

members that occupy different habitats. Indeed, they may often suffer less. We do not suggest that habitat similarity among relatives within a clade alone permits reliable predictions of vulnerability of that clade to environmental change. Other factors such as the use of particularly endangered habitats, slow life histories, restricted range, or species-level ecological specialization are probably more important (reviewed in Pimm, 1991; Colles *et al.*, 2009). We do suggest, however, that there exists a detectable relationship between habitat similarity among species within clades and their vulnerability, and this relationship provides a new perspective for evolutionary ecology.

Specifically, if past evolutionary or biogeographical constraints on diversification of habitat use among species within a clade affect the species' present assembly into local communities, the species' microevolution and their extinction vulnerability, this would be a case of an eco-evolutionary feedback (see Post & Palkovacs, 2009; Gerhold *et al.*, 2015; Mittelbach & Schemske, 2015; for conceptual development); one that has, to our knowledge, not previously been identified. We suggest that low evolutionary diversification of habitats within clades ultimately reduces the vulnerability of species. Such a phenomenon would explain why some clades in which habitat use was constrained in the evolutionary past have succeeded in persisting in spite of past environmental change (Hermant *et al.*, 2012).

We conclude that clades whose species occupy similar habitats might decline less than other clades. We argue that species in such clades often co-occur and interact in a way that increases the species' capacity to respond to environmental change. However, there are cases where this conclusion does not appear to hold. Decline in plant species in South Africa, for instance, does not seem to vary among clades (Davies *et al.*, 2011). Also, many of the genera depicted in Fig. 4(a,b) do not follow the overall relationship between habitat similarity among congeners and decline. We see five possible levels at which our argument may not hold in some cases.

First, occupation of similar habitat niches may not necessarily increase local co-occurrence within habitat patches. Local co-occurrence may be hindered, for instance, if habitat patches are ephemeral or dispersal among patches is limited. Fig. 4(c, right) shows that genera whose species occupy similar habitats without co-occurring may decline just as much as genera whose species occupy dissimilar habitat niches.

Second, co-occurrence among clade members may not trigger the interactions that increase the capacity to respond to environmental change but rather those that have the opposite effect. Recruitment of mutualists, for instance, may not be favoured by co-occurring clade members if mutualists are highly generalist and can interact with any clade – or if mutualists are specialized to a single plant species. Hybridization may not be favoured if species reproduce vegetatively. Conversely, competition may be favoured in undisturbed and productive habitats (Grime, 2001; Huston, 2014). Herbivore pressure may be favoured by co-occurring clade members if herbivores are specialists of that clade, and if the herbivores are little controlled by their enemies, for example, in sites of intermediate productivity or high fragmentation (Fretwell, 1987; Kruess & Tschardtke, 1994).

Third, co-occurrence among clade members may not only be the cause of interactions but also their result. Members of a given clade might co-occur because they cannot cope with the asymmetric competition from other clades, because they are the only species to persist in the face of local predation pressure, because they have been co-dispersed by mutualists, because they have a broad niche, or because they are hybrids having retained their parent species' habitat preferences.

Fourth, interactions among clade members that in theory favour the capacity to respond to environmental change may in practice be of little importance. Support from mutualists, for instance, may be required mainly under nutrient-poor conditions. Variation of habitat niches may be required only if environmental change affects habitats rather than, for instance, the ecotoxicological conditions or the disturbance regime within habitats.

Finally, the current vulnerability of species to environmental change also might be independent of the similarity of habitat use among relatives within clades, an aspect that we did not develop in this study. Specifically, species may be able to track environmental change in space or time or by shifts in metabolomic composition or function (the chemical processing of metabolites). We detail these aspects in Notes S4 and Fig. S4. We explore whether and how clade members using similar habitats track small-scale environmental change in space and time and stress that the phenotypic, epigenetic and metabolic mechanisms behind the capacity of species to shift and expand environmental optima remain to be identified (Fig. S5).

Overall, there might be distinct situations under which the mechanisms we suggest may not apply, but rather those we suggest as 'implications from mainstream ecology', or there is no relationship at all. Future research should systematically test whether such situations may explain why sometimes our conclusion does not hold. The situation of co-occurrence among clade members as a consequence, rather than a cause, of interactions could be excluded by assembling co-occurring clade members experimentally. To some degree this has already been done unconsciously in numerous diversity experiments, which could now be meta-analysed. This future research should include regions that are larger, or more species rich or more pristine than the Netherlands. Research should also account for the effect of co-occurrence among clade members on ecosystem functioning, such as more efficient decomposition of plant litter (Pan *et al.*, 2015). Ultimately, this future research will help us to understand eco-evolutionary questions. Which ecological situations have favoured the persistence of 'specialist' clades composed of species occupying similar habitats, thereby contributing to the frequently reported pattern of niche conservatism (Wiens *et al.*, 2010, phylogenetic signal *sensu* Losos, 2008)? Which ecological situations have contributed to the persistence of generalist species within these 'specialist' clades?

Addressing these questions requires a combination of expertise from ecological, evolutionary and molecular biology, integrating macroevolutionary patterns with local interactions among species in ecosystems. We recommend pursuit of a feedback perspective rather than a unidirectional perspective in which macroevolutionary patterns are given priority to explain local processes or the

inverse (Cornell & Lawton, 1992; Ricklefs, 2004). We hope that this review will assist in improving the integration of often disconnected disciplines by joint study of global macroevolutionary patterns and local interactions among and microevolution within species. Integration of these fields implies integration of varied data and we recommend profiting from extensive published databases that are becoming available for an increasing number of regions of the globe, covering local community assembly across the tree of life.

Acknowledgements

We thank two anonymous referees, Steve Chown, Ary Hoffmann, Dietrich Ober, William DiMichele and Gudrun Bornette for very useful comments on a previous version of the manuscript. Valérie Briand provided technical support. F.H., I.V.B. and A.P. were funded by a CNRS ATIP grant and the French Polar Institute (IPEV) programme no. 136; I.V.B. was supported by a Purkine Fellowship from the Academy of Sciences of the Czech Republic; and P-E.C. has current support from the Swiss National Science Foundation (grant PZ00P3-136651). W.A.O. was supported by the Dutch Science Foundation (NWO Biodiversity Works). A.P., I.V.B. and W.A.O. were supported by The Synthesis Centre for Biodiversity Sciences (sDiv). This is contribution 246 of the Evolution of Terrestrial Ecosystems Consortium at the National Museum of Natural History, in Washington, DC, USA.

References

- Abbott R, Albach D, Ansell S, Arntzen JW, Baird SJE, Bierne N, Boughman J, Brelsford A, Buerkle CA, Buggs R *et al.* 2013. Hybridization and speciation. *Journal of Evolutionary Biology* 26: 229–246.
- Ackerly DD. 2004. Adaptation, niche conservatism, and convergence: comparative studies of leaf evolution in the California chaparral *American Naturalist* 163: 654–671.
- Adler PB, Salguero-Gomez R, Compagnoni A, Hsu JS, Ray-Mukherjee J, Mbeau-Ache C, Franco M. 2014. Functional traits explain variation in plant life history strategies. *Proceedings of the National Academy of Sciences, USA* 111: 740–745.
- Aguilee R, Shaw FH, Rousset F, Shaw RG, Ronce O. 2012. How does pollen versus seed dispersal affect niche evolution? *Evolution* 67: 792–805.
- Ainouche ML, Fortune PM, Salmon A, Parisod C, Grandbastien MA, Fukunaga K, Ricou M, Misset MT. 2009. Hybridization, polyploidy and invasion: lessons from *Spartina* (Poaceae). *Biological Invasions* 11: 1159–1173.
- Alcantara JM, Jaime R, Bastida JM, Rey PJ. 2014. The role of genetic constraints on the diversification of Iberian taxa of the genus *Aquilegia* (Ranunculaceae). *Biological Journal of the Linnean Society* 111: 252–261.
- Alpert P. 2006. Constraints of tolerance: why are desiccation-tolerant organisms so small or rare? *Journal of Experimental Biology* 209: 1575–1584.
- Anacker BL, Klironomos JN, Maherali H, Reinhart KO, Strauss SY. 2014. Phylogenetic conservatism in plant-soil feedback and its implications for plant abundance. *Ecology Letters* 17: 1613–1621.
- Armbruster WS. 2012. Evolution and ecological implications of "specialized" pollinator rewards. In: Patiny S, ed. *Evolution of plant-pollinator relationships*. Cambridge, UK: Cambridge University Press, 44–67.
- Baldwin IT, Hamilton W. 2000. Jasmonate-induced responses of *Nicotiana sylvestris* results in fitness costs due to impaired competitive ability for nitrogen. *Journal of Chemical Ecology* 26: 915–952.
- Bartish IV, Ozinga WA, Bartish MI, Wamelink GWW, Hennekens SM, Prinzing A. 2016. Different habitats within a region contain evolutionary heritage from different epochs depending on the abiotic environment. *Global Ecology and Biogeography* 25: 274–285.

- Bever JD, Dickie IA, Facelli E, Facelli JM, Klironomos J, Moora M, Rillig MC, Stock WD, Tibbett M, Zobel M. 2010. Rooting theories of plant community ecology in microbial interactions. *Trends in Ecology and Evolution* 8: 468–478.
- Brändle M, Brandl R. 2006. Is the composition of phytophagous insects and parasitic fungi among trees predictable? *Oikos* 113: 296–304.
- Brändle M, Prinzing A, Pfeifer R, Brandl R. 2002. Dietary niche breadth for Central European birds: correlations with species-specific traits. *Evolutionary Ecology Research* 4: 643–657.
- Breed AC, Plowright RK, Hayman DTS, Knobel DL, Molenaar FM, Gardner-Roberts D, Cleaveland S, Haydon DT, Kock RA, Cunningham AA *et al.* 2009. Disease management in endangered mammals. In: Delahay RJ, Smith GC, Hutchings MR, eds. *Management of disease in wild mammals*. Berlin, Germany: Springer, 215–239.
- Brennan EB, Weinbaum SA. 2001a. Performance of adult psyllids in no-choice experiments on juvenile and adult leaves of *Eucalyptus globulus*. *Entomologia Experimentalis et Applicata* 100: 179–185.
- Brennan EB, Weinbaum SA. 2001b. Stylet penetration and survival of three psyllid species on adult leaves and 'waxy' and 'de-waxed' juvenile leaves of *Eucalyptus globulus*. *Entomologia Experimentalis et Applicata* 100: 355–363.
- Burke JM, Arnold ML. 2001. Genetics and the fitness of hybrids. *Genetics* 35: 31–52.
- Burns JH, Strauss SY. 2011. More closely related species are more ecologically similar in an experimental test. *Proceedings of the National Academy of Sciences, USA* 108: 5302–5307.
- Cardillo M, Mace GM, Jones KE, Bielby J, Bininda-Emonds ORP, Sechrest W, Orme CDL, Purvis A. 2005. Multiple causes of high extinction risk in large mammal species. *Science* 309: 1239–1241.
- Cavender-Bares J, Kozak KH, Fine PVA, Kembel SW. 2009. The merging of community ecology and phylogenetic biology. *Ecology Letters* 12: 693–715.
- Cavieres LA, Hernandez-Fuentes C, Sierra-Almeida A, Kikvidze Z. 2016. Facilitation among plants as an insurance policy for diversity in Alpine communities. *Functional Ecology* 30: 52–59.
- Christin PA, Osborne CP. 2014. The evolutionary ecology of C₄ plants. *New Phytologist* 204: 765–781.
- Cipollini D, Purrington CB, Bergelson J. 2003. Costs of induced responses in plants. *Basic and Applied Ecology* 4: 79–89.
- Colles A, Liow LH, Prinzing A. 2009. Are specialists at risk under environmental change? – Neoeological, paleoecological and phylogenetic approaches. *Ecology Letters* 8: 849–863.
- Conran JG, Lee WG, Lee DE, Bannister JM, Kaulfuss U. 2014. Reproductive niche conservatism in the isolated New Zealand flora over 23 million years. *Biology Letters* 10: 20140647.
- Cornell HV, Lawton JH. 1992. Species interactions, local and regional processes, and limits to the richness of ecological communities: a theoretical perspective. *Journal of Animal Ecology* 61: 1–12.
- Courty PE, Buée M, Diedhiou AG, Frey-Klett P, Le Tacon F, Rineau F, Turpault MP, Uroz S, Garbaye J. 2010. The role of ectomycorrhizal communities in forest ecosystem processes: new perspectives and emerging concepts. *Soil Biology and Biochemistry* 42: 679–698.
- Courty PE, Munoz F, Selosse MA, Duchemin M, Criquet S, Ziarelli F, Buée M, Plassard C, Taudière A, Garbaye J *et al.* 2016. Into the functional ecology of ectomycorrhizal communities: environmental filtering of enzymatic activities. *Journal of Ecology* 104: 1585–1598.
- Craine JM. 2005. Reconciling plant strategy theories of Grime and Tilman. *Journal of Ecology* 93: 1041–1052.
- Crisp MD, Arroyo MTK, Cook LG, Gandolfo MA, Jordan GJ, McGlone MS, Weston PH, Westoby M, Wilf P, Linder HP. 2009. Phylogenetic biome conservatism on a global scale. *Nature* 458: 754–756.
- Daszak P, Cunningham AA, Hyatt AD. 2000. Emerging infectious diseases of wildlife – threats to biodiversity and human health. *Science* 287: 443–449.
- Davies TJ, Smith GF, Bellstedt DU, Boatwright JS, Bytebier B, Cowling RM, Forest F, Harmon LJ, Muasya AM, Schrire BD *et al.* 2011. Extinction risk and diversification are linked in a plant biodiversity hotspot. *PLoS Biology* 9: e1000620.
- Dayan T, Simberloff D. 2005. Ecological and community-wide character displacement: the next generation. *Ecology Letters* 8: 875–894.
- Dhondt AA, Badyaev AV, Dobson AP, Hawley DM, Driscoll MJL, Hochachka WM, Ley DH. 2006. Dynamics of mycoplasmal conjunctivitis in the native and introduced range of the host. *EcoHealth* 3: 95–102.
- Diekmann M. 2003. Species indicator values as an important tool in applied plant ecology – a review. *Basic and Applied Ecology* 4: 493–506.
- Donoghue MJ. 2008. A phylogenetic perspective on the distribution of plant diversity. *Proceedings of the National Academy of Sciences, USA* 105: 11549–11555.
- Doyle JJ, Flagel LE, Paterson AH, Rapp RA, Soltis DE, Soltis PS, Wendel JF. 2008. Evolutionary genetics of genome merger and doubling in plants. *Annual Review of Genetics* 42: 443–461.
- D’Rozario A, Labandeira CC, Guo WY, Yao YF, Li CS. 2011. Spatiotemporal extension of the Eurasian *Psaronius* component community to the Late Permian of Cathaysia: *in situ* coprolites in a *P. housuensis* stem from Yunnan Province, Southwest China. *Palaeogeography, Palaeoclimatology, Palaeoecology* 306: 127–133.
- Durka W, Michalski SG. 2012. Daphne: a dated phylogeny of a large European flora for phylogenetically informed ecological analyses. *Ecology* 93: 2297.
- Eiserhardt WL, Borchsenius F, Plum CM, Ordóñez A, Svenning J-C. 2015. Climate-driven extinctions shape the phylogenetic structure of temperate tree floras. *Ecology Letters* 18: 263–272.
- Ellenberg H, Weber HE, Düll R, Wirth V, Werner W, Paulissen D. 1992. Indicator values of central European plants. *Scripta Geobotanica* 18: 1–258.
- Francis JE, Hill RS. 1996. Fossil plants from the Pliocene Sirius Group, Transantarctic Mountains: evidence for climate from growth rings and fossil leaves. *Palaios* 11: 389–396.
- Frank D, Klotz S. 1990. Biologisch-ökologische Daten zur Flora der DDR. *Wissenschaftliche Beiträge der Martin-Luther-Universität Halle-Wittenberg* 32: 1–167.
- Fretwell SD. 1987. Food-chain dynamics – the central theory of ecology. *Oikos* 50: 291–301.
- Fürstenberg-Hägg J, Zagrobelny M, Bak S. 2013. Plant defence against insect herbivores. *International Journal of Molecular Sciences* 14: 10242–10297.
- Gassman AJ, Hare JD. 2005. Indirect cost of a defensive trait: variation in trichome type affects the natural enemies of herbivorous insects on *Datura wrightii*. *Oecologia* 144: 62–71.
- Gerhold P, Cahill JF Jr, Winter M, Bartish IV, Prinzing A. 2015. Phylogenetic patterns are not proxies of community assembly mechanisms (they are far better). *Functional Ecology* 29: 600–614.
- Gerhold P, Partel M, Tackenberg O, Hennekens SM, Bartish I, Schaminée JHJ, Fergus AJF, Ozinga WA, Prinzing A. 2011. Phylogenetically poor plant communities receive more alien species, which more easily coexist with natives. *American Naturalist* 177: 668–680.
- Glennon KL, Ritchie ME, Segraves KA. 2014. Evidence for shared broad-scale climatic niches of diploid and polyploid plants. *Ecology Letters* 17: 574–582.
- Gossner MM, Chao A, Bailey RI, Prinzing A. 2009. Native fauna on exotic trees: phylogenetic conservatism and geographic contingency in two lineages of phytophages on two lineages of trees. *American Naturalist* 173: 599–614.
- Grime JP. 2001. *Plant strategies, vegetation processes, and ecosystem properties*, 2nd edn. New York, NY, USA: Wiley.
- Grubb PJ. 1977. Maintenance of species-richness in plant communities – importance of regeneration niche. *Biological Reviews of the Cambridge Philosophical Society* 52: 107–145.
- Hempel S, Götzenberger L, Kühn I, Michalski SG, Rillig MC, Zobel M, Moora M. 2013. Mycorrhizas in the Central European flora: relationships with plant life history traits and ecology. *Ecology* 94: 1389–1399.
- Hennekens SM, Smits NAC, Schaminée JHJ. 2010. SynBioSys Nederland versie 2. Alterra, Wageningen UR. [WWW document] URL <http://www.wageningenur.nl/nl/Expertises-Dienstverlening/Onderzoeksinstituten/Alterra/Faciliteiten-Producten/Software-en-modellen/Synbiosys-1.htm>
- Hermant M, Hennion F, Bartish IV, Yguel B, Prinzing A. 2012. Disparate relatives: life histories vary more in genera occupying intermediate environments. *Perspectives in Plant Ecology and Evolutionary Systems* 14: 281–301.
- Hill MO, Carey PD. 1997. Prediction of Rothamsted Park grass experiment by Ellenberg indicator values. *Journal of Vegetation Science* 8: 579–586.
- Hill SB, Kotanen PM. 2011. Phylogenetic structure predicts capitular damage to Asteraceae better than origin or phylogenetic distance to natives. *Oecologia* 166: 843–851.

- Huston MA. 2014. Disturbance, productivity, and species diversity: empiricism vs. logic in ecological theory. *Ecology* 95: 2382–2396.
- Jäger EJ, Werner K. 2005. *Rothmaler – Exkursionsflora von Deutschland* 4. Heidelberg, Germany: Spektrum Akademischer Verlag.
- Jetter R, Schaffer S, Riederer M. 2000. Leaf cuticular waxes are arranged in chemically and mechanically distinct layers: evidence from *Prunus laurocerasus* L. *Plant, Cell & Environment* 23: 619–628.
- Johnson MG, Granath G, Tahvanainen T, Pouliot R, Stenoien HK, Rochefort L, Rydin H, Shaw AJ. 2015. Evolution of niche preference in *Sphagnum* peat mosses. *Evolution* 69: 90–103.
- Johnson NC, Angelard C, Sanders IR, Kiers ET. 2013. Predicting community and ecosystem outcomes of mycorrhizal responses to global changes. *Ecology Letters* 16: 140–153.
- Karban R, Agrawal AA, Mangel M. 1997. The benefits of induced defenses against herbivores. *Ecology* 78: 1351–1355.
- Kellermann V, Loeschke V, Hoffmann AA, Kristensen TN, Flojgaard C, David JR, Svenning J, Overgaard J. 2012. Phylogenetic constraints in key functional traits behind species' climate niches: patterns of desiccation and cold resistance across 95 *Drosophila* species. *Evolution* 66: 3377–3389.
- Kessler A, Baldwin IT. 2001. Defensive function of herbivore-induced plant volatile emissions in nature. *Science* 291: 2141–2144.
- Kiers ET, Duhamel M, Beesetty Y, Mensah JA, Franken O, Verbruggen E, Fellbaum CR, Kowalchuk GA, Hart MM, Bago A *et al.* 2011. Reciprocal rewards stabilize cooperation in the mycorrhizal symbiosis. *Science* 333: 880–882.
- Kleyer M, Bekker RM, Knevel IC, Bakker JP, Thompson K, Sonnenschein M, Poschlod P, Van Groenendael JM, Klimes L, Klimesova J *et al.* 2008. The LEDA Traitbase: a database of life-history traits of Northwest European flora. *Journal of Ecology* 96: 1266–1274.
- Klotz S, Kühn I, Durka W. 2002. BIOFLOR – a database on biological and ecological traits of vascular plants in Germany. *Schriftenreihe für Vegetationskunde* 38: 1–334.
- Koricheva J. 2002. Meta-analysis of sources of variation in fitness costs of plant antiherbivore defenses. *Ecology* 83: 1761–1790.
- Kruess A, Tscharntke T. 1994. Habitat fragmentation, species loss, and biological-control. *Science* 264: 1581–1584.
- Kunster G, Falster D, Coomes DA, Hui F, Kooyman RM, Laughlin DC, Poorter L, Vanderwel M, Vieilledent G, Wright SJ *et al.* 2016. Plant functional traits have globally consistent effects on competition. *Nature* 529: 204–207.
- Labandeira CC, Phillips TL. 2002. Stem borings and petiole galls from Pennsylvanian tree ferns of Illinois, USA: implications for the origin of the borer and galling functional-feeding-groups and holometabolous insects. *Palaeontographica A* 264: 1–84.
- Lavergne S, Evans MEK, Burfield IJ, Jiguet F, Thuiller W. 2013. Are species' responses to global change predicted by past niche evolution? *Philosophical Transactions of the Royal Society B Biological Science* 368: 20120091.
- Le Gac M, Plucain J, Hindre T, Lenski RE, Schneider D. 2012. Ecological and evolutionary dynamics of coexisting lineages during a long-term experiment with *Escherichia coli*. *Proceedings of the National Academy of Sciences, USA* 109: 9487–9492.
- Lee DE, Lee WG, Mortimer N. 2001. Where and why have all the flowers gone? Depletion and turnover in the New Zealand Cenozoic angiosperm flora in relation to palaeogeography and climate. *Australian Journal of Botany* 49: 341–356.
- Liow LH. 2007. Does versatility as measured by geographic range, bathymetric range and morphological variability contribute to taxon longevity? *Global Ecology and Biogeography* 16: 117–128.
- Losos JB. 2008. Phylogenetic niche conservatism, phylogenetic signal and the relationship between phylogenetic relatedness and ecological similarity among species. *Ecology Letters* 11: 995–1003.
- Maherali H, Klironomos JN. 2007. Influence of phylogeny on fungal community assembly and ecosystem functioning. *Science* 316: 1746–1748.
- Martínez-Vilalta J, Lopez BC, Loepfe L, Lloret F. 2012. Stand- and tree-level determinants of the drought response of Scots pine radial growth. *Oecologia* 168: 877–888.
- Mayfield MM, Levine JM. 2010. Opposing effects of competitive exclusion on the phylogenetic structure of communities. *Ecology Letters* 13: 1085–1093.
- McCallum H, Dobson A. 1995. Detecting disease and parasite threats to endangered species and ecosystems. *Trends in Ecology and Evolution* 10: 190–194.
- McGuire KL. 2007. Common ectomycorrhizal networks may maintain monodominance in a tropical rain forest. *Ecology* 88: 567–574.
- McKinney ML. 1997. Extinction vulnerability and selectivity: combining ecological and paleontological views. *Annual Review of Ecology and Systematics* 28: 495–516.
- Mittelbach GG, Schemske DW. 2015. Ecological and evolutionary perspectives on community assembly. *Trends in Ecology and Evolution* 30: 241–247.
- Niinemets Ü, Valladares F. 2006. Tolerance to shade, drought, and waterlogging of temperate northern hemisphere trees and shrubs. *Ecological Monographs* 76: 521–547.
- Olalla-Tárraga MA, McInnes L, Bini LM, Diniz-Filho JAF, Fritz SA, Hawkins BA, Hortal J, Orme CDL, Rahbek C, Rodríguez MA *et al.* 2011. Climatic niche conservatism and the evolutionary dynamics in species' range boundaries: congruence across mammals and amphibians globally. *Journal of Biogeography* 38: 2237–2247.
- Opler PA. 1973. Fossil lepidopterous leaf mines demonstrate the age of some insect-plant relationships. *Science* 179: 1321–1323.
- Opler PA. 1974. Biology, ecology, and host specificity of microlepidoptera associated with *Quercus agrifolia* (Fagaceae). *University of California Publications in Entomology* 75: 1–83.
- Ozinga WA, Colles A, Bartish IV, Hennion F, Hennekens SM, Pavoine S, Poschlod P, Hermant M, Schaminee JHJ, Prinzing A. 2013. Specialists leave fewer descendants within a region than generalists. *Global Ecology and Biogeography* 22: 213–222.
- Ozinga WA, Römermann C, Bekker RM, Prinzing A, Tamis WLM, Schaminee JHJ, Hennekens SM, Thompson K, Poschlod P, Kleyer M *et al.* 2009. Dispersal failure contributes to plant losses in NW Europe. *Ecology Letters* 12: 66–74.
- Ozinga WA, Schaminee JHJ, Bekker RM, Bonn S, Poschlod P, Tackenberg O, Van Groenendael JM. 2005. Predictability of plant species composition from environmental conditions is constrained by dispersal limitation. *Oikos* 108: 555–561.
- Pan X, Berg MP, Butenschoten O, Murray PJ, Bartish IV, Cornelissen JHC, Dong M, Prinzing A. 2015. Larger phylogenetic distances in litter mixtures: lower microbial biomass and higher C/N ratios but equal mass loss. *Proceedings of the Royal Society B Biological Science* 282: 20150103.
- Parisod C, Salmon A, Zerjal T, Tenaillon M, Grandbastien MA, Ainouche M. 2009. Rapid structural and epigenetic reorganization near transposable elements in hybrid and allopolyploid genomes in *Spartina*. *New Phytologist* 184: 1003–1015.
- Parker IM, Saunders M, Bontrager M, Weitz AP, Hendricks R, Magarey R, Suiter K, Gilbert GS. 2015. Phylogenetic structure and host abundance drive disease pressure in communities. *Nature* 520: 542–544.
- Pearman PB, Guisan A, Broennimann O, Randin CF. 2008. Niche dynamics in space and time. *Trends in Ecology and Evolution* 23: 149–158.
- Peay KG, Kennedy PG, Davies SJ, Tan S, Bruns TD. 2010. Potential link between plant and fungal distributions in a dipterocarp rainforest: community and phylogenetic structure of tropical ectomycorrhizal fungi across a plant and soil ecotone. *New Phytologist* 185: 529–542.
- Pennington R, Lavin M, Prado DE, Pendry CA, Pell SK, Butterworth CA. 2004. Historical climate change and speciation: neotropical seasonally dry forest plants show patterns of both Tertiary and Quaternary diversification. *Philosophical Transactions of the Royal Society London B Biological Science* 359: 515–537.
- Perret M, Chautems A, Spichiger R, Barraclough TG, Savolainen V. 2007. The geographical pattern of speciation and floral diversification in the neotropics: the tribe Sinningieae (Gesneriaceae) as a case study. *Evolution* 61: 1641–1660.
- Petitpierre B, Kueffer C, Broennimann O, Randin C, Daehler C, Guisan A. 2012. Climatic niche shifts are rare among terrestrial plant invaders. *Science* 335: 1344–1348.
- Pimm SL. 1991. *The balance of nature? Ecological issues in the conservation of species and communities*. Chicago, IL, USA: University of Chicago Press.
- Piston N, Armas C, Schob C, Macek P, Pugnaire FI. 2015. Phylogenetic distance among beneficiary species in a cushion plant species explains interaction outcome. *Oikos* 124: 1354–1359.
- Post DM, Palkovacs EP. 2009. Eco-evolutionary feedbacks in community and ecosystem ecology: interactions between the ecological theatre and the

- evolutionary play. *Philosophical Transactions of the Royal Society London B Biological Science* 364: 1629–1640.
- Price PW. 1995. *Insect ecology*, 3rd edn. Chichester, UK: John Wiley & Sons.
- Prinzing A, Durka W, Klotz S, Brandl R. 2001. The niche of higher plants: evidence for phylogenetic conservatism. *Proceedings of the Royal Society London B Biological Science* 268: 2383–2389.
- Prinzing A, Powrie LW, Hennekens SM, Bartish IV, Ozinga WA. 2016. 'High-co-occurrence genera': weak but consistent relationships with global richness, niche partitioning, hybridization and decline. *Global Ecology and Biogeography* 25: 55–64.
- Prinzing A, Reiffers R, Braakhekke WG, Hennekens SM, Tackenberg O, Ozinga WA, Schaminee JHJ, Van Groenendael JM. 2008. Less lineages – more trait variation: phylogenetically clustered plant communities are functionally more diverse. *Ecology Letters* 11: 809–819.
- Purvis A. 2008. Phylogenetic approaches to the study of extinction. *Annual Review of Ecology, Evolution, and Systematics* 39: 301–319.
- Purvis A, Agapow PM, Gittleman JL, Mace GM. 2000. Nonrandom extinction and the loss of evolutionary history. *Science* 288: 328–330.
- Reich PB. 2014. The world-wide 'fast-slow' plant economics spectrum: a traits manifesto. *Journal of Ecology* 102: 275–301.
- Renwick KM, Rocca ME. 2015. Temporal context affects the observed rate of climate-driven range shifts in tree species. *Global Ecology and Biogeography* 24: 44–51.
- Ricklefs RE. 2004. A comprehensive framework for global patterns in biodiversity. *Ecology Letters* 7: 1–15.
- Rieseberg LH, Willis JH. 2007. Plant speciation. *Science* 317: 910–914.
- Rineau F, Courty PE. 2011. Secreted enzymatic activities of ectomycorrhizal fungi as a case study of functional diversity and functional redundancy. *Annals of Forest Science* 68: 69–80.
- Ross MA, Harper JL. 1972. Occupation of biological space during seedling establishment. *Journal of Ecology* 60: 77–88.
- Sargent RD, Kembel SW, Emery NC, Forrester EJ, Ackerly DD. 2011. Effect of local community phylogenetic structure on pollen limitation in an obligately insect-pollinated plant. *American Journal of Botany* 98: 283–289.
- Schaffers AP, Sykora KV. 2000. Reliability of Ellenberg indicator values for moisture, nitrogen and soil reaction: a comparison with field measurements. *Journal of Vegetation Science* 11: 225–244.
- Schaminée JHJ, Hennekens SM, Ozinga WA. 2012. The Dutch national vegetation database. *Biodiversity and Ecology* 4: 201–209.
- Scheffer M, Van Nes EH. 2006. Self-organized similarity, the evolutionary emergence of groups of similar species. *Proceedings of the National Academy of Sciences, USA* 103: 6230–6235.
- Schoonhoven LM, Van Loon JJA, Dicke M. 2005. *Insect-plant biology*. Oxford, UK: Oxford University Press.
- Schwartz MW, Simberloff D. 2001. Taxon size predicts rates of rarity in vascular plants. *Ecology Letters* 4: 464–469.
- Sedio BE, Wright SJ, Dick CW. 2012. Trait evolution and the coexistence of a species swarm in the tropical forest understorey. *Journal of Ecology* 100: 1183–1193.
- Selosse MA, Richard F, He X, Simard SW. 2006. Mycorrhizal networks: des liaisons dangereuses? *Trends in Ecology and Evolution* 21: 621–628.
- Shudo E, Iwasa Y. 2001. Inducible defense against pathogens and parasites: optimal choice among multiple options *Journal of Theoretical Biology* 209: 233–247.
- Siefert A, Lesser MR, Fridley JD. 2015. How do climate and dispersal traits limit ranges of tree species along latitudinal and elevational gradients? *Global Ecology and Biogeography* 24: 581–593.
- Siemens DH, Haugen R, Matzner S, Vanasna N. 2009. Plant chemical defence allocation constrains evolution of local range. *Molecular Ecology* 18: 4974–4983.
- Silvertown J, Dodd M, Gowing D, Lawson C, McConway K. 2006. Phylogeny and the hierarchical organization of plant diversity. *Ecology* 87: S39–S49.
- Sjöström A, Gross CL. 2006. Life-history characters and phylogeny are correlated with extinction risk in the Australian angiosperms. *Journal of Biogeography* 33: 271–290.
- Smith SE, Read DJ. 2008. *Mycorrhizal symbiosis*. Cambridge, UK: Academic Press.
- Sodhi NS, Koh LP, Peh KSH, Tan HTW, Chazdon RL, Corlett RT, Lee TM, Colwell RK, Brook BW, Sekercioglu CH *et al.* 2008. Correlates of extinction proneness in tropical angiosperms. *Diversity and Distributions* 14: 1–10.
- Strauss SY, Lau JA, Carroll SP. 2006. Evolutionary responses of natives to introduced species: what do introductions tell us about natural communities? *Ecology Letters* 9: 357–371.
- Stuart YE, Losos JB. 2013. Ecological character displacement: glass half full or half empty? *Trends in Ecology and Evolution* 28: 402–408.
- Teste FP, Simard SW. 2008. Mycorrhizal networks and distance from mature trees alter patterns of competition and facilitation in dry Douglas-fir forests. *Oecologia* 158: 193–203.
- Thuiller W, Lavorel S, Araujo MB, Sykes MT, Prentice IC. 2005. Climate change threats to plant diversity in Europe. *Proceedings of the National Academy of Sciences, USA* 102: 8245–8250.
- Tobias JA, Cornwallis CK, Derryberry EP, Claramunt S, Brumfield RT, Seddon N. 2014. Species coexistence and the dynamics of phenotypic evolution in adaptive radiation. *Nature* 506: 359–363.
- Vamosi JC, Wilson JR. 2008. Nonrandom extinction leads to elevated loss of angiosperm evolutionary history. *Ecology Letters* 11: 1047–1053.
- Van der Heijden MGA. 2002. Arbuscular mycorrhizal fungi as a determinant of plant diversity. In: Van der Heijden MGA, Sanders IR, eds. *Mycorrhizal ecology*. Berlin, Germany: Springer, 243–266.
- Van der Heijden MGA, Martin FM, Selosse MA, Sanders IR. 2015. Mycorrhizal ecology and evolution: the past, the present, and the future. *New Phytologist* 205: 1406–1423.
- Van der Meijden R, Odé B, Groen CLG, Witte JPM, Bal D. 2000. Endangered and vulnerable vascular plants in the Netherlands. Basic report with proposal for the Red List. *Gorteria* 26: 85–208.
- Van velzen E, Etienne RS. 2015. The importance of ecological costs for the evolution of plant defense against herbivory. *Journal of Theoretical Biology* 372: 89–99.
- Vergnon R, Van Nes EH, Scheffer M. 2012. Emergent neutrality leads to multimodal species abundance distributions. *Nature Communications* 3: 663.
- Vialatte A, Bailey R, Vasseur C, Matocq A, Goßner M, Everhart D, Vitrac X, Belhadj A, Ernoult A, Prinzing A. 2010. Phylogenetic isolation of host trees affects assembly of local Heteroptera communities. *Proceedings of the Royal Society London B Biological Science* 227: 2227–2236.
- Villalobos F, Rangel TF, Diniz-Filho JAF. 2013. Phylogenetic fields of species: cross-species patterns of phylogenetic structure and geographical coexistence. *Proceedings of the Royal Society of London B: Biological Science* 280: 20122570.
- Violle C, Nemergut DR, Pu ZC, Jiang L. 2011. Phylogenetic limiting similarity and competitive exclusion. *Ecology Letters* 14: 782–787.
- Walder F, Brulé D, Koegel S, Wiemken A, Boller T, Courty PE. 2015. Plant phosphorus acquisition in a common mycorrhizal network: regulation of phosphate transporter genes of the Pht1 family in sorghum and flax. *New Phytologist* 205: 1632–1645.
- Walder F, Niemann H, Mathimaran N, Lehmann MF, Boller T, Wiemken A. 2012. Mycorrhizal networks: common goods of plants shared under unequal terms of trade. *Plant Physiology* 159: 789–797.
- Wang SP, Chen AP, Fang JY, Pacala SW. 2013. Why abundant tropical tree species are phylogenetically old. *Proceedings of the National Academy of Sciences, USA* 110: 16039–16043.
- Wardhaugh CW. 2014. The spatial and temporal distributions of arthropods in forest canopies: uniting disparate patterns with hypotheses for specialisation. *Biological Reviews* 89: 1021–1041.
- Watanabe K, Murakami M, Hirao T, Kamata N. 2014. Species diversity estimation of ambrosia and bark beetles in temperate mixed forests in Japan based on host phylogeny and specificity. *Ecological Research* 29: 299–307.
- Webb CO, Ackerly DD, McPeck MA, Donoghue MJ. 2002. Phylogenies and community ecology. *Annual Review of Ecology and Systematics* 33: 475–505.
- Wiens JJ, Ackerly DD, Allen AP, Anacker BL, Buckley LB, Cornell HV, Damschen EI, Jonathan Davies T, Grytnes JA, Harrison SP *et al.* 2010. Niche conservatism as an emerging principle in ecology and conservation biology. *Ecology Letters* 13: 1310–1324.
- Wilkinson DM. 1998. The evolutionary ecology of mycorrhizal networks. *Oikos* 82: 407–410.
- Wise MJ, Rausher MD. 2013. Evolution of resistance to a multiple-herbivore community: genetic correlations, diffuse coevolution, and constraints on the plant's response to selection. *Evolution* 67: 1767–1779.

- Yguel B, Bailey R, Tosh ND, Vialatte A, Vasseur C, Vitrac X, Jean F, Prinzing A. 2011. Phytophagy on phylogenetically isolated trees: why hosts should escape their relatives. *Ecology Letters* 14: 1117–1124.
- Yguel B, Bailey R, Villemant C, Brault A, Jactel H, Prinzing A. 2014a. Enemy release of insect herbivores on phylogenetically isolated trees: why phytophages should follow plants escaping their relatives? *Oecologia* 176: 521–532.
- Yguel B, Courty PE, Jactel H, Prinzing A. 2014b. Below-ground mutualists support oaks growing in a phylogenetically distant neighbourhood. *Soil Biology and Biochemistry* 78: 204–212.
- Yoshimura J, Jansen VAA. 1996. Evolution and population dynamics in stochastic environments. *Researches on Population Ecology* 38: 165–182.
- Yu H, Nason JD. 2013. Nuclear and chloroplast DNA phylogeography of *Ficus hirta*: obligate pollination mutualism and constraints on range expansion in response to climate change. *New Phytologist* 197: 276–289.
- Zanne AE, Tank DC, Cornwell WK, Eastman JM, Smith SA, FitzJohn RG, McGlenn DJ, O'Meara BC, Moles AT, Reich PB *et al.* 2014. Three keys to the radiation of angiosperms into freezing environments. *Nature* 506: 89–92.
- Zavala JA, Patankar AG, Gase K, Baldwin IT. 2004. Constitutive and inducible trypsin proteinase inhibitor production incurs large fitness costs in *Nicotiana attenuata*. *Proceedings of the National Academy of Sciences, USA* 101: 1607–1612.

Supporting Information

Additional Supporting Information may be found online in the Supporting Information tab for this article:

Fig. S1 An example of variation in habitats among species within different angiosperm genera, based on the flora of the Netherlands.

Fig. S2 Relationship between competitiveness and habitat similarity within genera.

Fig. S3 Varying degrees of habitat similarity among ancestors and descendants in the fossil record of plant–insect interactions.

Fig. S4 Scheme summarizing the scenarios in which similarity in habitat use among species within clades has no consequences for the vulnerability of species to present environmental change.

Fig. S5 Habitat tracking as a function of the capacity for long-distance dispersal, adult life span and habitat similarity within genera.

Notes S1 Habitat similarity among species within each of the angiosperm genera in the Netherlands.

Notes S2 Relationship between competitiveness and habitat similarity within genera.

Notes S3 Habitat similarity among related species in fossil plant–insect relationships.

Notes S4 Species within clades occupying similar habitats: no consequences for the capacity to respond to environmental change.