



Do we need squirrels everywhere?

On the distinction between biodiversity and nature

B8

M.M. Gossner^{1,2}, T. Wohlgemuth¹

¹ Swiss Federal Institute for Forest, Snow and Landscape Research WSL, Birmensdorf, Switzerland

² Swiss Federal Institute of Technology ETH, Zürich, Switzerland

We observe an ongoing biodiversity decline and mitigating this biodiversity crisis is highly demanded by science, society, and politics. Therefore, multitrophic and evidence-based scientific approaches across multiple scales are needed. Biodiversity is affected by both natural disturbance and forest management at different temporal and spatial scales, with consequences for ecosystem functioning and human well-being. Accordingly, species diversity in forests is not evenly distributed, but dynamic across temporal and spatial scales. Stochastic processes of niche distribution are not only present in primeval forests but will also increasingly become important in managed landscapes due to climate change induced extreme events such as windthrow, fire, or drought. These build the basis for natural processes that beget biodiversity. The case of the Bavarian forest demonstrates that natural disturbances and subsequent successional pathways across large areas result in a dynamic occurrence of patches of low and high diversity. In naturally disturbed areas, as well as in managed landscapes, both rarity and commonness exist as consequence of species extinctions and biogeographic processes. The quality of naturally evolved biodiversity is, however, different from e.g. the maximised biodiversity fostered by biodiversity management. While we do not need every species everywhere, we do need strategies that guarantee the distribution of different successional stages at the landscape scale in a configuration that allows the coexistence of most species. This will be important to achieve resilient forest ecosystems and hence maintain ecosystem functions and services over a longer time-scale. We therefore advocate the need for large forest landscapes that eventually should be – against all odds – released from management in the future. Such large unmanaged landscapes not only will result in more dense populations of highly demanding species that also can spread out to managed landscapes, but will also serve as long-term references of natural forest processes and their benefits for biodiversity. One of the main challenges will be to develop strategies that aligns scientific evidence with the needs of society and policy makers.

On the rise of biodiversity

The industrial revolution of the eighteenth century with increasing exploitation of resources and

urbanisation was followed by the growing societal awareness of the need to protect the nature that was being lost. During this time, many people such as Charles Darwin and Alexander von Humboldt were fascinated by the diversity of living organisms and their forms and functions and travelled all over the world collecting organisms and naming them. Over time there was a growing realisation that this biodiversity is vulnerable, e.g. by Rachel Carson in 'Silent Spring' (Carson 1962). The digital revolution beginning at the end of the twentieth century has subsequently led to the now broad awareness of the loss of biodiversity (Sala *et al.* 2000). In particu-

< Fig. B8.1. The necessity to provide habitats for forest species, incl. such beautiful coral fungi (*Ramaria* spp.) is more than a moral commitment. The value of often unimpressive species for ecosystem functioning is usually little known among society but undisputed in the scientific community. However, it can be difficult to convince the broad society of the need to sustain biodiversity as a whole and not only spectacular flagship species (Photo: Ulrich Wasem).

lar, it made available to most people those things that had previously only been available to the privileged: diversity of food, knowledge and data, and forms of communication. Global markets and digital interconnection have made many of the demanded things more easily accessible, e.g. by reducing costs (food), or by increasing availability (free knowledge through the internet, or unlimited communication through digital networks). Population growth led to increased human demand for natural resources that caused overexploitation, with e.g. intensified agriculture being still a prevalent threat to biodiversity (Maxwell *et al.* 2016). The strong growth of world-wide resource exchanges has additionally caused the extinction of species (the so-called ‘Holocene extinction’ or ‘Anthropocene extinction’ or ‘sixth mass extinction’) (Ripple *et al.* 2017). The increasing loss of species has also been discussed as a threat to the future of humankind. This insight resulted in the inclusion of the term ‘biological diversity’ in the Convention on Biological Diversity (CBD), which was opened for signature at the Rio Earth Summit in 1992. The CBD defines biological diversity as:

... the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems. (UNEP 1994)

The concept was further developed in the Millennium Ecosystem Assessment (Hassan *et al.* 2005) and the reports of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES 2019).

The digital revolution opened up new possibilities for data storage and data reproducibility. For example, data on species occurrences and distribution enabled us to quantify whether species are common or rare, the basis of nature conservation. Therefore, the importance of biodiversity has continuously grown and is increasingly being linked to the issue of most concern nowadays: climate change.

Changes in habitat quantity, quality, and diversity often correspond with the variation of individual species or species groups, in particular as a consequence of the species-area relationship and related fragmentation effects (e.g. Tscharnkte *et al.* 2012). Beyond the evolutionary and biogeographic

constraints that frame regional floras and faunas, both natural disturbance and forest management change the resource availability and diversity as well as microclimatic properties in multiple ways on temporal scales of years to centuries and in particular at smaller spatial scales. As a consequence, species diversity in forests is not evenly distributed but varies in time and space. On the one hand, many species are associated with undisturbed old-growth forest patches and some of these are generally low in abundance. On the other hand, there are old-growth forest species that only occur in open forest patches such as forest gaps and their abundance depend on the frequency and size of such disturbances; these species might partly occur also in more open and disturbed areas in managed landscapes as long as the required resources and habitats are available at sufficient temporal and spatial scales. Nowadays, primeval forests are mostly restricted to small patches, which are often too small to allow diverse natural gap dynamics. As a consequence, the richness of iconic species in old-growth forests, not disturbed by humans, is often low and transiently high in patches where disturbance has taken place. As a result, stochastic processes of niche distribution are not only present in primeval forests but will also increasingly become important in managed landscapes because of climate change induced extreme events (e.g. windthrow, fire, drought). These build the basis for natural processes that beget biodiversity.

Natural vs. biodiverse forests

Nature is the variety of life and of non-human landscape features and includes “*all the animals, plants, rocks, etc. in the world and all the features, forces, and processes that happen or exist independently of people, such as the weather, the sea, mountains, the production of young animals or plants, and growth*” (Cambridge Dictionary). Biodiversity can be thought of in a more neutral or more scientific way as the expression ‘nature’ (UNEP 1994; Thompson 2010). Thinking of ‘nature’ conjures up thoughts of ‘virgin, pristine, or unmanaged land’, ‘wilderness’, and ‘unplanned’ processes. In contrast, thinking about ‘biodiversity’ also means ‘quantifying’, ‘measuring’, and ‘planning’. Unmanaged ecosystems underlie unplanned disturbances that shape natural processes. The absence of management



Fig. B8.2. (a) European spruce bark beetle (*Ips typographus*) has generated large-scale disturbances in Norway spruce (*Picea abies*) forests in Europe. Because of its functional impact, creating habitats for many species, it is considered an ecosystem engineer (Lawton and Jones 1995; Müller *et al.* 2008) (Photo: Beat Wermelinger). (b) After the spruce bark beetle outbreak in the Bavarian Forest National Park, the high availability of spruce stumps colonised by, for instance, the polypore *Fomitopsis pinicola*, greatly increased the populations of the threatened flat bug *Aradus obtectus* (6.3–9.5 mm) (male-top, female-bottom; Photo: Martin Gossner), which sucks on hyphae of *F. pinicola*. *Aradus obtectus* is also quite abundant in the primeval forest of Scatlè (Grisons, Switzerland), where there is a large amount of suitable habitat.

does not mean an absence of dynamics, but rather the absence of planning, i.e. the absence of humans. This is the story of the forest in Bavaria, Germany. In the 1990s, this forest was affected by infestations of bark beetles and windthrows (fig. B8.2) across large areas with widespread tree death in some stands; rather than taking steps to limit the extent of the infestation, the forest was allowed to develop and regenerate without human intervention. The area affected increased rapidly, reaching a maximum of >800 ha per year in 1996 and comprised a total area of 40 % in the older park of the national park (Müller *et al.* 2008, Lehnert *et al.* 2013). The unplanned processes have allowed millions of visitors to see nature ‘striving’ (Mayer 2014), and have resulted in increases in populations of species that used to be locally threatened (figs B8.2, B8.6; Thorn *et al.* 2017). Such processes contrast with most of the managed, and hence controlled, forests in Europe; by such processes, we can ask questions about the meaning of nature and biodiversity.

The case of the Bavarian forest nicely illustrates the relevance of dynamics in natural ecosystems for species variation over a longer time-scale. Biodiversity varies enormously over space and time leading to patches of low biodiversity as well as patches of high biodiversity at particular points in time; this also applies to unmanaged forests. Maximising biodiversity at the local scale is thus not necessarily

related to a natural reference. This raises the question of whether increasing biodiversity should be an ultimate target irrespective of a natural reference (which is mostly not available in a cultural landscape like Central Europe) and irrespective of spatial and temporal dynamics (see also section “To protect as much as possible: scale matters”). However, the question ‘Which biodiversity should be targeted in nature conservation?’ is not only a matter of natural references such as any natural ecosystem; the question also requires consideration of our normative nature, i.e. our wish to ‘re-establish’ or ‘rescue’ biodiversity to the extent that we think existed at some unspecified time, and in particular to when biodiversity presumably peaked; this is often referenced to a time 100–200 years ago (Landolt 1991; Korneck *et al.* 1998). All this has to be considered in conservation and management strategies. In any case, we need adaptive approaches as biodiversity and processes are generally dynamic.

Biodiversity is more than just species richness

Many recent studies have shown that biodiversity should not only be preserved because of its intrinsic value, but also because it is related to important ecosystem functions and services. There is growing evidence that greater biodiversity increases ecosys-

tem functioning (Isbell *et al.* 2011; Ratcliffe *et al.* 2017) and promotes the services that nature provides for humans, e.g. cultural services (BEF: Biodiversity-Ecosystem Functioning Relationships; van der Plas 2019). For example, in a global study Liang *et al.* (2016) found that there was a correlation between plant diversity in forests and forest productivity (in terms of tree volume), with less diverse forests having a lower productivity; the association was likely due to both a sampling effect (i.e. diverse forests more likely have a functionally important, e.g. productive, species in place than species-poor forests) and because there is a more complete resource use when there is a greater diversity of plants that occupy different parts of the functional trait space (Roscher *et al.* 2012). Functional traits are generally defined as morphological, physiological, and behavioural characteristics that affect species fitness (Violle *et al.* 2007). In the perspective of BEF relationships these are properties of species that are linked to a particular function. For example: (1) a variety of root systems among plant species allow for a more effective resource use and thus higher productivity (Forrester and Bauhus 2016); (2) a variability in the periods when foragers are active (e.g. daily, seasonal, active during sunshine or rain) and in morphological characteristics of mouth parts (e.g. trait matching between tongue length and flower morphology) of pollinators lead to more effective pollination (Blüthgen and Klein 2011); and (3) the variability in resource use by herbivores (e.g. feeding on different plant parts) intensifies nutrient cycling (Blüthgen and Klein 2011). In conclusion, beside species richness per se, also species functional characteristics and identity as well as the abundance of each species and thus the evenness of species (i.e. how equal the community is numerically in terms of individuals) might be crucial for the outcome of an ecosystem process or function (Wilsey and Potvin 2000; Hooper *et al.* 2005). Despite the general importance of BEF relationships, the correlation might be weak in cases where an ecosystem process is driven by a few abundant key species, as is the case for crop pollinating bees (Kleijn *et al.* 2015), but might be underestimated in more natural systems (Garibaldi *et al.* 2011; Garibaldi *et al.* 2013). There is a huge variation in pollination efficiency (Jauker *et al.* 2012; Eraerts *et al.* 2020) among the >700 bee species in Switzerland (Westrich 1990; Michener 2007). While the domesticated honey bee (*Apis mellifera*) and a few other species might be

sufficient to pollinate the vast majority of crops, we need many more bee species to ensure that we have diverse plant communities that provide ecosystem services of productivity, erosion control, resistance to environmental change, and not to forget diverse habitats for a diverse invertebrate community (Garibaldi *et al.* 2013). This becomes even more important in the context of ecosystem resistance and resilience (see also Chapter B9, Lindner *et al.*, this book). Resistance is the ability for an ecosystem to remain unchanged when subjected to a disturbance. Resilience is the ability and rate of an ecosystem to recover from a disturbance and return to its pre-disturbed state (Ghazoul *et al.* 2015; Oliver *et al.* 2015). It is generally believed that more diverse systems have a higher resistance and resilience to environmental perturbations because there is more likely to be functional redundancy in a diverse ecosystem, i.e. if one species becomes extinct in a particular area after a disturbance, there are more likely to be other species that can fulfil the ecological roles previously played by the newly extinct species ('insurance hypothesis') (Mori *et al.* 2013; Pillar *et al.* 2013; Silva Pedro *et al.* 2015). Climate change will most likely further increase temperature and the frequency of extreme events, and this will further challenge ecosystem resistance and resilience. Maintaining a high functional diversity, which is currently under threat (Hallmann *et al.* 2017; Seibold *et al.* 2019; van Klink *et al.* 2020), is thus crucial for mitigating climate change effects.

To protect as much as possible: scale matters

To protect biodiversity requires an understanding of how biodiversity is organised in space and time; such understanding enables prediction of the impact of environmental change across different scales. Many recent studies have demonstrated that a focus on the biodiversity at the forest stand level (known as α -diversity) is not sufficient to predict biodiversity at larger spatial scales (landscape-scale diversity: γ -diversity; figs B8.2 and B8.3) because species turnover among forest stands and regions (β -diversity) might be even more important and driven by other factors (Müller and Gossner 2010; Gossner *et al.* 2013; Schall *et al.* 2018).

Ecological theory predicts that environmental heterogeneity generally promotes biodiversity

(Heterogeneity-Diversity Hypothesis; MacArthur and MacArthur 1961; Wilson 2000). However, the general validity of this relationship has been questioned recently as high heterogeneity might reduce the available area per species and, thus, population size to a degree where stochastic events can lead to local extinctions of species (Area-Heterogeneity Trade-off Hypothesis; Allouche *et al.* 2012). Species seem to show variable responses and a recent comprehensive study did not find consistent support for a generalisable mechanism determining patterns of heterogeneity–diversity relationships (Heidrich *et al.* 2020). From an applied perspective, it is not yet fully understood how scale and, thus, the spatial grain of management within one management system affects biodiversity at larger spatial scales, and how this is shaped by the composition and configuration of management systems.

Beech (*Fagus sylvatica*) forests are restricted to Europe and are one of the most important natural European forest types, covering an area of about 14–15 million ha (Brunet *et al.* 2010). Therefore, Europe has a responsibility to conserve forests dominated by this species (Knapp and Spangenberg 2007). Nature conservation policy and forest policy advocate fine-grained, so-called ‘close-to-nature’ management systems over traditional coarse-grained shelterwood systems in beech forests. This is driven by the idea that these management systems well mimic the prevailing natural disturbance regime characterised by small-scale gap dynamics (e.g. primeval beech forest remnants of eastern Central Europe, Hobi *et al.* 2015). Management systems emulating small-scale gap dynamics have also been promoted by forest policy and nature conservation in temperate forests of other parts of the world (Coates and Burton 1997; Ott and Juday 2002). While there are strong indications for the positive effects on biodiversity of selection cutting compared to large-scale clearcutting in boreal forests (Hjältén *et al.* 2017; Joelsson *et al.* 2018), there is little empirical evidence that uneven-aged (selection cutting) compared to even-aged (shelterwood system) management in beech forests actually increase biodiversity. Shelterwood systems in beech forests operate at much smaller scales than clearcutting systems in boreal forests. A recent study in European beech forests for instance has shown that even-aged forest management at a scale of 4–8 ha management units promotes a higher diversity of most taxonomic and trophic groups relative to une-

ven-aged fine-grain management. This was not only an effect of disturbance indicators or open land species as forest specialists were similarly affected (Schall *et al.* 2018). Moreover, mixing even-aged with uneven-aged and young unmanaged forests – in contrast to unmanaged beech forests in eastern Europe, the majority of unmanaged beech forests in Central Europe have been set aside relatively recently and largely lack old-growth features – in a landscape seems not to benefit biodiversity (Schall *et al.* 2020).

It underlines the importance of environmental heterogeneity at larger spatial scales for biodiversity levels, and the role of spatial grain (either management or disturbance) as a trigger to vary the abundance and diversity of specific species groups. On one hand, the spatial grain of management defines which species may establish viable populations given stochastic events that likely cause local extinction (Allouche *et al.* 2012). On the other hand, genetic exchange among populations is reduced by landscape fragmentation by roads and other built environments, but also by management units that are too large in even-aged management systems. To allow the survival of species requires continuous availability of suitable habitats at sufficiently close proximity to sustainably allow dispersal and establishment. Limited dispersal capacities in fragmented habitats mean that regional and internationally agreed measures to ensure survival of populations of threatened species are required, e.g. the establishment of corridors and stepping-stones. These measures have already been applied in biodiversity conservation strategies that aim to promote biodiversity such as the strategy implemented in the Steigerwald Forest in Germany where active deadwood enrichment is combined with the protection of habitat trees and protection of forest patches of different sizes (Bollmann and Braunisch 2013; Doerfler *et al.* 2017; Doerfler *et al.* 2018; Bürgi *et al.* Chapter B2 in this book).

Scale issues need to be addressed when considering management systems and related forest structural properties, i.e. the scale at which particular resources are altered in terms of amount and diversity. For example, whether habitat amount or habitat connectivity is more important is the subject of lively debate (Fahrig 2013), with an increasing number of studies supporting the habitat amount as being a more important factor (Seibold *et al.* 2017; Komonen and Müller 2018). Also, dispersal

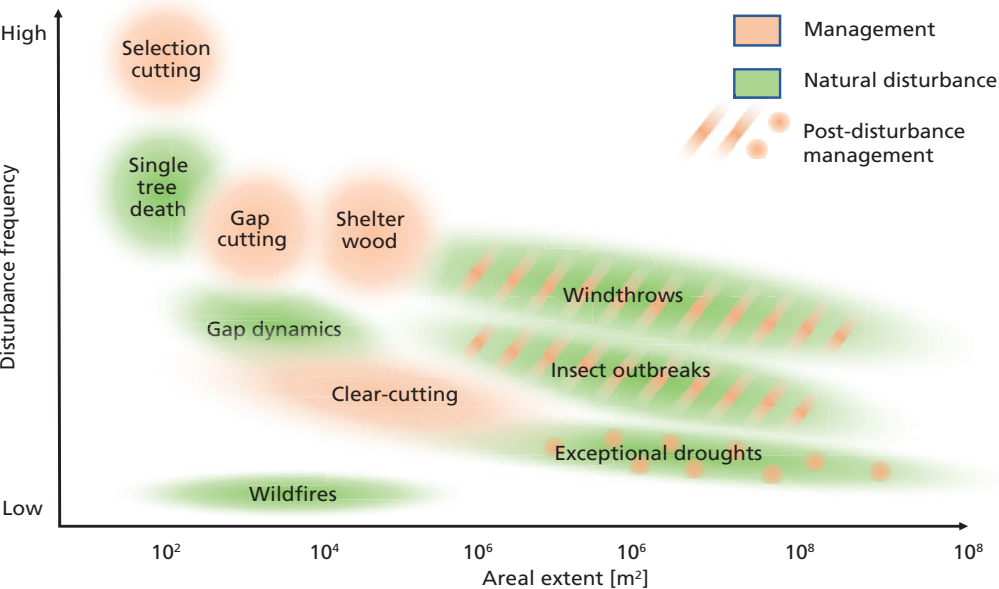


Fig. B8.3. Comparison of disturbances to forests in space and time (based on Spies and Turner [1999]; Leverkus *et al.* [2018]). The post-disturbance management includes, in particular, ‘salvage logging’ that is often applied on a large scale after windthrows and insect outbreaks and that threatens biodiversity (Thorn *et al.* 2018) and ‘sanitary fellings’ that are applied after exceptional droughts that mostly cause scattered tree death.

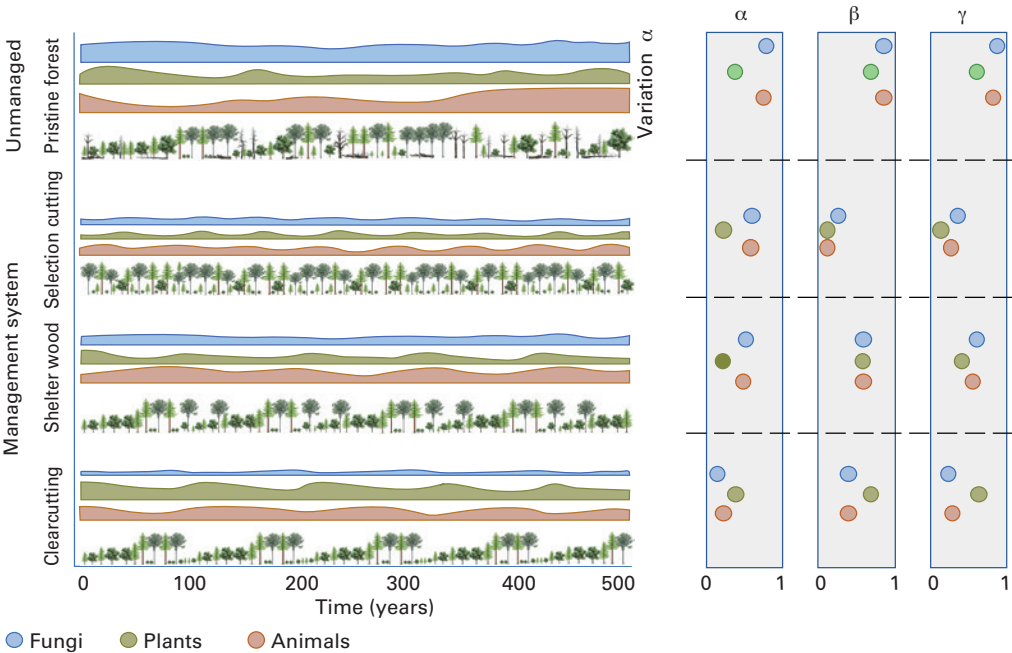


Fig. B8.4. Comparison of managed and natural forest systems over several hundred years and with respect to local average (α), species turnover among sites (β), and landscape-scale (γ) diversity for fungi, plant, and animal species. The schema refers to beech and mixed broadleaved-coniferous forests in Central Europe. Left: α -diversity across time in relation to changes in forest structure, Right: diversities in unmanaged forests and different management systems across different successional or management stages. Based on Scherzinger (1996), Kuuluvainen (2009); Hilmers *et al.* (2018), and Schall *et al.* (2018).



Fig. B8.5. Fruits of strawberry blite (*Blitum virgatum*) (a). The seeds can endure in soils for hundreds of years and germinate after soil perturbation or forest fire (Moser *et al.* 2006) (Photo: Barbara Moser). (b) So-called pyrophilous species such as the flat bug *Aradus lugubris* (4.5–6.6 mm), mostly black-coloured, can detect rapidly burned deadwood infested by a particular host fungus, i.e. *Daldinia loculata* for *A. lugubris* (Wikars 2001) after forest fires (Photo: Martin Gossner).

limitations seem to be apparent in some species (Sverdrup-Thygesen *et al.* 2017).

In fact, both resource amount and resource diversity in combination with different environmental conditions might be important for many species, and consequently for forest resilience. This leads back to the great significance of natural disturbances which generates a stochastically evolved mosaic of conditions promoting biodiversity in a dynamic way. We should make use of the presumed increase in disturbances with climate change such as bark beetle outbreaks, storms, forest fire, and drought effects (Seidl *et al.* 2017) that contributes to the geomorphologically defined heterogeneity by creating additional structures on which many species depend (Müller *et al.* 2008; Thorn *et al.* 2019). The predicted changes in disturbances will likely differ in size and frequency, and this will promote heterogeneity (fig. B8.3). In the frame of an adaptive forest management such areas could be taken out of management for a particular time (Bollmann and Braunisch 2013) to promote the first successional stages (fig. B8.5) which are in particular important for biodiversity (Wermelinger *et al.* 2017; Hilmers *et al.* 2018).

However, in this context two aspects need to be considered. First, the increased frequency and intensity of disturbances destroys structures and environments that are only facilitated by less intense and less frequent disturbance. Many species are already facing too much disturbance of

their habitat, and thus the application of novel disturbance regimes requires careful consideration of the implications (Felton *et al.* 2016). Second, a large number of species depend on later successional stages and large trees (Lindenmayer and Laurance 2016) as well as the long-term processes of growth and decay that allow for the development of the structures, resources, and habitats on which the species depend (fig. B8.4). Thus, we need strategies that guarantee the distribution of different successional stages at the landscape scale in a configuration that allows the coexistence of most species.

Do we need a squirrel? Do we need a lynx? The rarity template

Commonness and rarity vs. endangerment

The Red Lists of flora and fauna around the globe give a similar picture: near threatened, vulnerable, endangered, and regionally extinct species usually make up 50 % to 75 % of all species of a regional flora or fauna (e.g. <http://www.iucnredlist.org>). However, it has been suggested that rarity type should not be used as a surrogate for extinction risk (Reed *et al.* 2020). Rabinowitz (1981) defined seven types of rarity based on the geographic range (large vs. small), habitat specificity (wide vs. narrow), and local population size (large, dominant in a particular location vs. small, non-dominant) of a species. Given any region, a list of species strongly

reflects the availability of their habitats in this region, with fewer widespread species representing the widespread habitats and the wide habitat specificity of many of these species (habitat generalists) and many rare species using the rare habitats and exhibiting mostly narrow habitat specificity (habitat specialists) of that region. Red Lists report mainly the rarity in the landscape in combination with human-induced drivers. The abundance of many of the currently rare species in a region has declined because of the recent habitat loss caused by land-use change. However, there are also species that are rare because they persist in a rare habitat, e.g. rocky outcrops, raised bogs or rare deadwood structures (fig. B8.6). Conservation strategies should aim to reduce extinction risks for those threatened species for which a decline is due to definable human causes (human-caused rarity), but often include charismatic species that are rare but not threatened when using ecological criteria (natural rarity; Habel *et al.* 2020). If both land use and forest management have reduced or strongly homogenised natural habitats, and abundances of habitats are shifting, then species frequencies and abundances are also changing, resulting in, for example, longer Red Lists. Any conservation strategy will result in a shift of rarity and commonness. Managed and unmanaged regions differ in both the number and the type of disturbances, which, for some taxonomic groups such as vascular plants, corresponds to higher species diversity in regions

with managed land in contrast to biodiversity in untouched natural systems (Paillet *et al.* 2010; Boch *et al.* 2013). Other taxonomic groups, such as mammals requiring large untouched habitats, are more frequent in unmanaged land.

Iconic species

There is no way to establish a complete inventory of all organisms of a region; however, we do know a lot about locally occurring iconic species. In forests, woodpeckers, squirrels (fig. B8.7), deer, and wild boars are iconic animals. A sighting of such a species brings joy and excitement for the viewers, and so, the lack of such a species in a habitat, feels like a loss; more so than for losses of other less conspicuous species. The extinction of iconic species (or rather big species), has started thousands of years ago and can be viewed as the start of the sixth mass extinction. Obviously, nature adapts to the loss of such species, and so do humans. We know that we can live without certain species. For example: woolly mammoths (*Mammuthus primigenius*; once widespread across the tundra in Europe, Asia and North America, but now extinct); brown bears (*Ursos arctos*; once widespread across the whole of Europe but, in Europe, now restricted to a few fragmented populations); and northern bald ibis (*Geronticus eremita*; once widespread across Southern and Central Europe but which disappeared from these areas completely over 300 years ago). People adapt to nature or, more generally, to



Fig. B8.6. Lemon-coloured *Antrodiella* (*Antrodiella citrinella*) (a) is a rare fungi inhabiting deadwood (Bässler and Müller 2010). It is considered an indicator of pristine forests and requires deadwood amounts of $> 140 \text{ m}^3/\text{ha}$ (Photo: Josef Hlasek). Other highly demanding species also only occur in large protected areas where suitable habitats are present in the long term. For instance, the saproxylic click beetle *Danosoma fasciata* (b; Photo: Simon Thorn) has survived in the Bavarian forest and its distribution is now expanding after the designation of the National Park in 1970. The saproxylic *Peltis grossa* (c; Photo: Jörg Müller) was considered extinct in the Bavarian forest for over a hundred years and was recently rediscovered.



Fig. B8.7. The red squirrel (*Sciurus vulgaris*) is widespread in forests from Europe to Siberia (Photo: Andreas Rigling). In continental Europe, red squirrels are not threatened and forests do not depend on this species; however, people enjoy meeting this iconic species in forests, parks, and gardens (black variant on this photo).

change and not all species on earth are necessary to maintain the functioning of our ecosystems. However, anyone who has ever been in a pristine forest or, more likely, walked through an uncleared windthrow patch or an untreated burned forest, will experience the excitement of natural dynamics, will observe natural rarity and commonness, will find a source of interest and life that makes life worth living. This excitement does not only arise from seeing iconic species but also from experiencing the ecosystem consisting of so many species that coexist and interact (Thompson 2010). Nature and species diversity have an often underestimated positive effect on human health (Irvine *et al.* 2019).

Do we need squirrels everywhere? Do we need forest management everywhere?

Our society is characterised by its need to control everything. In Central Europe, we know the habitat requirements for many species and consequently can design, at least theoretically, how much biodiversity we want in any particular place. Therefore, in theory we could design our forests according to our desires; for instance, we could have more red squirrels, more capercaillie, or more light-demand-

ing, or more deadwood-related species. Across Europe, there are thousands of people working to transform disturbed ecosystems into species rich ecosystems; the fields of restoration ecology and ecological restoration are flourishing. Eventually, society (or governments) defines, how much forest and what type of forest management we want, and how much of the forest should remain as untouched ecosystems by taking forest areas out of management and restricting access. (fig. B87). Protection of species and biodiversity is now one of the important considerations that informs these policies and choices. Protecting biodiversity means investing in species, and in increasing or maintaining biodiversity in disturbed habitats ('cultural landscapes') such as open land that would otherwise be covered by forests. Red squirrels are not threatened, but are a highly attractive and typical element of forests. They feed on the seeds of coniferous and broadleaved forest trees, and thus occur in many forest types across Europe and northern Asia. The species is also found in parks and gardens. While squirrels depend on seed trees, forests do not necessarily depend on squirrels. In most forests on the Mediterranean islands, squirrels have been

absent since evolutionary times, and the niche has – at least on some islands – been occupied by crossbills (*Loxia* spp.) (Benkman and Parchman 2011).

In contrast to designing our environment, rescuing natural processes by non-intervention not only is less spectacular, but it is also less labour-intensive, although difficult to achieve in densely populated landscapes. There are, however, examples of initiatives of forests being released from management to let natural processes rule. Thanks to an initiative of a former city-forester of Zürich, management in the highly productive Sihl forest (German: Sihlwald) was stopped in 2000 (Kasper 2012). For 20 years, natural disturbances have been the only dynamics in the forest that now serves as a wilderness area open to the public.

Conclusions

Natural disturbance regimes in pristine or in old-growth forests are important study objects to evaluate natural processes in space and time; they serve as evidence of, and a blueprint for, optimum habitat requirements regarding biodiversity. Also, research in managed forests serve to demonstrate effects of increased frequency of natural disturbances as a consequence of climate change and disturbances by human interventions on biodiversity. Such information on a multitude of different disturbance regimes at different temporal and spatial scales can help to inform and optimise management strategies by integrating naturally occurring disturbances and targeted interventions into adaptive management systems. On the one hand, this allows us to promote species diversity in forests by providing sustainable habitat availability at the landscape scale. On the other hand, common efforts across municipality, state, and even national borders are needed to achieve sustainable biodiversity aims. This will be challenging owing to the complex organisation of forestry across Europe, but is needed to promote a high diversity given the high environmental heterogeneity and related species turnover (β -diversity). Importantly, additional socio-cultural aspects need to be considered, for instance the beauty of iconic species such as the red squirrels as part of cultural ecosystem services. We advocate the need for large forest landscapes that eventually should be – against all odds – released from management in the future. Such large unman-

aged landscapes not only will result in more dense populations of highly demanding species that also can spread out to managed landscapes, but will also serve as long-term references of natural forest processes and their effects for biodiversity. Ecosystem functioning in the best interest of the societies in Europe can only be guaranteed, in the long term, by a combination of nature (i.e. large strict reserves) including natural processes and the promotion of biodiversity in managed land.

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