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ORIGINAL PAPER

Rapid biodiversity assessment of arthropods for monitoring average local species richness and related ecosystem services

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Abstract Rapid biodiversity assessment (RBA) is proposed as an affordable indicator for monitoring local species richness of arthropods and sustainability of related ecosystem services. The indicator is based on strictly standardised sampling procedures and the identification of parataxonomic units (morphospecies) instead of species identification. The collection of arthropods was optimized with regard to trap types, time and length of collecting period, selection of four out of seven weekly samples, and choice of counted taxa and trophic guilds. By measuring arthropod activity, RBA is an indicator for functional diversity. Over a period of 8 years, average yearly numbers of morphospecies were assessed in Switzerland in 15 agricultural habitats, 15 managed forests, and in 12 unmanaged habitats ranging from protected lowland wetlands to Alpine meadows. The yearly RBA-trend in unmanaged habitats is used for assessing the influence of climate and weather on biodiversity, and as a reference for measuring the relative influences of recent management changes in agriculture and forestry. The average number of morphospecies per sampling station per year depends on temperature, and was only marginally significantly increasing over time in agriculture, but not in forestry or unmanaged areas. Three RBA indices considered to be relevant for maintaining ecosystem services were calculated from the average number of morphospecies per location per year: (1) indicator for ecological resilience and sustainability (all morphospecies); (2) indicator for pollinator diversity (taxa with a majority of pollinators) and (3) indicator for biocontrol diversity (ratio between carnivore and herbivore guilds).

Keywords Arthropods · Biodiversity indicator · Ecosystem services · Insects · Morphospecies · Resilience

Introduction

Average local species richness (alpha-diversity) has no emotional appeal for nature protection and is not consistently considered as a valuable and pertinent aspect or entity of

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biodiversity. However, ecological resilience (Peterson et al. 1998) and sustainability of ecosystem services (Hooper et al. 2000, 2005; Kremen 2005; Loreau 2000), such as pollination (Kremen et al. 2007), pest control (Cardinale et al. 2003; Moonen and Barberi 2008) and preventing invasions, may depend very much on local species richness as well as genetic variability within species. This dependency may become aggravated in the prospect of major global environmental changes such as global warming and management changes in agriculture and forestry (Allison 2004; Kassar and Lasserre 2004; Loreau 2000; Loreau et al. 2003; Petchey et al. 1999).

There are numerous publications on the requirements of a good biodiversity indicator (e.g. European Academies Science Advisory Council 2005). McGeoch (1998) gives an excellent overview and lists 32 criteria advocated by the authors of 11 publications on biodiversity indicators. The main requirements (cost efficient, effective, sampled and sorted easily, correlating with trophic levels and functional groups, show a well defined and measurable distribution over a range of habitats, representative of related and unrelated taxa, etc.) cannot all be combined in reality, because there is always the compromise between the inherent complexity of biodiversity and the simplicity of what in fact is affordably measurable (Schmeller 2008).

In international, national, or even in regional monitoring programmes there is always a tough compromise between spatial representativity and organismal representativity: Existing programmes are either restricted to a few well-known taxa such as birds or butterflies, or to a small number of sampling stations (Henry et al. 2008; McGeoch 1998). The EU-indicators for the Countdown 2010 of the Convention on Biological Diversity (CBD, European Environment Agency 2007) focus on the extent of biomes, ecosystems and habitats, coverage of protected areas, and on trends in abundance and distribution of selected species. These indicators, usually based on species poor taxa, hardly represent organismal biodiversity, and even less so functional biodiversity. For most ecosystem services, loss of local species richness is more significant and pertinent than the national or regional loss of flagship or red list species.

While we might be able to decide by the year 2010 whether we were able to halt the loss of species richness in birds and vascular plants, there are no standardised data available for the monitoring of average species richness, ecological resilience, or ecosystem services such as pollination and the potential for biological control of pest organisms.

Arthropods make up for the largest proportion of species richness at any spatial scale (Hammond 1992) so they are more representative for wholesale organismal biodiversity than any other group of organisms. Arthropods, especially insects, play a major role in ecosystem services. One important reason for not using arthropods in large monitoring projects is the fact that specialists for the identification of many taxa are scarce, for some taxa even non-existing (Noss 1996; Whitehead 1990) and that the effort and costs for arthropod species identification are much higher than for plants or birds. On the other hand, the use of a specious group such as the arthropods is tempting for biodiversity assessment, because no other group represents so much of overall species richness. A first attempt to use ‘morphospecies’ instead of species was made in Australia, where the number of undescribed species in samples can still be considerable (Cranston and Hillman 1992; Oliver and Beattie 1993, 1996). For assessing species diversity in selected arthropod groups, Bolger et al. (2000) and Kerr et al. (2000) have used ‘parataxonomic units’. Both terms describe a group of biological organisms that differs in some morphological respect from all other groups (Allaby 1999).

The purpose of this study was to find a monitoring scheme and a biodiversity indicator for the average local species richness of arthropods, and to test and operationalize it in a mid-scale (time and space) field experiment. We propose a rapid biodiversity assessment (RBA) monitoring programme that is standardised with respect to reliable trapping devices, the optimum sampling period, and the most cost efficient way to obtain testable estimates for overall local species richness (alpha-diversity). Specifically we reduced expenses for the identification of species by applying an adapted morphospecies approach.

Three management regimes dominate Swiss landscapes and were thus selected to be sampled in the process of developing the RBA programme into applicable form: Agricultural landscapes, managed forests, and unmanaged habitats (mostly protected areas) such as unmanaged forests, wetlands, or alpine meadows. The basic idea is that in a time series unmanaged habitats reflect the influence of climate change and natural succession on biodiversity, while the influences of management change in agriculture or forestry become visible in comparison with the calibration curve in unmanaged habitats.

With all the caveats going along with a reduction in time and costs, special care had to be given to the aspects of variability and interpretation capacity (valuation). In the conclusions we will discuss the potential and the limitations of the proposed indicator set.

Materials and methods

The development of methodologies used followed the course of experiences in a multitude of projects, employing progressively more elaborate methods for the assessment of arthropod diversity. Optimization involved the evolution of standardized trap designs, the selection and narrowing down of a seasonal sampling time window, and the design of an economic scheme for the identification of species richness, all of which will be detailed in the following (Table 1).

Trapping devices for aerial and epigeal arthropods

Several standardized, passive methods were established to sample arthropods, like e.g. window traps or coloured traps for aerial organisms, and pitfall traps for epigeal arthropods (Mühlenberg 1989; Southwood 1978). The experiences with different collecting methods used in three previous projects (Duelli and Obrist 1998; Duelli et al. 1990, 1999, 2002; Flückiger et al. 2003), made us choose one pitfall trap and one window interception trap in combination with a yellow pan trap as a standard sampling unit. The latter two were combined into a so called combi-trap, placed at a height of 1.5 m above ground (Duelli et al. 1999) to sample flying and flower visiting species. Surface dwelling species (e.g. spiders, carabids, other epigaeic beetles, myriapods, isopods) were sampled with pitfall traps. A pitfall trap consisted of a plastic funnel recessed into the soil (opening diameter of 15 cm) and mounted on top of a plastic bottle containing 2% formaldehyde solution. A roof 10 cm above the traps provided protection against rain. For details and limits of the method, see Duelli et al. (1999) and Obrist and Duelli (1996).

Each trap station consisted of one pitfall trap and one combi-trap (Fig. 1). The probability of an animal being caught in these traps is a function of the trap diameter, the animals activity and the species' abundance. The risk of using only one trap of each type per location was balanced by the choice of sampling period and processing protocol (see below).

Table 1 Methodological framework of the RBA method evaluation followed throughout the text

Topic	Focus	Method	Motivation
Evaluation of trapping devices	Aerial arthropods	Comparison of catching success of different trap models; fusion of window trap and yellow pan	Sampling of flying arthropods; combination trap for passive flight-interception and attractive for flower visiting species; servicing economics
	Epigeal arthropods	Comparison of catching success of different trap models; optimised funnel for all species	Avoidance of species specificity; servicing economics
Time evaluation	Seasonal position	Database analysis of previous projects sampled all year	Find period with highest percentage of full year's species pool
	Duration	Database extractions of temporal subsets of previous projects sampled all year	Minimize time window to maximise ratio of species per individuals; minimize sorting effort
Evaluation of identification method	Group selection	Sorting catch in obvious groups of taxonomic differences	Achieve an analog of functional groups; facilitate differentiation in morphospecies
	Species differentiation	Morphospecies approach; correlation against species identification; effect of processing person	Reduce costs for identification; quantify quality of measures; quantify observer effects
Implementing the method	Three habitat types	Sampling according to emerging optimized method	Establishing evaluated method
	Eight years	Sampling standardized at same location over time	Time series analysis to evaluate method for biodiversity monitoring

Evaluation of optimal collecting time: when?

To define the best period for representative sampling all over Switzerland, we consulted our database containing the results of all former projects, in which the period of standardized weekly sampling extended over much of the vegetation season, and where a majority of arthropods had been identified to the species level. Based on the data of six earlier projects in agricultural landscapes (Duelli 1997; Duelli and Obrist 1998; Duelli et al. 1999), windthrow forests (Duelli et al. 2002; Wermelinger et al. 2002), and forest edges (Flückiger and Duelli 1997; Flückiger et al. 2003) the best period for assessing arthropod species richness was calculated. With a moving window approach we identified for each of the six projects a period of 7 weeks, which contained 7 weeks (see below) yielding the highest proportion of species of the total seasons catch.

Evaluation of optimum collecting period: how long?

To reduce identification costs, the samples were minimized without loosing too many species present: only the material of the most productive (abundance) of the seven weekly catches were used. This subselection also allowed to introduce a redundancy with the remaining three weekly samples, to optionally compensate for unexpected losses during the seven sampling weeks. As bad weather with snowfall can reduce arthropod activity for several weeks e.g. in the Alps, we prolonged the overall collecting period after a pilot project from 5 to 7 weeks and the subset from 3 to 4 weeks to cover enough of the seasons

Fig. 1 Trap station at Celerina, Grisons (1,730 m a.s.l.), consisting of a flight trap (combination of plexiglass interception trap and a yellow water pan) and a covered pitfall trap in the ground

spectrum of species. We modelled the outcome of this procedure again with data from earlier projects.

In two of the above mentioned former projects, arthropods had been sampled in transects covering a total of 53 trap stations through various types of habitats (Duelli and Obrist 1998; Flückiger et al. 2003). Here too, the weekly collected material of one full year had been identified to the species level for a majority of the taxa. Based on these weekly samples at the 53 trap stations, the performance in terms of species numbers of an optimized selection of 4 weeks (out of seven of sampling) in relation to the full year's yield was assessed (see "Results", Fig. 4). Technically it is a correlation, but we wanted to know the representativity of the 4-week sample for the trap stations total yearly collected biodiversity. Accordingly, we do not correlate the 4 weeks with the rest of the weeks, but with all the weeks.

Selection of the four "best" weeks

The focussing on four out of 7 weeks, to reduce identification costs, and the identification of the arthropods proceeded in several steps as given below.

First, the weekly catches were cleaned from worms, snails, plants and debris. Occasional vertebrates were kept in a separate vial per location and year.

Then, for any given trapping location the seven couples of vials (from the two trap types) of the weekly catches were arranged in a double row, starting with the first catch in June and ending with the seventh (last) catch in August.

The first selections consisted of the first vial (June) and the last vial (August) of both trap types, in order to cover optimally the seasonal spectrum of species composition. If, however, one of those catches were obviously lessened by factors such as trap damage, bad weather, lack of water in the funnel or flooded pitfall trap, or if for unknown reasons the volume of the catch is less than two-thirds of that in the neighbouring weeks, the latter were chosen instead.

Of the remaining five weekly samples, the two with most material (volume) were chosen, independently for the two trap types. It was assumed that more individuals in 1 week also meant more species. In the rare exceptions to that, e.g. in cases of species outbreaks or swarming events, the next fullest vial was chosen. Only the material from the selected 4 weeks was processed any further, but the remaining 3 weeks were kept as reserves. From the material of the four selected weeks, some of the taxa were identified to species level by voluntary specialists with faunistic aims.

The morphospecies approach

Following the first attempt to use morphospecies instead of species made in Australia (Cranston and Hillman 1992; Oliver and Beattie 1993, 1996), we adapted the identification of parataxonomic units (Bolger et al. 2000; Kerr et al. 2000).

The specimens in the eight selected vials per location, trap type and year were processed consecutively, one vial after the other. This facilitated the sorting of the voluminous catches into orders and families. Pooling the catches at the beginning made sorting much more tedious. For any given sampling location, the four weekly catches were sorted into 14 vials of distinct taxonomic groups (see Table 2). Diptera and Collembola were completely omitted from the count, as their sorting into morphospecies by non-specialists proved too costly and unreliable in a pilot project.

Table 2 Assignment of separately counted taxonomic groups to the three functional guilds carnivores, herbivores and pollinators

	Carnivores	Herbivores	Pollinators
Lepidoptera		X	X
Carabidae	X		
Cerambycidae		X	X
Buprestidae		X	X
Other Coleoptera			
Aculeata			X
Other Hymenoptera	X		
Heteroptera			
Homoptera		X	
Arachnoidea	X		
Thysanoptera		X	
Neuroptera	X		
Psocoptera		X	
Other groups			

Species can be both herbivores (e.g. as larvae) and pollinators (as adults) and thus contribute to two ecosystem services

Each taxonomic group was counted for morphospecies separately. Two or more specimens belonged to the same morphospecies, if an entomologically trained person (but non-specialist for these groups) could not see any external morphological differences. With such a definition many species with sexual dimorphism went as two morphospecies, whereas sibling or even more so cryptic species were lumped into one morphospecies.

The 14 counts of morphospecies per taxonomic group were protocolled and added up to yield the "RBA-index" of a specific location and year. Additionally, selected groups (Table 2) were used for assessing rough estimates of the potential for maintaining ecosystem services such as pollination or biological control of pest organisms.

The 14 taxonomically separated vials per trap location and year were filled with 70% alcohol, labelled, sealed, and stored in a dark and cool place.

Reliability and consistency of morphospecies counts

A main concern with the morphospecies approach is the capability and consistency of non-specialists in sorting and counting morphospecies (Krell 2004). All our material of the implementation phase (see below) was to be processed by two biologists (one botanist, one vertebrate zoologist) with basic entomological training. To assess their accuracy, their temporal identification consistency and the reproducibility of the morphospecies approach, we presented eight samples to them, randomly selected from different years and locations, to re-process in a double blind test: they were not allowed to communicate on their results, and the samples were anonymised, so even the person distributing the samples and gathering the results did not know the origin of the samples. At the same time, the performances of two additional people with less experience in sorting and counting arthropods were compared to those of our two experts. Mean, standard deviation and coefficient of variation (CV) were calculated for the morphospecies counts of each of the eight samples.

Representativeness of the RBA-index for total species numbers

The RBA-catches of the years 2000 and 2005 were used to assess the performance of the morphospecies approach. After being sorted and counted as morphospecies, the same material was identified by specialists at the species level. Not for all taxa was it possible to find specialists willing to identify material soaked first in water or formaldehyde solution in the traps, then stored in 70% alcohol. Species–morphospecies relations were compared with a regression analysis through the origin (0:0), as zero species must result in zero morphospecies.

Implementation of the RBA method for a Swiss monitoring scheme: trapping locations and time

To operationalize and test our RBA approach, as defined above, we set up a sampling design for a monitoring scheme spanning 8 years (2000–2007) and covering most parts of Switzerland. With respect to the authorities providing financial support to the project, the trap stations were not placed randomly throughout all of Switzerland, hence the results are not representative for the country (Fig. 2). However, the 15 trap stations in managed forest are predominantly linked to the national LWF-programme (Kräuchi 2007), which intends to cover the main forest types of Switzerland. The 15 agricultural trap stations were placed in accord with a project analysing the impact of ecological compensation measures in

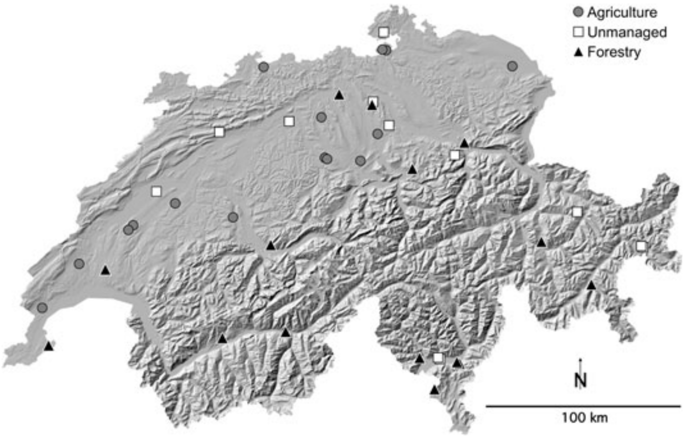

Fig. 2 Distribution of the 42 trap stations in Switzerland: 15 in agricultural habitats, 15 in managed forests, 12 in unmanaged habitats. GIS layers were obtained from the Swiss Federal Office of Topography (dhm25 © 2010 swisstopo, 5704 000 000)

Swiss agroecosystems, geographically limited to the Swiss Plateau (Aviron et al. 2009). Finally, 12 locations for traps in unmanaged habitats ranged from lowland insubric forest south of the Alps (200 m a.s.l.) to protected wetland sites and montane forests, up to alpine meadows at 2,500 m a.s.l.

In the RBA programme we present here we collected during 7 weeks per year, always starting on calendar week 24, mid-June (the summer peak for all evaluated model locations, see "Results"). However, at the highest alpine locations, accessibility limited by snow sometimes lead to a slightly deferred collecting period. All traps were emptied weekly and processed according to the procedures described above.

Possible applications: ecosystem services

The morphospecies approach, by ignoring the names of single species, relies completely on the numbers of separated morphs. While this approach is not of great interest for species conservation, it offers a number of indicators of ecological relevance. The first is the number of species as such. The concepts of ecological resilience, of the "balance of nature", and thus of sustainability are based on the "insurance hypothesis" (Naeem and Li 1997; Yachi and Loreau 1999), where a higher number of species has a higher potential for filling new or empty niches after an environmental impact or in times of change. We will derive more indicators for maintaining the potential for ecosystem services (pollination, biological control) from the RBA process by assembling selected taxa from the list of the 14 sorted groups shown in Table 2.

Statistical analyses

ANOVA, regression and correlation analyses were performed with Data Desk 6.2.1 (Data Description, Inc., Ithaca, NY, USA) or R 2.8 (R Development Core Team 2008). Coefficients of variation were compared with a variance ratio test and slopes of regression with a modified *t*-test (Zar 1984).

Results

In the following, we present the outcome of the evaluation of the methods and subsequently will introduce first results and possible applications of the implementation of the RBA-method in a monitoring scheme.

The best collecting period

The purpose of this analysis was to identify a seasonal time period, where trapping promised to produce the highest species numbers. We plotted the percentage of species of the total season's catch contained in four out of seven catching weeks (see "Materials and methods") versus the starting point of each 7 week period (Fig. 3). The curves all levelled off between calendar week 22–26 and peaked around week 24, in average yielding slightly more than 40% of the annual catch in species numbers. In some of the projects in the lowlands, in week 14 (Fig. 3) a spring peak for species richness appeared, which could even surmount the summer hump (Duelli et al. 1999). For those habitats earlier collecting or two separated collecting periods could be superior to the 7 weeks sampling in a row. However, in early April (week 14) weather conditions can still be very harsh and alpine regions are mostly inaccessible. Thus, for practical, and environmental as well as economic reasons we focused on the summer peak, and started collecting in calendar week 24, which also coincides with stable and highest average temperatures.

Testing the representativity of the selected 4 weeks for a whole year's catch

With the weekly catch data from two former projects mentioned above (Duelli and Obrist 1998; Flückiger et al. 2003), together comprising 53 trap stations in various habitats (see "Materials and methods"), we related the number of species that would have been caught

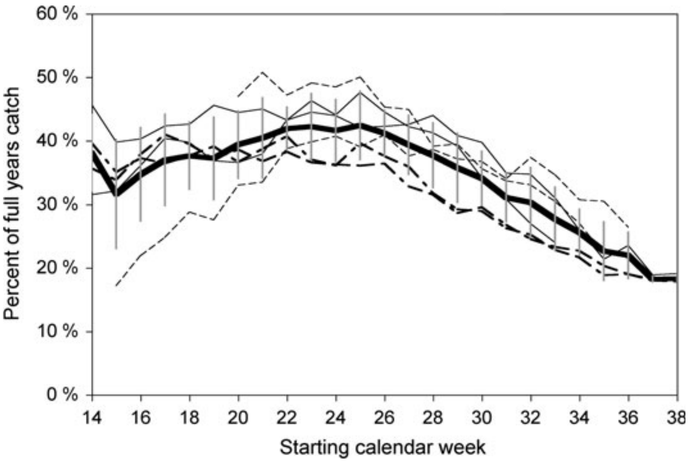

Fig. 3 Proportion of species numbers collected in 4 weeks, selected from a 7 weeks collecting period according to the RBA procedure, in comparison to the species numbers over the whole vegetation period (mid-April to mid-September). Figures are calculated from data of six earlier projects (for refs. see text). The *fat solid line* shows the average percentage of species richness (±SD), with a flat peak between calendar week 22 and 26. *Thin solid lines*: agriculture; *evenly dashed lines*: windthrow forest; *irregularly dashed lines*: forest edge transects

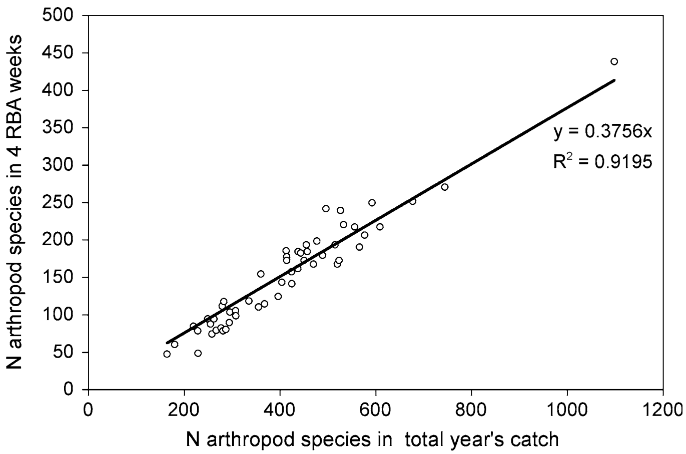

Fig. 4 Representativity (regression) of the catches of 4 weeks, selected according to RBA procedure, for the catches of the entire year. Figures calculated from identified material of 53 standardised trap stations from two earlier projects, one in agricultural habitats (Duelli and Obrist 1998), one at five forest edges (Flückiger and Duelli 1997; Flückiger et al. 2003)

with the RBA standardized optimum collecting period to the total year's catch in a site-wise manner.

The two values nicely corresponded ($R^2 = 0.92$) and the inclination of the regression line indicated that a proportion of about 38% of the annual catch was sampled with the RBA scheme (Fig. 4). This value underlined the yield of 40% found in the temporal analysis above.

Reliability and consistency of morphospecies counts

To assess the quality of the morphospecies assignment, four people each sorted and counted the same eight samples in turn in double blind manner. The eight samples from different habitat types all over Switzerland varied in richness of morphospecies between 118 and 478 (Table 3). The average morphospecies counts per person are listed in the right

Table 3 Performance of four non professional entomologists in a double blind test, sorting and counting the same eight RBA samples in turn

Person numbers	Sample numbers								MN
	1	2	3	4	5	6	7	8	
1	385	425	180	339	456	267	195	119	295.8
2	384	435	184	356	453	288	197	121	302.3
3	439	453	195	348	488	308	165	114	313.8
4	446	459	204	376	513	340	169	116	327.9
MN	413.5	443.0	190.8	354.8	477.5	300.8	181.5	117.5	309.9
SD	29.1	13.6	9.4	13.7	24.7	26.9	14.6	2.7	16.8
CV	7.0%	3.1%	4.9%	3.9%	5.2%	8.9%	8.0%	2.3%	5.4%

Mean (MN), standard deviation (SD) and coefficient of variation (CV) of morphospecies counts are given for each sample. The list of persons is sorted with increasing splitter tendency column, with increasing numbers of morphospecies. Person 4 with the highest counts was the only non-biologist. An ANOVA controlling for site differences identified a significant effect of person on the number of identified morphospecies ($F_{3,29} = 5.061$, $P = 0.006$). However, a Scheffe post hoc test showed, that the effect was due to the single difference between person 4 and 1 ($P = 0.013$). The average CV of all samples is 5.4%.

The eight samples were also re-evaluated by the same person, that had treated the samples initially after collection. An ANOVA controlling for site differences, showed a significant difference between re-identifications ($F_{2,14} = 13.789$, $P < 0{,}001$), which was due to the re-identification in 2007 of samples from 2003 ($P < 0.001$). Samples from 2006 were re-identified with no significant difference in 2007 ($P = 0.422$). In average, their original and repeat counts varied with a CV of 5.7%, which is not significantly different from the overall CV of 5.4% (Variance ratio test: $F_{8,8} = 1.083$, $P = 0.919$). Thus, repeatability of RBA counts by the same person and reproducibility by alternate persons had the same accuracy.

Correlation of morphospecies versus real species numbers

The predictive value of the RBA-index for real species richness is of great interest. The taxonomic groups, for which a comparison between morphospecies and true species numbers was possible, are shown in Table 4. For the two years (2000, 2005) where we had the samples identified to species level by taxonomists, we related total morphospecies numbers (RBA-index) to true species numbers for all 42 trap stations (Fig. 5).

At higher species richness, the morphospecies approach slightly underestimated the true species numbers. With higher species numbers chances likely increased that two species looked very similar and thus were lumped into one morphospecies. The effect was more pronounced in the samples from the year 2000 than in those of 2005. The slopes of the two regression lines differed significantly ($t = 2.375$, df = 80, $P = 0.019$), indicating 6% more morphospecies identified per species in the year 2005 compared to the year 2000, which might represent a learning effect.

Of 636 samples (vials per taxonomic group, site, and year) which were identified to species, over 90% showed very restricted ratios of morphospecies to species between 1:1.4

Table 4 Regression analyses (without intercept) between morphospecies counts and identified species richness in taxonomic groups, for which experts for species identification were available

Taxonomic group	Coeff.	R^2	*N* sp.	*N* morph.
All groups	0.960	0.986	142.4	144.2
Other Coleoptera	0.945	0.976	58.1	56.1
Aculeata	0.890	0.972	21.7	20.5
Cerambycidae + Buprestidae	0.998	0.963	3.5	3.6
Heteroptera	0.782	0.955	11.5	9.8
Araneae	0.990	0.941	20.2	20.5
Carabidae	0.968	0.939	10.1	10.3
Homoptera	1.042	0.907	13.6	13.9
Neuroptera	0.675	0.739	1.5	1.3
Orthoptera	3.235	0.674	1.7	7.3

Lines are sorted by decreasing R^2. Average number of species (*N* sp.) and morphospecies (*N* morph.) are indicated

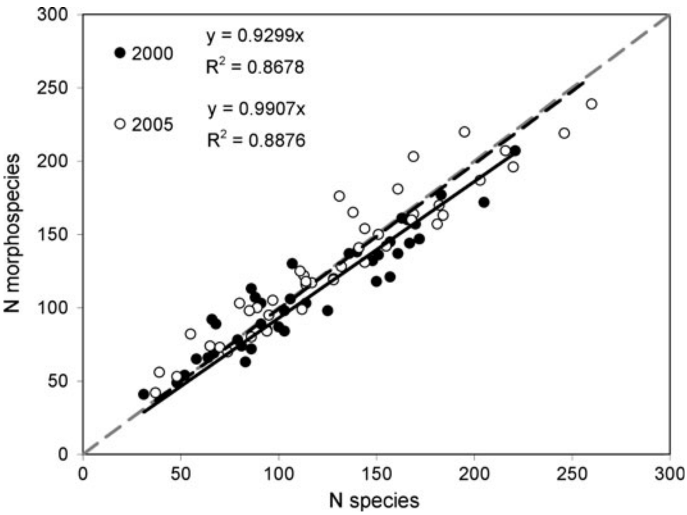

Fig. 5 Regression (without intercept) between morphospecies numbers and identified species for a majority of the taxonomic groups (see Table 4) counted for RBA in the years 2000 (*solid dot*, *black line*) and 2005 (*circle*, *stippled line*). The *grey stippled line* indicates a 1:1 ratio

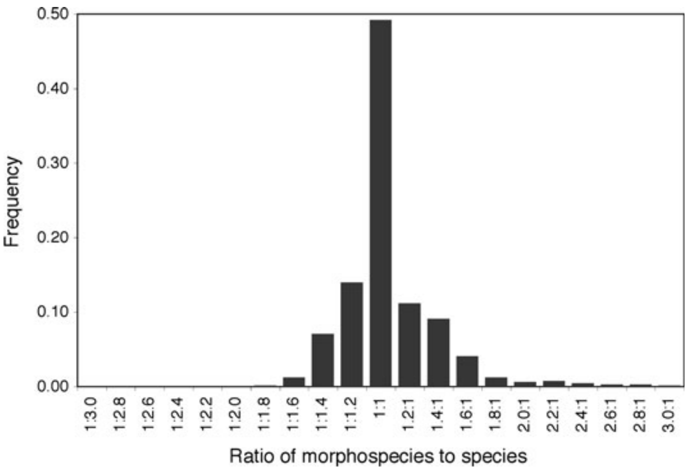

Fig. 6 Splitting (ratio >1) and lumping (ratio <1) of species in separating morphospecies. Groups included are: Araneae, Aculeata, Carabidae, Cerambycidae and Buprestidae, other Coleoptera, Heteroptera, Homoptera, Neuroptera, Psocoptera and Thysanoptera

(lumping) and 1.4:1 (splitting). Splitting and lumping average out nicely producing in roughly 50% of all cases a 1:1 relation (Fig. 6).

Monitoring implementation: trends at 42 stations between 2000 and 2007

We consider the sequence of the yearly average morphospecies numbers for all 42 trap-stations (rhombus in Fig. 7) to be a fairly accurate indicator for the development of local species richness (alpha-diversity) of arthropods in Switzerland.

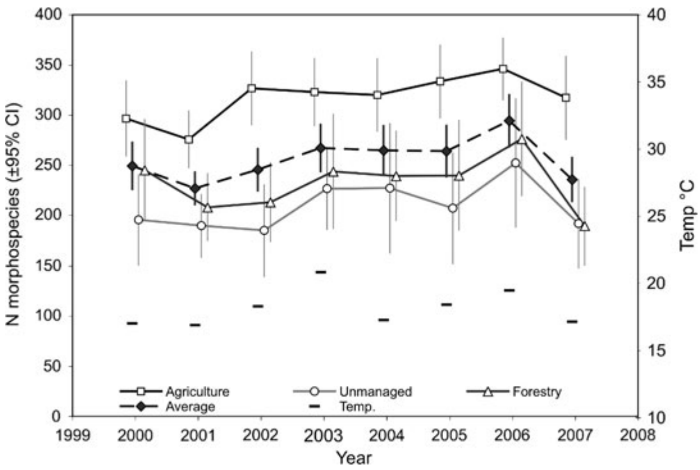

Fig. 7 Trends (±95% CI) of average morphospecies numbers per trap station per year (*dashed fat line*) as well as in agriculture, forestry and unmanaged habitats (wilderness). *Horizontal bars* below the *trend lines* indicate average daily temperatures in June and July at 24 official recording stations

Morphospecies numbers as a whole did not significantly differ between years. However, the numbers in agricultural habitat show a marginally significant increase between 2000 and 2007 (Regression: $t = 2.29$, $P = 0.062$) but no significant trends could be found for the managed forests, or for the unmanaged (wilderness) habitats (Fig. 7).

Morphospecies numbers depended on temperature, and thus on altitude. The average species richness in the tested agricultural areas (lowlands) was higher than that of alpine wilderness areas. Moreover, closed forests harbour insects both on the ground and in the canopy. Our standardized traps were on the ground, so we only collected part of the forest fauna. It is therefore not pertinent to compare the morphospecies numbers of forests directly with those of agriculture. But we could compare the changes between years or trends over time.

There were two diversity peaks in 2003 and 2006, when summer temperatures were higher than in the other years. The average number of morphospecies per year slightly depended on the deviation from the mean temperatures in June and July in the same year ($R^2 = 0.44$, $P = 0.068$), indicating an increase of 10 morphospecies per degree temperature rise.

Popular or ecologically important groups could be treated separately, if there was a good correlation between the number of morphospecies and real species numbers. We show examples of the trends for Carabidae (Fig. 8, top), bees, wasps and ants (Fig. 8, middle), and Lepidoptera (Fig. 8, bottom). Carabid beetles were most species rich in agriculture, where they showed no peak in 2003, probably because it was too dry for them. This groups showed a marginally significant increase in species numbers over the 8 years (Regression: $t = 2.22$, $P = 0.069$) while Araneae ($t = 3.83$, $P = 0.009$) and Homoptera ($t = 3.61$, $P = 0.011$) followed this trend more pronouncedly (not shown on graphs). Together they contributed to the marginally significant increase in total RBA numbers in this habitat type (Fig. 7).

The Lepidoptera reached their highest species numbers in managed forests, where they appear to have profited from the two warm summers. The Aculeata, on the other hand, reacted differently in the two warm summers. None of the other groups showed a visible upward or downward trend in any habitat over the 8 years.

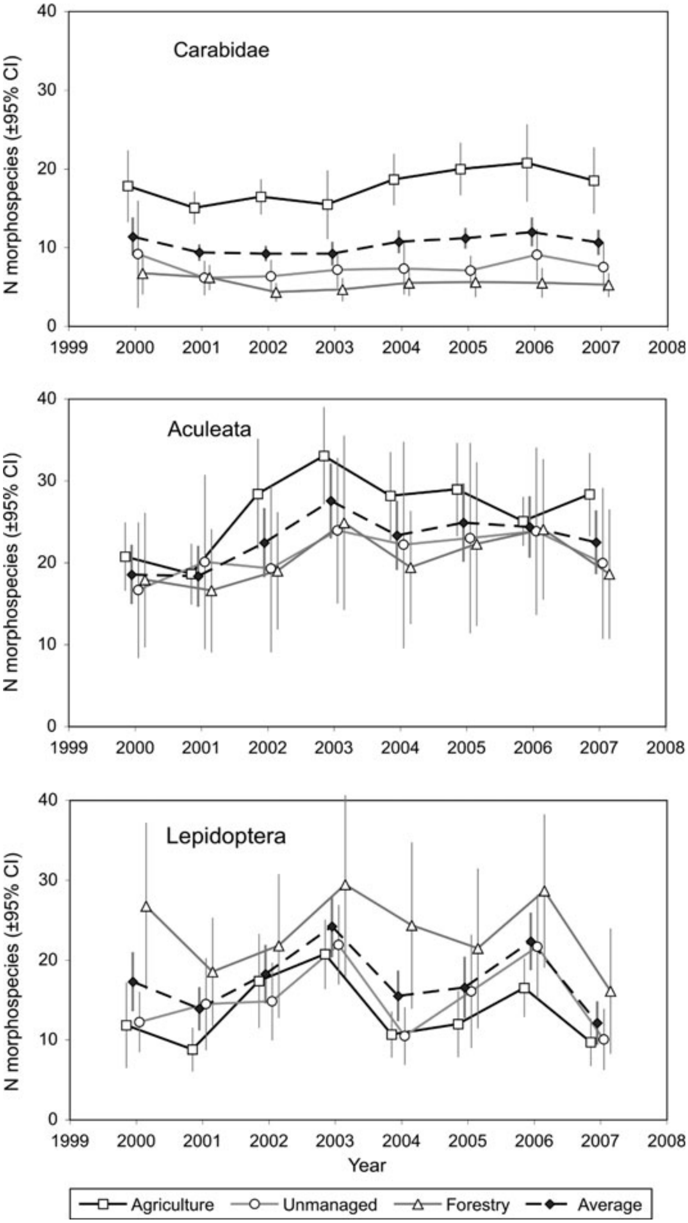

Fig. 8 RBA trends (±95% CI) of three groups of arthropods (carabid beetles, aculeate Hymenoptera, Lepidoptera) in the three investigated habitat or management types

Ecosystem services

The morphospecies approach results in numbers of morphs which is not suited for species conservation. However, the RBA process allows to derive indicators of ecological relevance besides the raw number of species, i.e. the alpha diversity (dashed line in Fig. 7). We assembled selected taxa from the list of the 14 sorted groups into three groups: carnivores, herbivores and pollinators (Table 2).

Pollination

The morphospecies numbers of four groups were added up for a rough estimate of the species richness of pollinators (Table 2). It is a crude and relative measure, because many other groups of insects also contain pollinators, and not all species in the chosen four groups are pollinators. This "RBA index for pollinator diversity" does of course not indicate the quantitative performance of pollination, which is anyway dominated by honeybees, but it indicates the development of pollinator diversity over time (Fig. 9). During the implementation of the RBA monitoring, the pollinator diversity clearly increased in all habitat or management types in the two exceptionally warm years of 2003 and 2006.

Biological control of pest organisms

We calculated a "RBA index for biocontrol diversity" by dividing the morphospecies number of carnivorous arthropods by the number of herbivore morphospecies. The taxonomic groups used to form the guilds of herbivores and carnivores are shown in Table 2. Again, many species in other groups are carnivorous or herbivorous, but they are in taxonomic groups with mixed trophic traits. A better separation into trophic guilds, notably in the Coleoptera and Heteroptera, is possible (Sattler et al., in press), but it requires better expertise than usually available with non-entomologists. The "RBA index for biocontrol diversity" is not a measure for the amount of potential pest organisms eaten by carnivores, which is a function of the abundances in the two trophic guilds. The index shows the adaptive potential for resilience in predator/prey relationships after sudden impacts, with immigration pressure, or along with general environmental changes.

The trends from 2000 to 2007 of herbivore and carnivore diversity differed slightly between agriculture, forestry and unmanaged habitats. While managed forests showed similar trends as unmanaged habitats, the trends in agriculture for carnivores and thus for the biocontrol diversity index were different. Carnivore diversity in agriculture did not increase in the two warm years 2003 and 2006, whereas those in managed forests and unmanaged habitats did so quite substantially. In 2006 the ratio between carnivores and herbivores (dashed line in Fig. 9) dropped in all three habitat types, whereas in 2003, when heat was combined with drought, it only dropped in agriculture, where the herbivores seem to have profited more from the heat.

Discussion

The main goal of the developed RBA programme was to be able to compare the development of biodiversity in the two dominant landscape management regimes in Switzerland, agriculture and forestry, with that in ecosystems without management pressure. Eight years of monitoring were not enough time to see drastic divergence between the three management regimes (Fig. 7). In all three habitat types, average alpha diversity was more strongly influenced by climate and weather than by the considerable management changes in agriculture and forestry over the last decades. In the exceptionally hot and dry summer of 2003 the extreme dryness lead to a reduced activity of insects in agriculture, because a

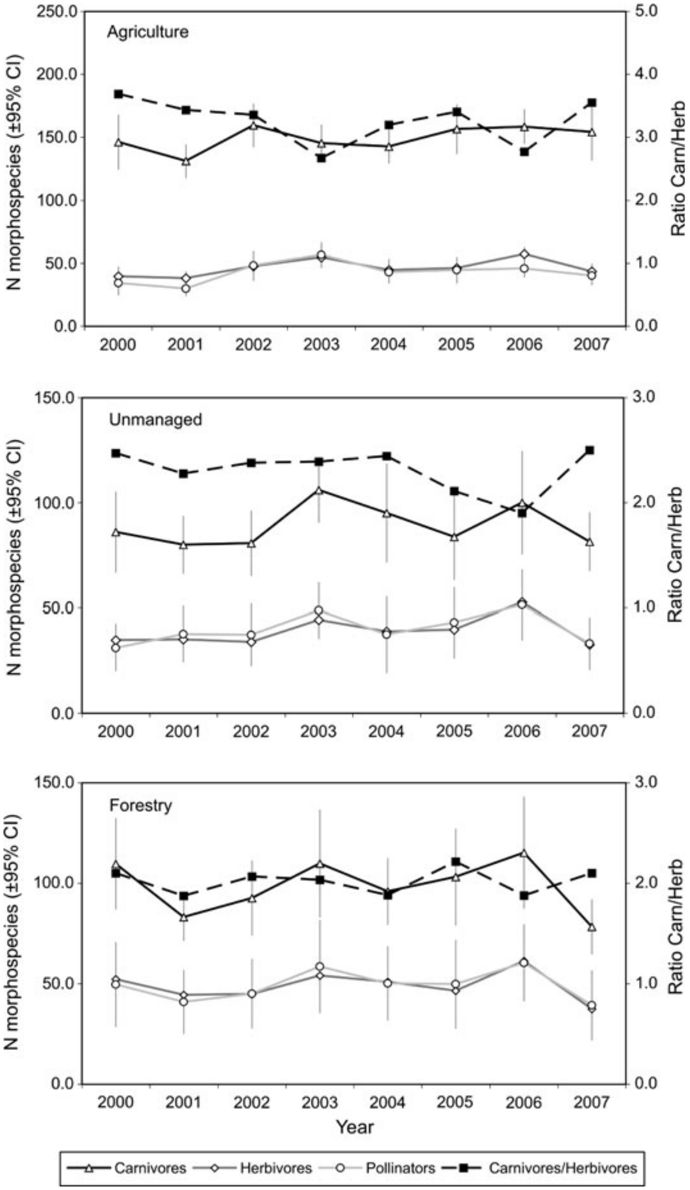

Fig. 9 RBA trends (±95% CI) of trophic guilds and the ratio between carnivores and herbivores ("index of biocontrol diversity") in the three investigated habitat or management types

lot of the crops and vegetation close to ground were dry and had few flowers. In cooler, moist or shaded habitats such as alpine meadows, wetlands and forests, the dryness was not able to hamper insect activity driven by high temperatures.

While at the single species level, which is the main concern of nature protection, notable changes in insect biodiversity may have taken place (local species extinctions, immigration of exotic species), no such changes can be found in forests or unmanaged areas. However, in agriculture some increases showed up in selected groups, possibly reflecting ecological measures taken in the last decade (Aviron et al. 2009). No such changes can be seen for average alpha diversity, local pollinator diversity, or the ratio between herbivore and carnivore diversity.

Another important goal of a RBA monitoring programme can be to provide a reference baseline for evaluating changes in short term research projects. If a doctoral thesis with 2 years of field work samples forest insects in 2006 and 2007, the resulting impressive downward trend is not a sign of a general decay of biodiversity, or the result of detrimental forest management, but simply reflects the impact of different weather in the two summers.

The RBA approach based on morphospecies is useful in long-term monitoring programmes where a broad spectrum of arthropod taxa can indicate overall biodiversity at moderate costs. With an average of 256 (SD $\pm$ 91), ranging from 69 to 522 morphospecies per trap station per year, the RBA index mapped a wide range of species richness of selected invertebrate groups and was a very good correlate to overall local species diversity (Fig. 5; Duelli and Obrist 1998). Similarly, in a case study on wild flora, Abadie et al. (2008) found that the number of parataxonomic units correlated well with the number of species identified by specialists.

The identification of large numbers of arthropods is expensive, and for many invertebrate taxa few or no specialists are available. Therefore, in most large scale monitoring programmes the arthropods are lacking completely, or only a few species-poor groups of organisms are sampled, with species identification and abundance. However, in the final report the information on faunistic diversity is usually boiled down to species richness of these groups, which certainly is a less accurate indicator of overall biodiversity than the RBA index. As the RBA indices are based on the activity of mobile, short lived organisms, they are much more sensitive to environmental changes than indicators based on sessile organisms such as e.g. vascular plants, mosses and gastropods, making up the indicator for local species richness in the Swiss Biodiversity Monitoring Programme (Z9 indicator, Weber et al. 2004).

With the RBA programme the main disadvantage of the compromise between spatial repesentativity and organismal representativity is that without species identification it is not possible to calculate beta- or gamma-diversity. Also, no information is available on the population trends of single species, unless the sorted and carefully stored material is later identified to the species level by specialists, or compared on single morphospecies level over the years. This way also a sudden outbreak of an invasive species could be traced back to the time and place of immigration. Counting the individuals per morphospecies would allow to estimate abundance related effects of ecosystem services, but at the same time increase the total costs of a RBA monitoring programme. The estimate would still be biased by the catching success, which also depends on insect activity, not only abundance.

Krell (2004) stated that parataxonomic units are useless for selection of areas of conservation concern and inventories in conservation evaluation, and of limited value for studies on species turnover. This is certainly true, but none of the authors critical about the use of morphospecies or parataxonomic units was ever concerned with monitoring the potential for ecosystem services or ecological resilience. In contrast to Abadie et al. (2008), we did not use a defined key for identification of the morphospecies. The richness of morphs in arthropods is prohibitive to such an endeavour and would raise complexity and effort to similar levels as species identification. Thus, unless a simplified identification key is devised and tested, beta diversity cannot be derived from RBA values. For implementing the proposed RBA programme on a national level, we suggest two methodological improvements: (1) For indicating the potential for maintaining ecosystem services, a more refined separation into trophic guilds, notably for Coleoptera and Heteroptera, is preferable (Sattler et al., in press), and the counting of abundance could be envisaged. (2) The sampling stations have to be distributed in a representative way over the area to be sampled (Weber et al. 2004). We further recommend to employ entomologically trained biologists for RBA-monitoring.

Conclusions

The "countdown 2010" (European Environment Agency 2007) will bring forward a lot of data and trends regarding the loss of biodiversity. The focus will be on conservation issues, the prevention of the loss of rare and threatened species. A number of indicators have been chosen internationally (European Environment Agency 2007), to decide whether the nations having signed the CBD have accomplished their obligation to "halt the loss of biodiversity by the year 2010".

While there is substantial knowledge on population declines in some popular groups of organisms such as birds, amphibia, flowering plants, and diurnal butterflies, documented in National Red Lists, there is very little information on trends of other aspects or entities of biodiversity, such as on most invertebrate taxa. Particularly lacking are indicators for functional biodiversity. How can we decide, in the year 2010, whether losses in the potential for ecosystem services had taken place and now have been halted? What is known on the ecologically relevant trends in average alpha-diversity? The "countdown 2010" will disclose what is known to date and what should be known for the next countdown.

Our RBA-programme complements the set of conservation centred indicators, with information on trends of alpha-diversity (local species richness of arthropods, which make up for about two-thirds of overall species richness, Hammond 1992). From the average morphospecies numbers of the separately counted taxa, guilds for particular ecosystem services can be formed. With these, very rough but affordable estimates can be calculated on trends of pollinator diversity and the potential for resilience in biological control against established pest organisms or invasive species.

All indices derived from the morphospecies approach are relative measures. They give no absolute values on species numbers or abundances present in a given habitat, because the probability for a species to get trapped depends on its abundance, activity and mobility. While most biodiversity indicators measure or estimate the presence or abundance of species in a given perimeter, the RBA-indices measure activity-density at the site of the trap. By this, they also integrate species originating from surrounding habitats, using the sampled habitat temporarily. The RBA indices thus incorporate information on the local habitat complex. For ecological resilience, and particularly for ecosystem services, this broad-spectrum type of functional alpha-diversity is more important than the presence of rare and threatened species. Or, to put it more bluntly, the more interesting an invertebrate species is for conservation (red list category), the less important it is for ecosystem services (Duelli and Obrist 2003).

We therefore advocate for future biodiversity monitoring programmes to include an indicator set of functional biodiversity and a tested correlate of overall alpha-diversity into the set of conservation centred compositional indicators.

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