Conceptual guidelines for the implementation of the ecosystem approach in biodiversity monitoring

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Abstract. The theoretical concept of the ecosystem approach (EA) aims at assessing ecosystem function based on integrative assessments of multiple levels of biological organization. Since the United Nations Convention on Biological Diversity in 1992, the EA has been increasingly integrated into environmental policy and legislation, but to date, its practical implementation remains vague with a lack of universal guidelines and concrete recommendations for its use across ecosystem boundaries. On the basis of a review of scientific literature, worldwide environmental legislation and existing monitoring approaches, we identified the most important factors which hamper the feasibility of the EA. We propose a generally applicable methodology for implementing the EA in ecological and environmental monitoring across different ecosystems and habitat types. Successful application of the EA largely depends on adequately standardized and synchronized sampling designs for all abiotic and biotic components, appropriate depth of taxonomic identification, and sufficient spatial and temporal replication. The proposed step-by-step guidelines for using the EA are valid across ecosystem types, geographic regions, and for a variety of data types, making them promising tools for ecological monitoring.

Key words: biodiversity; ecosystem change; environment; impact assessment; multiple taxonomic groups; multivariate data integration; restoration.

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Introduction

Ecosystems can be highly dynamic in space and time, caused by changes in natural factors and human activities. Today, natural ecosystem changes are mostly overlaid by anthropogenic impacts such as transformation of land and sea through the alteration of major biogeochemical cycles and the introduction or removal of species (Lubchenco 1998). An understanding of ecosystem change and its consequences is central for the success of conservation and restoration of ecosystems, especially of the services they provide (Millennium Ecosystem Assessment 2005). Monitoring activities are gaining increasing support and enforcement from politicians and society. Ecological monitoring was originally mainly focused on single taxonomic groups (Cairns and Pratt 1993), but there is growing evidence from different aquatic and terrestrial habitats of low congruency between different indicator groups in their response to environmental change (e.g., European and North American streams and lakes: Allen et al. 1999, Declerck et al. 2005, Heino 2010, Mueller et al. 2011, tropical rain forests in the Amazon basin: Landeiro et al. 2012). Consequently, there is an increasing need for more holistic and integrative concepts of ecological monitoring

(Geist 2011, 2015), including the simultaneous consideration of several indicator groups from different trophic levels (ecosystem approach, EA). However, current monitoring is often limited in terms of sampling designs considering multiple taxonomic groups as well as in the use of integrative methods for data analysis. Consequently, it is difficult to obtain a comprehensive and representative picture of the ecosystem as a whole which is crucial in terms of monitoring environmental change, including the effects of habitat destruction, pollution, or restoration and the associated ecosystem functions and services. Meta-analyses provide a promising tool to circumvent these problems, relying on the deduction of ecosystem level effects from multiple studies, each of which considered single taxonomic groups. However, they are often limited due to unknown data quality and methodological inconsistency as well as publication bias toward studies showing pronounced effects (Stewart 2010, Geist 2015).

This study reviews scientific literature, environmental legislation and practical monitoring approaches in the context of the ecosystem approach. While existing literature on this topic is either limited to one specific type of ecosystem (e.g., terrestrial ecosystems in Schmeller et al. 2015) or specific aspects of the EA (e.g., spatial standardization to reduce uncertainty in Magnusson 2014), the objective of this paper is to promote the applicability of the EA by a holistic view on the topic. This spans from the detection of current deficits—by providing a detailed and worldwide review of related legislation, monitoring protocols, and scientific literature—to practical recommendations that may increase the relevance and applicability of the EA. The main objectives were to identify existing deficits in the practical implementation of the ecosystem approach, and to deduce concrete and universally applicable conceptual guidelines considering (1) sampling design and methodology, (2) depth of taxonomic identification, (3) multivariate data integration, and (4) the transferability to different ecosystem types, investigation scales, and data types.

Definition and History of the Ecosystem Approach

The "ecosystem approach" (EA) is the primary framework for all actions in environmental monitoring, impact assessment, and management under the convention on biological diversity (United Nations 1992). In contrast to species-focused conservation strategies, the complex and dynamic nature of ecosystems, including all trophic levels and their interactions among each other, with the nonliving environment and humans is recognized within the framework of the EA (United Nations Environment Programme 1998). Since biological diversity is inextricably linked to ecosystem processes, function, and resilience (United Nations Environment Programme 1998), the EA requires a holistic view of biodiversity that extends from classical species diversity to functional diversity, diversity of ecosystem processes, and habitat diversity. The term EA had been referred to in scientific literature long before its first application in a policy context in 1992. It dates back to an article published in Ecology by Odum (1957), who suggested the study of ecosystems in university ecological field courses. He proposed that students should sample and analyze as many ecosystem components (which can be taxonomic or trophic units) and physicochemical variables as possible. Odum (1957) named this concept "ecosystem approach in teaching ecology". In the following decades, the EA appeared in publications on environmental toxicology (e.g., Metcalf 1977, Lakshman 1979), before the term was extended to socioeconomic and political dimensions in the 1990s (e.g., Rowe 1992, Spence and Hughes 1996, Sherman and Duda 1999). There are numerous definitions of what EA means (Laffoley et al. 2004). Rowe (1992) considers the idea of EA to "shift the focus from parts to wholes, from the interest to the capital, from trees and other plants, animals, stream flow, esthetics and whatever else the earth's surface yields to the three dimensional landscape ecosystems and waterscape ecosystems that produce these valuable things", and establishes an understanding with a strong focus on ecosystem services. Cury et al. (2005) consider the EA in fisheries in a very general way, being an approach that "deals with ecosystems instead of individual stocks", while De Jonge et al. (2012) specifically include community structure and function as well as environmental variables into their definition of

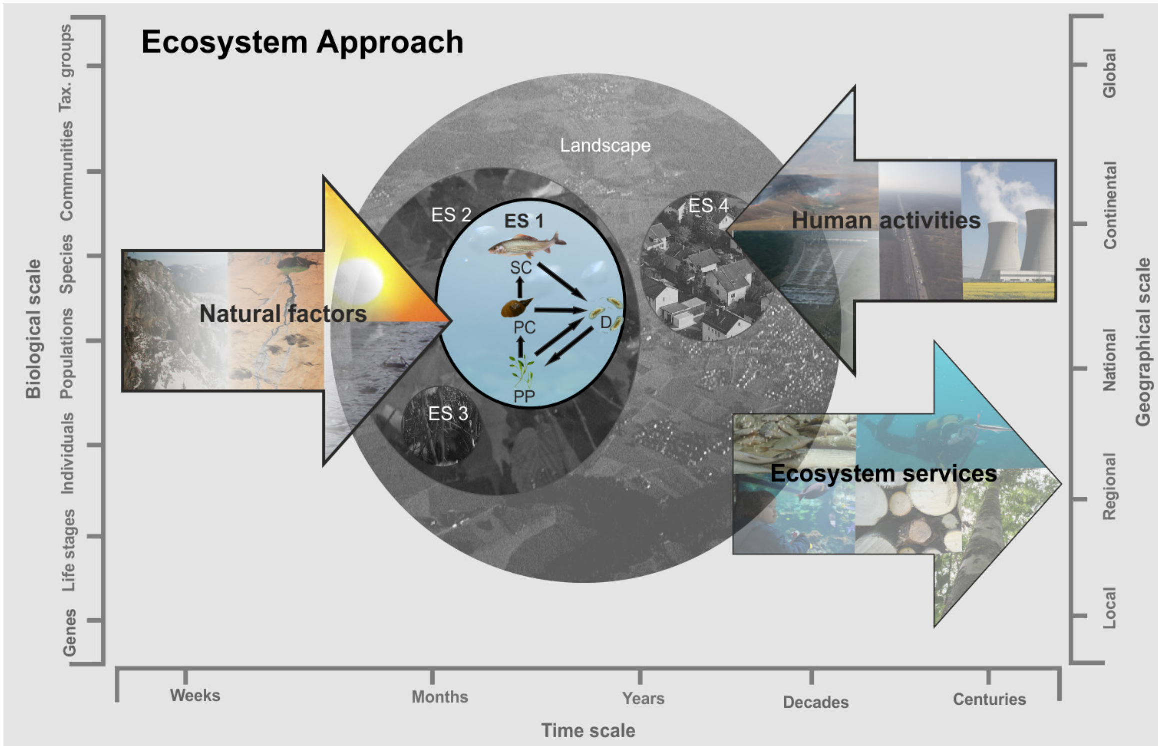

Fig. 1. Multiple dimensions of the ecosystem approach. Nested circles symbolize different ecosystems that interact with each other (figure part modified from United Nations Environment Programme 1998). ES 1 = ecosystem 1, e.g., freshwater ecosystem; ES 2 = ecosystem 2, e.g., alluvial forest; ES 3 = ecosystem 3, e.g., meadow; ES 4 = ecosystem 4, e.g., urban area. All these different ecosystems are situated within a landscape. PP = primary producers; PC = primary consumers; SC = secondary consumers; D = destruents. Arrows symbolize different factors that affect or emerge from ecosystems: Human activities = all human induced alterations of ecosystems which change their structure, function, and composition, e.g., hydropower plants or agricultural land use; Natural factors = physicochemical characteristics resulting from climate (e.g., solar radiation, temperature, rainfall) and geology (e.g., topography, lithology); Ecosystem services = provisioning, regulating, cultural, and supporting services provided by ecosystems. The scale bars on the left, right, and bottom indicate the different biological, geographical, and temporal scales at which the setup in the middle of the figure can be investigated.

EA: "environmental conditions should be assessed on the basis of the structure and functioning of the biological part of the ecosystem in response to the sum of natural variation and human induced stresses". Beaumont et al. (2007) provide a more management- and conservation-oriented definition of the EA, being a "strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way". However, all present definitions of the EA are based on a universal understanding of an ecosystem. An ecosystem comprises different components, representing different trophic levels (Fig. 1), including primary production (e.g., algae, plants), primary consumption (e.g., invertebrates), secondary and higher levels of consumption (e.g., insectivore or carnivore mammals and fishes), as well as destruents (e.g., bacteria) (Campbell et al. 2009). The interaction of different ecosystem components with each other and with natural environmental factors as well as human activities, determines the resulting ecosystem services (Fig. 1). Ecosystems can be analyzed at many different geographical, biological, and temporal scales which are hierarchically structured and can be nested into each other (Fig. 1). For instance, the investigation of ecosystem components can be conducted for a single ecosystem, from a wide range of ecosystem types (e.g., terrestrial, marine, and freshwater biomes), as well as at different geographical scales (e.g., local, regional, national, continental, and global), biological complexity (e.g., genes, individuals, populations, species, and communities of each trophic level) and temporal scales (e.g., months, years, and decades), and also for multiple ecosystems situated in an entire landscape (Fig. 1). Consequently, monitoring based on the principle of the EA also should be conducted on multiple scales. The minimum requirement to meet the EA in monitoring is to cover components from all trophic levels of a single ecosystem at the local scale.

Several decades after defining the EA and integrating it into the Convention on Biological Diversity (CBD) objectives, the question is no more whether the EA is required (Cury et al. 2005), but how to implement this complex concept. This still constitutes a key challenge in ecological research and management, even at the minimum level of single ecosystems (De Bello et al. 2010). Research into more integrative solutions to ecology and biodiversity conservation is now frequently postulated by scientists, and biodiversity conservation managers, and is increasingly enforced through legislation (Leslie and McLeod 2007, Levin et al. 2009, Geist 2011, Irschick et al. 2013, Pander and Geist 2013).

Methods

In a first step, we screened existing information in the context of the EA with the objective of identifying current limitations in its applicability, which is essential in deducing recommendations. Different sources of information were considered in assessing the application of the ecosystem approach in science, environmental legislation, and monitoring. We undertook web searches using the terms "ecosystem approach", "monitoring different taxonomic groups", "monitoring multiple taxonomic groups", "monitoring several taxonomic groups", "effects different taxonomic groups", "effects multiple taxonomic groups", and "effects several taxonomic groups". Searches for peer-reviewed literature were performed in "ISI Web of Knowledge" and "Google Scholar" in January 2015. The first 200 results for each search term per search engine were analyzed, excluding review papers that only analyzed already published studies on multiple taxonomic groups to avoid duplication. If several publications from the same authors on the same data sets were found, only the chronologically first publication was considered. The search resulted in a total number of 173 studies that were considered for the analyses presented in Fig. 2. A complete list of these studies is supplied in supplemental material (Appendix S1). In addition to the search for scientific studies, worldwide environmental legislation and monitoring protocols available in English (including all European, Asian, North American, Australian, and African protocols) or Spanish (including South American protocols) were also screened for EA, performing searches in "Google" as well as the websites of environmental authorities in January 2015. The search resulted in a total of 25 environmental policies from seven continents (Table 1), as well as in 13 monitoring protocols from five continents (Table 2).

Scientific studies were analyzed concerning the year of publication, the ecosystem type (marine, freshwater, or terrestrial), continent, purpose of study, and method of data integration. Environmental policies were analyzed according to the ecosystem type as well as the type of monitoring or impact assessment. Monitoring protocols were analyzed concerning the related environmental policy, ecosystem type, considered ecosystem components and number of taxonomic groups, sampling design (before-after-control-impact, reference condition approach) and analyses methods (multivariate or univariate statistics, multimetric single score indices, data integration methods).

International Environmental Legislation and the Ecosystem Approach

Legal implementation of the objectives of the CBD has been achieved on various international levels. The main focus of legislative action consists of environmental protection in general as well as laws protecting aquatic resources (Table 1). However, only few of these legal regulations (water laws in Europe, Argentina,

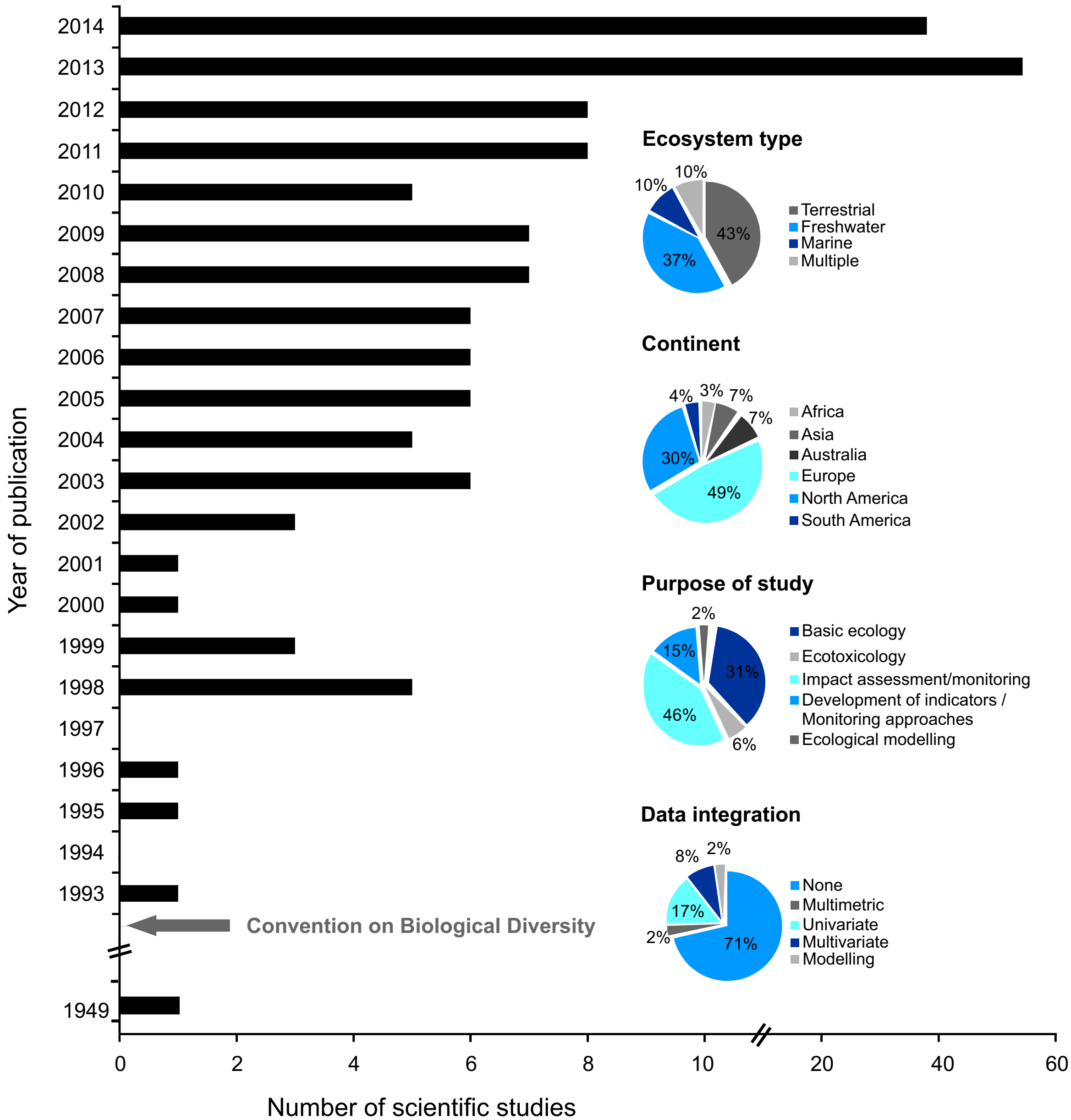

Fig. 2. Literature review of ecological studies including multiple taxonomic groups. Left part: Barplot indicating the number of scientific studies considering more than one taxonomic group per year. Right part: pie charts indicating the percentage of all reviewed studies (total: 82 studies) from different ecosystem types and continents, as well as with different overall study purpose and data integration methods.

and Namibia) directly refer to the EA in terms of declaring the integration of several biological, physical, and chemical ecosystem components into environmental monitoring mandatory (Table 1). The European Water Framework Directive (WFD, European Parliament 2000) most specifically regulates the monitoring of aquatic ecosystems by clearly defining the chemical, physical, and biological quality elements, the way of presenting the results and a time frame for surveillance monitoring in Annex V (European Parliament 2000), though

Table 1. Examples of worldwide environmental legislation considering ecological monitoring and impact assessment. PU = political unit; Environ. Law = environmental law; ES Type = ecosystem type. EA indicates reference to the ecosystem approach: + = explicit reference to defined ecosystem components; ± = mentioning of ecosystem components in general; − = no reference to the EA. EIA = environmental impact assessment. EU = European Union; USA = United States of America; CDN = Canada; AUS = Australia; ZA = South Africa; NAM = Namibia; KEN = Kenya; KOR = Korea; IND = India; CHN = China; RU = Russia; AR = Argentina; CHI = Chile; PAN = Panama. Note that only those environmental policies were considered that are available in English or Spanish language on the web.

Continent	PU	Environ. Law	ES Type	EA	Monitoring	EIA
Europe	EU	Habitats Directive (HD, European Parliament 1992)	Multiple	±	Surveillance monitoring	Regulated in European Parliament (2011)
Europe	EU	Water Framework Directive (WFD, European Parliament 2000)	Aquatic	+	Physical, chemical, and biological components	Adapted monitoring in case of bad condition
North America	USA	Environmental Policy Act (EA, 91st United States Congress 1970)	Multiple	±	Surveillance monitoring	Expert estimation
North America	USA	Clean Water Act (CWA, 92nd United States Congress 1972)	Aquatic	−	Surveillance monitoring	Not specified
North America	CDN	Environmental Protection Act (EPA, Government of Canada 2000)	Multiple	±	Environmental quality monitoring	Expert estimation, monitoring
Australia	AUS	Environmental Protection and Biodiversity Conservation Act (EPBCA, Australian Government 1999)	Multiple	+	Voluntary and scientific monitoring, financial support	Expert estimation
Australia	AUS	Water Act (WA, Australian Government 2007)	Aquatic	±	Quality and quantity of water resources	Expert estimation
Africa	ZA	Environmental Conservation Act (ECA, Republic of South Africa 1989)	Multiple	−	Not specified	Expert estimation
Africa	ZA	Water Act (WA, Republic of South Africa 1998*b*)	Aquatic	±	Quality and quantity of water resources	Expert estimation
Africa	ZA	Forest Act (FA, Republic of South Africa 1998*a*)	Terrestrial	±	Multiple components, including biodiversity	Expert estimation
Africa	NAM	Environmental Management Act (EMA, Republic of Namibia 2007)	Multiple	+	Functional integrity and biodiversity	Literature research, field work, monitoring
Africa	NAM	Water Resources Management Act (WRMA, Republic of Namibia 2004)	Aquatic	+	Physical, chemical, and biological components	Expert estimation
Africa	KEN	Environmental Management Act (EMCA, Parliament of Kenya 1999)	Multiple	−	Environmental changes	Environmental audit
Asia	KOR	Natural Environmental Conservation Act (NECA, Republic of Korea 1997*b*)	Multiple	±	Not specified	Regulated in Republic of Korea (1997*a*)
Asia	KOR	Water Quality and Ecosystem Conservation Act (WQECA, Republic of Korea 1997*c*)	Aquatic	−	Water pollution, water quality, aquatic ecosystems quality	Regulated in Republic of Korea (1997*a*)
Asia	IND	Biological Diversity Act (BDA, Parliament of India 2003)	Multiple	+	Areas rich in biological resources	Expert estimation
Asia	CHN	Environmental Protection Law (EPL, Republic of China 1989)	Multiple	−	Establishment of monitoring system	Expert estimation
Asia	CHN	Water Law (WL, Republic of China 1988)	Aquatic	−	Dynamic monitoring of water resources	Comprehensive scientific survey

Table 1. Continued.

Continent	PU	Environ. Law	ES Type	EA	Monitoring	EIA
Eurasia	RU	Law on Environmental Protection (LEP, Russian Federation 2002)	Multiple	+	Environmental condition in areas with man made effects	Not specified
Eurasia	RU	Forest Code (FC, Russian Federation 1997)	Terrestrial	±	Observations, assessments and forecasts	Conservation measures as compensation
Eurasia	RU	Water Code (WC, Russian Federation 2006)	Aquatic	−	Qualitative and quantitative indicators of waterbody state	Efficiency control for implemented measures
South America	AR	Law on National Environmental Policy (LGA, Republica de Argentina 2002*a*)	Multiple	±	Scientific investigations in the field of biodiversity conservation	Expert estimation
South America	AR	Law on the Use of Public Waters (RGA, Republica de Argentina 2002*b*)	Aquatic	+	Physical, chemical, and biological components	Expert estimation
South America	CHI	Environmental Law (LMBA, Congreso Nacional de Chile 2010)	Multiple	±	Not specified	Expert estimation, monitoring
South America	PAN	General Environmental Law (LGA, Republica de Panama 1998)	Multiple	+	Environmental quality, support of scientific studies	Regulated in Republica de Panama (2009)

not directly referring to the EA (Vlachopoulou et al. 2014). In contrast, details on data collection and analysis methods remain unclear. The development and the implementation of a monitoring strategy are left to the member states (Scheuer 2006), with no centralized system of data storage and accessibility (Geist 2014). In general, reference to EA within the framework of most national and international environmental legislation is limited to indirect regulations, for example, legal achievements such as the protection of the ecosystem with all its components without further specification of implementation, monitoring or enforcement regulations. This lack of precise implementation strategies in environmental legislation leads, especially in countries where access to scientific knowledge and technology is limited, to weak enforcement and implementation of the EA on the federal level (Alshuwaikhat 2005, Sands and Peel 2012). However, the increasing number of international legal regulations creates an urgent need for ecosystem-based environmental monitoring throughout the world. To improve precision and enforcement of environmental legislation particularly in developing countries, research in the field of standardized monitoring concepts that are adaptable to these regions is essential.

Current Implementation of the Ecosystem Approach in Science and Monitoring Practice

The review revealed a strong increase in the number of studies considering multiple ecosystem components over time, especially during the last few years (Fig. 2). The majority of the studies reviewed (88%) clearly support a "shopping basket approach" to ecosystem monitoring, that is, the use of a suite of taxa instead of single indicators (Pullin 2002). According to a review by Siqueira et al. (2015), the main research focus in ecological studies is on terrestrial ecosystems. Specifically looking at scientific studies and monitoring protocols that include multiple taxonomic groups, we found ecological research and applied monitoring to be equally distributed in aquatic and terrestrial ecosystems, with most of the aquatic studies addressing freshwater habitats (Fig. 2). Considering continents, multigroup research and monitoring is mostly carried out in Europe and North America, whereas implementations in other parts of the

Table 2. Examples of environmental monitoring systems. REL = respective environmental law (abbreviations from Table 1). ES Type = Ecosystem type: T = terrestrial; FW = freshwater; M = marine. Monitoring comp. = Monitoring components: SE = socioeconomic factors; PP = primary producers; PC = primary consumers; SC = secondary consumers and higher trophic levels; D = destruents; EV = environmental variables. TG = Number of taxonomic groups. AT = Analysis Type: UV = univariate statistics; MV = multivariate statistics; SA = statistics in general; MM = multimetric; d = descriptive. DI = method of data integration: SSI = single score index; SRP = sunray plots. RF = type of reference: RC = reference condition; BACI = before after control impact design; nsp. = not specified.

				ES Type			Monitoring comp.									
Continent	PU	Monitoring Protocol	REL	T	FW	M	SE	PP	PC	SC	D	EV	TG	AT	DI	RF
Europe	GER	Natura 2000 Monitoring (Sachtleben 2010)	HD	x	x	x		x	x	x		x	6	d	-	nsp.
Europe	GER	PHYLIB (Schaumburg et al. 2007), PERLODES (Meier et al. 2006), FiBS (Diekmann et al. 2005)	WFD		x			x	x	x		x	4	MM	SSI	RC
North America	USA	Rapid Bioassessment Protocols (Barbour et al. 1999)	CWA		x			x	x	x		x	4	MM	SRP	RC
North America	CAN	Marine and Estuarine Biodiversity Monitoring Protocols (Environment Canada 1997)	EPA			x		x	x	x			5	UV, MV	-	BACI
North America	CAN	Field Manual Wadable Streams (Environment Canada 2012)	EPA		x				x	x		x	1	nsp.	-	RC
North America	CAN	Framework for Monitoring Biodiversity Change (Roberts-Pichette 1995)	EPA	x				x	x	x		x	7	MM, MV	-	nsp.
Australia	AUS	Methods for Ecological Monitoring of Coral Reefs (Hills and Wilkinson 2004)	AUS-EPBCA, AUS-WA			x		x	x	x		x	3	SA	-	BACI
Australia	AUS	AUSRIVAS (Nichols et al. 2000)	AUS-EPBCA, AUS-WA		x				x	x		x	1	MV	-	RC
Africa	ZA	River Health Programme (Dallas et al. 2008)	ZA-WA		x			x	x	x		x	4	MM, MV	SSI	RC
Africa	NAM, ZA	BIOTA Africa (Jürgens et al. 2012)	ECA, EMA	x			x	x	x	x	x	x	7	SA	-	nsp.
Asia	KOR	Aquatic Ecological Monitoring Program (Lee et al. 2011)	WQECA		x			x	x	x		x	3	MM	-	RC
Asia	IND	CLEAN India (http://www.cleanindia.org)	BDA	x	x		x	x	x	x		x	2	d	-	nsp.
South America	AR	Monitoreo Ambiental Rural (Zaccagnini et al. 2007)	LGA, RGA	x	x		x	x	x	x		x	10	nsp.	nsp.	nsp.

world are scarce (Fig. 2 and Table 2). A large percentage of the scientific literature on ecosystem-scale assessments addresses basic ecological questions (Fig. 2), such as community congruency (e.g., Allen et al. 1999, Ficetola et al. 2007, Mykrä et al. 2008, Larsen et al. 2012, Padial et al. 2012) and biodiversity research (e.g., Niemelä and Baur 1998, Fabricius et al. 2003, Bailey et al. 2007). However, since 2013, there is also a stronger focus on environmental monitoring considering biological responses to ocean acidification (Harvey et al. 2013, Kroeker et al. 2013), change in land use (Birkhofer et al. 2014, Johnson and Angeler 2014, Luescher et al. 2014), and the introduction of neobiota (Jackson and Grey 2013, Kumschick and Richardson 2013, Galiana et al. 2014). Although primary producers, primary consumers, and higher trophic levels are well represented, micro- and meiobiota (e.g., bacteria, archaea, fungi, protozoa) are largely excluded from ecosystem-based studies and monitoring protocols (Thompson et al. 2012, see also Fig. 2 and Table 2). However, they play a key role in decomposition and thus link trophic levels (Townsend et al. 2009). While standardized data collection methods are available and used for most disciplines (e.g., electrofishing or net fishing: freshwater fish [EN 14962], Braun-Blanquet (1932); plant coverage estimation method: terrestrial vegetation, pitfall traps: terrestrial vertebrates), there are typically no standards used for the study design (i.e., selection of included ecosystem components, spatial and temporal replication) and the integration of data from different ecosystem components (Dabrowski et al. 2011). Consequently, data are typically analyzed and presented separately for each taxonomic group both in science as well as in applied monitoring, using a range of different methods, including descriptive methods, univariate statistics, modeling approaches, and multivariate statistics (Fig. 2, Table 2). Even in cases such as the European Water Framework Directive, which requires an integration of different biological and physicochemical quality elements (e.g., Geist 2014), samples for different taxonomic groups are not taken at the same sites at the same time, questioning the validity of data integration. General assessment results are mostly realized by the calculation of single score indices (e.g., Crosswhite et al. 1999, Maes and Dyck 2005, Lougheed et al. 2007, Schouten et al. 2009), bearing the risk of losing information content inherent in the multiple dimensions (i.e., large number of response variables) of ecosystem level data (Caroni et al. 2013, Dahm et al. 2013). To date, only few authors apply multivariate ordination methods (Canonical Correspondence Analysis, CCA; NMDS) integrating data from different taxonomic groups (Thompson and Townsend 2000, Guerra-García et al. 2006, Martínez-Crego et al. 2010, Mueller et al. 2011, 2014a). Meta-analyses are a recently emerging method of data integration in which the results of single studies on different ecosystem components can be combined (e.g., Kroeker et al. 2013, Slatyer et al. 2013, Deikumah et al. 2014, Stevens et al. 2014, Tuck et al. 2014), potentially ignoring differences in temporal and spatial resolution, sampling techniques as well as data processing.

To the best of our knowledge, the only common data integration method practiced in applied monitoring is the simplification of the complex data to a single score multimetric index (e.g., European WFD Monitoring, South African River Health Programme, Table 2). Most monitoring protocols leave data analysis up to the user, resulting in scarce statistical validation of the results (less than half of the monitoring protocols reviewed, Table 2) and low comparability between studies. Monitoring approaches in general are highly specific to certain geographic regions, since they strongly rely on reference conditions and the occurrence of certain indicator taxa for index calculation. This also limits the transferability of monitoring approaches between ecosystem types and geographic regions, and the comparability of their results. However, an examination across ecosystems and geographic regions would be essential for the sustainable management of biodiversity at the global level.

Conceptual Monitoring Strategies Based on the Theoretical Principle of the EA

Comprehensive ecosystem studies that consider multiple dimensions of variation (multiple taxonomic groups, temporal, and spatial resolution) are very time and cost intensive. The importance of including adequate numbers of spatial and temporal replicates in environmental monitoring is often underestimated (Pander and Geist 2013), often hampering statistical validation of the results. As a consequence, ecosystem-based management decisions are currently often hampered by oversimplification (Caroni et al. 2013, Geist 2015) and a lack of estimation of uncertainty. To improve ecosystem management and conservation, a conceptual standard for the selection of ecosystem components and the analysis of the resulting data that is applicable across ecosystems is needed. In this context, the establishment of a methodological framework for coupling and integratively analyzing abiotic data and biological community data eliminating the shortcomings of the recent index-based approaches is crucial (Lindenmayer and Likens 2010, De Jonge et al. 2012). This methodological framework for ecosystem monitoring should ideally consider ecological relevance, statistical credibility, cost-effectiveness, and transferability to other systems, which are considered as the most important criteria for successful ecological monitoring programs (Caughlan and Oakley 2001).

Selection of ecosystem components

Over the past decades, monitoring of biodiversity has mainly focused on species richness of certain, intensively studied biotic groups (e.g., birds, butterflies, stream macroinvertebrates) and systematically neglected and oversimplified the complexity of ecosystems as well as functional interactions between different components (De Leo and Levin 1997, Fischer et al. 2004). A global harmonized biodiversity observation system with common "Essential Biodiversity Variables" (EBVs) has not yet been established (Pereira et al. 2013). Some authors see this as the main reason for the failure of appropriate monitoring in the context of biodiversity conservation targets (Sachs et al. 2009, Rands et al. 2010, Pereira et al. 2013). A monitoring program following the theoretical principle of the EA should address the variety of possible reactions of different taxonomic groups found by other authors (e.g., Niemelä and Baur 1998, Maes and Dyck 2005, Larsen et al. 2012) and the strong interactions among trophic levels (Campbell et al. 2009). To achieve this, the selected set of ecosystem components should include at least one representative community per trophic level and habitat type. In aquatic and terrestrial habitats these could be algae or vascular plants for primary producers, invertebrates for primary consumers, vertebrates for secondary consumers, and bacteria for destruents. At a level of finer resolution, integration of different life stages can also become important, particularly if the case of presence–absence data or abundance data is not representative of the status of community components.

Sampling design and methodology

A standardized spatial and temporal sampling design is crucial to align data from different taxonomic groups of a common spatial area and time frame (Magnusson 2013, 2014, Pereira et al. 2013, Schmeller et al. 2015). The problem is that field sampling methods for different taxonomic groups and physicochemical characteristics typically differ in terms of the spatial scale of the sampling area. Secondary consumers, which include large and mobile species, are mostly assessed based on an area covering several m^2 due to their larger home ranges. In contrast, primary consumers, primary producers, or destruents (e.g., soil bacteria) occur at scales from one m^2, down to mm^2. A possible solution for this problem is to apply a nested stratified-randomized sampling design. The sampling area of larger taxa is covered by a collective sample of smaller taxa (Fig. 3), which can ideally be integrated into a "before after control impact" (BACI) design (Underwood 1992). In this context, the most crucial step is to determine the spatial and temporal replicates necessary to representatively cover all habitat types and seasonal conditions of the investigated ecosystem, and take into account autocorrelation (e.g., using geostatistical methods, Braun et al. 2012). The spatial extent of the lowest common sampling unit should ideally be based on the scales at which management decisions will be made (Fig. 3). To maintain statistical power, large numbers of replicates, allowing for spatial allocation to specific habitat structures, are needed. Thus, covering the area of interest with a larger number of spatial and temporal replicates in a stratified-randomized way is advantageous compared to relying on one large sample for each discipline (Schmeller et al. 2015). In the context of ecosystem services, ecosystem productivity is a very important aspect in the evaluation of ecosystem changes

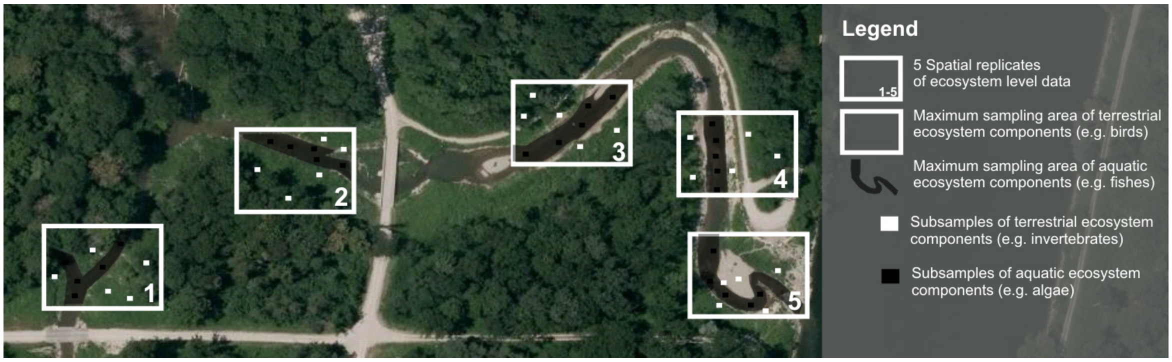

Fig. 3. Schematic of a nested stratified-randomized sampling design considering multiple ecosystem components.

(Thorp and Delong 1994). Therefore, defined sampling units should be assigned for each taxonomic group, for example, standardized sampling time, area, transect length, volume, or mass that allow the calculation of catch per unit effort (CPUE, Gulland 1964). Furthermore, it is necessary that the sampling of different ecosystem components at a site is conducted simultaneously.

Depth of taxonomic identification

The inclusion of multiple taxonomic groups in ecosystem assessments generally involves a considerable amount of laboratory work for taxonomic identification. Especially for difficult taxonomical groups, such as algae and some invertebrate groups (e.g., chironomids, terrestrial beetles). This can strongly increase monitoring cost and error rates (Haase et al. 2010). Testing the applicability of coarser taxonomic levels as surrogates for species-level data is highly effective in reducing effort and monitoring costs in terrestrial, freshwater, and marine ecosystems (Pik et al. 1999, Bevilacqua et al. 2012, Mueller et al. 2013, Timms et al. 2013). Moreover, consideration of several taxonomic groups at coarse taxonomic resolution tends to be more effective for the analysis of ecological change than analyses of single groups at a fine level of taxonomic resolution (Mueller et al. 2013). Statistical threshold levels for the applicability of coarse taxonomic resolution seem to be dependent on the taxonomic group rather than on the type of ecosystem (Bevilacqua et al. 2012, 2013, Mueller et al. 2013). Calibration lines (Bevilacqua et al. 2012, Mueller et al. 2013) as well as the R-based tool "BestAgg" (Bevilacqua et al. 2013), allow for an easy determination of the most appropriate aggregation level based on the average taxonomic breadth of a species list in previously untested habitats or taxonomic groups. Nevertheless, scientists should still pay attention to species-level identification when it comes to studies into the effects of subtle environmental gradients. Furthermore, functional characteristics can often only be assigned at fine taxonomic resolution, and the consideration of specific target species also makes species identification essential.

Data integration and analysis

The integration of data gained by different methods and techniques is subject to several constraints. First, the sampling of multiple taxonomic groups produces large, multidimensional data sets with each species constituting one dimension. The reduction in multiple data dimensions to one single dimension, for example, through the calculation of single score indices, can severely reduce precision and sensitivity (Norris 1995, Caroni et al. 2013, Dahm et al. 2013, Mueller et al. 2014*a*). Data integration methods should therefore allow multivariate analyses of the overall data set to conserve the multiple dimensions of ecosystem components and avoid the leveling out of opposite responses of different taxonomic groups (Reynoldson et al. 1997, Mueller et al. 2014*a,b*). Discipline-specific differences in quantity (e.g., full quantitative data, relative abundance or presence–absence

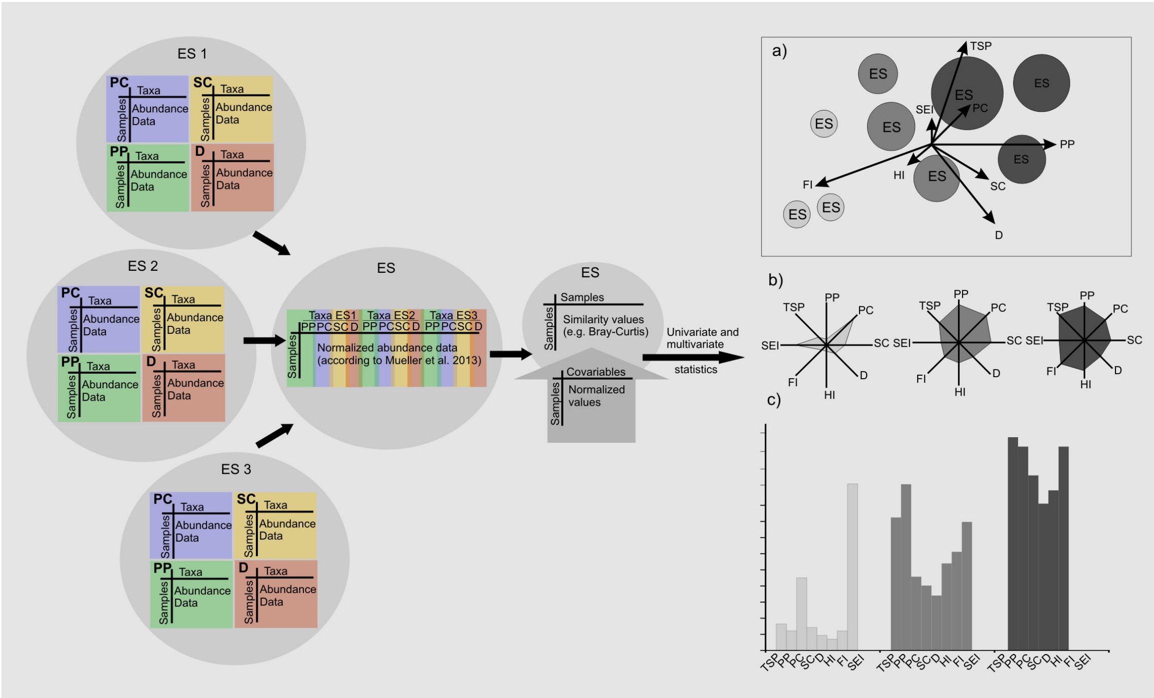

Fig. 4. Guideline for the multivariate integration of data from different taxonomic groups and ecosystems. ES1-3 = different ecosystems, compare Fig. 1. Ecosystem components: PP = primary producers, PC=primary consumers, SC = secondary consumers, D = destruents. Samples = single replicates for different treatments at highest common spatial and temporal sampling resolution for all taxonomic groups and ecosystems. Abundance data = individual counts for each taxonomic group. Similarity values=any measure of similarity that is appropriate for the used data type. Covariables = parameters that are associated with changes in community composition (normalized values; e.g., HI: physicochemical habitat characteristics, FI: functional traits, TSP: target species and life stage-specific variables or SEI: socioeconomic values). (a), (b), and (c) show different possibilities of data presentation, adaptable to the purpose of the study and the end user: (a) comprehensive scientific presentation of overall community composition as ordination plot (e.g., NMDS). Covariables are correlated onto the ordination plot and displayed as vectors of different size and direction, indicating the strength of the correlation and the direction of gradient, (b) sunray plots including selected criteria of special management interest, (c) barplots including selected criteria of special management interest. Different shades of gray indicate different sampling periods or treatments.

data) and numerical magnitude (i.e., natural differences in numbers of individuals of different groups) should be accounted for prior to data integration, without losing important information on differences in productivity between habitats as caused by the calculation of relative abundance.

Mueller et al. (2014*b*) proposed a multivariate data integration method, which is based on the normalization of the entire abundance data matrix of each studied group. This procedure can be universally applied to different taxonomic groups from various ecosystem types. It ensures an equal contribution of each taxonomic group to the overall results, but at the same time conserves information on productivity. In Fig. 4, we propose a step-by-step guideline for data integration. Essential prerequisites for following this guideline include the identification of a common spatial sampling unit for all components to be included beforehand. This common spatial unit can be applied to different treatments, such as impacted and unimpacted parts of the studied system, and constitutes the "samples" in Fig. 4. The basis for data integration and subsequent multivariate analyses are separate matrices containing abundance data for each investigated abiotic and biotic ecosystem component which can be extracted from different, discipline-specific data bases. In this context, the taxa can either be species or any other level of taxonomic resolution which applies to the group-specific statistical reliability and the purpose of the study. After a normalization step (Mueller et al. 2014*b*), individual abundance data matrices can be combined to one overall matrix in a spreadsheet such as Microsoft Excel. This matrix can be further used in common multivariate community analyses (e.g., nonmetric multidimensional scaling, analysis of similarities, permutational multivariate analysis of variance, biota-environmental matching). This provides a statistically verifiable, diagnostic assessment of the quantity and gradient directions of effects independent of conjectural expert judgment and coarse categories of impact (Baird and Hajibabaei 2012).

Despite the high complexity inherent to comprehensive ecosystem investigations and multivariate ordination plots, the most management relevant variables can be selected according to the case-specific conservation targets, and presented in a way that is easily accessible for conservation practitioners (e.g., barplots, sunray plots accompanied by the results of statistical analyses, Fig. 4). In addition to the integrative plots indicating overall ecosystem change, equivalent plots can be produced for single taxonomic groups to provide information on complementarity which is often required in systematic conservation biology (Magnusson 2014).

Transferability to different data types

In general, multivariate community analysis methods depend on some kind of nomenclature for the single elements of the community under study. However, it is not essential for the names to follow Linnean taxonomy. The names of the taxa in the abundance data matrices can be substituted by operational taxonomic units or numbers that individually mark and distinguish each taxon. Accordingly, also measures of functional processes, gene activity, or metabolites can be considered. The covariables included can be extended from traditional measurements of physicochemical habitat characteristics to target species and life stage-specific factors, functional traits, ecosystem processes, or socioeconomic measures of ecosystem services. Furthermore, the number of variables and covariables included is not restricted as long as the available statistical software can handle the size of the data set. This is of particular relevance for the future integration of data gained from modern approaches in the field of molecular biology, such as the currently arising environmental DNA (eDNA) metabarcoding (Ficetola et al. 2008) or nontargeted metabolomics (Suhre and Schmitt-Kopplin 2008). Often, scientists in these fields use similar data analysis methods as those applied in taxonomy-based community ecology (e.g., Nylund et al. 2011, Gonzalez and Knight 2012, Lefévre et al. 2013, Rocha et al. 2013), but currently information from traditional and molecular approaches are hardly combined in ecosystem studies. Applying the universally applicable "normalizing" procedure for multivariate data integration provided in Mueller et al. (2014*b*), classical community data and information gained from molecular approaches can be normalized independently of data structure and analyzed integratively in future ecosystem assessments.

In any case, access to original data (e.g., NCEAS, DATA ONE) has become a key requirement for publication of results to ensure the possibility of quality control. In general, the publication of monitoring results and data is considered to be essential for the success of conservation and restoration (Geist 2015), as well as for the applicability of the EA on a meta-level. However, for practical reasons giant data bases including data from all disciplines are not desirable. Scientists should rather consider the opportunity to extract abundance matrices suitable for data integration at the stage of data base design and upload their data bases in public data repositories. Given the high number of international standardization efforts, it would also be desirable to consider minimum standards for data integration from multiple taxonomic groups and sampling, for example, based on a similar number of replicates and the choice of the same sampling sites.

Transferability across ecosystems and geographic regions

Since ecological community data are usually structured similarly throughout ecosystem types (taxa × sites matrices), the presented methods for data combination can be universally applied to integratively assess community change in multiple taxonomic groups from all types of habitats. The matrix normalization allows the integration of an unlimited and flexible number of taxonomic groups (as long as there is a minimum common sampling resolution) that can simultaneously comprise different ecosystems or habitats, such as a stream system and the surrounding alluvial forests and wetlands, or the entire catchment landscape (Fig. 1). The multivariate consideration of taxa abundance data based on the similarity measure for all ecosystem types allows the comparative quantification of effect size across ecosystems (e.g., comparison of the effect size of floodplain restoration measures on the alluvial forest and aquatic habitats) and geographic regions. Furthermore, this approach is independent of minimum abundances of certain indicator taxa and can also include neobiota, which can both be critical constraints that limit the applicability of traditional monitoring approaches in heavily altered ecosystems (e.g., weir-influenced river sections, Mueller et al. 2011, Arndt et al. 2009, Mueller et al. 2014*a*) and their transferability to other geographic regions. Unfortunately, systematic scientific evidence supporting taxonomic sufficiency methods in other ecosystems is still restricted to single taxonomic groups (marine macroinverteberates, terrestrial invertebrates), making the application of the ecosystem approach very costly in terms of labor (Bevilacqua et al. 2009). The applicability to a broad range of data types, together with the high transferability between ecosystem types and geographic regions makes the new data integration and analysis approach presented herein a highly flexible and promising tool for the future development of ecological monitoring.

Conclusions

In conclusion, there is no doubt about the usefulness of the EA in ecological monitoring that aims at holistic assessments of ecosystem changes. Currently, applicability of the EA is mostly hampered by deficits in standardized sampling designs, and a lack of transdisciplinary collaboration that could result in common databases. Implementation of the ecosystem approach in biodiversity monitoring ultimately depends on the representative integration of all trophic levels and taxonomic groups. In particular, data on micro- and meiobiota should be better considered. As a general rule for data acquisition, greater emphasis should be placed on trophic and taxonomic representation than on resolution of taxonomic detail within specific functional groups and species. Moreover, placing stronger emphasis on a larger number of spatial and temporal replicates that remain the same for different ecosystem components is essential, yet often not considered in classical monitoring programs. A greater degree of standardization of sampling designs, data analyses and data storage, for example, through international standardization bodies and committees, is a prerequisite for improving the usefulness of this approach in meta-analyses.

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