What is ecosystem multifunctionality?

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Preface

Recent years have seen a surge of interest in ecosystem multifunctionality, a concept that has developed in the largely separate fields of biodiversity-ecosystem function and land management research. Here we discuss the merit of the multifunctionality concept, the advances it has delivered, the challenges it faces, and solutions to these challenges. This involves the redefinition of multifunctionality as a property that exists at two levels: ecosystem function multifunctionality and ecosystem service multifunctionality. The framework presented provides a road map for the development of multifunctionality measures that are robust, quantifiable and relevant to both fundamental ecological science and ecosystem management.

Introduction

The idea of holistic 'whole ecosystem' properties and measures has a long history in ecology[1]. However, research into the ability of ecosystems to simultaneously provide multiple ecosystem functions and services (multifunctionality) has become increasingly common in recent years, as comprehensive datasets and model outputs from multidisciplinary, collaborative projects have become available[2-8]. Multifunctionality has been defined in several ways, including 'the overall functioning of an ecosystem[2], 'the simultaneous provision of several ecosystem processes'[9], the 'provision of multiple ecosystem functions and services at high or desired levels'[10], and 'the potential of landscapes to supply multiple benefits to society'[11], to name a few. However, underlying these seemingly simple definitions are complex and unresolved issues regarding the conceptualisation and measurement of multifunctionality[9-11,] and the overall utility of the multifunctionality concept in practice[12-15]. Research on multifunctionality has been carried out within two largely separate research fields: one that sought to understand how biotic attributes of ecological communities (mainly biodiversity) are related to overall ecosystem functioning (biodiversity-ecosystem functioning research), and the other which concerns how landscapes can be managed to deliver multiple, alternative land-use objectives (land management research). Accordingly, these two fields have defined and measured multifunctionality in very different ways.

In this article, we first discuss the potential benefits of the multifunctionality concept, and the advances it has enabled, before discussing the risks and drawbacks of current approaches to studying multifunctionality. We show how more explicit definitions of multifunctionality are required to overcome these hurdles and to answer both fundamental and applied research questions. In light of these challenges we propose a new general framework that defines multifunctionality at two levels. The first, *ecosystem function multifunctionality*, is most relevant to fundamental research into the drivers of ecosystem functioning, which we define as the array of biological, geochemical and physical processes that occur within an ecosystem. The second, *ecosystem service multifunctionality*, we define as the co-supply of multiple ecosystem services relative to their human demand, and is most relevant for applied research in which stakeholders have definable management objectives. These ideas are illustrated with worked examples from European forests. We conclude by showing how this framework can be extended to measure multifunctionality at the larger spatial and temporal scales where it is most relevant.

Benefits of the multifunctionality concept

Traditional studies of ecosystem functioning within the field of ecosystem ecology typically involve detailed investigations into how individual functions relate to their drivers. Moreover, by quantifying functions in a standardised way (e.g. soil carbon fluxes, biomass production) these measures can be compared amongst ecosystems and studies[15]. However, ecosystem functioning is inherently multidimensional and so multifunctionality measures can potentially complement this approach by summarising the ability of an ecosystem to deliver multiple functions or services simultaneously. Just as aggregated community-level properties such as species richness, evenness and functional diversity[16-17] have provided great insight into broad ecological patterns at a higher level of organisation, multifunctionality research could generate an integrative understanding of ecosystem functioning and ecosystem service provision.

The concept of ecosystem multifunctionality has recently gained traction with the publication of several studies that assessed the relationship between biodiversity and ecosystem functioning within experimental systems[2,3,18,19]. Overall conclusions from these studies have been largely consistent: the relationship between biodiversity and ecosystem functioning becomes stronger when multiple functions are considered. This has been attributed to different species promoting different functions[2,20,21], but recent work shows that such positive biodiversity-multifunctionality relationships can also be driven by the effect of diversity on individual functions and statistical averaging effects[22]. An increasing number of studies have also shown positive relationships, but of varying strength, between biodiversity and the multifunctionality of non-experimental 'real-world' (i.e. natural, semi-natural and human-dominated) ecosystems, where management and abiotic drivers additionally affect functioning[10,23-28].

The multifunctionality concept used in biodiversity-ecosystem functioning research overlaps with ideas developed in research fields related to landscape-level management of ecosystem services, where there is a long history of studying the drivers of 'multifunctional landscapes', although the term multifunctionality itself is not always used. The motivation for such work is that a growing and resource-hungry human population is placing increasing pressure on dwindling land resources[29] resulting in a need to design and manage landscapes that can reliably provide multiple ecosystem services simultaneously. For example, the concept of landscape multifunctionality permeates discussions over the design of landscapes in which food and bioenergy production, carbon storage, flood regulation and biodiversity conservation are all goals[7,8,30]. Landscape multifunctionality is also central to the 'land sparing' versus 'land sharing' debate, which focuses on the relative merits of managing for biodiversity and food production within the same or separated land areas[31,32].

Measurement of multifunctionality

To date there has been no single accepted definition of multifunctionality, nor any agreed means of measuring it. In biodiversity-ecosystem functioning studies the main methods for quantifying ecosystem-level multifunctionality are the 'averaging' (or sum) approach and the 'threshold' approach. The averaging approach takes the average, or sum, of the standardised values of each function[28,33]. In contrast, the threshold approach[9,18] counts the number of functions that have passed a threshold, or a range of thresholds, usually expressed as a percentage of the highest observed level of functioning in a study[9,18,23,27,34]. The conceptual and mathematical merits of these approaches have been discussed and reviewed from the viewpoint of biodiversity-ecosystem function research[9,22,35] but their relevance to other fields of fundamental ecological research, and to the management of 'real-world' ecosystems, has not.

Averaging- and threshold-multifunctionality measures are now being related to a wide range of other ecosystem drivers, including climate[25,28,34], soil conditions[36], habitat diversity[37], land cover changes[38], nitrogen enrichment[12,39], invasive species[40], and management actions, such as agricultural intensification[10], pasture and green roof planting schemes[41,42] and crop planting systems[39,43,44]. These advances have blurred the line between the multifunctionality concepts used in the biodiversity-functioning and land management research fields. In the latter, multifunctionality is defined more broadly than it is in biodiversity research, and it can even encompass social factors such as employment and benefits provided by human infrastructure (e.g. transport systems) in addition to ecosystem components[45,46]. Furthermore, multifunctionality is typically considered at much larger (landscape) scales than in most biodiversity research, and there is sometimes consideration of both the demand for ecosystem services (the level of service provision desired by people[47]) and their supply (the capacity of an ecosystem to provide a given ecosystem service[47]). Maps of multiple ecosystem service supplies are often overlain to assess trade-offs and synergies between them[48,49], to identify ecosystem service bundles, i.e. a set of services with a similar pattern of supply[50-52], or to find hotspots of multiple ecosystem services that can be prioritised for conservation[48,49]. These approaches could be extended to create more explicit measures of ecosystem-service multifunctionality that can inform a diverse range of ecosystem management decisions, with potential applications including the setting of restoration targets, invasive species management, forest planting and the design of agri-environment schemes. Multifunctionality measures can also indicate the overall benefit provided by an ecosystem to a range of stakeholder groups, thereby helping to minimise trade-offs and conflicts between them[10].

Multifunctionality risks

While the concept of multifunctionality can be useful in both fundamental and applied ecology, its measurement is extremely challenging. Any multifunctionality measure will always be comprised of a subset of all possible functions or services and so will only capture a fraction of "true" multifunctionality. Unfortunately, so far, few researchers have carefully defined what their subset of functions represents and what it omits. It is also clear that the definition of multifunctionality determines how it is measured, and vice versa. Hence, the different perspectives in biodiversity and land management research and the intermingling of these fields mean that a better conceptualisation of multifunctionality is required.

As with any aggregated measure, multifunctionality metrics simplify reality, and can obscure important information about variation in individual functions and their drivers[12]. Many drivers have contrasting effects on the component functions of a multifunctionality measure, meaning that trade-offs between ecosystem functions and services are common, and it is impossible to maximise all functions simultaneously. For example, promoting soil nutrient turnover often results in the release

154 of carbon dioxide, thus boosting one ecosystem service (crop production) while diminishing another
155 (carbon storage)[39]. Where such trade-offs exist, there is therefore uncertainty in how well measures
156 of multifunctionality reflect mechanistic relationships[12-14]. A new method for measuring
157 multifunctionality, the Multivariate Diversity-Interactions framework[35], overcomes some of these
158 limitations by testing the relative importance of drivers across functions and identifying trade-offs
159 between them. This provides considerable insight into the drivers of each function but the method
160 does not provide a measure of overall multifunctionality and its complexity and reliance on detailed
161 data may limit its widespread adoption.
162
163 Current standard practice in both averaging and threshold-based approaches is to include all
164 available measures of ecosystem functions and services, to include a mix of state, rate and indicator
165 variables, and to weight all variables equally[12,23,25-27,36]. It is also common for multiple closely related
166 variables to be included in multifunctionality measures. This causes the up-weighting of certain
167 aspects of ecosystem functioning or particular ecosystem services, biasing the multifunctionality
168 measurement, especially if other important ecosystem functions are not measured. Furthermore,
169 such measures assume that all functions are equally important, which may be a false assumption in
170 many cases, as ecosystem managers typically prioritise certain functions or services in particular
171 contexts. To address this issue, a recent study in European grasslands[10] weighted functions according
172 to their presumed importance to different management objectives, such as agricultural production
173 or tourism. This demonstrated that the identity and importance of the drivers of multifunctionality,
174 such as land-use intensification and biodiversity, depended greatly on how multifunctionality was
175 defined. To extend this approach, realistic measures of how different stakeholders value each
176 ecosystem service are required.
177
178 It has been argued that the threshold approach is the most informative of the current approaches,
179 especially when metrics are calculated for multiple thresholds[9]. A notable benefit of the threshold
180 approach is that it avoids assumptions regarding the substitutability of functions and services that
181 the averaging approach does not. However, it does not reflect the significance of particular functions
182 or services, as it treats all functions passing an arbitrary threshold as equivalent. Furthermore,
183 threshold-based metrics are highly sensitive to the means of standardisation and the number of
184 functions included[22]. Specifically, the method of standardisation affects the mean and distribution of
185 function values, and achieving 100% multifunctionality becomes increasingly unlikely as the number
186 of functions increases[22]. Furthermore, different studies, using both averaging and threshold
187 approaches, include different numbers and sets of ecosystem functions, which are standardised
188 according to different local maxima[10,23,53]. This renders comparisons of multifunctionality measures
189 across studies extremely challenging[22]. The mixing of functions and services also means that many
190 multifunctionality measures are difficult to interpret from both a fundamental or applied
191 perspective.
192
193 A final issue is that multifunctionality is rarely measured at the large spatial scales relevant to most
194 management decisions: almost all multifunctionality measures have been calculated at the 'plot'
195 scale (<1ha). In some cases, the delivery of multiple ecosystem services is required at these small
196 scales, e.g. in smallholder subsistence farms, but landscape-level multifunctionality is often the
197 priority for land managers, e.g. when managing watersheds[54]. Initial investigations into the drivers of
198 landscape-level multifunctionality show that it is driven by factors other than those determining
199 local-scale multifunctionality, such as the spatial turnover in species composition[53], and the variety
200 and identity of different land uses and habitat types[37,55]. In land-management research there is a
201 plethora of frameworks for assessing patterns in landscape multifunctionality, which frequently
202 highlight the need to understand trade-offs and synergies between ecosystem services as key to
203 maximising landscape multifunctionality[46,56]. Although earlier attempts to measure landscape
204 multifunctionality (*sensu lato*) have been made[57], the frameworks of land-management research

205 tend to lack explicit procedures for quantitatively measuring overall landscape multifunctionality[11].
206 For example, the delivery of multiple individual services is described[6,49], or hotspot approaches are
207 used to identify locations where several services are at high supply, but not whether this supply
208 exceeds or falls short of demand. It may be possible to represent multifunctionality as the total
209 economic value of the ecosystem, but such approaches are demanding and typically fail to account
210 for certain ecosystem values (e.g. those of cultural ecosystem services), or to represent the non-
211 equivalence of ecosystem service values between stakeholder groups[58,59].

213 In summary, a lack of conceptual clarity in the definition of multifunctionality has led to
214 multifunctionality measures that are subjective and difficult to interpret. Accordingly, the use of
215 such measures could lead to erroneous conclusions about the drivers of ecosystem functioning and
216 to poor management decisions.

Redefining multifunctionality

220 We propose that studies should clearly differentiate between 1) measures of multifunctionality
221 including only ecosystem functions, which therefore constitute a metric of the overall performance
222 of an ecosystem, which we term ecosystem-function multifunctionality (hereafter *EF-
223 multifunctionality*), and 2) measures which include ecosystem services and where multifunctionality
224 is defined and valued from a human perspective, which we term ecosystem-service
225 multifunctionality (hereafter *ES-multifunctionality*). A key distinction between these measures is that
226 EF-multifunctionality attempts to objectively represent overall ecosystem functioning without any
227 value judgement regarding the desired level or types of functions, while ES-multifunctionality
228 represents the supply of ecosystem services relative to human demand. These two
229 multifunctionality types need to be calculated according to different procedures, which we outline
230 below (see also Boxes 1 and 2). Throughout the process of measuring multifunctionality, we
231 recommend the use of standardized definitions of ecosystem functions and services[60,61], which
232 would increase comparability between studies.

Ecosystem-function multifunctionality

236 A standardised approach to defining and measuring multifunctionality is desirable in fundamental
237 research on the drivers of ecosystem functioning, and for long-term monitoring of ecosystem
238 conditions. In the following section, we describe calculation methods for calculating EF-
239 multifunctionality that are designed to be as objective as possible and at the same time repeatable.
240 The first barrier to achieving standardised and comparable measures is that there is little consensus
241 on the definition of ecosystem functioning, and on what can be considered high levels of function[62].
242 A truly standardised and comparable measure of EF-multifunctionality is not likely to be possible
243 until ecologists resolve long-running debates regarding the nature of ecosystem function, including
244 whether states, rates and processes should all be considered functions. As a full discussion of this
245 topic is outside the scope of this article, we work here from the basis that ecosystem functioning
246 should ideally be defined solely on processes rates, i.e. those involving fluxes of energy and matter
247 between trophic levels and the environment, with high functioning being defined by fast rates. High
248 stocks of energy and matter (e.g. soil carbon stocks, algal biomass) can also be considered indicators
249 of process rates over the long term, as they represent the net balance of inputs and outputs.
250 However, care should be taken in interpreting them as they may either represent high rates of
251 accumulation or low rates of biological activity, and it is important to clearly justify why a high or low
252 stock indicates high or low functioning. Alternatives to this approach, in which ecosystem
253 functioning or multifunctionality is defined relative to specific or desired levels, immediately take the
254 measure outside of objective fundamental sciences and into the more subjective realm of ES-

255 multifunctionality (see below). This approach suggested in this section avoids such value
256 judgements.
257
258 The next step towards the development of standardised EF-multifunctionality measures is to assess
259 which variables represent independent aspects of ecosystem functioning. To date, many
260 multifunctionality metrics have attempted to represent overall ecosystem functioning by including
261 as many different types of functions as possible[3,23,26,28,53,63]. However, ecosystem functions are
262 numerous and interrelated via networks of interactions and shared drivers (e.g. those related to
263 nutrient cycling and productivity). Accordingly, EF-multifunctionality measures should avoid bias
264 caused by overweighting certain categories of function. As researchers will differ greatly in their
265 definitions of these subsets, we suggest that these subsets are defined as objectively as possible, by
266 applying a cluster analysis to all ecosystem function data, after first standardising the variables to
267 make them comparable (Fig. 1a).
268
269 Once the clusters are identified they can be used to define weightings in threshold-based
270 multifunctionality measures. In contrast to ES-multifunctionality measures (see below) there is no
271 particular level of each function which is desired by people, so we consider threshold-based
272 approaches[6] to be appropriate as long as each cluster is weighted equally in the EF-
273 multifunctionality measure, irrespective of the number of functions within each cluster. This will
274 prevent the overrepresentation of many similar functions. Prior to this analysis, a standardised
275 maximum for each function should be defined (e.g. using existing data) and used to place the
276 function data on a standardised scale, thus making studies comparable. As the indicator functions,
277 and the means of measuring them, are likely to differ according to ecosystem types, standardisation
278 should be performed at the level of major ecosystem types (e.g. grassland, forest, dryland, urban,
279 cropland, wetland, lake, river, coastal, or open ocean), or relative to the likely maximum potential
280 function given local conditions, if this can be determined. As certain clusters or functions may be of
281 particular interest we also suggest that users report results for individual functions and clusters
282 separately.
283
284 As the clustering method is sensitive to the identity of the functions used in the analysis, this process
285 will produce system-specific measures for the time being. However, as studies accumulate, certain
286 common groupings of functions are likely to become recognisable. This in turn, may allow us to
287 identify standard indicators of multifunctionality in the future, for which rapid and standardised
288 ecosystem assessments[64] can be developed. The identification of standard indicator functions and
289 EF-multifunctionality measures would be greatly accelerated by the collation and analysis of
290 ecosystem function data at a global level. To achieve a fully comprehensive and comparable
291 measure of multifunctionality, we need to evaluate how many, and which, functions are necessary
292 to measure to obtain a good representation of overall ecosystem functioning (i.e., the
293 dimensionality of ecosystem functioning). In such an initiative the dimensionality of ecosystem
294 functioning can be assessed by identifying associations between a fully comprehensive set of
295 ecosystem functions (e.g. with principal components analysis), measured across a very wide range of
296 conditions. Fundamental axes of ecosystem variation could then be identified and causes of
297 variation along these will become better understood, in a process similar to what has been achieved
298 for broad plant functional strategies, where fundamental axes of variation across plant species and
299 communities are broadly accepted[65].
300
301 Delivering a set of accurate, comparable and easily measured indicators of ecosystem function, that
302 have been validated across a wide range of conditions, is clearly a non-trivial task, yet it has the
303 potential to provide significant insight into the drivers of ecosystem functioning and to help in
304 identifying fundamental trade-offs and synergies between ecosystem functions. Such standardized
305 measures are not without precedent as they are being used to monitor spatio-temporal changes in

306 ecosystem functioning at continental scales worldwide[66], and they are roughly analogous to the use
307 of indicator taxa in conservation monitoring, or to the measurement of a few plant traits to
308 represent major axes of functional trait variation[65]. Furthermore, standard EF-multifunctionality
309 indicator measures could be linked to related schemes to monitor climate and biodiversity change
310 via 'essential variables'[67].
311
312 In the short-term, we advise a cautious approach to the use of EF-multifunctionality measures,
313 which should acknowledge the mathematical and conceptual sensitivity of these measures to the
314 functions included, and which is transparent in reporting any biases in selecting variables. We also
315 recommend reporting the degree of trade-off between functions (e.g. as a correlation matrix) and
316 the maximum EF-multifunctionality present within a study. Ideally, this should be related to a
317 theoretical or standardised maximum, so that cases where high EF-multifunctionality is impossible,
318 e.g. due to strong trade-offs between functions, are identified. Regardless of the wider property that
319 an EF-multifunctionality measure represents, it is imperative that researchers justify their choice of
320 ecosystem function measures and understand the implications of these choices in driving their
321 conclusions. We also recommend that EF-multifunctionality scores are compared to null
322 expectations, given their sensitivity to the form of standardisation and number of contributing
323 functions, and given that tools exist for their computation[22].
324
325 **Ecosystem-service multifunctionality**
326
327 As ecosystem services are defined in relation to human needs, the definition and measurement of
328 ES-multifunctionality requires a different approach. The first step is to define which ecosystem
329 services (including material, regulating and non-material relational values[68]) are desired, and the
330 level and scale at which they are to be delivered. This requires consulting stakeholders[69,70]. As
331 priorities differ depending on stakeholder identities, and local socio-economic and ecological factors,
332 a single ES-multifunctionality measure would not be globally meaningful. Instead, bespoke ES-
333 multifunctionality measures are needed to reflect the supply of ecosystem services relative to their
334 demand with respect to various groups and organisations (Box 2, Fig. 2). This should be done in a
335 two-stage process using social-science methodologies. First, the identity of important stakeholder
336 groups and the services they value are identified qualitatively (e.g. via interview and discourse),
337 before the weightings of these services are derived quantitatively (e.g. by deriving stated
338 preferences from stakeholder questionnaires in which the importance of different ecosystem
339 services are ranked on an ordinal scale[70]).
340
341 Once the main ecosystem services and their relative importance have been defined, the next step is
342 to describe the functional relationship between the supply of each service and the benefit delivered
343 in terms of a relevant measure of wellbeing (e.g. economic benefit, health, security or equity), which
344 we term the supply-benefit relationship. The threshold approach[9,18] is a particular case of this
345 relationship that assumes an abrupt shift from zero to full benefit at a particular level. Previous work
346 on ecosystem services has found that such relationships can take a wide range of forms, e.g.
347 threshold, asymptotic or linear. This emphasizes the need to construct ES-multifunctionality
348 measures in which the supply-benefit relationship is derived for each service[71] (Box 2, Fig. 2). We
349 suggest that many locally relevant, regulating services show a threshold relationship in which there
350 are definable safe levels (e.g. a safe maximum threshold for nitrate in drinking water), while
351 ecosystem services that operate at very large scales (e.g. climate regulation via carbon storage) can
352 show a linear relationship with benefits at local scales. Ecosystem services with direct economic
353 benefits, on the other hand, might show a 'threshold-plus' relationship, characterised by a break-
354 even point, beyond which increasing levels of a service deliver increasing benefits (e.g. there is a
355 minimum crop yield that will be profitable, beyond which further yields generate further profits, see
356 Appendix S2 for further examples). The supply-benefit relationship can be defined using a range of

357 techniques, many of which were developed in economics[69,71], and - where relevant - they may be
358 defined separately for different stakeholder groups. Where it is difficult to determine the supply-
359 benefit relationship, or it is uncertain, we suggest exploring the sensitivity of ES-multifunctionality
360 metrics to a range of possible relationships (see Example 2, Appendix S1).
361
362 As a next step, ecosystem services need to be quantified. The services described by stakeholders will
363 generally denote broad categories, so effort is required to convert these to quantifiable properties.
364 In certain cases, they can be measured directly, e.g. carbon stocks[72]. However, many other services
365 do not have generally applicable metrics, and so locally relevant indicators, ideally with direct links
366 to the final service, need to be identified. Furthermore, multiple indicators may be required in cases
367 where services have several components (Fig. 2, Example 2). Once identified and measured,
368 indicator variables should then be transformed to service values using mathematical transfer
369 functions that are appropriate for the function-service relationship[7,8] (see Example 2, Appendix S1).
370 Then, the standardised values can be multiplied by the stakeholder-derived weightings (see Box 2)
371 and finally be summed to generate ES-multifunctionality measures. With this method issues with
372 substitutability[9], and with applying the same supply-benefit relationship (e.g. a 50% threshold) to all
373 services, are largely avoided. Also, the preliminary assessment of stakeholder needs means that all
374 important services for each area should be included, thus providing a comprehensive measure of ES-
375 multifunctionality. This ensures that measures are comparable within a study, even where the
376 number of services differs.
377
378 Once ES-multifunctionality measures have been calculated, their relationship to biotic (e.g. the
379 presence of a keystone species) and abiotic (e.g. climate or land-use) drivers can be investigated for
380 a range of stakeholder groups (Fig. 3) and the resulting knowledge can inform landscape
381 management. For example, simulating changes in the most important drivers may allow for the
382 prediction of future changes in ES-multifunctionality to different stakeholder groups, or the costs
383 and benefits of different management actions. Such information is compatible with existing
384 environmental decision-making frameworks, such as the Driving Forces-Pressures-States-Impacts-
385 Responses (DPSIR) framework used by the European Environment Agency[73] or the Conceptual
386 Framework of the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES)[74]. These
387 recommended ES-multifunctionality measures advance upon existing approaches[50,51] by delivering
388 an integrated measure of the supply of ecosystem services relative to their demand from a wide
389 range of stakeholders, rather than simply indicating the supply of multiple ecosystem services[6], or
390 coarsely estimating their total value[59]. In addition to ES-multifunctionality, the response of individual
391 underlying services should also be reported for transparency and to allow individual practitioners to
392 assess the data. This can be summarised concisely in the form of flower diagrams and radar
393 charts[6, 51,75].
394
Landscape-scale multifunctionality
396
397 In the previous sections we assumed that multifunctionality is measured at small spatial scales
398 (often <1 ha). However, as mentioned earlier, high levels of ES-multifunctionality are often desired
399 at much larger scales (often >1 ha), where factors such as beta diversity, connectivity, and landscape
400 configuration may become important drivers of multifunctionality[37,53,76,77]. There have been previous
401 attempts to measure landscape multifunctionality within biodiversity-ecosystem function research,
402 where it has been quantified as the number of functions exceeding a threshold in at least one part of
403 a landscape, and also as the average of standardised function measures across a landscape[53,78].
404 These previous studies measured multifunctionality by aggregating properties of plot-level
405 measures, however, and thus were not able to consider spatial interactions between organisms and
406 landscape features, which can strongly influence some ecosystem functions, particularly in
407 heterogeneous and complex landscapes[45,76,75]. Where such interactions occur, simple extrapolation

408   of existing knowledge of the drivers of local-scale multifunctionality to larger scales is not
409   recommended as it is highly likely that whole landscape functioning is not equal to the sum of the
410   functioning of small landscape units. In this section we suggest possible approaches to address this
411   challenge and to quantify ES-multifunctionality at the landscape scale.
412
413   The first steps towards the measurement of landscape ES-multifunctionality are to ensure that the
414   landscape is divided into analytically manageable units, e.g. even-sized grid cells, or patches
415   undergoing uniform management, such as fields, which can then be used in upscaling calculations.
416   Next, appropriate scaling functions should be applied to each ecosystem service of interest to
417   calculate its overall level within the landscape (Fig. 5). For certain services, simple upscaling methods
418   - in which the supply of a service is estimated from the properties of each landscape unit and then
419   summed or averaged across the landscape - will be appropriate, e.g. carbon storage, which can be
420   estimated from simple local measures or remote-sensing proxies[72]. However, many services and
421   their underlying functions involve spatial exchanges of matter and organisms, e.g. nutrient leaching,
422   pollination services or pest control[75,76,79]. These will be strongly influenced by surrounding features,
423   making direct upscaling from local-level measures unreliable. Therefore, the quantification of such
424   services will require spatially explicit algorithms in which the levels of an ecosystem service in each
425   landscape unit are modified by features of the local environment. Finally, some important
426   ecosystem services are not observable at local scales at all and so require landscape-level
427   assessment, or estimation from the aggregated properties of smaller landscape units. Examples are
428   landscape beauty, habitat suitability for organisms with large range sizes (e.g. many charismatic
429   vertebrates) or landslip risk (Fig. 5). Ecosystem services can be attributed to these categories of
430   upscaling method by combining expert knowledge with quantitative assessment of which local level
431   services are influenced by surrounding features[75]. Such assessments could also provide the
432   algorithms required to upscale each function or service (e.g. from spatially-explicit statistical
433   models).
434
435   The next step in measuring landscape ES-multifunctionality is to define the supply-benefit
436   relationship spatially, i.e. to define the location and level required for each service. Certain services
437   may be required at very high levels, but only in certain locations (e.g. recreation, avalanche control),
438   while for others only their overall landscape level is important (e.g. carbon storage). This spatial
439   supply-benefit relationship should be defined by a range of stakeholders because they may differ in
440   their spatial pattern of demand[80]. For example, a landscape formed of small subsistence farms
441   requires multiple benefits in many landscape positions, while land belonging to a single owner (e.g. a
442   large private company or conservation charity) may require larger scale ES-multifunctionality, with
443   large areas dedicated to a small number of services. Once the spatial pattern of supply relative to
444   demand is determined for each service, landscape level ES-multifunctionality can be quantified as
445   described previously (Fig. 5).
446
447
448   **Future avenues**
449
450   Given the complexity and diversity of ecosystem functions and services, it is conceivable that the
451   framework presented here may require adaptation for certain circumstances. It is also clear that
452   several gaps in knowledge and data, e.g. the identity of the best indicators within clusters of related
453   ecosystem functions, or the spatial patterns of ecosystem-service benefits, need to be addressed
454   before EF- and ES-multifunctionality can be quantified with confidence. Temporal aspects also bring
455   further complexity to the measurement of multifunctionality, which may explain the paucity of
456   knowledge on this subject. Nevertheless, such aspects are essential for understanding the stability,
457   resistance and resilience of overall ecosystem performance and its long-term benefits for human
458   well-being. Time-series data give the potential to extend multifunctionality measures, e.g. by

quantifying the number of years in which an ecosystem had high levels of multiple functions, thus merging measures of stability[77,81] and multifunctionality[9,18] to give measures of multifunctional stability. Future linkages between ecological and socio-economic systems are also encouraged, and are possible through the extension of the framework presented here, e.g. by quantifying ES-multifunctionality using monetary or life-satisfaction[82] units.

Conclusions

Multifunctionality is a simple but nebulous concept with many potential applications. It is increasingly studied in fundamental biodiversity and ecosystem science, whilst also becoming a common objective for ecosystem management and landscape-scale policy. There is therefore a pressing need to define it clearly and to provide useful multifunctionality metrics. With careful consideration of the issues raised here, multifunctionality metrics will become well founded, thus giving them the potential to provide important insights in ecosystem science and to support environmental decision-making. The recommendations made in this article often require greater resources and effort than current approaches, and it is still unlikely that all can be implemented within a single study. However, data-intensive methods are becoming increasingly possible thanks to large collaborative projects[2-8] and data-sharing, opening the possibility to identify general indicators of ecosystem functions and services, which may then be applied widely. By focusing research efforts on well-designed sampling protocols that include the most relevant and easy-to-measure functions and services, we can further accelerate this process. Even before such protocols are devised, increased awareness of the issues covered here will help to prevent inappropriate conclusions from being drawn from multifunctionality studies. Producing new and more reliable measures of EF- and ES-multifunctionality is not a trivial challenge, but a highly worthwhile one, given their great potential to provide insight into whole ecosystem functioning and to guide ecosystem management in an era in which dwindling natural resources are placed under increasing pressure.

Acknowledgments

Author contributions

PM conceived the study and wrote the initial draft, which was developed and revised by all other authors. PM and FvdP designed and performed analyses.

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Box 1. Measurement of *ecosystem-function multifunctionality*

1. Using a cluster analysis of ecosystem function data, *n* clusters of closely related functions are identified and given equal weight.

2. EF-multifunctionality is then quantified according to the threshold method (see references 9 and 18 for details). Prior to this analysis function measures are standardised according to regionally standardised maxima for each ecosystem type.

3. Each cluster is then assigned equal weight in the threshold based measure (e.g. 1 one each) and functions within the cluster are weighted equally (e.g. 0.25 each if the cluster contains four functions). This avoids the overweighting of certain aspects of overall ecosystem functioning.

4. Alongside the overall measure of EF-multifunctionality, individual ecosystem function values, the response of individual clusters of interest, the maximum observed EF-multifunctionality and the degree of trade-off between functions should also be reported.

Example 1: Forest ecosystem function multifunctionality.

EF-multifunctionality was calculated using data collected in forests as part of the FunDivEUROPE project[51]. This dataset contains 21 ecosystem functions and services measured in 209 forest plots across six European countries. These plots were selected to differ in the diversity and composition of dominant tree species.

To calculate EF- multifunctionality from this data we first excluded variables which cannot be considered ecosystem functions (e.g. cultural service indicators such as bird diversity) and those which are not measures of the rates of ecosystem processes or major stocks of energy and matter (e.g. drought resistance). Next, we performed an agglomerative -cluster analysis of the remaining functions and found that four clusters was the appropriate number (see tutorial and Fig. 1a). The data were then scaled according to the maximum values observed across the whole dataset and EF-multifunctionality was calculated using a 50% threshold, where each cluster of ecosystem function had the same weight in the overall EF-multifunctionality measure. The resulting scores were then related to European region and the proportion of conifer trees (Fig. 1b). This showed that conifer cover promoted EF-multifunctionality (i.e. ecosystems with high levels of these four clusters of function in Fig. 1a) in some regions (e.g. Poland) but affected it negatively in others (e.g. Germany). The maximum and minimum values observed were 0.87 and 0 respectively, with a theoretical maximum of 1. See Appendix S1 for methodological details and a tutorial.
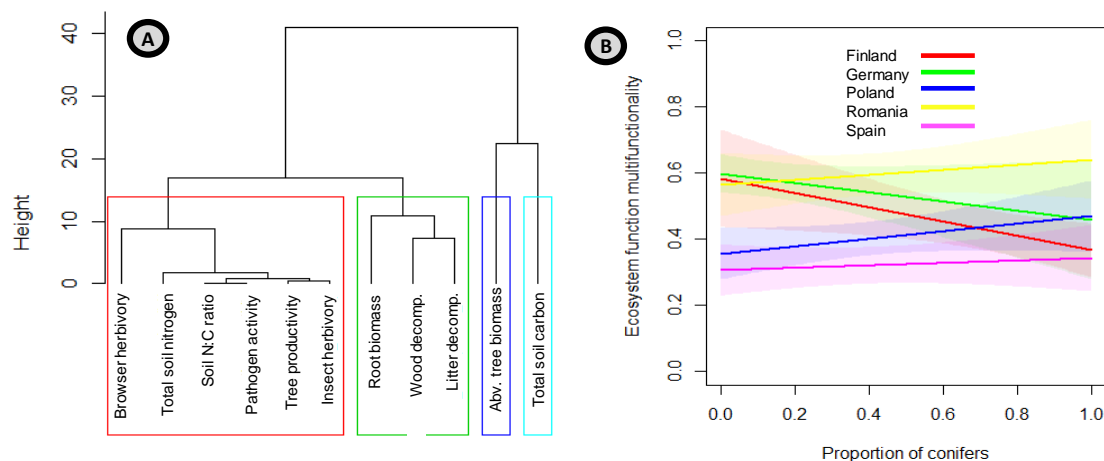

Figure 1. The quantification of EF multifunctionality in European Forests. (A) A dendrogram of ecosystem functions showing four main clusters, two related to fertility and turnover (red and green) and two related to the main stocks of energy and matter above- and belowground (blue and cyan) (B) The effect of forest region and conifer abundance on EF-multifunctionality. See Appendix S1 for details.

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Box 2. Measurement of *ecosystem-service multifunctionality*

1. First, important ecosystem services can be identified and weighted according to their relative importance, via consultation of a representative range of stakeholder groups within the focal area, thus ensuring that a full range of perspectives are represented (Fig 2).

2. The relationship between supply levels of an ecosystem service and the benefit it provides to humans (supply-benefit relationship) is also defined, e.g. via expert knowledge, economic methods, or stakeholder consultation (Fig 2).

3. Next, the levels of each service are measured using indicators or direct measurements (e.g. Fig. 2)

4. Indicator measures are then standardised to the same scale using the supply-benefit relationship (Fig 3).

5. Finally, the scaled measures can be multiplied by their stakeholder weightings and summed to quantify ES-multifunctionality. Stakeholder weightings should sum to 1 so that ES-multifunctionality metrics are comparable.
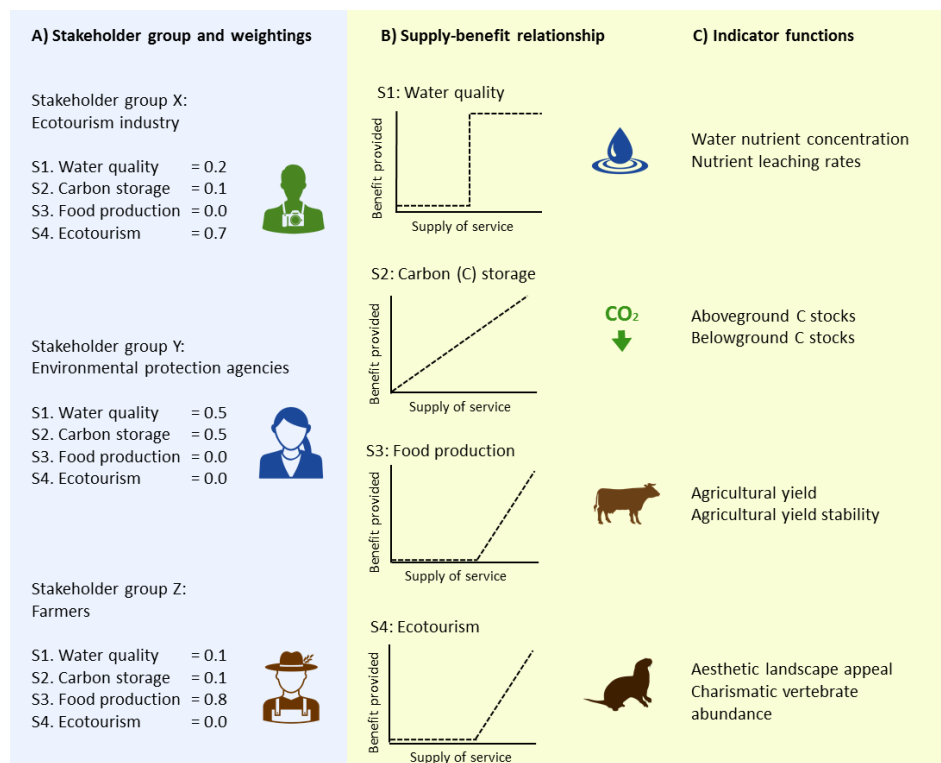

Figure. 2. Precursor stages to the measurement of ecosystem-service multifunctionality. Weighting of four example services according to different stakeholder perspectives (A). These services differ in the form of their supply-benefit relationships (B); for example, water is either legally safe to drink, or not; thus displaying threshold behaviour (S1), while at local level carbon storage has a linear relationship with global climate regulation. In contrast, a minimum amount of food production is required before agriculture becomes economically viable (S3), and ecosystems need to be in reasonable condition to attract tourists (S4). The values of these services increase linearly beyond these thresholds as greater profits are realised. Example indicator functions for each service are provided (C), and in for water quality these need to be transformed to a negative scale (i.e. high nutrient concentration is low water quality).
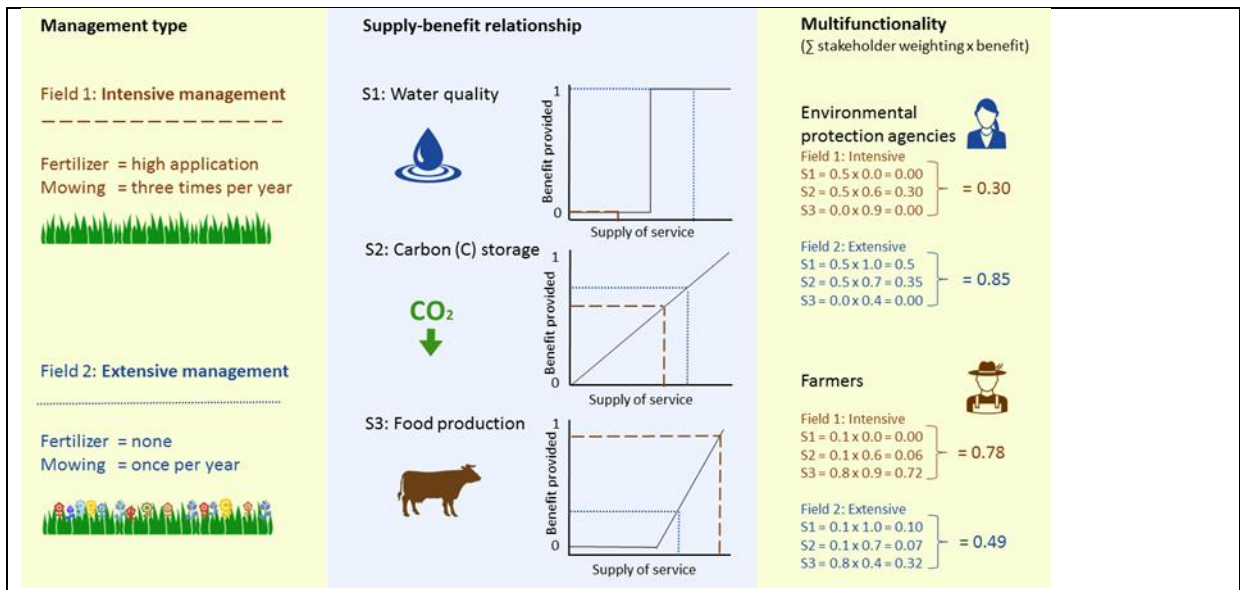
Management type

Field 1: **Intensive management**
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Fertilizer = high application
Mowing = three times per year

Supply-benefit relationship

S1: Water quality

Benefit provided (0 to 1)
Supply of service

S2: Carbon (C) storage

CO_2

Benefit provided (0 to 1)
Supply of service

Field 2: **Extensive management**
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Fertilizer = none
Mowing = once per year

S3: Food production

Benefit provided (0 to 1)
Supply of service

Multifunctionality
($\sum$ stakeholder weighting x benefit)

Environmental protection agencies

Field 1: Intensive
S1 = 0.5 x 0.0 = 0.00
S2 = 0.5 x 0.6 = 0.30   = 0.30
S3 = 0.0 x 0.9 = 0.00

Field 2: Extensive
S1 = 0.5 x 1.0 = 0.5
S2 = 0.5 x 0.7 = 0.35   = 0.85
S3 = 0.0 x 0.4 = 0.00

Farmers

Field 1: Intensive
S1 = 0.1 x 0.0 = 0.00
S2 = 0.1 x 0.6 = 0.06   = 0.78
S3 = 0.8 x 0.9 = 0.72

Field 2: Extensive
S1 = 0.1 x 1.0 = 0.10
S2 = 0.1 x 0.7 = 0.07   = 0.49
S3 = 0.8 x 0.4 = 0.32

Figure 3. An example of how levels of ecosystem-service multifunctionality depend on stakeholder preferences and how they can be compared between ecosystems subject to differing management regimes.

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Example 2. Ecosystem service multifunctionality of forest ecosystems

We demonstrate the calculation of ES-multifunctionality by using the FunDivEUROPE forest data (see Example 1)[53]. The first step was to obtain estimates of different ecosystem service values from stakeholders. As such data was not available for FunDivEUROPE, we took values from a stakeholder consultation conducted in Germany[83], where one of the FunDiv regions is located. Here, three stakeholder groups, managers of public owned forests (Public), managers of private owned forests (Private) and environmental organisations (Environmental group) stated differing priorities for four primary ecosystem services: timber production, biodiversity conservation, water supply and carbon sequestration. Timber production was given greater priority by the public and private groups. To represent these services with quantifiable measures we selected 1-3 indicator variables for each service from the 21 service and function variables available, and weighted them according to the stated stakeholder preferences. Each indicator variable was scaled relative to local or continental maximum and minimum values in a manner relevant to the demand of the ecosystem service (e.g. biodiversity relative to local maxima and carbon relative to continental maxima). As data for supply-benefit relationships were not available, we tested the sensitivity of ES-multifunctionality measures to a range of these relationships: linear, a 50% threshold, and 25, 50 and 75% threshold plus relationships. See Appendix S1 for details and a tutorial.

We found that a positive relationship between conifer abundance and ES-multifunctionality in German forests is broadly consistent across scenarios (Fig. 4). However, the slope of this relationship depended on stakeholder identity and the particular supply-benefit relationship. These differences are great enough to drive management decisions. For example, conifer planting would boost multifunctionality to public and private owners in the case of a 50% threshold-plus relationship, but would not promote multifunctionality from the perspective of the environmental organisations. This demonstrates the importance of using appropriate stakeholder weightings and supply-benefit relationships in ES-multifunctionality measures.
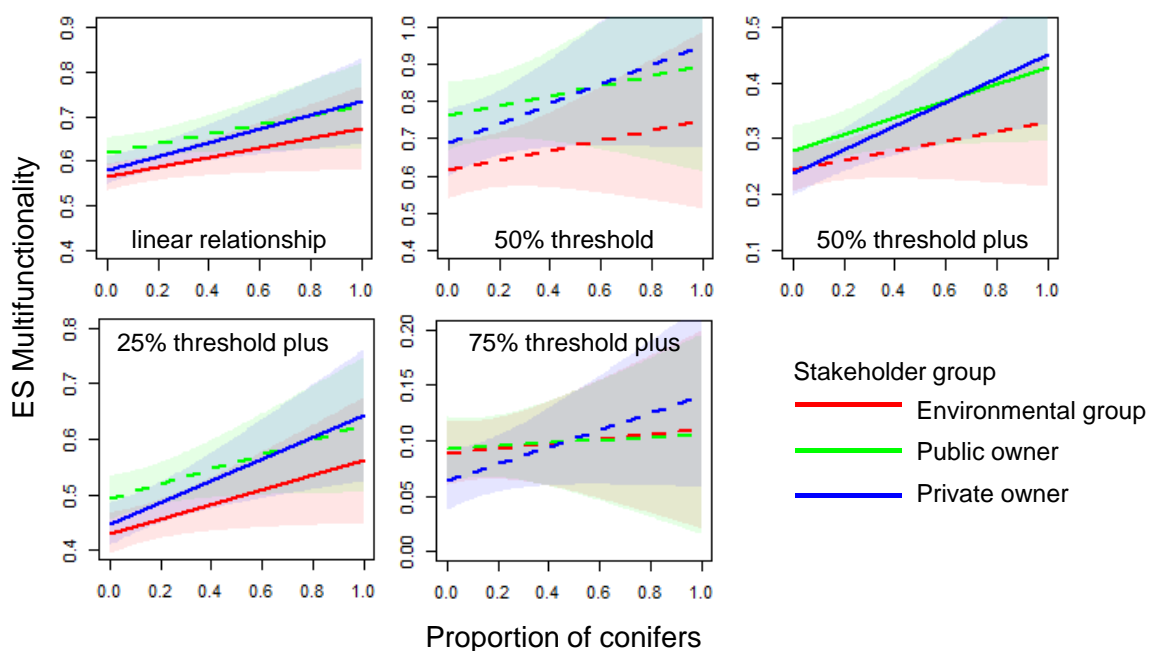

Figure 4. Dependency of ecosystem service multifunctionality on the supply-benefit relationship, stakeholder preferences and conifer abundance in German forests. Dashed lines indicate non-significant slopes ($p>0.05$) and solid lines significant slopes ($p<0.05$). Note the wide range in absolute ES multifunctionality values between the different supply-benefit relationships.
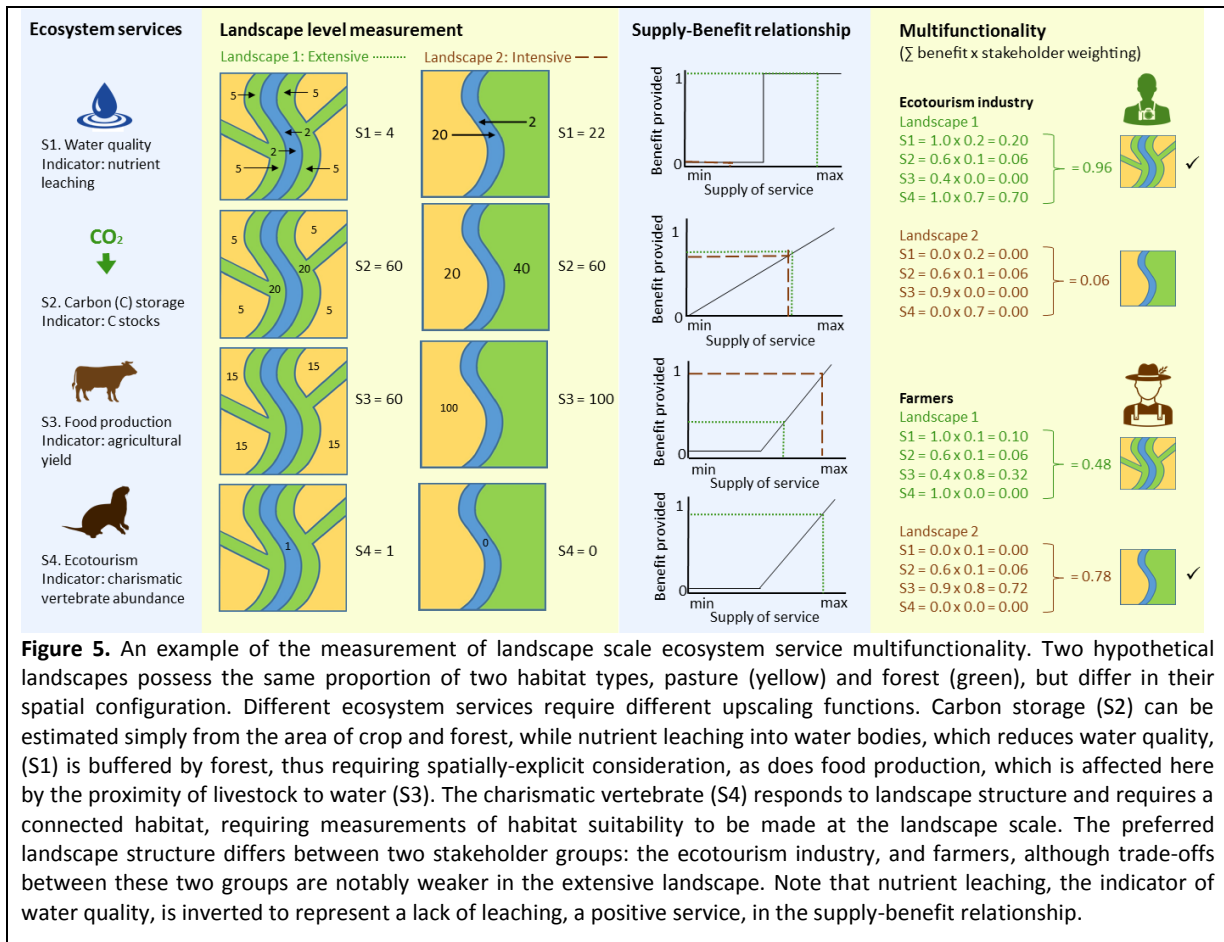
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Figure 5. An example of the measurement of landscape scale ecosystem service multifunctionality. Two hypothetical landscapes possess the same proportion of two habitat types, pasture (yellow) and forest (green), but differ in their spatial configuration. Different ecosystem services require different upscaling functions. Carbon storage (S2) can be estimated simply from the area of crop and forest, while nutrient leaching into water bodies, which reduces water quality, (S1) is buffered by forest, thus requiring spatially-explicit consideration, as does food production, which is affected here by the proximity of livestock to water (S3). The charismatic vertebrate (S4) responds to landscape structure and requires a connected habitat, requiring measurements of habitat suitability to be made at the landscape scale. The preferred landscape structure differs between two stakeholder groups: the ecotourism industry, and farmers, although trade-offs between these two groups are notably weaker in the extensive landscape. Note that nutrient leaching, the indicator of water quality, is inverted to represent a lack of leaching, a positive service, in the supply-benefit relationship.

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