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Mapping of ecosystem services: Missing links between purposes and procedures

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A B S T R A C T

The literature on ecosystem services mapping presents a diversity of procedures whose consistency might question the reliability of maps for decision-making. This study aims at analyzing the correspondence between the purpose of maps (e.g. land use planning) and the procedures used for mapping (e.g. benefit transfer, ecological transfer). Fifty scientific studies published between 2005 and 2012 were selected and analyzed according to 19 variables, applying independence tests over contingency tables, ANOVA and regression analysis. The results show that most studies declared a decision-making purpose (82%), which in 50% of the cases, was land use planning. Only few relationships were found between variables selected to describe the purpose of the maps and those selected to describe the mapping procedures. Thus for example, maps aimed at supporting land use planning did not include any level of stakeholder participation or scenario analysis, as it would have been expected given this purpose. Likewise, maps were based on either economic value or biophysical transfers, regardless of the spatial and temporal scales of mapping. This generally weak relation between map's purposes with the used procedures could explain the still restricted incidence of ES on decision-making by limiting the transmission, comparison and synthesis of results.

1. Introduction

Explicit mapping of ecosystem services (henceforth ES) is recognized as a key step for the implementation of the ecosystem services framework in decision-making (Daily and Matson, 2008; Daily et al., 2009; Burkhard et al., 2011; Seppelt et al., 2011; Hauck et al., 2013; Villamagna et al., 2013). In recent years a range of procedures have been proposed for ES mapping (see for instance, Troy and Wilson, 2006; Egoh et al., 2008; Naidoo et al., 2008; Willemen et al., 2008; Nelson et al., 2009; Tallis and Polasky, 2009; Seppelt et al., 2011; Crossman et al., 2013). Nelson et al. (2009) categorized these procedures under three general types. The first category consists of broad-scale assessments of multiple ES used to extrapolate a reduced number of value estimates, based on habitat types, regions, or the planet (see examples in Costanza et al., 1997; Troy and Wilson, 2006; Viglizzo and Frank, 2006; Turner et al., 2007). While simple, this transference of benefits has two restrictions: (i) it is based on the simplified assumption that every hectare of a given habitat type is of equal value – regardless of its quality, rarity, size, spatial configuration, neighboring land uses, proximity to ES beneficiaries and population centers, or the prevailing social practices and values; and (ii) it does not allow for analyses of service provision and changes in value under new scenarios.

The second type consists in modeling the provision of a single or few services in a small area using mechanistic models of ecosystem processes or fitting empirical responses to ecosystem variables ("ecological production functions") (see examples in Kaiser and Roumasset, 2002; Ricketts et al., 2004) that relate

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ES fluxes with local ecological variables. While this approach probably brings more reliable results than benefit transfer, it is generally restricted to provisioning services and tends to lack both the scope (number of services) and scale (geographic and temporal) for most policy matters (Nelson et al., 2009 and references therein). Finally, the third and most recent type is the social mapping of ES which emphasizes social perceptions, values and priorities over economic and ecological indicators. These methods commonly incorporate informants who are given a preliminary list of ES and then asked to associate values with landscape areas. An important issue that emerges from social mapping is the potential effect of "super-mappers" (*sensu* Ambrose-Oji and Pagella, 2012) since "when no limits are placed on the number of ES "markers" that can be placed on maps, some participants tend to place many more markers than others" (Ambrose-Oji and Pagella, 2012). This has noticeable implications in terms of the representativeness of the maps produced using these techniques. Examples of social mapping can be found in Raymond et al. (2009), Sherrouse et al. (2011), Fagerholm et al. (2012), and Plieninger et al. (2013).

Despite the important progresses in the development of mapping procedures, studies published in the last years (see for example, Seppelt et al., 2011; Martínez-Harms and Balvanera, 2012; Hauck et al., 2013; Crossman et al., 2013; Nahuelhual et al., 2013a) comment on the lack of consistency and adequacy between procedures and assessment purposes, which might question the reliability of maps for decision-making. For example, benefit transfer as an economic technique is applied for biodiversity conservation, for the design of payments for ecosystem services (PES) and for land use planning alike. In turn, land use planning is assumed to be equally supported by ecological assessments of functions and services, benefit transfer, social value mapping or mixed techniques (Nahuelhual et al., 2013a). The weak relation between a map's purpose and the attributes of the procedures used, could explain in part the still limited incidence of ES spatial assessment on decision-making (Villamagna et al., 2013).

In this context, the objective of this work was to analyze the correspondence between the purpose of maps (e.g. land use planning) and the procedures used for mapping (e.g. benefit transfer, ecological transfer), through a review of selected published studies that spatially assessed ES. The manuscript expects to contribute to the ES mapping literature by pointing at the main issues behind the insufficient consistency of mapping techniques, which still limits the transfer, comparison and synthesis of mapping results at different scales.

2. Methods

2.1. Search of scientific studies

Given the purpose of the study, the following search profile was applied to titles, keywords and abstracts: ("*ecosystem functions*" OR "*ecosystem service*" OR "*landscape service*" OR "*environmental service*" OR "*ecosystem good*" OR "*ecosystem benefit*" OR "*ecosystem services vulnerability*") AND ("*mapping*" OR "*map*" OR "*land use change*"). The selected material included original articles and key monographs obtained from SCOPUS database, published in English. The selection was narrowed to terrestrial ecosystems excluding seascapes. In this way, 99 studies published between 2000 (date of the first article retrieved by the search profile) and 2012 were preliminary selected. The final collection of studies for the analysis was obtained based on two criteria. The first one was directed to avoid the influence of earliest and mostly exploratory studies. Therefore, 2005 was chosen as the starting year for the following reasons. From this year on, there was an exponential growth in published studies on ES mapping (Martínez-Harms and Balvanera, 2012) and significant contributions were released which prompted the development and use of mapping procedures, such as for example "*The Economics of Ecosystems and Biodiversity*" (TEEB, 2010) and the "*Partnership for European Environmental Research*" (PEER) Report (Maes et al., 2011). Additionally, after 2005 specific software were developed and released such as Integrated Valuation of Environmental Services and Tradeoffs (InVEST) (Nelson et al., 2009; Tallis and Polasky, 2009), *Social Values for Ecosystem Services* (SolVES) (Sherrouse and Semmens, 2012) and Artificial Intelligence for Ecosystem Services (ARIES) (Villa et al., 2009). In this way, studies published between 2005 and 2012 captured 95% of the total studies retrieved between 2000 and 2012, while probably filtering too early mapping procedures. The second selection criterion was the development in the research article of at least one map (showed or only declared) of an ecosystem function, ecosystem service or benefit, in order to rule out papers that mapped other landscape features (e.g. land cover map; biodiversity map). Over this preliminary filtering, a random selection was conducted to finally choose 50 studies, which came from international indexed journals and international organization's key reports (i.e. UNEP). The sources comprised several aspects of ecology, environmental science and other natural sciences, as well as environmental modeling, economy and environmental policy.

2.2. Analysis of the selected studies and data base construction

The analysis of the studies was conducted around the following criteria: (i) the correspondence between a map's purpose and used procedures; (ii) the consistency among methodological variables of each procedure; and (iii) the relationships of mapping purposes and procedures with map quality. To achieve this, 19 variables were selected to characterize purpose, methodological procedures, and quality of maps, as detailed in Table 1. These 19 variables were created and agreed upon by the authors as those that best represented these three criteria.

The purpose of maps was characterized according to the author's statement of specific mapping objective/s, the type of decision the study declares to support, the geographic or political scale of mapping, and the existence or not of a recognized private or public stakeholder need. Consistency of procedures was understood as the existence of association patterns between different methodological variables (in contrast to their independent adoption). Quality of ES maps as a confidence tool for decision-making, was represented by two variables: (i) the "distance" between what is mapped and what is needed for informed decision-making (*distance to decision making*, Table 1), and (ii) the integrality of the mapping approach, which was understood as the level of adoption of a sequence of logical procedures, capable of connecting the ecosystem biophysical properties with capture and valuation of the consequent benefits (*cascade integration*, Table 1). This logical sequence implies mapping the different elements suggested by the ecosystem services framework as presented by Turner and Daily (2008) and backed up by "*The Economics of Ecosystems and Biodiversity*" (TEEB, 2010) and synthesized in the "*services cascade*" model (*sensu* Haines-Young and Potschin, 2010).

A data base was created where each row was one of the 50 selected study and the columns were the 19 variables described in Table 1. Relations among variables were explored using different types of univariate analysis according to variable types. Associations among categorical variables belonging to the same group or among groups of variables, were analyzed by the Fisher Exact Test on contingency tables. Relations among categorical vs. continuous (*mapped area*) or discrete (*number of components mapped*, *number of ES mapped*) variables were tested by comparing continuous or discrete variables among categories using ANOVA. Finally,

Table 1
Variables used for the characterization of 50 selected studies of ecosystem services mapping and distribution of percentages for categories or classes within each variable. Classes for continuous (*Mapped area*) and discrete variables (*Number of ES mapped*) were defined only for descriptive purposes.

Criteria	Variables		
	Name	Definitions and categories	Percentages
Map purpose	*Link to decision*	The study declares a specific mapping objective (i.e. private or public decision)	Yes: 82 No: 18
	Decision type	Type of decision the study declares to support: (a) Payment for ecosystem services (b) Land use planning (c) Sustainability assessment (d) Bio-centric conservation goals (e) Other goals	 (a) 4 (b) 50 (c) 14 (d) 26 (e) 6
	Institutional mapping scale	Geographic or political scale of mapping (a) Sub-continental to planetary (b) Country (c) Municipality or city (d) Region (e) District, province or state (f) Watershed (g) Protected area (h) Other institutional scales (different combinations of the other categories)	 (a) 12 (b)12 (c) 8 (d) 22 (e) 12 (f) 20 (g) 8 (h) 6
	Stakeholder demand	The study responds to a known private or public stakeholder need	Yes: 20 No: 80
Methodological procedures	*Mapped area*	Total area mapped (ha) (a) 0–100,000 ha; (b) 100.001–1,000,000 ha; (c) 1,000,001–10,000,000 ha; (d) 10,000,001–200,000,000 ha; (e) more than 200,000,000 ha.	(a) 24 (b) 26 (c) 10 (d) 20 (e) 20
	General mapping procedure	Main procedure used is: (a) Economic assessment (b) Ecological assessment: ecological evaluation of ecosystem functions and ES where these components are estimated as functions of ecosystem attributes or by means of "*proxies*" or biophysical indicators. (c) Mixed assessment: ecological production functions are combined with stakeholder valuations. (d) Stakeholder assessment: stakeholders participate in mapping ES flows and benefits.	 (a) 8 (b) 70 (c) 16 (d) 6
	Type of ecological procedure (N=46, corresponding to the sum of categories b, c and d within	If general mapping procedure is ecological, it uses: (a) Ecological production function: where ES are estimated as functions of structural attributes of ecosystem and landscapes or ecosystem processes (b) Ecological transfer: where mean values of ES obtained from other studies are transferred homogeneous cover units (e.g. biomes).	 (a) 58 (b) 42
	Stakeholder participation level	(a) Stakeholders are not consulted at any stage of ES mapping. (b) Stakeholders are consulted to validate results from mapping, or they are engaged in identification, valuation or mapping. c) Stakeholders participate in two or more of the stages listed in (b).	(a) 72 (b) 22 (c) 6
	Data type	(a) Quantitative (b) Qualitative (c) Both	(a) 68 (b) 8 (c) 24
	Map aggregation level	(a) Only ES independently (b) The study offers a map where a range of ES or benefits are aggregated	(a) 48 (b) 52
	Spatial scales integration	Mapping is conducted at different spatial scales	Yes: 16 No: 84
	Temporal scales	Type of temporal assessment: (a) No temporal assessment is conducted. (b) Retrospective analysis (e.g. ES mapping under past land use change). (c) Simulation of changes in ecosystem functions, or services or benefits under future scenarios (e.g. ES mapping under future land uses and covers). (d) Projection of current trends.	 (a) 74 (b) 4 (c) 8 (d) 14
	Number of components mapped	Number of ecosystem functions, ES or benefits, are mapped without a distinction among components (a) 1–2; (b) 3–6; (c) 7 or more	(a) 34 (b) 48 (c) 18
	Service concept	The concept mapped is: (a) Ecosystem service: it only considers local attributes (at the mapping scale, e.g. pixels, patch, or administrative unit). (b) Landscape service: it considers biophysical and social attributes of the spatial context. (c) Does not specify.	 (a) 74 (b) 22 (c) 4
	Number of ES mapped	The number of different ES types is based on the ES cascade criteria (Haines-Young and Potschin, 2010). Therefore, ecosystem functions and/or benefits derived from ES were not taken into account. A variable representing the total number of ES, ecosystem functions and benefits was discarded after the analysis of results because of its high redundancy with the *Number of ES mapped*. Discrete variable. (a) 0–1 ES types (b) 2–6 ES	 (a) 44 (b) 40

Table 1 (*continued*)

Criteria	Variables		
	Name	Definitions and categories	Percentages
		(c) >6 ES	(c) 16
	Thresholds	Mapping considers ecosystem thresholds	Yes: 8 No: 92
	Sustainability adjustments	Maps are adjusted by sustainability criteria (e.g. carrying capacity in case of recreation opportunities; ecological flows in case of water provision).	Yes: 2 No: 98
Map quality	*Distance to decision making*	Distance is given by the components that are mapped (e.g. only ecosystem functions, longer distance; vulnerability, shorter distance):	
		(a) Structural attributes ("*proxy*" of functions)	(a) 10
		(b) Ecosystem functions or intermediate ES	(b) 4
		(c) Final ES	(c) 38
		(d) Benefits or wellbeing indicators	(d) 16
		(e) Benefits, opportunity costs, *tradeoffs*, *hotspots*, *cold spots*, synergies, vulnerability, or risks and opportunities for conservation for multiple uses.	(e) 32
	Cascade integration	Cascade components that the mapping integrates:	
		(a) Only "*proxies*" of ecosystem functions or ES	(a) 52
		(b) Ecosystem functions	(b) 26
		(c) ES, weighted by social or economic values	(c) 14
		(d) (b) and (c)	(d) 4
		(e) (c) or (d) with benefits	(e) 4

relations among continuous and/or discrete variables were explored by linear regression analysis. All the analysis were performed in R software (R Core Team, 2013).

3. Results and discussion

3.1. Overall relationships between purpose, procedures and map quality

Variables describing purpose (e.g. *link to decision*, *decision type*, *stakeholder demand*) showed six significant relationships out of 52 (11%) with the variables describing the procedures (e.g. *mapped area*, *type of ecological procedures*, *temporal scales*) (Fig. 1). In turn, independence of most of the variables describing the procedures with the two variables accounting for map quality (*distance to decision making* and *cascade integration*) could not be significantly rejected (only one significant relationship among 26 tested relationships). In the following sections, significant associations among variables describing purpose, procedures and map quality are further analyzed. Results of ANOVAs are shown in Table A1, whereas results of Fisher Exact Tests on contingency tables are shown in Table A2, but because of space limitations, only some of these results are illustrated in Figs. 2 and 3. Results of regression analysis are presented within the text.

3.2. Relationships between purpose and procedure's variables

Purpose's variables showed associations with some procedure's variables such as *mapped area*, *stakeholder participation level*, consideration of *temporal scales*, *type of ecological procedure*, and *service concept* (Fig. 1). In turn, none of the four purpose's variables was significantly related with *general mapping procedure*.

According to ANOVA, mean *mapped area* was significantly lower for studies which declared to support a specific decision (*link to decision*=Yes) than studies which did not state a purpose (*link to decision*=No) (Fig. 1; AppendixTable A1), probably reflecting that most of decision-leaden objectives are more related to local than regional or global issues.

In turn, Fisher Exact Test revealed that studies conducted in response to a specific demand from a sector of society or from the government (*stakeholder demand*=Yes), were significantly associated to three procedure's variables: (i) *stakeholder participation*

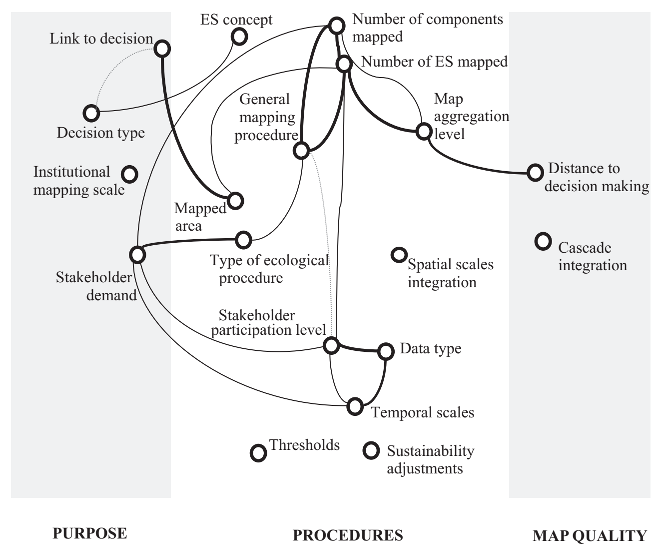

Fig. 1. Diagram of the associations between variables describing purpose, procedures, and map quality. The lines represent significant associations and their thickness represents the significance level (according to Fisher's Exact Test), in which the thick, intermediate and thin lines, correspond to $p<0.001$, $0.001<p<0.01$, and $p<0.05$, respectively. The trivial associations between *link to decision* and *decision type*, and between *general mapping approach* and *stakeholder participation level* are indicated with dotted lines and in gray. The arrows orientation indicates theoretical causality. See description of variables in Table 1.

level (Fig. 2a); (ii) the *temporal scale* of the maps (Fig. 2b); and (iii) the *type of ecological procedure* (result not shown). In the first case (i), when there was a stakeholder demand, the number of observed cases with medium level of participation (*stakeholder participation level*=b) was higer than expected by the independence hypothesis (examples of this association are provided by Lautenbach et al., 2011; Swetnam et al., 2011; Van Jaarsveld et al., 2012). In the second case (ii), when there was a stakeholder demand, the number of observed cases with *projection of current trends* (*temporal scale*=d) was higher than expected by independence. Studies like Kienast et al. (2009) and Reyers et al. (2009) are examples of the association between a response to a concrete *stakeholder demand* and *projections of current trends*. Studies which did not state a concrete stakeholder demand (*stakeholder demand*=No) did not consider *temporal scales* (*temporal scales*=a)

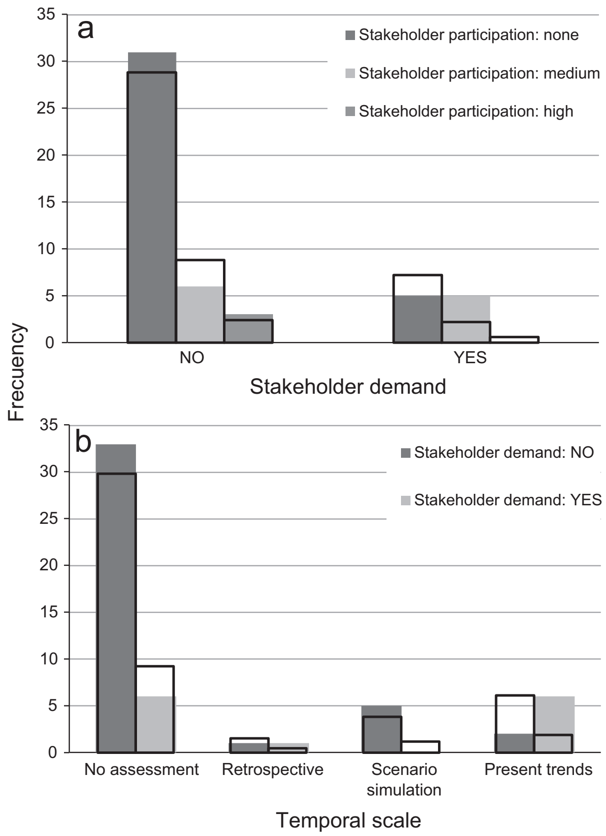

Fig. 2. Distribution of observed and expected frequencies (number of studies) under the hypothesis of independence between variables describing purpose and variables describing mapping procedures: (a) *stakeholder demand* frequencies along classes of *stakeholder participation level* and (b) *temporal scale* along classes of *stakeholder demand*. Filled and bordered bars represent the observed and expected frequencies, respectively.

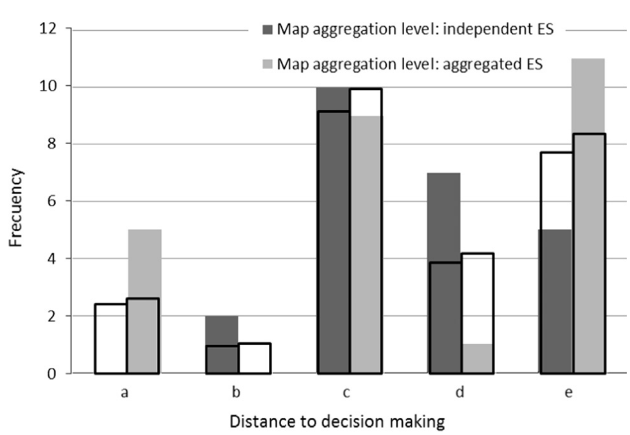

Fig. 3. Distribution of observed and expected frequencies under the hypothesis of independence between variables describing mapping procedures and variables describing map quality: *distance to decision making* vs. *map aggregation level.* Filled and bordered bars represent the observed and expected frequencies, respectively.

such as in the studies by Zhenghua et al. (2007), Pert et al. (2010), and Koschke et al. (2012). In the third case (iii), studies conducted in response to a concrete demand, showed a frequency of mapping based on ecological transfer (*type of ecological procedure*=b) twice the one expected by indepence, while mapping based on ecological production functions (*type of ecological procedure*=a) showed the opposite trend (result not shown).

The lack of correspondence between purpose's variables and the *general mapping procedure* suggests that similar techniques were applied to diverse purposes and, reciprocally, the same problem was analyzed through different mapping techniques. For instance, while benefit transfer was applied to generate maps for both biodiversity conservation and land use planning, the latter was supported through ecological assessments of functions and ES, benefit transfer, the participation of groups of interest or mixed procedures alike. However, if land use planning is seen as an integrative and participatory process, it should demand procedures with specific variables such as for example, high level of stakeholder engagement, consideration of ecological thresholds, and analysis of past and future scenarios (Randolph, 2004).

Of the observed associations between purpose's variables and those of the used procedures, the link between *stakeholder demand* and *stakeholder participation level* is worth noting. The studies by Raymond et al. (2009) and Reyers et al. (2009) show examples of this association. In Raymond et al. (2009), the authors present a procedure to map community values based on the concept of natural capital and ES to link local perception to a broader measure of environmental values at the landscape level. In Reyers et al. (2009), the authors aim to respond to beneficiaries and administrators questions, by mapping multiple ES and thus contributing to incorporate ES in land use planning at a local level.

From the decision-making point of view, the participation of groups of interest is deemed relevant not just for their capacity to give more realism to maps and their social validation, but also because the early engagement of these groups can, in a way, guarantee their commitment in posterior stages of ES assessments (Reed, 2008; Urgenson et al., 2013). In fact, the first stage of the ecosystem service framework (see Turner and Daily (2008) and Morse-Jones et al. (2011) for a decription), requires a detailed characterization of stakeholders' engagement. However, the results suggest that this consideration is still limited; of the reviewed studies, only 20% responded to a concrete stakeholder demand, and of these only 50% considered some degree of participation of interest groups (Fig. 3).

3.3. *Relationships between variables of the procedures*

Unlike the scarce relationships found between the purpose and procedure's variables, the results suggest a relatively higher number of relationships within the group of procedure' variables, such as those observed for *general mapping procedure vs.* the *number of components mapped* and the *number of ES mapped*, as well as those observed for *stakeholder participation level vs. number of ES mapped*, *data type* and *temporal scales* (Fig. 1). Therefore, while *general mapping procedure* and *stakeholder participation* seems to be conditioning other methodological variables, other variables describing methodological procedures like *spatial scales integration*, consideration of *thresholds* and *sustainability adjustments* did not show any significant relationship with the rest of variables.

Independence tests between pairs of categorical variables showed only some significant associations between procedure's variables. Among them, *general mapping procedure* was significantly associated with *mapped area*. In turn, *mapped area* was positively associated with *economic assessment*, where the frequency of studies conducted at larger extensions (*mapped area*=e) resulted higher than the expected by independence (result not shown). Chen et al. (2009), Deng et al. (2011) and UNEP *World Conservation Monitoring Centre* (2008) represent examples of economic approaches (particularly benefit transfer) for mapping of extensive areas (larger than 900 million hectares). The opposite tendency was observed for general mapping approaches based on mixed assessments and stakeholder participation, in which the frequency of observed cases for the largest areas resulted lower than expected. This association suggests a limited engagement of stakeholders at larger spatial scales, as well as relevance of this procedure for mapping of ES at small extensions (smaller than 5 million hectares), as illustrated by Raymond et al. (2009), Maynard et al. (2010), and Fagerholm et al. (2012). However, some specific associations were different from expected. Thus for example, it would have been expected that *mapping area*, *number of components mapped* and *number of ES mapped* differed among categories of *general mapping procedure*, but no significant differences were detected by ANOVA (Appendix Table A1).

The observed frequency of *stakeholder participation level* was lower than expected according to the independence hypothesis, both for the *ecological assessment* as well as for the *economic assessment*, but the opposite occurred with the *mixed assessment* and, logically, with the *stakeholder assessment*. In turn, the *type of ecological approach* had a negative association with medium and high ranks of *stakeholder participation level* (result not shown).

Due to the high correlation between *number of components mapped* and *number of ES mapped* ($R=0.84$, $R^2=0.73$, $p<0.001$), these variables showed similar relationships with other procedures' variables. Regarding the significant differences in *number of components mapped* and *number of ES mapped* along different *general mapping procedures*, it is noticeable that all studies based on stakeholder engagement mapped a higher number of components (ecosystem functions, services and/or benefits) than other *general mapping procedures* (e.g. Raymond et al., 2009; Maynard et al., 2010; Fagerholm et al., 2012), whereas for those based on an economic assessment, exactly the opposite happened (e.g. Chen et al., 2009; Baerenklau et al., 2010; Deng et al., 2011).

Regression analysis of *mapped area* on *number of components mapped* and *number of ES mapped* did not show any significant relationship ($p>0.05$, $R^2=0.003$ and 0.009, respectively).

The significant association observed between *temporal scales* and *data type* fundamentaly reflects a possitive relationship between the *projection of current trends* and the use of *qualitative data* (result not shown). Not surprisingly, the *stakeholder participation level* was associated to the use of qualitative data (*data type*=b); specifically, studies where stakeholders were consulted at different stages of ES evaluation and mapping (*stakeholder participation level*=c) showed a frequency of use of qualitative data higher than expected under the independence hypothesis (result not shown). By definition, participatory mapping not only requires the engagement of one or some stakeholder types, but also the participation of the widest array of stakeholder types as possible. Most stakeholders can better provide and understand qualitative than quantitative data, so this result seems to be reflecting that necessary leveling of different stakeholders' capacities for participatory mapping usually is more feasible by the utilization of qualitative than quantitative data.

3.4. Associations between procedure's variables and map quality

Distance to decision making showed a significant association with *map aggregation level.* Specifically, cases where structural variables ("*proxy*" of functions) (*distance to decision making*=a) as well as those where benefits, opportunity costs, and other components were mapped (*distance to decision making*=e), were based on aggregated ES maps (*map aggregation level*=a) with a higher frequency than expected by independence (Fig. 3). However, for intermediate levels of distance (*distance to decision making*=b to d), the opposite happened. Examples of the first case (*distance to decision making*=e; *map aggregation level*=b) can be found in Li et al. (2006), Bowker et al. (2008), Lavorel et al. (2011) and Busch et al. (2012), whereas examples of the second case (*distance to decision making*=a; *map aggregation level*=b) are found in Bailey et al. (2006), Metzger et al. (2006), and Egoh et al. (2008).

3.5. Overall consistency between purposes, procedures and map quality

The obtained results suggest that even while most of the revised studies declare that the maps produced are aimed at supporting a concrete decision (82% of the studies), significant associations between type of purpose and variables of the different procedures used were not verified. Likewise, similar procedures were used to generate maps aimed at supporting disparate purposes regarding decision-making. It is worth mentioning as well, that the dominant mapping procedure continues to be strongly ecologically-based (*general mapping procedure*=b; 70% of the cases) regardless of the purpose.

The studies which showed the highest consistency among purpose, procedures and map quality were Bryan et al. (2011) and Newton et al. (2012). Both studies display an effort for mapping ES and benefits using the expected procedures. This is further complemented in the case of Newton et al. (2012), by remarking different antagonisms, through a scenario prospection. However, despite being the most consistent works, none of them included thresholds or sustainability adjustments in order for maps to reflect the ES benefits that can be sustainably captured by society without compromising natural capital. Some recent studies have suggested the importance of considering adjustment of maps for sustainability criteria (Maes et al., 2012; Nahuelhual et al., 2013b).

Overall, the results suggest that different decisions (e.g. land use planning, payments for ecosystem services) require different general mapping techniques and distinct procedure's attributes. For example, ecological assessment alone generally lacks scope (number of ES) and scale (both geographic and temporal) to be able to address most questions related to decision-making and policy design (Nelson et al., 2009), and demand a high level of knowledge about ecological processes that define functions and ES fluxes. These limitations are being overcome by means of biophysical process models. These models not only make it possible to functionally integrate the main variables of a local ecosystem – that define their capacity to provide an ES – but also enable the consideration of the spatial context (landscape) that can modify that functional capacity of local ecosystems (Laterra et al., 2011). In turn, economic valuation methods have been criticized for their disconnection with the underlying biophysical processes, as for the possibility of them leading to social asymmetries in access to benefits derived from ecosystem functioning (Viglizzo et al., 2012). It is worth noting that the lack of relation with the biophysical base is a limitation that can also be shared by social value mapping that includes stakeholder participation, since these approaches focus on local perceptions on ES importance. Finally, the approaches associated to the participation of groups of interest, generally share with ecological assessment – at big scales – the lack of representativeness, and face the difficulty of an effective inclusion of all relevant stakeholders groups. Even so, the participation of groups of interest is considered as an appropriate approach to relate ecosystem functions to human wellbeing (Ananda and Herath, 2009).

In this context, tending to mixed procedures as decisions become more complex (e.g. from public protected areas creation to land use planning) seems coherent with the requirements of the ecosystem service framework (Turner and Daily, 2008).

4. Conclusions

The results obtained in this research are in line with other studies that recognize the still insufficient coherence of mapping procedures, which presents a problem for decision makers, as well as for researchers on the topic, by limiting results' credibility and reducing their comparability. This in turn, could explain at least in part the still limited incidence of ES approach in decision-making.

The results obtained in this study suggest that the mapping methods used are still unable to offer all the inputs required for the implementation of the ES approach due to the complexity of the task. This in turn supports the need for more integrated approaches and adapted to local scales, where most of the decisions related to ecosystems use and management take place. However, it is important to consider that even if complex decisions require more integration of the cascade elements and procedures, the high requirements of information and analysis techniques for ES mapping, complicate or narrow the possibility of achieving such integration levels. Nevertheless, it is important to acknowledge that ES maps have other purposes besides decision-making, such as their heuristic use in initiating discussions about solutions and alternatives, and their pedagogic use to explain to people the relevance of ES and biodiversity.

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Appendix A

See Tables A1 and A2.

Table A1
Mean values, standard errors, and significance of ANOVA tests performed on continuous (*mapped area*) or discrete (*number of components mapped*, *number of ES mapped*) variables along categories of different categorical variables.

	Mapped area (km^2)		Number of components mapped		Number of ES mapped	
	Mean	SE	Mean	SE	Mean	SE
Cascade integration						
p-value	0.74		0.89		0.81	
Categories						
(a) Only “proxies” of ecosystem functions or ES	13,130,000	(5,935,000)	4.5	(1.1)	2.7	(0.7)
(b) Ecosystem functions	1,890,000	(10,560,000)	6.2	(1.9)	4.2	(1.3)
(c) ES, weighted by social or economic values	220,000	(12,890,000)	4.4	(2.3)	3.1	(1.6)
(d) (b) and (c)	120,000	(22,210,000)	3.5	(4.0)	2.5	(2.8)
(e) (c) or (d) with benefits	1678	(22,210,000)	4.0	(4.0)	2.5	(2.8)
Data type						
p-value	0.998		0.453		0.315	
Categories						
(a) Quantitative	5,768,290	(4,868,676)	4.5	(0.9)	2.7	(0.6)
(b) Qualitative	49,635,297	(17,098,230)	6.3	(2.9)	4.5	(2.0)
(c) Both	1,858,331	(9,847,383)	5.7	(1.9)	3.9	(1.3)
Decision type						
p-value	0.541		0.382		0.485	
Categories						
(a) Payment for ecosystem services	7000	(21,410,000)	3.5	(4.0)	2.5	(2.9)
(b) Land use planning	1,540,000	(9,709,000)	5.2	(1.8)	3.5	(1.3)
(c) Sustainability assessment	2,240,000	(13,220,000)	1.6	(2.5)	0.0	(1.8)
(d) Bio-centric conservation goals	12,310,000	(7,819,000)	5.6	(3.4)	3.5	(1.0)
(e) Other goals	50,950,000	(18,060,000)	6.7	(1.5)	4.0	(2.4)
Distance to decision making						
p-value	0.525		0.448		0.232	
Categories						
(a) Structural attributes (“proxy” of functions)	3927	(13,350,000)	4.4	(2.4)	3.2	(1.6)
(b) Ecosystem functions or intermediate ES	1,402,927	(24,970,000)	1.5	(4.4)	5.9	(3.1)
(c) Final ES	8,896,927	(15,000,000)	4.7	(2.7)	3.3	(1.8)
(d) Benefits or wellbeing indicators	24,123,927	(17,480,000)	2.8	(3.0)	0.9	(2.1)
(e) Benefits, opportunity costs, tradeoffs, hotspots, cold spots, synergies, vulnerability, or risks and opportunities for conservation for multiple uses.	1,567,927	(15,290,000)	6.6	(2.7)	4.3	(1.9)
General mapping procedure						
p-value	0.098		**< 0.001**		**< 0.001**	
Categories						
(a) Economic assessment	41,910,000	(14,270,000)	2.5	(1.8)	0.3	(1.1)
(b) Ecological assessment: ecological evaluation of ecosystem functions and ES.	5,800,000	(15,090,000)	4.1	(1.9)	2.7	(1.2)
(c) Mixed assessment: ecological production functions are combined with stakeholder valuations.	100,000	(17,480,000)	3.3	(2.2)	2.1	(1.4)
(d) Stakeholder assessment: stakeholders participate in mapping ES flows and benefits.	20,000	(21,800,000)	20.3	(2.7)	14.7	(1.7)
Institutional mapping scale						
p-value	0.543		0.563		0.142	
Categories						
(a) Sub-continental to planetary	31,750,000	(12,210,000)	6.2	(2.2)	3.2	(1.4)
(b) Country	3,240,000	(17,270,000)	2.8	(3.1)	1.2	(2.0)
(c) Municipality or city	90,000	(19,310,000)	5.8	(3.4)	5.5	(2.3)
(d) Region	13,570,000	(15,180,000)	7.5	(2.7)	5.3	(1.8)

Table A1 (*continued*)

	Mapped area (km^2)		Number of components mapped		Number of ES mapped	
	Mean	SE	Mean	SE	Mean	SE
(e) District, province or state	480,000	(17,270,000)	3.2	(3.1)	2.2	(2.0)
(f) Watershed	200,000	(15,770,000)	4.5	(2.8)	3.3	(1.8)
(g) Protected area	390,000	(19,310,000)	2.0	(3.4)	0.5	(2.3)
(h) Other institutional scales (different combinations of the other categories)	1162	(21,150,000)	3.7	(3.8	0.7	(2.5)
Link to decision						
p-value	**0.001**		0.604		0.764	
Categories						
Yes	1,490,000	(10,260,000)	1.0	(2.0)	0.4	(1.4)
No	38,080,000	(9,388,000)	4.0	(1.8)	2.8	(1.3)
Map aggregation level						
p-value	0.159	**0.030**	< **0.001**			
Categories						
(a) Only ES independently	13,571,000	(8,372,000)	3.2	(1.4)	1.2	(0.9)
(b) The study offers a map where a range of ES or benefits are aggregated	1,601,000	(5,859,000)	6.4	(1.0)	4.9	(0.6)
Service concept						
p-value	0.474		0.953		0.804	
Categories						
(a) Ecosystem service: it only considers local attributes (at the mapping scale, e.g. pixels, patch, or administrative unit).	9,444,547	(4,937,946)	4.8	(0.9)	3.3	(0.6)
(b) Landscape service: it considers biophysical and social attributes of the spatial context.	1,635,036	(10,705,212)	4.7	(1.9)	3.0	(1.3)
(c) Does not specify.	5394	(21,805,380)	5.5	(3.9)	0.0	(2.7)
Spatial scales integration						
p-value	0.131		0.681		0.680	
Categories						
Yes	21,992,000	(11,290,000)	4.1	(2.1)	3.6	(1.4)
No	4,632,000	(4,561,000)	5.0	(0.8)	3.0	(0.6)
Stakeholder demand						
p-value	0.499		**0.016**		0.073	
Categories						
Yes	1,743,160	(10,556,197)	8.4	(0.8)	5.0	(1.3)
No	8,932,764	(4,768,804)	4.0	(1.8)	2.7	(0.6)
Stakeholder participation level						
p-value	0.725		0.138		**0.014**	
Categories						
(a) Stakeholders are not consulted in any stage of ES mapping.	9,620,617	(18,070,000)	3.9	(3.1)	2.2	(2)
(b) Stakeholders are consulted to validate results from mapping, or they are engaged in identification, valuation or mapping.	2,643,617	(19,570,000)	7.4	(3.4)	5.3	(2.3)
(c) Stakeholders participate in two or more of the stages listed in (b).	1617	(17,340,000)	6.7	(3.0)	6.3	(2.0)
Sustainability adjustments						
p-value	0.802		0.729		0.974	
Categories						
Yes	436	(30,217,475)	3.0	(5.4)	3.0	(3.8)
No	7,621,020	(4,316,782)	4.9	(0.8)	3.1	(0.5)
Temporal scales						
p-value	0.900		0.859		0.897	
Categories						
(a) No temporal assessment is conducted.	9,324,984.55	(4,996,137)	4.8	(0.9)	3.3	(0.6)
(b) Retrospective analysis	3218	(22,062,343)	−2.8	(3.9)	−1.3	(2.8)
(c) Simulation of changes in ecosystem functions, or services or benefits under future scenarios	1,106,629.23	(15,995,442)	1.2	(2.8)	−0.6	(2.0)
(d) Projection of current trends.	2,725,332.83	(13,374,973)	0.4	(2.4)	−1.0	(1.7)
Thresholds						
p-value	0.620		0.744		0.834	
Categories						
Yes	326,160	(15,573,563)	4.0	(2.8)	3.5	(2.0)
No	8,100,105.3	(4,449,590)	4.9	(0.8)	3.1	(0.5)

Table A2
p-Values of independence tests between pairs of categorical variables belonging to the same group or among groups of variables according to Fisher Exact Test on contingency tables.

	Cascade integration	Data type	Decision type	Distance to decision making	General mapping procedure	Instit. mapping scale	Link to decision	Map aggregation level	Service concept	Spatial scales integration	Stakeh. demand	Stakeh. particip. level	Sustain. Adjust.	Temporal scales	Thresholds	Type of ecological procedure
Cascade integration		0.855	0.583	0.285	0.441	0.308	0,549	0.092	0.594	0.512	0.561	0.824	0.080	0.396	0.471	0.665
Data type	0.855		**0.055**	0.552	0.246	0.318	0.302	0.801	0.462	0.999	0.392	**0.001**	0.999	**0.005**	0.999	0.609
Decision type	0.583	**0.055**		0.679	0.377	0.255	**0.019**	0.545	**0.020**	0.763	0.727	0.709	0.999	0.564	0.650	0.470
Distance to decision making	0.285	0.552	0.679		0.460	0.510	0.353	**0.005**	0.476	0.346	0.287	0.694	0.620	0.997	0.508	0.928
General mapping procedure	0.441	0.246	0.377	0.460		0.401	0.467	0,064	0.812	0.469	0.198	0.021	0.999	0.999	0.999	**0.011**
Institutional mapping scale	0.308	0.318	0.255	0.510	0.401		0.573	0.108	0.543	0.336	0.208	0.204	0.999	0.375	0.463	0.448
Link to decision	0.549	0.302	**0.019**	0.353	0.467	0.573		0.999	0.774	0.623	0.174	0.427	0.999	0.251	0.999	0.609
Map aggregation level	0.092	0.801	0.545	**0.005**	0.064	0.108	0.999		0.999	0.250	0.728	0.266	0.480	0.565	0.611	0.057
Service concept	0.594	0.462	**0.020**	0.476	0.812	0.543	0.774	0.999		0.999	0.793	0.798	0.260	0.054	0.342	0.841
Spatial scales integration	0.512	0.999	0.763	0.346	0.469	0.336	0.623	0.250	0.999		0.999	0.999	0.999	0.781	0.514	0.865
Stakeholder demand	0.561	0.392	0.727	0.287	0.198	0.208	0.174	0.728	0.793	0.999		**0.049**	0.999	**0.021**	0.571	**0.006**
Stakeholder participation level	0.824	**0.001**	0.709	0.694	**0.021**	0.204	0.427	0.266	0.798	0.999	**0.049**		0.999	0.017	0.659	0.863
Sustainability adjustments	0.080	0.999	0.999	0.620	0.999	0.999	0.999	0.480	0.260	0.999	0.999	0.999		0.060	0.999	0.999
Temporal scales	0.396	**0.005**	0.564	0.997	0.999	0.375	0.251	0.565	**0.054**	0.781	**0.021**	**0.017**	0.060		0.999	0.331
Thresholds	0.471	0.999	0.650	0.508	0.999	0.463	0.999	0.611	0.342	0.514	0.571	0.659	0.999	0.999		0.801
Type of ecological procedure	0.665	0.609	0.470	0.928	**0.011**	0.448	0.609	0.057	0.841	0.865	**0.006**	0.863	0.999	0.331	0.801	

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