Making research relevant?

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Making research relevant? Ecological methods and the ecosystem services framework

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Key Points:

- The ecosystem services framework has good industrial applications to decision making. Ecologists should understand what those uses are.
- The ecosystem services framework is not a theory of ecological interactions, or societal valuation, and should not be used that way.
- Many methodologies that combine empirical research with the ecosystem services framework produce data that is not useful to problem-solving.

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Abstract We examine some unexpected epistemological conflicts that arise at the interfaces between ecological science, the ecosystem services framework, policy, and industry. We use an example from our own research to motivate and illustrate our main arguments, while also reviewing standard approaches to ecological science using the ecosystem services framework. While we agree that the ecosystem services framework has benefits in its industrial applications because it may force economic decision makers to consider a broader range of costs and benefits than they would do otherwise, we find that many alignments of ecology with the ecosystem services framework are asking questions that are irrelevant to real-world applications, and generating data that does not serve real-world applications. We attempt to clarify why these problems arise and how to avoid them. We urge fellow ecologists to reflect on the kind of research that can lead to both scientific advances and applied relevance to society. In our view, traditional empirical approaches at landscape scales or with place-based emphases are necessary to provide applied knowledge for problem solving, which is needed once decision makers identify risks to ecosystem services. We conclude that the ecosystem services framework is a good policy tool when applied to decision-making contexts, but not a good theory either of social valuation or ecological interactions, and should not be treated as one.

Plain Language Summary The ecosystem services framework is a policy tool that aims to help industrial actors and economic decision makers to appreciate the true costs of environmental destruction. In industry standards that incorporate ecosystem services, decision makers should consider the harms to a wide range of actors who may suffer if the ecosystems that they depend on are disrupted by a project. When used this way, the ecosystem services framework appears to be good and effective. However, in the rush to show that their research is relevant to society, and to attract funding, many ecologists have tried to use the ecosystem services framework as if it were a theory of ecological interactions and ecosystem functioning. We show how ecologists, including ourselves, have developed "hybrid" methods that mix traditional ecological science with an ecosystem services perspective. We have come to the conclusion, however, that these approaches often answer questions that are not relevant to applied contexts, and provide data that are not useful for solving real-world problems. These possible mistakes can be avoided, and we discuss approaches to make ecology useful to society without misusing the ecosystem services framework

1. Introduction

The ecosystem services framework has successfully become a major organizing rubric intended to translate ecological research into terms comprehensible to policy makers. Firmly established by the Millennium Ecosystem Assessment [*Millenium Ecosystem Assessment (MEA)*, 2007], ecosystem services represent groups of ecosystem processes as contributing to normative outcomes for humanity that can be given instrumental market values. Some conservationists and others are already protesting the primacy of instrumental values, especially economic valuation of nature according to its services to humanity, in conservation discourse [e.g., *Büscher et al*., 2012; *Tallis et al*., 2014; *Chan et al*., 2016]. However, the ecosystem services framework has been adopted by major policy and funding instruments (e.g. G8 + 5's TEEB study

http://ec.europa.eu/environment/nature/biodiversity/economics/index_en.htm; the WAVES Partnership (wavespartnership.org); the EU2020 Biodiversity Strategy; the Intergovernmental Platform on Biodiversity & Ecosystem Services; the International Finance Corporation [*IFC*, 2012]) and is thus an important mechanism for making ecology relevant to society and decision makers. Providing measurements of the ecosystem processes underlying ecosystem services has become a major concern of ecological research.

The rapid emergence and dominance of the ecosystem services framework has transformed the interfaces between pure ecology research, applied ecology research, policy, and industry. These transformations, and the translation of data across theoretical assumptions, across scales, and across technologies of representation have not been given the critical analysis they require. Many ecologists position themselves to be relevant to the ecosystem services framework in order to attract funding. Our guiding question for this review is "what principles, methods and data *are* relevant to the ecosystem services framework?"

To motivate our approach in this essay we first draw on a failed experiment seeking to quantify ecosystem services. We borrow from science and technology studies (STS) and history of science a focus on the details of the actual practices of most ecologists using the ecosystem services framework. Our intended audience is also primarily ecologists. We thus focus on understanding practices and the knowledge that they produce, and on the interplay between theory and real-world feasibility and necessity. As will be seen from our analysis of the kind of research facilitated and framed by ecosystem services, and its applied uses, we do not believe that there are any best-practice recommendations to make for the ecosystem services framework. We recommend that ecologists follow other socioecological research frameworks.

2. Learning From Failure: Tree Recruitment in a Silvopastoral Savanna

Our case study is an experiment on tree recruitment, the habitat factors promoting it, and its contributions to ecosystem services. The study investigated carbon stocking and hydrological cycling services in a silvopastoral habitat in central Chile. We sought to experimentally manipulate tree recruitment, a key ecological process underlying ecosystem service production, and to produce an analysis of how these services were distributed. However, because no tree recruitment was observed, nothing could be concluded. We wondered what lessons to draw about how the experiment ought to have been framed, what methods should have been chosen, and what form of result would have been easier to obtain as well as useful.

Our field plots represented a low-density sampling strategy across a medium-sized area in community ecology terms (one 100 m^2 plot per ha, times 80 plots, see Figure 1). There was only one kind of tree observed at the site, *Acacia caven* [*Root-Bernstein and Jaksic*, 2015; *Root-Bernstein et al*., 2017]. By measuring where *Acacia caven* seeds were produced, we formed a map of "source," "upstream," or "supply" plots where seeds originate. We also measured functional traits and characteristics of plots where recruitment could take place (Figures 1 and 2), allowing us to potentially relate these factors to "sink," "downstream," or "demand" plots. Sheep are grazed on the site and were expected to be seed dispersers.

The failure of the study to produce either observational or experimental results—seedling establishment did not occur prior to the experiment starting or in any experimental condition—can be attributed to detection and measurement problems. High-resolution techniques using DNA genotyping or LiDAR might have better tracked and detected seed dispersal and establishment [*Falkowski et al*., 2009; *Cho et al*., 2012], but were not available to us due to cost. The sheep may have dispersed the seeds in a net outflow from the studied site. We also may have missed that seedling establishment was blocked by some unmeasured interactions with the rodent *Octodon degus* or European rabbits [*Fuentes et al*., 1986; *Fleury et al*., 2015]. A decadal time frame may have been necessary to assess fluctuation in establishment rate due to climatic variation [*Holmgren et al*., 2006; *de la Maza et al*., 2009].

We could have done a different project, one that did not attempt to combine the ecosystem services framework with a traditional ecological approach. For example, an ecological experiment in planting *Acacia caven* seeds or seedlings in different microsites to determine their rate of growth, would have narrowed down the possible reasons for failure of seedling establishment. Alternatively, we could have followed the ecosystem services framework more closely to map the stock of the ecosystem service outcomes in each year, and asked whether the rate of tree establishment predicts these outcomes. Framed in this way, we can already conclude that rate of tree establishment and its proxies are not good predictors of carbon storage and hydrological cycling outcomes.

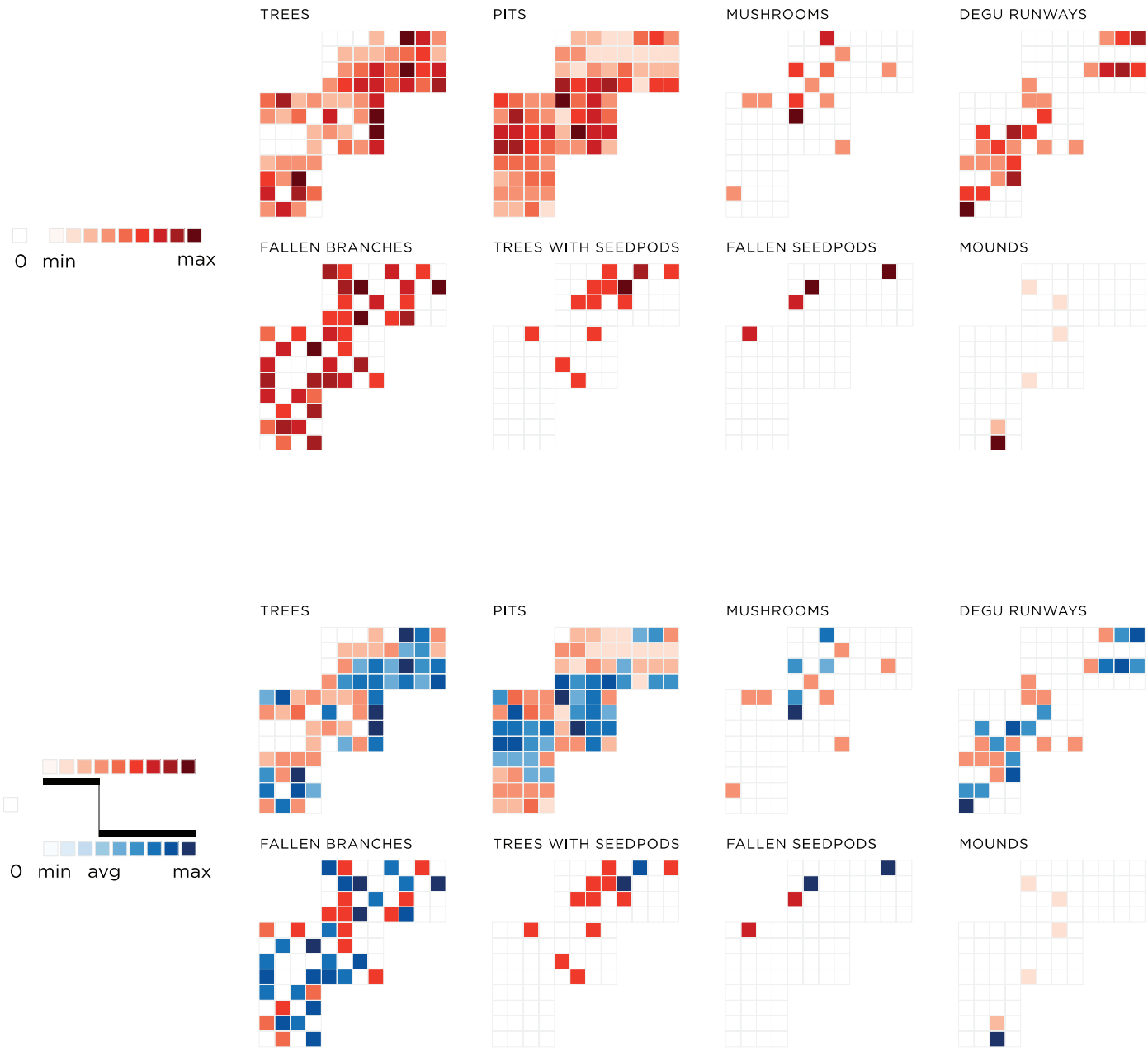

Figure 1. Two heat maps of collected variables in a savanna habitat in central Chile. Each square represents a 100 m^2 plot, located at the NE corner of a 1 ha grid. Top, each variable range is scaled from "min" to "max". Bottom, in an alternative view of the same data, blue values are greater than the median and red values are lesser than the median.

We believe that the difficulty with our research project resides critically in the way we attempted to combine questions about causality, interactions, spatial distributions, and ecosystem service production. Our combination of field ecology and ecosystem services does not merely emphasize the social and applied relevance of ecology research. It does something else too. Fitting into the ecosystem services framework creates important problems for the interpretation and meaningfulness of the ecological data produced. We review three such problems. First, we consider different ideas about what and how to measure when measuring the origins or causal interactions underlying ecosystem processes and ecosystem services. Second, we consider the implications for data collection, interpretation, and analysis when ecosystem processes and services are represented as either conceptual models or as spatial maps. Finally, we ask how policy makers and economic actors use the ecosystem services framework, and thus how to gather and present data to most effectively communicate with them.

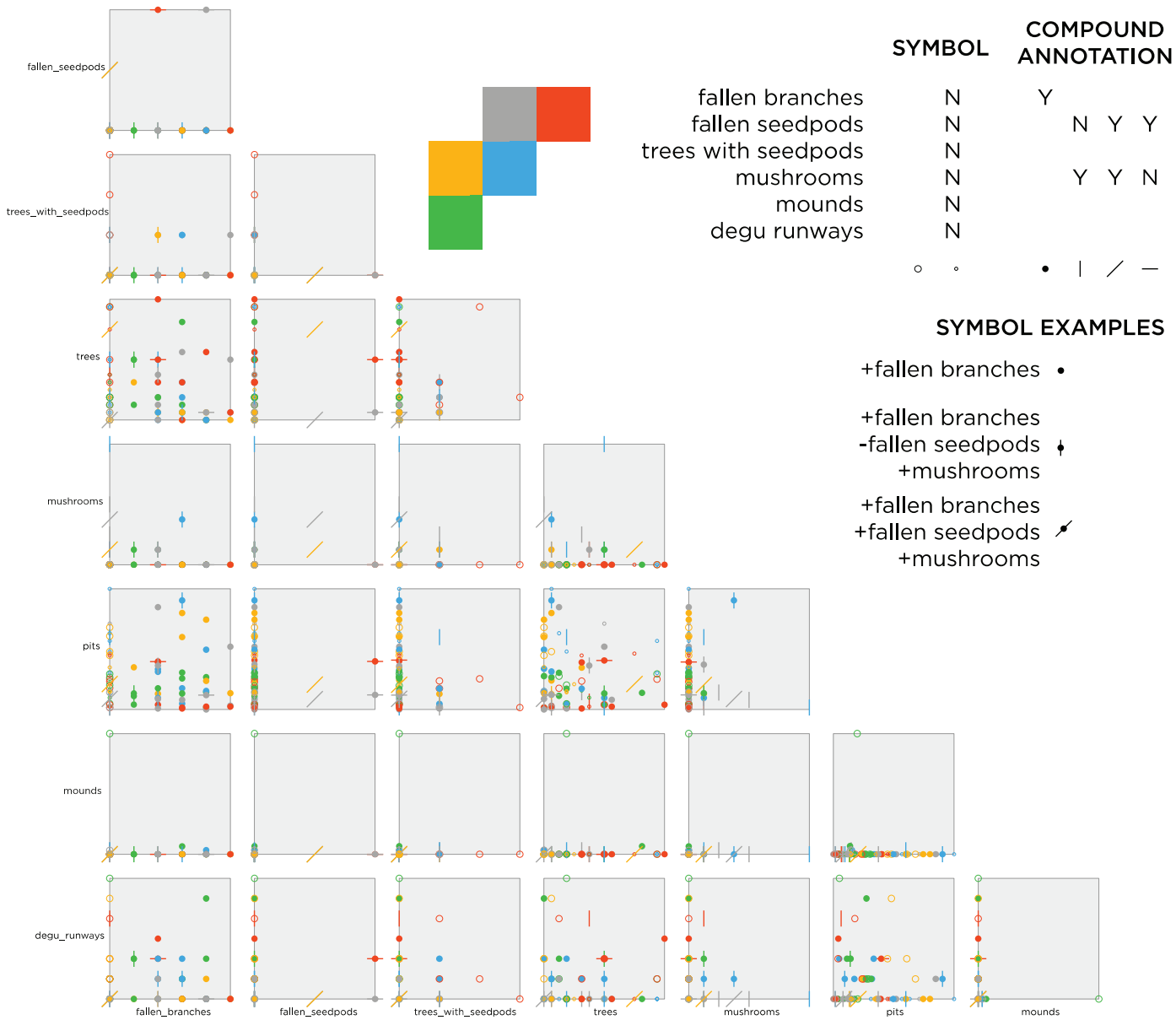

Figure 2. Co-occurrence of variables across plots. The key should be read vertically, and not all features are encoded in compound annotations. Variables measured included "pits" (formed by rabbits and birds), "degu runways" (made by the rodent *Octodon degus*), "mounds" (made by the rodent *Spalacopus cyanus*), and "fallen branches" (thorny branches of Acacia caven), and "mushrooms" (species not identified).

3. Origins, Causality, and Scales of Measurement for Ecosystem Processes and Services

Ecosystem services are the societally prefered outcomes provided to humanity through physical work, actions, or transformations resulting from ecosystem processes [*Costanza et al*., 1997; *Balmford et al*., 2002; *Kremen and Ostfeld*, 2005; *MEA*, 2007). Ecosystem processes are the biogeochemical, successional, and food-web assembly transformations and cycles characteristic of ecosystems. For example, erosion, cessation of erosion, and reversal of erosion through soil formation are all ecosystem processes, but erosion control is the normative ecosystem service. Since "ecosystem processes" is an ecological concept and "ecosystem services" is a policy concept, there is no one-to-one relation between them.

The identification of the units of origination of ecosystem processes is considered essential to underpin the ecosystem services concept [*Kremen*, 2005; *Guariguata and Balvanera*, 2009]. This idea is linked with the

literature on functional traits. The functional traits approach identifies key phenotypic traits and assesses how their abundance, richness, or diversity, usually at regional scales, correlate with ecosystem process or service outcomes [*Dıaz and Cabido*, 2001; *de Bello et al*., 2010; *Vandewalle et al*., 2010; *Lavorel*, 2013]. The functional trait approach disaggregates traits from their phenotypic species-level "packaging" to see how traits are distributed independently of species [*Eviner and Chapin*, 2003; *Westoby and Wright*, 2006]. Traditional (pre-ecosystem service framework) research on ecosystem processes claims that they are controlled by functional groups or guilds of species, or even by habitat types [*Lawton*, 1994; *Blondel*, 2003; *Rouget et al*., 2006; *De Deyn et al*., 2008; *Iglesias Briones et al*., 2009; *Luck et al*., 2009], in interaction with spatial heterogeneity and disturbance regimes [*Cardinale et al*., 2000]. Unlike functional traits approaches, functional group or guild-based research aggregates traits, by constructing multispecies functional groups ad hoc to reflect interpretations of the ecology at play [*Reynolds et al*., 2002; *Blondel*, 2003; *Voigt et al*., 2007].

These different approaches to what should be counted together are related to debates about the identification and explanation of patterns that have been at the core of ecology since its inception [*McCook*, 1994]. In addition, the argument between functional traits and ad hoc functional groups reflects a tension between the need for databases of standardized quantitative measures, which facilitate the creation of methods and maps of reference for and by policy makers, and a traditional community ecologist's approach focusing on scientific interpretation and the interactions between factors that shape particular ecosystems. For the policy maker, the independent functional trait is easily reified into a "plug-and-play" variable that can be employed across multiple planning contexts (see Section 5).

A different kind of measurement problem emerges when we consider what happens between the origin of an ecosystem process and its outcome. The analogy from economics is that ecosystems ("stocks") produce spatial and temporal "flows" of ecosystem services [*Bagstad et al*., 2013; *Burkhard et al*., 2013]. Flows are what are directly experienced by end-users and beneficiaries. The stock is a measure of accumulation, for example, the amount of a natural resource, while the flow is a measure of change over time, for example, inflow minus outflow. This view, however, is widely and variously reinterpreted by ecologists in an effort to make it work with ecological concepts of flow [e.g. *Boyd and Banzhaf*, 2007; *Fisher and Turner*, 2008; *Bagstad et al*., 2013].

Change over time of functional traits, if those are the sources of ecosystem "stocks," can be thought of as their spatial and temporal variation in expression or presence. In ecology, spatial and temporal variation in functional traits refers generally to gradients of traits over space, and to temporal changes in the expression or functionality of static traits. This is because the functional trait literature is mainly concerned with plants, which conveniently restrict the majority of their movement to the seed life-stage [*Chambers and MacMahon*, 1994; *Vander Wall*, 2001]. This makes the determination of *where* a function in an ecosystem process originates relatively simple. In this case "inflow minus outflow" does not refer to physical movement of the trait-bearing species.

There are specific ways in which temporal variation is usually considered in the functional trait literature. A popular perspective is the changes in function under environmental drivers such as climate change, formalized in the effect-and-response framework, where functional traits are divided between traits that predict a change in the plant community under climate change and traits that affect ecosystem processes under climate change [*Suding et al*., 2008; *Iglesias Briones et al*., 2009; *Laliberté et al*., 2010]. This correlative mapping approach fits well with perceived policy needs (see Section 4). However, how good functional traits are at predicting the spatial distribution of process outcomes depends on the temporal patterns of their functions. Some patterns are cyclical, for example, tree phenology controls tree productivity [*Richardson et al*., 2010; *Van Leeuwen et al*., 2013]. Many other ecosystem processes are strongly influenced by stochastic or highly variable inputs [e.g., *Holmgren et al*., 2006]. Such ecosystem processes often yield difficult to predict outputs, making a predictive/correlative approach like the effect and response framework poorly suited. Flows of ecosystem services are also difficult to understand as the arithmetic approximation "inputs minus outputs" when other factors create lags, bottlenecks, and changes in conversion rates.

Animal-mediated ecosystem processes may also have complicated spatial and temporal patterns, because animals may show both strong temporal and spatial variation, even within their range and distribution at any given period [*Grinnell*, 1917; *Haslett*, 1997, 2001; *Ellis*, 2003; *Lundberg and Moberg*, 2003; *Wilcove and*

Wikelski, 2008]. Animal-mediated ecosystem processes may also create problems in calculating or predicting ecosystem process outputs and flows.

Another tricky issue is the identification of the units across which to measure ecosystem process flow. In economics, flow is often an abstract concept. In ecology, however, the primacy of spatial dynamics and landscapes or habitats as structural units leads many researchers to think of flow as a spatial movement between an origin with a georeference and an outcome or beneficiary with a georeference [*Kontogianni et al*., 2010; *Syrbe and Walz*, 2012; *Potschin and Haines-Young*, 2013]. The identification of spatial units includes finding the right spatial scale, the areas where flows pass, and their directionality.

Ecosystem services tend to have characteristic scales. For example, the shade from one tree can have a direct effect on microsite temperature regulation under its canopy, while the contribution of the tree's growth to carbon storage is an extremely small contribution toward ameliorating a regional or global scale climate change phenomenon. Notably, the contributions from that individual tree to ameliorated climate change do not accrue as benefits directly or exclusively to the same individuals benefiting from the tree's shade, because the atmosphere is well-mixed. The choice of units thus depends partly on the ecosystem service in question.

The choice of units also depends on the methodology and the framing of the questions. Though the functional trait literature accepts that the gridded unit formed by the resolution of remote sensing is highly problematic for assessing ecosystem processes, it relies on those units. By contrast, literature that focuses not on functional traits but on particular processes is concerned with determining the relevant landscape- or ecosystem-specific boundaries of inputs and outputs, determined as a function of the physical landscape. Flows in a landscape context can be constrained by ecological or physical boundaries, or can change in rate and direction along corridors [*Cadenasso and Pickett*, 2001; *Laurance et al*., 2001; *Puth and Wilson*, 2001; *Cadenasso et al*., 2003a, 2003b; *McIntire*, 2004].

These ecological interpretations of flow complicate measuring it. If the definition of flow is accepted to be the economic "change in stock," then it is unnecessary to know where the stock came from, what route it took, its interactions with other ecosystem processes, or the correct scale at which to capture that movement. Only variables measuring the stock—or proxy data estimating or correlated to the stock—at two time points are required [e.g., *Naidoo et al*., 2008; *Bagstad et al*., 2013; *Gamfeldt et al*., 2013; *Costanza et al*., 2014]. Thus, the issues discussed above of functional traits versus ad hoc units, spatial scales, landscape barriers, ecosystem process interactions, or causality, can be seen as distractions if not actual misunderstandings of the ecosystem services framework. We will return to the assumption that an assessment of ecosystem service stocks by itself is a meaningful piece of information later on. Here, we emphasize that measurements of changes in stock ignore a large number of complications emphasized by traditional ecology, and consequently appear simpler to calculate.

4. Models and Maps of Ecosystem Processes and Services

Generally speaking, maps and models are used to communicate different types of information. You do not look at a map to understand underlying functional interactions, and you do not look at a flow diagram or algorithm structure to appreciate the spatial heterogeneity of an end product. In this section, we do not wish to formulate a taxonomy of when flows and fluxes should be mapped and when they should be modeled. Rather, we trace the assumptions and uses of those two technologies of representation in actual practice as applied to ecosystem services. In many cases, maps and models are difficult to inter-convert and involve very different assumptions.

We return to the idea of flows or fluxes of ecosystem processes. Energy flow through ecosystems has long been a key concept in ecology [*Teal*, 1962; *Odum*, 1968]. In this case, energy "flows" between abstract functional components of the ecosystem, for example, between trophic levels: flow occurs *within a model*. By contrast, water, air, and nutrients (ions) within them, clearly flow in a material and spatial sense, and thus can be represented both as a literal flow, across maps, or as an abstract flow, between categories [e.g., *Valiela et al*., 1978; *Cole and Rapp*, 1980; *Jordan et al*., 1983; *Hope et al*., 1994; *Rockström et al*., 1999; *Morales et al*., 2005]. Nutrients, once they leave the literal flow of fluids and gases, may be distributed across and cycled through biota, a process less amenable to being understood as literal flow and rather studied by stoichiometric and metabolic ecology and represented through models [*Reiners*, 1986; *Brown et al*., 2004]. Some

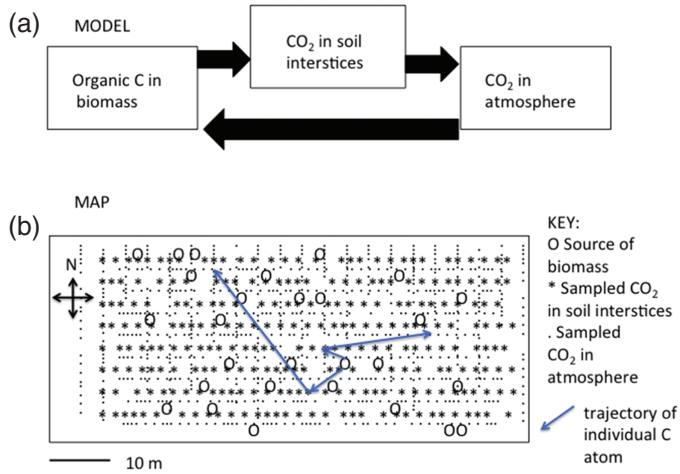

Figure 3. (a) The decomposition ecosystem process modeled as the movement of carbon atoms between different categories (top two arrows, bottom arrow represents respiration). Flow between categories is indicated with the large black arrows, and could be measured to give a rate of flow (flux) between categories. (b) A hypothetical example of the decomposition ecosystem process mapped as the positions of sampled decomposing biomass sources, CO_2 in soil interstices, and in the atmosphere. The flows of two individual carbon atoms are additionally mapped, showing their trajectories from source to two positions where they were sampled. It is not (hypothetically) impossible to map decomposition, but the map is not very legible or informative, and would be less so if all measured carbon atom flows were shown. Note that the ability to track the same molecules in their flow back to biomass through respiration is also extremely limited, due to the spatial extent potentially required to be shown.

researchers argue that ecosystem processes, flow via organismal movements [*Puth and Wilson*, 2001; *Lundberg and Moberg*, 2003], for example, mobile-agent-based ecosystem services [*Kremen et al*., 2007]. Here, the mapping aspect—the spatial distribution and distances across which links are made—is emphasized over the abstract categorical connections.

Many ecosystem processes lack a clear direction of flow in a literal spatial sense. If we consider maps to be spatially explicit, while models constitute abstract categories, it is not clear how the flow of decomposition can be *mapped*. We can measure the rate of decomposition of, say, a dead tree, but where does it go? The incoming decomposition agents (e.g., detritivores) each have trajectories and thus might be said to flow through the space around the dead tree. Similarly, the molecules in the tree are redistributed through nutrient cycling. However, the unclear characteristic scale and net velocity of these components make difficult a meaningful map of decomposition flow. On the other hand, it is easy to *model* the abstract flow of decomposition processes through ecological categories (e.g., movement of carbon through plants, soil organisms, soil organic matter, etc.). See Figure 3.

In a few cases, it is easy to both map and model the same process. One of the attractions of watersheds as study objects is that because they have a clearly bounded spatial unit determining a literal material and spatial flow, the map of the watershed translates easily to models of ecosystem processes in watersheds and vice versa. But even for ecosystem processes within watersheds such as the hydrological cycle or nutrient cycles, flux is not always easy to measure in practice [*Baldocchi et al*., 2001; *Schimel et al*., 2001]. A range of sophisticated sensors and techniques specific to detecting particles and their velocities in air or water have been developed to make increasingly accurate measures of their flux [e.g., *Baldocchi et al*., 2001; *Lomas et al*., 2013; *Moreira-Turcq et al*., 2013; *Gatti et al*., 2014].

For processes that depend on the movement of animals, for example, seed dispersal, the most sophisticated and high-resolution techniques have been developed by Jordano et al. [*Herrera et al*., 1994; *Jordano et al*., 2007; *González-Varo et al*., 2014]. They have used DNA barcoding techniques to trace the plant of origin of seeds dispersed by birds and small carnivores and deposited in their feces [*Godoy and Jordano*, 2001; *García et al*., 2007; *González-Varo et al*., 2014]. These methods allow researchers to determine the scales of the contributions of different dispersal functional groups to woody plant seed dispersal and forest regeneration [*García et al*., 2007; *Jordano et al*., 2007]. However, technical difficulties and sampling problems limit the ability to use a DNA barcoding approach for tracking the flow of wind-dispersed seeds,

pollen, spores, invertebrate life-stages, dead wood and other organic particles in streams, and many other biological elements that move.

Approaches to understanding ecosystem processes based on models have developed different methods. Model-based approaches emphasize that ecosystem processes interact and share steps and components [e.g., *Jordano and Herrera*, 1995; *Boumans et al*., 2002; *Borrett et al*., 2006; *Fath and Halnes*, 2007]. For example, the nitrogen cycle passes through parts of the decomposition process (mineralization), is moved across landscapes by the water cycle and hydrology [*Band et al*., 2001], and is influenced by the presence of certain habitats or land-cover classes, the distribution of nitrogen fixers due to successional processes [*Vitousek et al*., 1989], the grazing regime [*Schmitz*, 2008], and the stoichiometry of complementary nutrient cycles [*Reiners*, 1986; *Brown et al*., 2004]. The model-based approaches use a variety of analytical and representational formalisms, such as networks [*Fath and Patten*, 1999; *Ulanowicz*, 2004; *Borrett et al*., 2006; *Fath and Halnes*, 2007].

Environmental policy is often dependent on spatial zoning as a mode of implementation: land-use zonation, protected areas, and corridors, for example, are key conservation tools [e.g., *Kremen et al*., 2008; *Klein et al*., 2010]. This favors the creation of ecosystem service *maps*, and an emphasis on measuring the spatial distributions of stocks or indices of stocks, such as land-use, habitat type, or biodiversity [*Mace et al*., 2012]. So, one answer to why we would want to quantify an ecosystem service stock at a single point in time is because it is the easiest way to get half of the data required to make a map using the economic definition of flow.

5. Translation of Data Requirements Across Ecology, Policy, and Industry

Notions of the utility of ecosystem processes can be traced back to the thinkers of antiquity [for reviews see *Mooney et al*., 1997; *Gómez-Baggethun et al*., 2010]. From the 1970s onward, the first formulations of the ecosystem services concept had applied goals and were integrated into policy actions [*Mooney et al*., 1997; *Gómez-Baggethun et al*., 2010]. The current ecosystem service framework is used as an applied interface with economic operators, the industrial sector, and policy makers [e.g., *Tallis et al*., 2008; see Section *Introduction*]. Policy needs to justify itself with data, and science needs to justify itself to policy makers [see *Hoag*, 2011]. Thus, as we have seen above, the demands of mappability are incorporated into ecology research, static proxies are favored over dynamic interactions, outcomes are measured but causes are ignored. Here, we look at how these data are understood and used by their intended recipients. We first review the kinds of value that economic decisions trade-off between, and how ecosystem services fit into this set of values. We then ask how economic agents include ecosystem services in their decisions in practice.

According to ecosystem service proponents, a key step in making ecosystem services relevant is giving them an economic valuation or monetization [*Kremen and Ostfeld*, 2005; *Burkhard et al*., 2013]. There are numerous technical challenges associated with monetizing ecosystem services, including spatial and dynamic considerations [*Bagstad et al*., 2013], multifunctionality of ecosystem processes [*de Groot et al*., 2002; *Luck et al*., 2009; *Haslett et al*., 2010], and scale [e.g., *Guariguata and Balvanera*, 2009; *Pataki et al*., 2011]. In addition, ecosystem processes are highly context-dependent [*Womble and Doyle*, 2012; *Potschin and Haines-Young*, 2013]. However, due to the complexity of addressing these problems, many researchers find easier work-arounds. Examples include not attempting to relate processes to services even when both are measured, treating the process-to-service conversion as a black box [e.g., *Boumans et al*., 2002], or by assessing bundles of ecosystem services provided by habitats, ecosystems, or land uses, thus avoiding having to attribute value to each service independently [e.g., *Costanza et al*., 1997; *Schröter et al*., 2005; *García-Nieto et al*., 2013].

Sagoff [2011] argues that markets are efficient aggregators of relevant information. He points out that the assumption that the ecosystem service concept is necessary to make the "true" value of the environment clear implies inaccurately that markets lack information about the environment. Markets integrate the complex interactions between ecosystem processes through the perceptions of costs and benefits by and to decision makers. There are clearly problems with how markets integrate information, described as market failures, for example, short time horizons and externalities. An additional problem (for ecosystem service monetization) is that not all ecosystem services are in a market [*Fullerton and Stavins*, 1998]. Outside

the context of a specific market decision that implies potential changes in ecosystem services, ecosystem services have no inherent monetary value.

A quick review of how economics understands value illustrates the kinds of value that ecosystem services can have outside a market. As an analogy, a consultant might charge $50 an hour, but also does housework in their own home. Economics distinguishes between the marginal opportunity cost of 1 h of housework, which is $50 in this example, and the marginal benefit of an hour of doing housework, which has a monetarily undetermined and intrinsic benefit. Similarly, the housework of a stay-at-home husband has an intrinsic benefit but no opportunity cost unless he becomes a consultant, enters the market and has the option to be paid for work. In addition, one can also hire a maid to do the housework at a certain monetary cost, which should be distinguished from the opportunity cost and the intrinsic benefit described above. Rather than a single concept of value, economics would consider trade-offs between monetary costs, monetary benefits (income), opportunity costs, and monetarily undetermined benefits such as intrinsic benefits.

A forest, by its existence, provides ecosystem services that have no instrumental value in terms of monetary cost, monetary benefit, or opportunity cost. The ecosystem services of the forest, like housework, acquire instrumental value in terms of marginal costs and benefits, that is, when there is a decision to make about converting the forest into a marketable resource. The monetary cost of replacing the forest's services with a technological alternative (e.g., compensation) can be considered as part of a trade-off with other marginal costs and benefits. It is also critical to note that decision contexts determine the costs and benefits of the potential loss in ecosystem services, and thus the trade-off between them that economic actors will make. Factors that can affect costs and benefits include both supply and demand for ecosystem services. On the supply side, factors include the scale of the decision—does it affect just this forest or other forests as well; and history—how many forests are left. On the demand side, factors include the state of the environment, for example, air quality, and thus the need for certain ecosystem services; population growth—more individuals accrue more harm and more benefit; and economic growth—in purely economic terms richer individuals incur greater marginal costs and also value intrinsic benefits more via revealed preferences. Consequently, the value of a forest can only be assessed in light of these factors affecting economic decisions, and will change as these conditions change. The implication of this is that maps of the stock at a particular time of an ecosystem service [e.g., *Naidoo et al*., 2008; *Gamfeldt et al*., 2013] *tell us nothing about the geographical distribution of ecosystem service value*. To determine the monetary costs and benefits of maintaining a forest, we must assess potential changes to ecosystem services in scenarios with two specified features: a particular (e.g,. future) economic, population, development, and environmental context, and a particular decision being made. Only a small number of studies do this [e.g., *Bateman et al*., 2013; *Ruckelshaus et al*., 2015].

The imperative to monetize ecosystem services ignores that many ecosystem services are provided as "housework." It is simply incorrect that monetization is necessary to allow economic decision-making, as the housework example explains. Decision-making occurs over many forms of value.

The corporate and policy demands relating to the use of the ecosystem services framework reflect this insight from economic theory, in that they include ecosystem services in decision-making without, in the vast majority of cases, requiring monetization (or often, any kind of quantified data). This does not reflect the laziness, lack of environmental conscience, or intransigence of companies and governments. Rather, we argue, it reflects a theoretically supported and pragmatic understanding of how economic decisions are made. Here we review the main best-practice approaches and requirements for incorporating ecosystem services into the decisions of policy makers and industrial operators.

The International Finance Corporation, member of the World Bank and the largest development institution focusing on the private sector in developing countries, requires its clients to show that they have identified risks and impacts on priority ecosystem services, and to minimize or mitigate adverse impacts on those ecosystem services [*IFC*, 2012]. Their Guidance Notes do not stipulate a method [*IFC*, 2012]. The *Corporate Ecosystem Services Review*, which presents guidelines for best practice to corporations, recommends a qualitative assessment of which ecosystem services are relevant to a corporation, followed by a qualitative assessment of risks and opportunities [*Hansen et al*., 2012]. The World Business Council for Sustainable Development's *Guide to Corporate Ecosystem Valuation* notes, "As suggested in the methodology,

CEV [Corporate Ecosystem Valuation] should generally begin with a qualitative assessment to identify priority ecosystem services. Based on this information, a quantitative assessment can be undertaken, and finally a monetary valuation may be carried out for some or all of the ecosystem costs and benefits identified. There will, however, sometimes be situations where qualitative or quantitative assessments will suffice to inform the business decision to be made" [*Anonymous*, 2011, p. 25]. The most commonly used ecosystem valuation techniques recommended by the *Guide to Corporate Ecosystem Valuation* include revealed preference techniques (if a consultant gives up 1 h of working at a wage of $50 to spend an hour playing with his children, playing with his children is worth $50 to him); cost-based approaches in which one estimates the market cost to avoid damage or provide a replacement service; stated preference approaches such as "willingness to pay" questionnaire methods; and value or benefit transfer which refers to using existing data from previous valuations in similar situations [*Anonymous*, 2011]. None of these methods requires the level of precision in measures of flow discussed in Section 4, since even in the cost-based approach, the relevant cost to be calculated is that of the known anthropogenic and market activities that can avert or replace ecosystem service loss. So, if company A is concerned that the use of a certain chemical kills vultures which reduces detritivore services and risks poor outcomes for human or livestock health, then the costs they need to compare are the cost of switching chemicals to a safe one, versus the cost or risks associated with paying or not paying for sanitary control of uneaten carcasses in the landscape. To monetize the latter issue, company A might need to assess the quantity of large carcasses produced every year over a certain area. However, understanding the flow, stock, or distribution of the existing vulture detritivore services is not actually relevant.

The EU Biodiversity Strategy for 2020 (http://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm) more fully integrates explicit mapping and economic valuation of ecosystem services conceived of as "flows" with supplies and demands, stating, "Member States … will map and … assess the economic value of [ecosystem] services … by 2020." The Mapping and Assessment of Ecosystems and their Services framework, developed to address this requirement, recommends a set of "plug-and-play" variables as proxies for ecosystem services, which can be mapped by land use/ecosystem type. They then propose using natural capital accounting to economically value each mapped service, following the System of Environmental-Economic Accounting (SEEA) framework developed by the UN [*Gocheva and Petersen*, 2014]. The SEEA provides requirements (in SEEA-CF) for mapping assets and stocks of traditional extractive resources and pollutants, and has methodological recommendations (in SEEA-EEA) for assets including carbon and biodiversity, and flows of provisioning, regulation and maintenance, and cultural ecosystem services [*Gocheva and Petersen*, 2014]. As in the private sector as discussed above, the attribution of monetary value in the SEEA framework is done via assessing costs (e.g., avoided costs or restoration costs), revealed preferences or stated preferences [*Gocheva and Petersen*, 2014]. A Technical Report from 2014 describes a number of conceptual and technical problems with mapping ecosystem services by "plug-and-play" proxy data [*European Union (EU)*, 2014]. The rapid expansion of such a methodology outside the EU is unlikely, since many if not most countries lack the mappable "plug-and-play" data proxies making it feasible [e.g., *Naidoo et al*., 2008]. But most importantly, the valuation mechanisms applied to those maps are the same as those discussed above, which do not require any knowledge of the dynamics of the underlying ecological processes, their flow, stocks, or distribution.

Finally, Payment for Ecosystem Services and compensation (offset) markets involve particular stocks (e.g., carbon) or activities (e.g. a specific type of smallholder farming) intended to safeguard or offset biodiversity and its contributions to ecosystem services [e.g., *Ruckelshaus et al*., 2015]. There are many issues of proper accounting in REDD+ and other compensation markets (http://carbonmarketwatch.org/category/redd/; *Ituarte-Lima et al*., 2014; *Kollmuss et al*., 2008). There are multiple kinds of carbon compensation schemes in which forests and other habitats can be conserved or restored (LULUCF or land use, land-use change, and forestry projects) [*Kollmuss et al*., 2008]. Many carbon compensation schemes, however, do not accept LULUCF projects due to the difficulty of assessing their additionality (i.e., would the habitat definitely be destroyed if the project does not occur), as well as the difficulty in assessing the baseline carbon storage stock and the expected gain from the proposed project [*Kollmuss et al*., 2008]. According to the IPCC LUCF Good Practice Guidelines [*IPCC (Intergovernmental Panel on Climate Change)*, 2003], the carbon storage stock should be calculated using specified forestry parameters. Where species-level data are missing, the guide provides "default values." The calculations require, at a minimum, knowledge of the area of a particular habitat type to be included in the project, and an estimate of the change in land use from one time point to

another. Notably, this best practice is not based on correlations with functional trait data or dynamic drivers of ecosystem processes.

Ecological science, in short, is mainly irrelevant to how the ecosystem service framework is put into practice. The LULUCF carbon compensation schemes, with by far the most sophisticated accounting of an ecosystem service, are limited to carbon storage, estimate only stocks, and are often based on default values. Industrial operators only need to know that an ecosystem service exists in order to make all the economic and strategic calculations that they require. They do not require, for example, a quantified mapping of erosion control ecosystem services in order to realize that the risk of flooding represents a large potential cost. Once they consider this risk, traditional (pre-ecosystem services) terrain maps and hydrological, ecological, and geomorphological models and associated field practices inform place-based solutions. In other words, the ecosystem services framework goes to great lengths to provide economic decision makers with *the wrong kind of data*.

There are, of course, benefits of the ecosystem services framework. The incentive or requirement for economic and industrial operators to consult with various stakeholders, and to consider longer time horizons and a broader range of potential beneficiaries and victims when assessing risks, helps to correct market failures. These requirements are thus the most valuable aspect of best-practice ecosystem service guidelines.

6. Conclusions

We now return to our failed project that elicited these reflections. We set up the research from the perspective of a *model* but incorporated spatially explicit elements of a *map*; we emphasized measuring *interactions* between ecological processes and services rather than *stocks*; and we substituted measured data for plug-and-play functional trait approximations. Our hybrid methods both failed to obtain the attribution of causality available from an experiment, as well as the correlational requirements for prediction. In addition, our original goal was to encourage people to conserve and restore the habitat, but we were ignoring the market. A landowner wishing to decide if he should convert an espinal to an irrigated fruit orchard is likely to make this decision based on the current revenue from silvopastoralism versus the investment, outlays, and projected revenue from conversion to producing fruit, or the availability of a carbon compensation scheme. Adoption of one of the best-practice guidelines might force him to negotiate with stakeholders with appreciation for the "housework" being done by the habitat. However, neither the landowner nor these other stakeholders need maps or models of ecosystem service flows. A traditional experiment with no link to the ecosystem services framework would have produced information that could be applied to land management, while a standard academic assessment of ecosystem stocks would not have management applications.

Increasingly, ecologists are aligning their research with the ecosystem service framework. This involves forming various hybrid methods and framings that are somewhere between civil service data collection and scientific enquiry. As we have tried to show, these hybrid studies often tie themselves in knots trying to resolve conceptual and technical issues in ways that ultimately have no relevance to policy needs or economic decisions. Using the ecosystem services framework to drive scientific research strikes us as misguided. The ecosystem services framework is not a theory of ecosystem process dynamics, nor of valuation, and should not be treated that way.

If researchers want to provide knowledge useful to solving real problems, they should concentrate on building up expertise in the complex interactions of real systems at landscape scales; if they want to affect economic decisions about land use change, they should ask which real, site-specific factors constrain the market and nonmarket valuations of their preferred ecological states. There are numerous approaches to the place-based value-centered study of socioecological systems that predate and are conceptually independent of the ecosystem services framework. Although here we do not have scope to review them, we highlight the existence of fields including the socioecological systems paradigm and the biocultural approach within the conservation literature, human ecology, anthropology of the environment, multispecies ethnography, environmental history, and ecological economics [for some examples see citations above in relevant sections as well as, here, some reviews and some examples of papers combining social sciences and ecology in these ways: *Becker and Ostrom*, 1995; *Scoones*, 1999; *Dove*, 2006; *Folke*, 2006; *Nazarea*, 2006; *Shepard and Ramírez*, 2011; *Ogden et al.*, 2013; *Palsson et al.*, 2013; *Frascaroli et al.*, 2014;

Kelly et al., 2014; *Poe et al*., 2014; *Ehrich et al*., 2016; *Palmer et al*., 2015; *Orr et al*., 2015; *Root-Bernstein and Frascaroli*, 2016]. In addition, there is an increasing interest in natural history as a method for generating disciplinary inspiration, interdisciplinary shared methodologies, and public outreach [*Root-Bernstein*, 2016; http://naturalhistoriesproject.org, http://naturalhistorynetwork.org, https://esanaturalhistory.wordpress.com]. All of these disciplines and approaches have robust histories of producing insightful research with concrete applications, and we recommend any of them as superior approaches to linking ecology and society.

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