King, E., J. Cavender-Bares, P. Balvanera, T. H. Mwampamba, and S. Polasky. 2015. Trade-offs in ecosystem services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities. *Ecology and Society* **20**(3): 25. http://dx.doi.org/10.5751/ES-07822-200325

Insight, part of a Special Feature on Ecosystem Service Trade-offs across Global Contexts and Scales

Trade-offs in ecosystem services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities

Elizabeth King [1,2], *Jeannine Cavender-Bares* [3,4], *Patricia Balvanera* [5], *Tuyeni H. Mwampamba* [5] *and Stephen Polasky* [3,4,6]

ABSTRACT. In efforts to increase human well-being while maintaining the natural systems and processes upon which we depend, navigating the trade-offs that can arise between different ecosystem services is a profound challenge. We evaluated a recently developed simple analytic framework for assessing ecosystem service trade-offs, which characterizes such trade-offs in terms of their underlying biophysical constraints as well as divergences in stakeholders' values for the services in question. Through a workshop and subsequent discussions, we identified four different types of challenging situations under which the framework allows important insights to clarify the nature of stakeholder conflicts, obstacles to promoting more sustainable outcomes, and potential enabling factors to promote efficiency and sustainability of ecosystem service yields. We illustrated the framework's analytical steps by applying them to case studies representing three of the challenging situations. We explored the fourth challenging situation conceptually, using published literature for examples. We examined the potential utility and feasibility of using the framework as a participatory tool in resource management and conflict resolution. We concluded that the framework can be instrumental for promoting pluralism and insightful analysis of trade-offs. The insights offered here may be viewed as hypotheses to be tested and refined as additional unforeseen challenges and benefits are revealed as the framework is put into practice.

Key Words: *biophysical constraint; conflict; ecosystem service; human values; participatory tool; production possibility frontier; sustainability; trade-off; utility*

INTRODUCTION

Many approaches in sustainability science utilize the concept of ecosystem services to characterize the interdependence of human well-being and the environment. Ecosystems simultaneously generate multiple services, although it is generally not possible to manage ecosystems to simultaneously maximize all services, and as a result trade-offs could occur (López-Ridaura et al. 2002, Polasky et al. 2008, Smith et al. 2012). We use the term "trade-off" to describe what happens when a land use or management decision leads to an increase in one service and a decrease in some other service or services. Trade-offs appear to be particularly common between provisioning services, whose benefits are derived from extracting some food, fiber, or other material product from the ecosystem or from transforming the ecosystem to foster the supply of a material product, and regulating services, which decline when the integrity of ecological processes are compromised as provisioning services are gained (Bennett et al. 2009).

Trade-offs among ecosystem services can generate conflicts in natural resource management, development, and planning. Trade-offs can occur because of inherent constraints of the biological, ecological, and physical system (called "biophysical" hereafter). Conflicts may then arise as a result of divergent preferences held by different service users and other stakeholders (Martín-López et al. 2012). An example of a biophysical constraint is the negative impact that timber harvesting may have on a regulating service such as groundwater recharge. Conflicts could arise between groups dependent on groundwater versus those dependent on timber. Navigating such conflicts entails recognizing the nature of biophysically based trade-offs and reconciling divergent stakeholder preferences over those services. Lack of explicit recognition of differences among stakeholders' values is a crucial challenge in efforts to appreciate the implications of trade-offs and to identify viable and sustainable solutions. Today, ecosystem service assessment approaches are more commonly recognizing the need to separate biophysical constraints from values as sources of conflict, as reflected in slightly different ways by the dichotomies described by Daily et al. (2009; services and values), Mouchet et al. (2014; supply-side and demand-side associations), and Yahdjian et al. (2015; supply and demand trade-offs).

Cavender-Bares et al. (2015) present a sustainability framework (SF) that characterizes ecosystem service trade-offs in terms of two dimensions of ecosystem service conflicts: biophysical constraints and divergent values. The SF was largely developed from first principles in ecology and economics, with the aspiration that it would prove useful in analyzing and understanding conflicts that arise in real-world situations involving ecosystem service trade-offs. In the course of a series of workshops we (1) explored the usefulness of the SF approach by applying it to a range of hypothetical and empirical situations and (2) developed a prototypical methodology for its practical use. We developed a step-by-step process for using the SF in real-life case studies, and the insights gained from the exercise were synthesized to identify potential types of trade-off situations for which the framework could serve as a particularly useful analytical tool.

METHODS

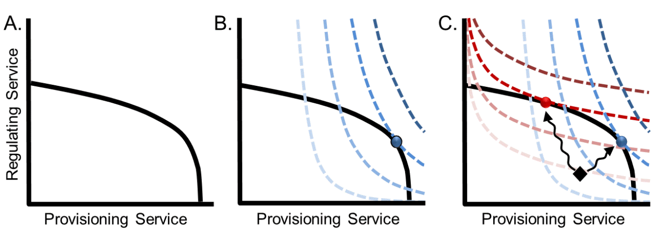
An environmental economic framework of ecosystem service trade-offs

The SF proposed by Cavender-Bares et al. (2015) in the lead article of this Special Feature, Ecosystem Service Trade-offs Across Global Contexts and Scales, characterizes trade-offs between two ecosystem services. The first "layer" of the SF uses the basic economic concept of a production possibility frontier (PPF). The

[1]Odum School of Ecology, University of Georgia, [2]Warnell School of Forestry and Natural Resources, University of Georgia, [3]Department of Ecology, Evolution and Behavior, University of Minnesota, [4]Institute on Environment, University of Minnesota, [5]Instituto de Investigaciones en Ecosistemas y Sustentabilidad, Universidad Nacional Autónoma de México, [6]Department of Applied Economics, University of Minnesota

PPF indicates the maximum amount of service B that can possibly be produced for each level of production of service A (Fig. 1A; Bator 1957). In resource economics, this has become an increasingly common tool for describing trade-offs between ecosystem services (Polasky et al. 2008, Kline and Mazzotta 2012, Smith et al. 2012). Cavender-Bares et al. (2015) illustrated that in the case of ecological processes and renewable resources, the frontier is the equilibrium solution to a system of equations representing ecological interactions that affect the yield of the two services in question. It would also be possible for the axes to represent bundles of services that are assessed using a synthetic metric, indicator, or proxy. For instance, percent forest cover may be positively correlated with water quality, carbon sequestration, and maintenance of biodiversity. The model does not necessitate monetary valuations of ecosystem services. Any quantitative measure of an ecosystem service can be plotted versus another, based on the theoretical or empirically observed relationship governing their joint production. It is important to recognize that management practices and technology shape the PPF. Exploring how technology or management regimes can change the PPF is indeed one of the ways we believe this framework can be useful in navigating trade-offs.

Fig. 1. Trade-off diagrams representing the basic steps of the analytical framework of Cavender-Bares et al. (2015). **A.** Black curve represents the production possibility frontier (PFF), or maximum sustainable rendering of regulating service (y-axis), for each level of provisioning service (x-axis) rendered. **B.** Blue dashed curves are isoclines or indifference curves of stakeholder utility values, with darker lines representing greater utility. The blue point represents the maximum sustainable utility that the stakeholder can garner, given the biophysical constraints that define the PPF. **C.** Red and blue dashed lines are utility indifference curves shown for two different stakeholder groups; red and blue points represent each group's goals for optimal utility. The black diamond represents the current levels of each ecosystem service being rendered in the system. The wavy arrows represent two potential trajectories for changing the levels of services rendered. Each pathway increases utility of one stakeholder group but decreases utility for the other. (Redrawn and modified from Cavender-Bares et al. 2015 with permission.)



The second "layer" of the SF assesses human preferences for different possible combinations of ecosystem services that trade off. Although all points on the PPF are efficient, they may not be equally desirable to all stakeholders. Each point in the trade-off parameter space offers a particular stakeholder a certain level of satisfaction, which economists call utility. These utility values can be graphically superimposed onto the trade-off parameter space; points of equal utility are drawn as isoclines or indifference curves (Fig. 1B). Where the isocline of greatest utility is tangential to the PPF, that particular stakeholder gains the greatest satisfaction possible from provision of ecosystem services. This point can serve as an optimal target for land use or management.

Groups of stakeholders may be characterized by a generalized, representative utility function that reflects their common preferences based on similarities in their livelihoods, cultural values, economic outlooks, and so forth. Preferences may be influenced not only by the direct benefits associated with the two services, but also by stakeholders' world views, values regarding nature, and the benefits of nature for a good quality of life. Norms and the influence of identity and history may also play a role in stakeholder preferences for different ecosystem states (Martín-López et al. 2012). Utility functions can therefore implicitly integrate other values and services that stakeholders associate with different levels of the trading-off services being plotted. Quantitative methods for assessing values, such as stated or revealed preferences, manifest stakeholder utilities and can thereby capture the suites of values and considerations that affect preference for different levels of the two trading-off services (De Groot et al. 2010, Chan et al. 2012). For instance, small-scale farmers in a forest-crop mosaic may be able to earn more money by converting their entire area to crops, but because they would lose their access to wild plant foods, sacred sites, or the value they hold for the aesthetic of their cultural landscape, their utility may be highest when a large fraction of the landscape remains forested. When utility functions for different stakeholder groups are mapped onto the first layer, i.e., the trade-off PPF, this can reveal that groups harbor quite divergent preferences regarding the optimal balance of services (Fig. 1C).

The two layers of the framework capture the state of the system with respect to biophysical and technological constraints, possible optimal efficiencies associated with trade-offs, and stakeholder preferences. The framework can be applied formally or graphically to assess the state of a given system. Onto this graphical representation, one can plot the actual level of both services currently rendered (Fig. 1C). The visualization of actual levels of services relative to desired optima by different stakeholders may offer an accessible tool to actors of diverse backgrounds for discussing barriers, enabling factors, and conflicting preferences for desired future trajectories.

Workshop-style analyses of trade-off situations

When the SF was initially developed, the same researchers held a workshop to more carefully explore the framework's practical usefulness and to develop a protocol for its implementation to facilitate analyses and management decision making. The participants included five faculty members and five graduate students from three academic institutions. Their ongoing research addressed ecosystem services across a range of social-ecological contexts.

The 1.5-day workshop began with a generalized discussion of the practical uses the SF could potentially provide. Then we devised five steps by which that utility might be effectively achieved in a participatory context with multiple stakeholders concerned with land use and management decisions. Four of the participants took turns working through the proposed process, using their primary research context as a case study. The presenters were asked to identify a trade-off of significance in their system. All cases explored the joint production of a provisioning service and a trading-off nonprovisioning resource. The case studies represented four different food production systems: cattle ranching in Mexico; avocado and maize agriculture in Mexico; subsistence pastoralism in Kenya; and shifting cardamom, maize, and cassava farming in Tanzania. Having identified a trade-off, the goal was for the presenter to provide key information and lead a collaborative process of characterizing their case study in terms of the five analytical steps, i.e., to derive the information presented in Figure 2.

Each presenter used different sources of information to derive a graphical representation of the putative PPF for their case study. In the Mexico cases, PPFs were drawn to reflect empirically measured trade-offs (González-Esquivel et al. 2015; F. Mora, P. Balvanera, E. García-Frapolli, A. Castillo, J. Trilleras, and D. Cohen, *unpublished manuscript*). The Kenya case PPF was a graphical representation of generalized threshold-exhibiting vegetation dynamics in drylands (May 1977, Scheffer et al. 2001). In the Tanzanian case, the PPF reflected measured farming practice–forest cover relationships (Mwampamba 2009, Mwampamba and Schwartz 2011) and qualitative representations of current trends in market values of different crops (T. Mwampamba, *personal communication*). Each presenter proposed putative utility functions for stakeholder groups, based on their knowledge of stakeholder values and ecosystem service preferences. The group iteratively discussed and refined the proposed protocol in response to challenges and new opportunities that arose. Two workshop participants authored a study that utilized the SF to analyze a case study involving midwestern U.S. corn production (Ewing and Runck 2015). Although we did not have time to work through their case study during the workshop, their article presented PPFs derived from empirical research and discussed other relevant components of the system. We interpreted their findings in terms of the five steps of the analytical procedure and corresponded with them to ensure we had characterized the system properly.

Interspersed with and following the case study presentations, we characterized classes of situations in which we anticipated that the framework would be particularly informative in planning and conflict resolution. We also deliberated over the practicality, advantages, and weaknesses of the approach. Discussions were informed and contextualized by conducting a literature search of other participatory and valuation methods using search terms such as participatory, ecosystem services, valuation, and multiple stakeholder preferences.

RESULTS

We envisioned the framework serving as a point of reference, implemented as a participatory tool for understanding trade-offs and for identifying obstacles and potential enabling factors to overcome them. Developing a protocol through which the framework could be implemented was the first key outcome of the project. Based on our "test drives" of the protocol using case studies and accompanying discussions, the next key outcome was the identification of four general types of challenging situations in which we believe the SF assessment protocol can yield constructive insights.

Proposed analytical protocol

The first three steps are the basis of the SF itself. The first step is identifying two ecosystem services of concern that have trade-offs. The second step is to derive and graph the PPF for those services as the first layer of the framework. The third step is identifying utility functions of different stakeholder groups, superimposing their respective indifference curves onto the graph, and identifying the point on the PPF that yields highest utility for each stakeholder group. At this stage, one can also plot the system's current yield of both services and visualize how trajectories of change would affect the utility for different stakeholder groups. To extend the SF to its intended application in resource management and conflict resolution, the fourth step is to reflect on the graph, evaluate whether stakeholders have divergent preferences for future changes to the system, and consider the limiting conditions and obstacles to achieving more efficient, sustainable, and mutually acceptable levels of trade-offs in the future. The fifth step is a creative evaluation of potential enabling factors and strategies that would resolve limitations and tensions revealed in step 4.

We applied the step-by-step protocol to four case studies, and the results are illustrated in Figure 2. In discussions of the case studies, we sought to determine how the framework helped us understand and potentially resolve challenges posed in each case study. In doing so, we identified three types of challenging situations that arose as generic, cross-cutting types of problems for which the protocol generated analytical insights. We also discussed a fourth type of challenging situation that was not exemplified in our case studies. We organized our presentation of the results according to the four types of challenging situations because they give a more generalized idea of where we believe the use of the protocol can be productive in understanding system dynamics and sustainability implications. We elaborate on the case studies in Figure 2 as appropriate to provide illustrations of the challenging situations. We illustrate the fourth challenging situation with examples from the literature. We recognize that they are not the only potential situations in which the framework may prove useful, but they stood out to us as salient challenges and useful for organizing the outcomes of our deliberations.

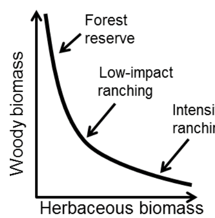
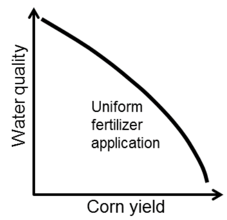
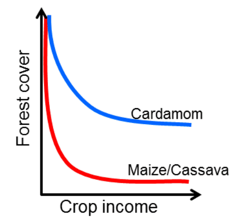
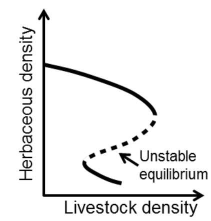
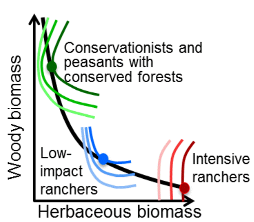
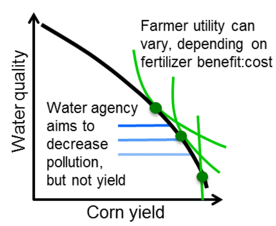
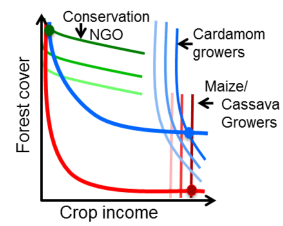
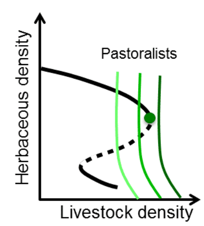
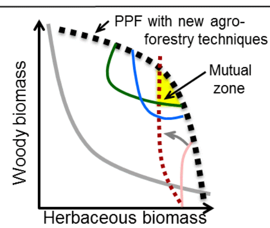
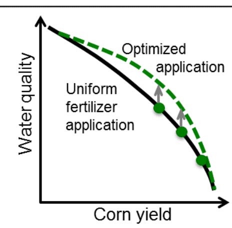
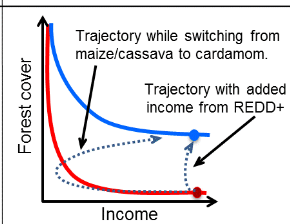
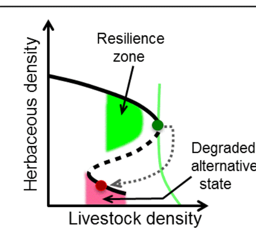
Types of challenging situations in which the trade-off framework provides insights to conflicts

Challenging situation 1: Stakeholder-dependent variation in utility functions

Challenging situation 1 is illustrated in case studies A (Mexico), B (midwestern United States), and C (Tanzania) in Figure 2. Within a region and spatial scale, different land users may have distinctly different priorities and, thus, utility functions. This challenge is what the SF was designed to graphically demonstrate, and it is visualized in the first three steps of the analytical protocol.

Case study A (Fig. 2A) provides one illustration: The SF was to visualize conflicting utility functions of different stakeholders in the dry forested regions of western Mexico, where ranchers use different land use practices. A single PPF describes the range of combinations of livestock fodder production and forest woody biomass that are potentially attainable at the spatial scale of plots (about 1-5 ha), given current agroforestry techniques (Fig. 2A, step 2). However, step 3 of the analysis revealed that all

Fig. 2. Four case studies (columns A-D) of provisioning-regulating service trade-offs explored in the workshop. Each case study was analyzed using the 5-step protocol, as indicated in the left column. Cases A, B, and C provide examples of challenging situation 1 (stakeholder divergence in utility functions) and challenging situation 2 (if and when win-win outcomes can be achieved). Case D represents challenging situation 3 (thresholds).

Case study:	**A) Mexico:** Tropical dry forest transformed to pastures for cattle ranching and secondary forest growth, considered at the plot level.	**B) Midwestern US:** Industrial-scale corn monoculture, where intense fertilization leads to nutrient pollution in waterways.	**C) Tanzania:** Shifting cultivation in montane forests, with divergent priorities for crops, biodiversity, and carbon storage.	**D) Kenya:** Subsistence pastoralism reliant on naturally occurring vegetation dynamics in semi-arid rangelands.
Challenging situation types represented:	*Situation 1*: Stakeholder-variation in utility functions *Situation 2*: Identifying if and how win-win solutions can be achieved.	*Situation 1*: Stakeholder-variation in utility functions *Situation 2*: Identifying if and how win-win solutions can be achieved.	*Situation 1*: Stakeholder-variation in utility functions *Situation 2*: Identifying if and how win-win solutions can be achieved.	*Situation 3*: Dynamic systems with zones of resilience
Step 1: Provisioning Service (x):	Herbaceous biomass (kg/ha): cattle fodder provision.	Corn production (kg/ha): income provision.	Cash crop harvest value (currency/ha): income provision.	Livestock density (cattle/ha): milk provision.
Regulating Service (y):	Woody vegetation biomass (kg/ha): contributes to biodiversity, soil, microclimatic, and hydrologic regulation.	Water quality (1-NO_3) : affects entire basin, e.g., algal blooms and hypoxic conditions that kill aquatic organisms.	Forest cover (% canopy cover or leaf area index): maintains habitat for biodiversity and carbon sequestration.	Herbaceous vegetation density (kg/ha): maintains ecosystem function and productivity through water-soil-vegetation dynamics.
Step 2: Production Possibility Frontier (PPF):	 Three land uses characterize different ranges of the tradeoff space.	 PPF expands if fertilizer application rates are optimized for each spatial unit, rather than uniformly applied.	 Cardamom grows best in partial shade, allowing more forest cover per crop yield than maize/cassava.	 Because of vegetation-herbivore feedbacks, system exhibits degradation threshold (see text).
Step 3: Stakeholder utility functions, shown as indifference curves:				
Step 4: Limiting conditions and obstacles	Highly concave PPF and strongly divergent utility functions preclude mutually acceptable solutions for ranchers and forest conservationists.	Lack of technology access, inertia, and fertilizer company-provided extension all limit farmers' likelihood of optimizing fertilizer application rates.	Growing cardamom benefits farmers and conservationists more than maize/cassava, but time lag to crop maturity is economic obstacle to switching to cardamom.	If livestock densities exceed the utility optimum, ensuing degradation will lead system to less productive alternative state.
Step 5: Enabling factors and strategies	 Improved silvopastoral practices can change PPF to alleviate severity of tradeoffs, and may ease ranchers' aversion to woody cover. If so, a zone of high efficiency and mutual acceptability emerges.	 Technology to determine temporal and spatial plant needs, buffer zones, and intercropping practices can maintain yields and improve water quality.	 REDD+ payments for reforestation can incentivize switching to cardamom by compensating farmers for loss of agricultural income during time lag to achieve cardamom productivity.	 Pursuing highest utility poises the system at a bifurcation point, which can lead to degradation. Maintaining livestock density in the resilience zone mitigates the risk of a degradation and poverty trap.
References:	Mora et al. *unpubl. manuscript* P. Balvanera, *pers. comm.*	Ewing and Runck 2015	Mwampamba et al. 2011 T. Mwampamba, *pers. comm.*	Rietkerk et al. 1996 E. King, *pers. comm.*

stakeholder groups aim to increase services, but they vary strongly in their preferred combination of services rendered along the PPF, resulting in highly contrasting targets (Fig. 2A, step 3). The different utility optima arise because stakeholder groups associate abundance of woody vegetation with different degrees of economic, normative, and cultural service values. In Figure 2, step 3, case studies B (midwestern United States) and C (Tanzania) also include utility functions for different stakeholder groups, illustrating similar potential tensions between stakeholder preferences.

Some stakeholder-dependent variation in preferences can arise when different groups' aims are focused on different spatial scales. A globally important example of this can be seen in the broad geographical trend of greater concentrations of biodiversity in less developed, tropical countries (Cavender-Bares et al. 2013). International conservation organizations use political pressure and incentives to encourage the protection of habitat for biodiversity conservation, whereas local communities, who in many cases are subsistence and small-scale agriculturalists, have strong preferences for increased agricultural productivity to meet minimum livelihood needs (Sunderlin et al. 2005, García-Barrios et al. 2009). Neglecting local preferences in larger policy decision making is an issue that arises repeatedly in struggles to achieve sustainable development (Muradian et al. 2013). The SF offers a means of visualizing such tensions via competing utility functions held by different stakeholders.

Challenging situation 2: Identifying if and how win-win outcomes can be achieved

Challenging situation 2 is illustrated in case studies A (Mexico), B (midwestern United States), and C (Tanzania) in Figure 2. The goal of sustainable development is to improve livelihoods while maintaining the ecological processes that support life on Earth. Translated to cases of two-service trade-offs, the goal of policy makers is often to increase provisioning and regulating services simultaneously, i.e., to generate win-win outcomes. Win-win can also refer to the simultaneous increase in utility for multiple stakeholder groups. This is an extremely important distinction, and this framework allows one to evaluate the feasibility of win-win proposals both in terms of biophysical constraints and stakeholder preferences.

In the generalized example illustrated in Figure 1C, the SF diagram illustrates that there is in fact available parameter space within the PPF to sustainably improve the yields of both services. However, the diagram also reveals that some shifts toward the PPF may actually cause a decrease in utility for some stakeholders. Importantly, the diagram can also demonstrate that in some cases, despite one stakeholder group's rhetoric, real, feasible win-win options may not in fact exist for the two services being examined, either because of the strong concavity of the PPF or strongly contrasting utility functions of different stakeholders. In such cases, side payments may offer a way to compensate for losses of utility experienced by some stakeholders, but with recognition that not all services can be substituted through financial mechanisms.

The SF helped to illustrate if and how win-win outcomes could be attained in three of our case studies. Returning to case study A from Mexico, step 4 used the state-of-the-system diagram to identify two potential obstacles: a highly concave PPF and highly contrasting utility functions for different stakeholders. Under these circumstances, none of the points along the efficiency frontier represent mutually acceptable outcomes. In step 5, however, we considered the potential impact of improved agroforestry practices, which maintain grass for livestock grazing and provide tree cover. This would ameliorate the severity of trade-offs, creating a more convex PPF (Fig. 2A, step 5, dashed black line), and it would open up the parameter space to allow higher woody biomass and high livestock forage production, thereby enabling a win-win outcome in terms of biophysical trade-offs. Many ranchers recognize that woody fodder can be very important in the dry periods of the year, can contribute to soil protection, and can provide shade for the cattle (F. Mora, P. Balvanera, E. García-Frapolli, A. Castillo, J. Trilleras, and D. Cohen, *unpublished data*). This offers a potential enabling factor; if these improved agroforestry practices were shared through information campaigns or peer-to-peer learning, the farmers' intense preferences could be reshaped to include the acceptance of more woody cover in pastures (Fig. 2A, step 5, dashed red line). In fact, agroforestry approaches are increasingly being considered by intensive ranchers as a viable win-win option. Through improved agroforestry techniques and raising awareness of the multiple benefits they provide, a zone would emerge that could provide each stakeholder group with acceptable levels of utility (Fig. 2A, step 5, yellow region).

Case study B, which consists of large-scale corn production in the midwestern United States, is another case in which the current PPF (Fig. 2B, step 2) could be expanded through technological advances that create more favorable trade-off parameter space. In their analyses of counties in Iowa and southern Minnesota, Ewing and Runck (2015) found that industrial agricultural operations were operating near or at the maximum efficiency attainable from their fertilizer application techniques. In other words, they were applying as much fertilizer as plants could benefit from, but not very much more because fertilizer costs are a key consideration in industrial agricultural management. Soil and natural resource conservation agencies aim to work with farmers to decrease excess fertilizer use and environmental impacts without compromising yields (Fig. 2B, step 3, horizontal blue utility curves), a strategy that may not be prioritized by extension agents who work for fertilizer companies. Precision technologies now allow fertilizer application rates that account for localized plant demand and soil conditions. A simulation model showed that if fertilizer application is spatially optimized, provisioning yields can be maintained or increased while improving the regulating ecosystem services rendered by higher quality surface water conditions (Fig. 2B, step 5). Improved technology offers a strategy to increase water quality without compromising crop yields.

In case study C, the SF is used to consider alternative production systems as possible win-win outcomes in a shifting cultivation system in Tanzania upland forest areas. In montane farming regions of Tanzania, farmers can grow cardamom or maize and cassava as cash crops (Fig. 2C). Because cardamom can be grown in shade, whereas maize and cassava yields decrease with shade, these contrasting crop production systems have markedly different trade-off curves with respect to biodiversity maintenance (Fig. 2C, step 2). Switching from maize to cardamom farming can generate more income and maintain more biodiversity, thus offering broader win-win parameter space. However, if we consider the temporal trajectory of a system switching from maize to cardamom, we encounter the challenge of low provisioning during the time necessary for cardamom shrubs and tree cover to mature (Fig. 2C, step 5, left dashed arrow). If farmers are eligible to receive payments for reafforestation during that lag, through programs such as Reduced Emissions from Deforestation and Degradation or Payments for Ecosystem Service, this can reduce their short-term costs of switching (Mwampamba 2009) and serve as an enabling factor.

Challenging situation 3: Dynamic systems with uncertainty, traps, and zones of resilience

Challenging situation 3 is illustrated in case study D (Kenya) in Figure 2. Ecological dynamics can play out at different time scales. In the short term, the biophysical frontier can actually be exceeded, but this represents an unsustainable level of benefits garnered. In systems where there is uncertainty regarding the location of the PPF, and in systems that exhibit threshold behavior and alternate stable states, the potential to overshoot the frontier merits some cautious consideration.

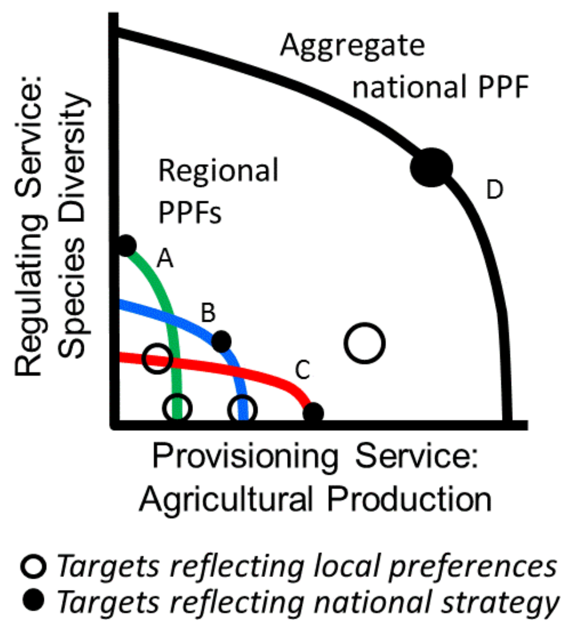
Case study D involves a semiarid rangeland (Figure 2D), an ecosystem well studied for its potential threshold behavior (Rietkerk et al. 1996, Scheffer et al. 2001, Bestelmeyer 2006). The provisioning service in this case is forage consumed by livestock, which is proportional to stocking density. The regulating service is the integrity of the remaining perennial vegetation, which must persist to capture rainfall and ensure regrowth of forage in the next season. If grazing pressure exceeds a threshold intensity, the ecosystem may cross a functional threshold, beyond which abiotic conditions (soil surface sealing, runoff) create ecohydrological feedbacks that result in much lower forage production, even if grazing intensity is reduced (King and Hobbs 2006, Turnbull et al. 2012). As illustrated by May (1977), the equilibrium solution (and thus the PPF) has two alternative domains at moderate stocking intensities: a prethreshold region where vegetation is fairly intact and a postthreshold, degraded state with very little vegetation (Fig. 2B, step 2). This situation is characterized as a discontinuous PPF, with the attainable state being path dependent on whether the threshold has been crossed.

Because the utility function of most ranchers and pastoralists is likely to favor the largest sustainable herd, the management target would be the point on the upper segment of the PPF with the greatest herd size (Fig. 2D, step 3). Unfortunately, that point is also a bifurcation point, which if exceeded, would elicit a threshold shift to lower productivity. For the system to recover, grazing intensity would have to be greatly reduced and might entail an unacceptably low level of livelihood provisioning from livestock. The bifurcation point is not necessarily fixed. It could shift inward under low rainfall conditions, leading to a collapse under grazing intensities that might have previously been supportable. This illustrates the prudence of establishing management goals within what we term a "zone of resilience" at some lesser grazing intensity than the preferred optimum (Fig. 2D, step 5). In a wide array of natural resources systems, we see surprises, i.e., unexpected shifts in ecosystems because of poorly understood nonlinearities, underestimations of human impacts, or a failure to account for temporal variability (Carpenter et al. 2009). The SF can help illustrate that maximizing efficiency may not be the most resilient strategy if exceeding a threshold carries the risk of shifting to a degraded state.

Challenging situation 4: Spatial and scale-dependent variation in biophysical trade-offs

Challenging situation 4 is illustrated with a literature example from Mexico and generically in Figure 3. There is wide recognition that the biophysical constraints on ecosystem service trade-offs vary greatly depending on the location and spatial scale at which they are assessed (Rodríguez et al. 2006, Costanza et al. 2007, Nelson et al. 2009). However, our workshop did not examine a case study that directly addressed PPFs at different scales. Therefore, our discussions of this situation drew from published examples.

Fig. 3. A stylized representation of challenging situation 4, spatial and scale-dependent variation in production possibility frontiers (PPFs), as illustrated with an example from Mexico (see text). Trade-offs between agricultural productivity and biodiversity maintenance are considered at local scales in three ecological regions in Mexico and at the broader national scale. Curves A, B, and C represent region-specific PPFs for the trade-off, and the black curve (D) represents the national PPF, aggregated over the included regions. Region A represents an area with low agricultural potential but high species richness, region C represents an area with high agricultural potential but low species diversity, and region B is intermediate. Small closed circles indicate local targets for achievement of a national land-sparing strategy to maximize both biodiversity and agricultural output at the national level. Small open circles represent local targets according to local values within each region. The large closed circle represents the national-level service rendering under "land sparing," which is optimally efficient at the aggregate, national level. The large open circle represents the national-level services that would be rendered if each region were managed according to local priorities, which are not optimal by national standards but maximize utility for stakeholders in each region. (See Appendix 1 for details regarding the assumptions and calculations used to the derive curves.)



In Mexico, for example, the karst soils of the Yucatan Peninsula limit agricultural productivity (Duch 1995), but these are regions of high biodiversity endemism (Ibarra Manriquez et al. 1995). In contrast, northwestern regions of Mexico have deep fertile soils and access to irrigation that can support high agricultural production, but have a lower proportion of endemic species (Arriaga et al. 1997). The two regions will have distinctly differently shaped PPFs (Fig. 3, curves A and C). Intermediate regions that hold a mosaic of valley bottoms with fertile soils and hills that host endemic biodiversity are found in central Mexico (González-Esquivel et al. 2015; Fig. 3, curve B). These regional PPFs can be aggregated to generate a national-level PPF, based on optimal assignment of land uses to different regions (Fig. 3, curve D; see Appendix 1 for calculation details). In effect, the most efficient service yields would be achieved by allocating large tracts of land to either agriculture or conservation, depending on their biophysical suitability to render provisioning and regulating services, a so-called "land-sparing" strategy (Fischer et al. 2008).

In the Mexico example, a land-sparing strategy at the country-level scale would entail massive conservation set-asides and minimal local food production in the Yucatan (Fig. 3, curve A), while intensive food production would be concentrated in transformed landscapes of northwest Mexico with reduced local regulating services (Fig. 3 curve C), and a mosaic of agricultural areas and high biodiversity areas would be maintained in the central part of the country (Fig. 3, curve B). Despite efficiency optimization at the national scale, the strategy may not be optimal to local residents, who may prefer land-sharing strategies that generate a local balance of regulating and provisioning ecosystem services, or who may prefer income maximization, regardless of their ecological region, even if it would not contribute optimally to national goals (García-Barrios et al. 2009, Perfecto and Vandermeer 2010).

Changes in trade-offs across space are readily represented using the first layer of the framework: identifying PPFs. In fact, a number of studies have adopted this approach to produce spatially optimized PPFs of biodiversity maintenance and provisioning services over a heterogeneous landscape (Nelson et al. 2008, Polasky et al. 2008, Kline and Mazzotta 2012, White et al. 2012). Because ecosystem service yields can show nonlinear increases with increasing spatial scale (Koch et al. 2009, Laterra et al. 2012), aggregating service yields may require a mathematical rather than qualitative application of the framework. Also, scaling and aggregating across regions require knowledge or assumptions about the substitutability of services across regions. For instance, if the regulating service of interest is being indexed as species diversity, the analysis would incorporate region-specific species-area curves and endemism for the taxa being considered and would consider adjacency of spatial units to calculate the area under similar land use practices (Nelson et al. 2009).

Applying the second layer of the framework, the overlay of utility functions, is more problematic when PPFs are drawn to reflect different spatial scales. Identifying the most desirable points for local stakeholders on their local PPF is graphically tractable when considering only one spatial scale. Understanding how their utility functions change across scales and reconciling those with utilities of larger scale agents present a substantial challenge. Local stakeholders' preferences may be focused only on local trade-offs, whereas larger scale stakeholders' preferences may incorporate, to varying degrees, both large-scale goals and concern for local well-being (Roe and Walpole 2010, Kari and Korhonen-Kurki 2013). With scale-dependent variability in stakeholder values and nonlinear scaling of ecosystem service yields, there is tremendous scope for complexity embedded in the exercise of associating values with trade-off outcomes at multiple spatial scales (Hein et al. 2006, Ernstson et al. 2010, Daw et al. 2011). Reconciling trade-offs at multiple spatial scales is a salient theme in a number of research programs that seek to promote the applicability of the ecosystem service concept in sustainability science and sustainable development (e.g., Hein et al. 2006, Biggs et al. 2007, Müller et al. 2010, Haase et al. 2012, Labiosa et al. 2013). Developing and evaluating the potential utility of this framework in that endeavor are areas of active ongoing study.

DISCUSSION

In this study we developed a methodology for implementing the SF to analyze ecosystem service trade-offs, applied the concepts to case studies, and identified four kinds of situations where the framework helps chart out the nature of trade-offs, highlighting conflicting preferences and identifying obstacles and enabling factors. We now reflect on our workshop experience with the SF and existing literature on participatory assessments to consider two topics: (1) putative guidelines for applying the framework in participatory settings and (2) key strengths, weaknesses, and limitations of the framework.

Using the framework in participatory planning and conflict resolution

In making recommendations for implementing the SF protocol in a real-world participatory process, we reflected on workshop experiences and literature on participatory assessments to offer putative guidelines for each step of the protocol. We envision that a participatory process would begin step 1 of the protocol by focusing on a known trade-off for which there exist research-derived and local knowledge bases that can be called upon to sketch or calculate efficiency frontiers and characterize stakeholder utility. In each of the workshop case studies we examined, at least some stakeholders were aware of and concerned about an ecosystem service trade-off in their system, and there was knowledge from scientific or economic research regarding that trade-off. We found that the selection of variables or indicators to use on the x- and y-axes required iterative discussion of the conflict itself, as well as the biophysical dynamics of the system, to identify variables that were meaningful and tractable in both contexts. In a participatory setting, we expect that facilitation by participants familiar both with the stakeholder divergences and the biophysical conditions would be useful and perhaps necessary to get the process started. We anticipate that facilitators would need to adjust the orientation process to account for levels of literacy of the engaged stakeholders and their familiarity with graphical representations of information.

For step 2 of the protocol, i.e., deriving the PPF, the framework can be used for a quantitative analysis with mathematical models (e.g., Nelson et al. 2009) or empirical data (e.g., González-Esquivel et al. 2015). Alternatively, we believe that using the framework for hypothetical or graphical analysis, as we did in the workshop, may also be instrumental in understanding conflicts and visualizing solutions. We envision that stakeholder groups, either separately or jointly, would work through a process of generating hypothesized PPFs, with the input of facilitators or participants who have ecological expertise. We expect that facilitation would be useful to initially explain and maintain focus on the concept of the frontier as representing the joint levels of services that can be sustainably garnered. The goal of cogenerating the PPF is to build a shared awareness of the balances of services that can potentially be achieved.

For step 3 of the protocol, i.e., proposing utility functions for stakeholders, we used hypothetical comparative stated preferences between different points in the parameter space, starting with points along the PPF. The presenters attempted to focus on holistic preferences, so that embedded values for other services and preferences were also reflected. They were essentially trying to guess the answer that a stakeholder would give if asked, "Given a future that looks like A and a future that looks like B, which would you prefer?" Then, using the same approach, they considered other parts of the parameter space to estimate where utility would remain equal. From this iterative exploration, utility indifference curves were sketched. We envision that this iterative, hypothetical approach would prove useful for eliciting preferences in a participatory context. Other participatory planning and analysis protocols use visioning and preference ranking approaches effectively (Lynam et al. 2007, Plieninger et al. 2013), so we expect that they could be used to help define utility functions in the context of the SF as well.

Once the two layers that characterize the state of the system are collaboratively generated, with the current service yields also plotted onto that diagram, the result, like other participatory planning tools, would provide a visual basis for the next steps of the protocol: exploratory discussions regarding goals, obstacles, and enabling factors for more sustainable trajectories (Lynam et al. 2007, Castella 2009, Grêt-Regamey et al. 2013). Because of the framework's two-layer construction, the participatory process can tease apart two important dimensions of enhancing sustainability: (1) biophysical constraints on directions in which the system can go and (2) value-based decisions, obstacles, and enabling strategies regarding where the system should go.

To initiate step 4, i.e., identifying obstacles, we propose participants first discuss the PPFs and the system's current location in the parameter space. If the system's current performance is well below the PPF, participants can explore and identify specific factors that pose obstacles to greater efficiency. Discussion would focus on social, political, economic, and technical barriers. Once obstacles are identified, discussion can focus on enabling factors that would overcome those obstacles. A second issue is the shape and location of the PPF itself. For instance, if the PPF is strongly concave, imposing sharp trade-offs, participants can explore what technological or management improvements would change the shape of the PPF so as to create more favorable, attainable yields of both services. Changes in land use, policy, or technology can expand the PPF itself, creating more "space" for sustainable, higher yielding resource use regimes.

In step 5 of the protocol, participants seek to identify opportunities and enabling factors. We propose a discussion that focuses on strategies to overcome obstacles and conflicts represented by second layer of utility divergences. Because many conflicts are rooted in divergent values, the framework can be used facilitate an open discussion of the disparate goals held by different stakeholders. With contrasting utility functions plotted for each stakeholder group, the participants can more clearly evaluate the degree and points at which their preferences in fact diverge. By identifying a region of the parameter space where utility functions overlap to promote multiple groups' values, a zone of acceptability may be identified as a mutual, consensus goal (as in case study A, Fig. 2A). The ability to identify potential space for conflict resolution is an important asset of this framework. Participants can explore how policies, incentive mechanisms, or educational initiatives could alter stakeholder perceptions of utility and create new, mutually beneficial parameter space (Bryan 2013).

Strengths, weaknesses, and limitations

SF, as presently developed, directly assesses trade-offs between two services

In developing the SF, the main intention was assessment of trade-offs between provisioning and nonprovisioning services. The graphical nature of the assessment owes its clear, simple explanatory power to the low dimensionality. But there are drawbacks to considering only two services, as well as some windows of opportunity for further model refinement. In the case studies explored, we used ecological conditions such as forest cover and herbaceous biomass as proxies for nonprovisioning services (y-axes), because the system's biophysical constraints arise from the ecological relationship between these state variables and provisioning. In each case, however, that ecological condition actually represented multiple nonprovisioning services, including cultural services, that are associated with the chosen metric (Fig. 2, Regulating Service row). The framework does not explicitly evaluate synergies between the multiple services associated with the chosen metrics. The framework does, however implicitly incorporate additional services.

To the extent that stakeholder preferences account for them, cultural services implicitly enter the evaluation via utility functions. Cultural services arise from ecological conditions that provide nonmaterial benefits that individuals value (Daniel et al. 2012). Because utility or preference is the sum of benefits gained from a point in the trade-off space, utility is not necessarily only because of economic values, but should reflect the cultural services, norms, history, and ethics that a stakeholder associates with a point in the trade-off space. Devising a clearer methodology to make these embedded values more explicit could help improve the SF's utility in conflict resolution and is a topic meriting further research (Chan et al. 2012).

SF distinguishes between biophysical and value-based obstacles

The clear distinction between the biophysical trade-offs in ecosystem services and stakeholders' divergent valuations of those services is expected to facilitate better fit between obstacles and potential solutions for more effective resource management. If the SF protocol reveals constraints to sustainability posed by the PPF itself, this directs the participants toward consideration of biophysical solutions. In other cases, if obstacles arise because of tensions between different stakeholders' perceptions of utility, solutions may use social processes such as participatory planning, compromise and negotiation, reconciliation, and appropriate incentive systems. In other cases, a long history of social, cultural, and institutional drivers has contributed to highly inefficient conditions. In case study A from Mexico, for example, we were able to link biophysical obstacles with potential technological enabling strategies, while the analysis pointed to knowledge-sharing strategies to overcome value-based obstacles posed by ranchers' intensive preferences for forest clearing. In case study C from Tanzania, financial incentives to overcome short-term obstacles could contribute to more desirable solutions.

Another way that we expect the distinct layers of the SF to prove useful is by drawing attention to the biophysical limits of the system from the beginning. Some desirable solutions may not be possible to attain sustainably. Centering the planning or conflict resolution process around a graphical representation of the biophysically attainable levels of services may help keep deliberations focused on achievable goals and necessary compromises.

SF can account for different stakeholder values for the same level of a service

A notable advantage of this framework is the clear distinction between, and simultaneous visualization of, the quantities of services yielded in a trade-off situation and the utility of those services. Most ecosystem valuation methodologies in current practice are economically based and tend to blur the distinction between quantity and utility by implicitly assuming that they are basically the same (De Groot 2006). This confusion may stem from the fact that we use the term "value" both for preferences (utilities) and quantification (quantities). Even in approaches that acknowledge the distinction between quantities and values, the issue of divergent values can remain obfuscated. For instance, in their framework for including ecosystem services in decision making, Daily et al. (2009) distinguished between biophysical and valuation processes, and they addressed the challenge of valuing nonprovisioning services, such as cultural services. However, they did not address the issue of divergent values for the same quantity of a service. The SF highlights how quantities and utility need not necessarily coincide, even for provisioning services that generate cash for livelihoods. Different utility optima arise because stakeholder groups associate different degrees of historical, ethical, and cultural service values with a given level of provisioning-service production. We believe that capturing such divergences is a critical advantage of the SF in diagnosing the nature of value-based conflicts that arise in association with ecosystem service trade-offs. The supply-side and demand-side trade-off typologies used by Mouchet et al. (2014) and Yahdjian et al. (2015) offer other new approaches for distinguishing between biophysical constraints and divergent stakeholder values, and both reports recognize that decision making entails reconciliation between the two. The simultaneous visualization of both dimensions of trade-offs in the SF may offer a tractable starting point for such reconciliation in participatory settings.

SF can be utilized through qualitative or mathematical methods, and can account for nonlinear and complex associations between ecosystem services

In the case studies examined, PPFs were derived using varying combinations of theoretical, empirical, qualitative, and quantitative information. We found this flexibility to be a strength in that each presenter was able to propose a PPF, using whatever forms of information they had. Although each case assessment began with the general notion that more provisioning would mean fewer regulating services, we found the nonlinearities in each system's biophysical dynamics led to differently shaped curves in each case, with different consequences for subsequent evaluation of divergences in stakeholder preferences.

SF is expected to stimulate and facilitate participatory stakeholder discussions, and help build shared mutual understanding

Although ecosystem services are analyzed and modeled extensively, there is so far relatively limited transfer of these methods to participatory processes (Etienne et al. 2011). However, there is extensive recognition of the importance of participatory processes in establishing common understanding of problems and finding solutions (Lynam et al. 2007, Reed 2008). Efforts to develop effective tools to facilitate both ecosystem service valuation and participatory planning/conflict resolution are intensifying. For a framework to work well as a valuation tool in a participatory context, it must offer a valid and useful representation of the system's dynamics, while also being simple and accessible enough for nonspecialists from different backgrounds to understand and engage with.

By integrating stakeholder knowledge and experience with scientific knowledge of ecological dynamics and constraints, this process can help participants build a shared mental model of causal relations that explain the nature of ecosystem service trade-offs. Establishing shared mental models is receiving increasing attention in natural resource management as an important step to facilitate cooperation, mutual understanding, and capacity for adaptive governance (Biggs et al. 2011, Jones et al. 2011, Stone-Jovicich et al. 2011, Grêt-Regamey et al. 2013). This framework provides a novel way to use both empirical data and stakeholder experiences to collaboratively form a mental model, and also to identify important information gaps. We expect that the visual nature of the assessment tool, which represents all stakeholders on a common playing field, will also help make the concepts more accessible to participants.

The framework's second layer, thinking through and mapping utility functions for different stakeholder groups, necessitates pluralistic thinking and can build a rich mutual understanding of different groups' motivations and points of dissatisfaction, making very clear the reasons why conflicts arise and persist. Because the framework explicitly recognizes that different stakeholders vary in the way they integrate economic, social, ethical, and other values, it becomes more evident that there may be no single "correct" value that can be assigned to an ecosystem service. However, ignoring or denying the legitimacy of other groups' values is often a key part of power dynamics in natural resource conflicts (Brosius 2010); powerful parties may resist the pluralistic approach of the framework's second layer.

Although this participatory process may help identify opportunities to create more mutually preferred parameter space, additional discussion of rights, priorities, and power is also likely to be necessary to mitigate conflicts in such cases. Power imbalances among stakeholders are likely to exist, and resource use decisions by the most powerful may negatively impact the utility of some stakeholders. To address this, the framework could be used to identify minimum service provisions and lower bounds of tolerable losses to utility for less powerful stakeholders, which if transgressed would critically threaten livelihoods (Leach et al.

2010). It is also valuable to recognize that scale and power dynamics are often coupled. Examples of this can be seen in the case of large-scale hydropower interventions in which the state, the energy sector, and private actors can set resource use priorities to enhance their utility, while multiple local, disempowered stakeholders are deprived of their land, livelihoods, and water access (Matthews 2012).

SF utilizes biophysical and economic understandings and ways of representing dynamics

In decision-making processes with participants from different disciplinary backgrounds, Smith and others (2012) found PPFs to be effective boundary objects, or points of reference that are shared between groups with otherwise very different conceptualizations of a system. Although the process is participatory in that stakeholders' inputs are part of the cogeneration of the PPFs, it is constrained a priori to utilize PPFs and not any other construct to characterize the system. PPFs themselves reflect the scientific and economic paradigms of thought from which they emerged. Although these paradigmatic approaches are fundamental to the field of sustainability science today (Clark 2007), they may neglect knowledge systems used by other stakeholders. Disparate nonscientific stakeholders may have never considered the dynamics of ecosystem service trade-offs in such an explicit manner; this approach may not be easy to understand. Additionally, for the PPF to hold credibility and legitimacy, those stakeholders would need to trust the empirical data (if they exist), trust the natural resource ecologists or economists contributing to the framework application, and trust one another as they reason through the causal relationships that determine the shape of the PPF. It remains an open question as to whether, or which, stakeholders will relate to and embrace the SF's approach to characterizing their system. Participatory visioning and planning processes are gaining salience in resource management, with proponents arguing that they are not only more effective but also more legitimate than top-down approaches (Etienne et al. 2011). In the same way that participatory processes have advanced through iterations of implementation and refinement (Castella 2009, Helming and Pérez-Soba 2011), we expect implementation of the SF to lead to new lessons learned and adaptations that improve its applied usefulness.

CONCLUSION

Cavender-Bares and others (2015) proposed a framework that is intended to clarify the distinctions between biophysical constraints and contrasting values in the evaluation of ecosystem service trade-offs. By "test-driving" the framework with a number of case studies, we concluded that the framework is likely to be useful when stakeholders have divergent values, when there may be thresholds in the system, and when participants seek to identify if and how win-win outcomes can indeed be achieved. The complex issues related to valuations across multiple scales are a challenging and critical field of inquiry to enhance the applicability of this framework, and are a vibrant topic of inquiry throughout sustainability research more broadly.

We envisioned the strengths and challenges of using the framework in a fully participatory context. We expect that it would help participants distinguish between challenges imposed by the biophysical constraints of the system versus challenges that arise because of divergent values. The framework invites pluralism and could conceivably be used with a wide number of monetary and nonmonetary valuation techniques for determining stakeholder utility. We recognize, however, that because the approach is grounded in economic and ecological perspectives on resource use conflicts, the concepts may require thoughtful introduction to participants who view human-environment relations from fundamentally different perspectives. As with any single framing tool used in a participatory setting, its effectiveness will depend on the trust and willingness of participants to work within the framework to visualize their sustainability challenges. We anticipate that much more will be learned about the framework's utility as our research team, and others, use the approach in the practice of natural resource management and conflict resolution.

Responses to this article can be read online at:
http://www.ecologyandsociety.org/issues/responses.php/7822

Acknowledgments:

We thank the support of the National Center for Ecological Synthesis and Analysis (NCEAS), which supported a Distributed Graduate Seminar that originally stimulated the ideas developed herein. We also thank three anonymous reviewers for constructive feedback and recommendations on the manuscript.

LITERATURE CITED

Arriaga, L., C. Aguilar, D. Espinosa, and R. Jiménez. 1997. *Regionalización ecológica y biogeográfica de México*. Comisión Nacional para el Conocimeinto y Uso de la Biodiversidad, México City, Distrito Federal, Mexico.

Bator, F. M. 1957. The simple analytics of welfare maximization. *American Economic Review* 47:22-59.

Bennett, E. M., G. D. Peterson, and L. J. Gordon. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12:1394-1404. http://dx.doi.org/10.1111/j.1461-0248.2009.01387.x

Bestelmeyer, B. T. 2006. Threshold concepts and their use in rangeland management and restoration: the good, the bad, and the insidious. *Restoration Ecology* 14:325-329. http://dx.doi.org/10.1111/j.1526-100X.2006.00140.x

Biggs, D., N. Abel, A. T. Knight, A. Leitch, A. Langston, and N. C. Ban. 2011. The implementation crisis in conservation planning: could "mental models" help? *Conservation Letters* 4:169-183.

Biggs, R., C. Raudsepp-Hearne, C. Atkinson-Palombo, E. Bohensky, E. Boyd, G. Cundill, H. Fox, S. Ingram, K. Kok, S. Spehar, et al. 2007. Linking futures across scales: a dialog on multiscale scenarios. *Ecology and Society* 12(1): 17. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art17/

Brosius, J. P. 2010. Conservation trade-offs and the politics of knowledge. Pages 311-328 *in* N. Leader-Williams, W. M. Adams, and R. J. Smith, editors. *Trade-offs in conservation: deciding what to save*. Wiley-Blackwell, Chichester, UK. http://dx.doi.org/10.1002/9781444324907.ch17

Bryan, B. A. 2013. Incentives, land use, and ecosystem services: synthesizing complex linkages. *Environmental Science & Policy* 27:124-134. http://dx.doi.org/10.1016/j.envsci.2012.12.010

Carpenter, S. R., C. Folke, M. Scheffer, and F. R. Westley. 2009. Resilience: accounting for the noncomputable. *Ecology and Society* 14(1): 13. [online] URL: http://www.ecologyandsociety.org/vol14/iss1/art13/

Castella, J.-C. 2009. Assessing the role of learning devices and geovisualisation tools for collective action in natural resource management: experiences from Vietnam. *Journal of Environmental Management* 90:1313-1319. http://dx.doi.org/10.1016/j.jenvman.2008.07.010

Cavender-Bares, J., J. Heffernan, E. King, S. Polasky, P. Balvanera, and W. C. Clark. 2013. Sustainability and biodiversity. Pages 71-84 *in* S. Levin, editor. *Encyclopedia of biodiversity*. Elsevier, Oxford, UK. http://dx.doi.org/10.1016/b978-0-12-384719-5.00390-7

Cavender-Bares, J., S. Polasky, E. King, and P. Balvanera. 2015. A sustainability framework for assessing trade-offs in ecosystem services. *Ecology and Society* 20(1): 17. http://dx.doi.org/10.5751/ES-06917-200117

Chan, K. M. A., A. D. Guerry, P. Balvanera, S. Klain, T. Satterfield, X. Basurto, A. Bostrom, R. Chuenpagdee, R. Gould, B. S. Halpern, et al. 2012. Where are *cultural* and *social* in ecosystem services? A framework for constructive engagement. *Bioscience* 62:744-756. http://dx.doi.org/10.1525/bio.2012.62.8.7

Chan, K. M. A., T. Satterfield, and J. Goldstein. 2012. Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics* 74:8-18. http://dx.doi.org/10.1016/j.ecolecon.2011.11.011

Clark, W. C. 2007. Sustainability science: a room of its own. *Proceedings of the National Academy of Sciences of the United States of America* 104:1737-1738. http://dx.doi.org/10.1073/pnas.0611291104

Costanza, R., B. Fisher, K. Mulder, S. Liu, and T. Christopher. 2007. Biodiversity and ecosystem services: a multi-scale empirical study of the relationship between species richness and net primary production. *Ecological Economics* 61:478-491. http://dx.doi.org/10.1016/j.ecolecon.2006.03.021

Daily, G. C., S. Polasky, J. Goldstein, P. M. Kareiva, H. A. Mooney, L. Pejchar, T. H. Ricketts, J. Salzman, and R. Shallenberger. 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment* 7:21-28. http://dx.doi.org/10.1890/080025

Daniel, T. C., A. Muhar, A. Arnberger, O. Aznar, J. W. Boyd, K. M. A. Chan, R. Costanza, T. Elmqvist, C. G. Flint, P. H. Gobster, et al. 2012. Contributions of cultural services to the ecosystem services agenda. *Proceedings of the National Academy of Sciences of the United States of America* 109:8812-8819. http://dx.doi.org/10.1073/pnas.1114773109

Daw, T., K. Brown, S. Rosendo, and R. Pomeroy. 2011. Applying the ecosystem services concept to poverty alleviation: the need to disaggregate human well-being. *Environmental Conservation* 38:370-379. http://dx.doi.org/10.1017/S0376892911000506

De Groot, R. 2006. Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning* 75:175-186. http://dx.doi.org/10.1016/j.landurbplan.2005.02.016

De Groot, R. S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity* 7:260-272. http://dx.doi.org/10.1016/j.ecocom.2009.10.006

Duch, G. J. 1995. Los suelos, la agricultura y vegetación en Yucatán. *La milpa en Yucatan* 1:97-107.

Ernstson, H., S. Barthel, E. Andersson, and S. T. Borgström. 2010. Scale-crossing brokers and network governance of urban ecosystem services: the case of Stockholm. *Ecology and Society* 15(4): 28. [online] URL: http://www.ecologyandsociety.org/vol15/iss4/art28/

Etienne, M., D. R. Du Toit, and S. Pollard. 2011. ARDI: a co-construction method for participatory modeling in natural resources management. *Ecology and Society* 16(1): 44. [online] URL: http://www.ecologyandsociety.org/vol16/iss1/art44/

Ewing, P. W., and B. C. Runck. 2015. Optimizing nitrogen rates in the midwestern United States for maximum ecosystem value. *Ecology and Society* 20(1): 18. http://dx.doi.org/10.5751/ES-06767-200118

Fischer, J., B. Brosi, G. C. Daily, P. R. Ehrlich, R. Goldman, J. Goldstein, D. B. Lindenmayer, A. D. Manning, H. A. Mooney, L. Pejchar, et al. 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming? *Frontiers in Ecology and the Environment* 6:380–385.

García-Barrios, L., Y. M. Galván-Miyoshi, I. A. Valdivieso-Pérez, O. R. Masera, G. Bocco, and J. Vandermeer. 2009. Neotropical forest conservation, agricultural intensification, and rural out-migration: the Mexican experience. *BioScience* 59:863-873. http://dx.doi.org/10.1525/bio.2009.59.10.8

González-Esquivel, C. E., M. E. Gavito, M. Astier, M. Cadena-Salgado, E. del-Val, L. Villamil-Echeverri, Y. Merlín-Uribe, and P. Balvanera. 2015. Ecosystem service trade-offs, perceived drivers, and sustainability in contrasting agroecosystems in central Mexico. *Ecology and Society* 20(1): 38. http://dx.doi.org/10.5751/ES-06875-200138

Grêt-Regamey, A., S. H. Brunner, J. Altwegg, M. Christen, and P. Bebi. 2013. Integrating expert knowledge into mapping ecosystem services trade-offs for sustainable forest management. *Ecology and Society* 18(3): 34. http://dx.doi.org/10.5751/ES-05800-180334

Haase, D., N. Schwarz, M. Strohbach, F. Kroll, and R. Seppelt. 2012. Synergies, trade-offs, and losses of ecosystem services in urban regions: an integrated multiscale framework applied to the Leipzig-Halle region, Germany. *Ecology and Society* 17(3): 22. http://dx.doi.org/10.5751/ES-04853-170322

Hein, L., K. van Koppen, R. S. de Groot, and E. C. van Ierland. 2006. Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics* 57:209-228. http://dx.doi.org/10.1016/j.ecolecon.2005.04.005

Helming, K., and M. Pérez-Soba. 2011. Landscape scenarios and multifunctionality: making land use impact assessment operational. *Ecology and Society* 16(1): 50. [online] URL: http://www.ecologyandsociety.org/vol16/iss1/art50/

Ibarra Manriquez, G., J. L. Villaseñor, and R. Duran Garcia. 1995. Riqueza de especies y endemismo del componente arbóreo de la Península de Yucatán, México. Species richness and endemisms of trees of Yucatan Peninsula, Mexico. *Boletin de la Sociedad Botánica de México* 57:49-77.

Jones, N. A., H. Ross, T. Lynam, P. Perez, and A. Leitch. 2011. Mental models: an interdisciplinary synthesis of theory and methods. *Ecology and Society* 16(1): 46. [online] URL: http://www.ecologyandsociety.org/vol16/iss1/art46

Kari, S., and K. Korhonen-Kurki. 2013. Framing local outcomes of biodiversity conservation through ecosystem services: a case study from Ranomafana, Madagascar. *Ecosystem Services* 3:e32–e39. http://dx.doi.org/10.1016/j.ecoser.2012.12.003

King, E. G., and R. J. Hobbs. 2006. Identifying linkages among conceptual models of ecosystem degradation and restoration: towards an integrative framework. *Restoration Ecology* 14:369-378. http://dx.doi.org/10.1111/j.1526-100X.2006.00145.x

Kline, J. D., and M. J. Mazzotta. 2012. *Evaluating tradeoffs among ecosystem services in the management of public lands*. General Technical Report PNW-GTR-865. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Koch, E. W., E. B. Barbier, B. R. Silliman, D. J. Reed, G. M. Perillo, S. D. Hacker, E. F. Granek, J. H. Primavera, N. Muthiga, S. Polasky, et al. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment* 7:29-37. http://dx.doi.org/10.1890/080126

Labiosa, W. B., W. M. Forney, A.-M. Esnard, D. Mitsova-Boneva, R. Bernknopf, P. Hearn, D. Hogan, L. Pearlstine, D. Strong, H. Gladwin, et al. 2013. An integrated multi-criteria scenario evaluation web tool for participatory land-use planning in urbanized areas: the Ecosystem Portfolio Model. *Environmental Modelling & Software* 41:210-222. http://dx.doi.org/10.1016/j.envsoft.2012.10.012

Laterra, P., M. E. Orúe, and G. C. Booman. 2012. Spatial complexity and ecosystem services in rural landscapes. *Agriculture, Ecosystems & Environment* 154:56-67. http://dx.doi.org/10.1016/j.agee.2011.05.013

Leach, M., I. Scoones, and A. Stirling. 2010. *Dynamic sustainabilities: technology, environment, social justice*. Earthscan, Bristol, UK.

López-Ridaura, S., O. Masera, and M. Astier. 2002. Evaluating the sustainability of complex socio-environmental systems. The MESMIS framework. *Ecological Indicators* 2:135-148. http://dx.doi.org/10.1016/S1470-160X(02)00043-2

Lynam, T., W. de Jong, D. Sheil, T. Kusumanto, and K. Evans. 2007. A review of tools for incorporating community knowledge, preferences, and values into decision making in natural resources management. *Ecology and Society* 12(1): 5. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art5/

Martín-López, B., I. Iniesta-Arandia, M. García-Llorente, I. Palomo, I. Casado-Arzuaga, D. G. del Amo, E. Gómez-Baggethun, E. Oteros-Rozas, I. Palacios-Agundez, B. Willaarts, et al. 2012. Uncovering ecosystem service bundles through social preferences. *PLoS ONE* 7:e38970. http://dx.doi.org/10.1371/journal.pone.0038970

Matthews, N. 2012. Water grabbing in the Mekong basin—an analysis of the winners and losers of Thailand's hydropower development in Lao PDR. *Water Alternatives* 5:392-411.

May, R. M. 1977. Thresholds and breakpoints in ecosystems with a multiplicity of stable states. *Nature* 269:471-477. http://dx.doi.org/10.1038/269471a0

Mouchet, M. A., P. Lamarque, B. Martín-López, E. Crouzat, P. Gos, C. Byczek, and S. Lavorel. 2014. An interdisciplinary methodological guide for quantifying associations between ecosystem services. *Global Environmental Change* 28:298-308. http://dx.doi.org/10.1016/j.gloenvcha.2014.07.012

Müller, F., R. de Groot, and L. Willemen. 2010. Ecosystem services at the landscape scale: the need for integrative approaches. *Landscape Online* 23:1-11.

Muradian, R., M. Arsel, L. Pellegrini, F. Adaman, B. Aguilar, B. Agarwal, E. Corbera, D. Ezzine de Blas, J. Farley, G. Froger, et al. 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters* 6:274-279. http://dx.doi.org/10.1111/j.1755-263X.2012.00309.x

Mwampamba, T. H. 2009. *Forest recovery and carbon sequestration under shifting cultivation in the Eastern Arc Mountains, Tanzania: landscape and landuse effects*. University of California Davis, Davis, California, USA.

Mwampamba, T. H., and M. W. Schwartz. 2011. The effects of cultivation history on forest recovery in fallows in the Eastern Arc Mountain, Tanzania. *Forest Ecology and Management* 261:1042-1052. http://dx.doi.org/10.1016/j.foreco.2010.12.026

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. Cameron, K. M. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, et al. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and trade-offs at landscape scales. *Frontiers in Ecology and the Environment* 7:4-11. http://dx.doi.org/10.1890/080023

Nelson, E., S. Polasky, D. J. Lewis, A. J. Plantinga, E. Lonsdorf, D. White, D. Bael, and J. J. Lawler. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proceedings of the National Academy of Sciences of the United States of America* 105:9471-9476. http://dx.doi.org/10.1073/pnas.0706178105

Perfecto, I., and J. Vandermeer. 2010. The agroecological matrix as alternative to the land-sparing/agriculture intensification model. *Proceedings of the National Academy of Sciences of the United States of America* 107:5786-5791. http://dx.doi.org/10.1073/pnas.0905455107

Plieninger, T., C. Bieling, B. Ohnesorge, H. Schaich, C. Schleyer, and F. Wolff. 2013. Exploring futures of ecosystem services in cultural landscapes through participatory scenario development in the Swabian Alb, Germany. *Ecology and Society* 18(3): 39. http://dx.doi.org/10.5751/ES-05802-180339

Polasky, S., E. Nelson, J. Camm, B. Csuti, P. Fackler, E. Lonsdorf, C. Montgomery, D. White, J. Arthur, B. Garber-Yonts, et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biological Conservation* 141:1505-1524. http://dx.doi.org/10.1016/j.biocon.2008.03.022

Reed, M. S. 2008. Stakeholder participation for environmental management: a literature review. *Biological Conservation* 141:2417-2431. http://dx.doi.org/10.1016/j.biocon.2008.07.014

Rietkerk, M., P. Ketner, L. Stroosnijder, and H. H. T. Prins. 1996. Sahelian rangeland development; a catastrophe? *Journal of Range Management* 49:512-519.

Rodríguez, J. P., T. D. Beard, Jr., E. M. Bennett, G. S. Cumming, S. J. Cork, J. Agard, A. P. Dobson, and G. D. Peterson. 2006. Trade-offs across space, time, and ecosystem services. *Ecology and Society* 11(1): 28. [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art28/

Roe, D., and M. Walpole. 2010. Whose value counts? Trade-offs between biodiversity conservation and poverty reduction. Pages 157-174 *in* N. Leader-Williams, W. Adams, and R. Smith, editors. *Trade-offs in conservation: deciding what to save*. Wiley-Blackwell, Oxford, UK. http://dx.doi.org/10.1002/9781444324907.ch9

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591-596. http://dx.doi.org/10.1038/35098000

Smith, F. P., R. Gorddard, A. P. N. House, S. McIntyre, and S. M. Prober. 2012. Biodiversity and agriculture: production frontiers as a framework for exploring trade-offs and evaluating policy. *Environmental Science & Policy* 23:85-94. http://dx.doi.org/10.1016/j.envsci.2012.07.013

Stone-Jovicich, S. S., T. Lynam, A. Leitch, and N. A. Jones. 2011. Using consensus analysis to assess mental models about water use and management in the Crocodile River catchment, South Africa. *Ecology and Society* 16(1): 45. [online] URL: http://www.ecologyandsociety.org/vol16/iss1/art45/

Sunderlin, W. D., A. Angelsen, B. Belcher, P. Burgers, R. Nasi, L. Santoso, and S. Wunder. 2005. Livelihoods, forests, and conservation in developing countries: an overview. *World Development* 33:1383-1402. http://dx.doi.org/10.1016/j.worlddev.2004.10.004

Turnbull, L., B. P. Wilcox, J. Belnap, S. Ravi, P. D'Odorico, D. Childers, W. Gwenzi, G. Okin, J. Wainwright, K. K. Caylor, et al. 2012. Understanding the role of ecohydrological feedbacks in ecosystem state change in drylands. *Ecohydrology* 5:174-183. http://dx.doi.org/10.1002/eco.265

White, C., B. S. Halpern, and C. V. Kappel. 2012. Ecosystem service trade-off analysis reveals the value of marine spatial planning for multiple ocean uses. *Proceedings of the National Academy of Sciences of the United States of America* 109:4696-4701. http://dx.doi.org/10.1073/pnas.1114215109

Yahdjian, L., O. E. Sala, and K. M. Havstad. 2015. Rangeland ecosystem services: shifting focus from supply to reconciling supply and demand. *Frontiers in Ecology and the Environment* 13:44-51. http://dx.doi.org/10.1890/140156

Appendix 1. Governing equations and parameters used to calculate production possibility frontiers (PPFs) in Figure 3, which provides a stylized representation of the tradeoffs between agricultural production and species richness at the national scale, and at local scales within 3 different ecological regions, in Mexico.

General species-area and agricultural production-area functions

The trade-off between the biodiversity or species richness (S) that can be sustained from land area in natural habitat (A_H) on the one hand, and the agricultural production (P) that can be derived from land area dedicated to crops (A_C) can be expressed mathematically as follows: The total land area (A_T) under consideration can be partitioned between habitat (A_H) and crop (A_C) production such that

$$A_T = A_H + A_C. \qquad \text{(A1.1)}$$

Both species richness and agricultural production are a function of area such that

$$S = \alpha A_H^z \qquad \text{(A1.2)}$$
$$P = \beta A_C, \qquad \text{(A1.3)}$$

where z is the slope of the log-log relationship between S and A_H, α is a constant (y-intercept) and β is the crop yield per unit area. The relationship between species richness (S) and agricultural production (P) can thus be written as:

$$S = \alpha(1-P/\beta)^z \qquad \text{(A1.4)}$$

Calculations of stylized PPFs for Figure 3

Figure 3 provides an example of an aggregated land area, in this case the country of Mexico, that is subdivided into three regions (i=1,2,3), each of which has different biophysical capacities to support agriculture and biodiversity. The full spatial extent of the aggregated region, A_T, is set to unity; m_i is the fraction of total land area apportioned to the regional subdivision i, and these values also sum to unity.

$$m_1 + m_2 + m_3 = A_T = 1 \qquad \text{(A1.5)}$$

The regional-level coefficient α_i is the y-intercept log-log species-area relationship for region i, and influences the total number of species that can accumulate in a given area of land in that region; z_i is the slope of the log-log species-area relationship for region i.

Crop area, A_{Ci}, for each region i is the total land area less the area conserved as habitat for biodiversity A_{Hi},

$$A_{Ci} = m_i - A_{Hi} \tag{A1.6}$$

and total crop area for the aggregated regions, rearranged from (A1.1) is $A_{CT} = A_T - A_{HT}$.

Each region has a different capacity to produce food, determined by the coefficient, β_i. Total crop productivity at the aggregate national scale, P_T, can be written as

$$P_T = \beta_1 A_{C1} + \beta_2 A_{C2} + \beta_3 A_{C3} \tag{A1.7}$$
$$P_T = \Sigma\left[\beta_i A_{ci}\right]$$

Each region has a different capacity for maintaining species diversity, determined by parameters α_i and z_i. The total number of species (S_T) that can accumulate in the nation if all land is conserved as habitat for biodiversity is the sum of each parcel:

$$S_T = a_1(m_1 A_H)^{z_1} + a_2(m_2 A_H)^{z_2} + a_3(m_3 A_H)^{z_3} \tag{A1.8}$$
$$S_T = \Sigma\left[a_i(m_i A_H)^{z_i}\right]$$

Table A1.1: Parameters used to simulate three distinct ecological regions (i=1,2,3) and generate PPFs in Figure 3.

Region (i)	Region label in Figure 3	α_i	z_i	β_i	m_i
1	A	20	0.3	10	0.25
2	B	10	0.26	15	0.35
3	C	5	0.27	20	0.4

R-code to run the example shown in the paper can be found at https://github.com/cavender/Trade-offs.