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Ferdinando Villa
Ikerbasque, the Basque Foundation for Science

Brian Voigt
University of Vermont

Jon D. Erickson
University of Vermont

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Author for correspondence:
Ferdinando Villa
e-mail: ferdinando.villa@bc3research.org

New perspectives in ecosystem services science as instruments to understand environmental securities

Ferdinando Villa[1], Brian Voigt[2] and Jon D. Erickson[2]

[1]Basque Centre for Climate Change (BC3), IKERBASQUE, Basque Foundation for Science, 48008 Bilbao, Spain
[2]Gund Institute for Ecological Economics, Rubenstein School of Environment and Natural Resources, University of Vermont, Burlington, VT 05405, USA

As societal demand for food, water and other life-sustaining resources grows, the science of ecosystem services (ES) is seen as a promising tool to improve our understanding, and ultimately the management, of increasingly uncertain supplies of critical goods provided or supported by natural ecosystems. This promise, however, is tempered by a relatively primitive understanding of the complex systems supporting ES, which as a result are often quantified as static resources rather than as the dynamic expression of human–natural systems. This article attempts to pinpoint the minimum level of detail that ES science needs to achieve in order to usefully inform the debate on environmental securities, and discusses both the state of the art and recent methodological developments in ES in this light. We briefly review the field of ES accounting methods and list some *desiderata* that we deem necessary, reachable and relevant to address environmental securities through an improved science of ES. We then discuss a methodological innovation that, while only addressing these needs partially, can improve our understanding of ES dynamics in data-scarce situations. The methodology is illustrated and discussed through an application related to water security in the semi-arid landscape of the Great Ruaha river of Tanzania.

1. Introduction

Societal demand for food, water, energy and other life-sustaining resources is growing at unprecedented levels [1]. Well-functioning ecosystems are essential to sustain the supplies of resources critical to health, livelihoods and production, such as water and food. As a result, governance directed to the maintenance or improvement of such supplies is often targeting maintenance or restoration of ecosystem function, with instruments that vary from direction regulation to financial incentives.

The complex and nonlinear dynamics of coupled human–natural systems [2] are of great concern in addressing the security of such essential supplies (referred to as *environmental securities* from now on). Complex systems exhibit thresholds and tipping points that make them notoriously difficult to predict and manage; changes that arise in the ecological system—such as a decline in pollinators owing to climate change—can propagate through societies and economies to cause catastrophic economic and social transitions [2]. Social systems can absorb and buffer changes until a threshold is reached, then react dramatically with behaviours (such as riots and political unrest) that determine structural changes of such entity to make any previous understanding useless. The nature of the dependence on ecosystems is different, both in meaning and in the implications of shortfalls, for the world's poor versus the rich, but no sector of society is invulnerable to the consequences of ecological change.

Predicting the effectiveness of environmental management in securing sufficient, fair and sustainable supplies is therefore difficult without adequate scientific understanding. Owing to the pressing need to support these securities, understanding the modes, rates and scales of the dependence of societies on natural ecosystems has become a crucial task for twenty-first century science.

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Among the many areas of science concerned with coupled human–natural systems, the relatively recent perspective of ecosystem service (ES) assessment [3,4] offers a joint consideration of (i) the biophysical processes of service provision; (ii) the economic outcomes of service uptake by society; and (iii) the social implications of service demand, utility and equitable distribution [5]. Delivering on the promise of robust environmental securities understanding through a fully quantitative and rigorous account of ES is an important test for the ability of science to meet societal needs. Carpenter *et al.* [2] issued a widely cited challenge to develop methods that can account for internal feedbacks, multiple scales and uncertainty in ES. Can this challenge be met in practice, and what are some practical scientific principles that can help us account for supplies of life-sustaining goods in a way that can usefully inform the management and decision-making aspects of environmental securities?

This article investigates the case for ES as a practical scientific framework for the study of environmental securities, and discusses the state-of-the-art and the unmet needs in the light of the possible application to this task. We describe the ARtificial Intelligence for Ecosystem Services (ARIES) [6–8] methodology as an example of a way forward to meet some of these needs, and discuss results of a preliminary application to water security in Tanzania as an example.

2. Ecosystem services science versus environmental securities

ES, the benefits humans obtain from ecosystems [3,4], have gained a central role as a conceptual framework for sustainable development. The ES notion became popular with the release of the United Nations 2005 Millennium Ecosystem Assessment (MEA) [9], a 4-year study involving more than 1300 scientists worldwide. Since then, ES have become central in the environmental policy discourse, and the research contributions on their quantification, valuation and significance in policy-making have multiplied at great speed [10,11]. Programmes, such as the UK-based Ecosystem Services for Poverty Alleviation (ESPA) [12], reflect the attention to ES from communities concerned with poverty alleviation and environmental securities.

The system dynamics of ES can be summarized as the interaction of the three processes of *production* (of beneficial goods or services at the ecosystem side), *use* (uptake by beneficiary groups in societies) and *flow* (transmission of benefits from nature to humans) [8]. Both the ES that supply essential resources ('provisioning' services, such as food or water supply) and those that prevent unwanted outcomes ('regulation' services, such as flood and erosion control by vegetation) can be conceptualized along these lines [8]. This system view lends itself well to the quantification and investigation of the mechanistic dynamics of the relationship between nature and society.

Discussing food security, the World Health Organization articulates the three 'pillars' of security along the dimensions of (i) *access* (ability to obtain appropriate foods), (ii) *availability* (sufficient quantities of food available on a consistent basis) and (iii) *use* (appropriate use for nutritional and health needs). These criteria, which are easily generalized to water and other critical resources, can be characterized as a society-centric view of the three elements of ES dynamics listed above. An ES-centric perspective appears, in our view, appropriate to providing a mechanistic foundation that can quantitatively describe the life cycle of critical resources, clarify the links between the systems involved and illuminate the trade-offs involved in the maintenance of the related environmental securities. The abundant research around ES also provides useful discussion of the direct and the indirect effects of economic incentives to sustainability [13], distributional equity [14] and trade-offs [15]. But are current ES theory and practice capable of handling such complexity?

We list below the aspects of a science of ES that we feel are necessary to adequately inform understanding, management and restoration of environmental securities, but are not emphasized to the necessary level in the current state of the art. We present them as a list of *desiderata* for ease of reference; the following sections will discuss the state of the art of ES science using these goals as a reference point.

(a) Goal 1: maintain focus on the coupled human–natural system

ES are expressed through a dynamic *transfer of benefits* from nature to society. The mode, rate and scale of this transfer are crucial to the understanding of environmental securities. In particular, it is impossible to account for ES-mediated securities without a full account of the ES *beneficiaries*, including addressing both *who* and *where*. A focus on beneficiaries is also necessary in choosing the proper scale for ES studies; in other words, the definition of the *benefit-shed* should be determined by the location of the beneficiaries and the scale of influence of the natural systems on them, on a case-by-case basis.

(b) Goal 2: provide appropriately quantitative information

Understanding and managing critical thresholds of life-sustaining resources is more important for addressing securities than establishing monetary or non-monetary value, whose relation to declining supply is highly nonlinear in the vicinity of critical thresholds [16,17]. Methods must be capable of providing sufficient quantitative accuracy in assessing supply and demand, so that critical situations can be anticipated before they are encountered. The quantitative accuracy must extend to the temporal dynamics of the resulting description, in order to capture thresholds and tipping points that are crucial to security [5]. Thresholds can be subtle, and the need for accuracy in their definition increases as the supply of ES approaches critical levels; accuracy in system description is therefore crucial to securities, whose zone of interest is around critical thresholds of supply.

(c) Goal 3: explicitly address both potential and actual values

In order to properly address issues of sustainability, ES analysis must not be limited to assessing the potential supply of services; it is essential that *actually accrued* benefits are differentiated from potential benefits in a quantitative and spatially explicit manner. In doing so, unused potential supplies can be identified and considered in an analysis of alternative management schemes for maximizing security. A spatially explicit approach is necessary not only to properly model the accrual of benefits by specific beneficiaries, but also to provide crucial information whenever issues of distributional equity between or among different stakeholders are to be addressed.

(d) Goal 4: address trade-offs in a dynamic, scale-aware perspective

The MEA had extensive discussion of trade-offs [18], which can arise from management decisions owing to interaction with natural or social processes other than those targeted, and cause conflicting effects for different beneficiary groups [15]. Many assessment methods claim to address trade-offs [10], but the mainstream approach is to compare static ES accounts generated by a single descriptor of change, often land cover type. The trade-offs of interest to the security debate, either between different ES or between different social groups in need of them, are deeply affected by system dynamics and change radically with varying temporal and spatial scales [15]. ES models of interest to environmental securities (and decisions based on them) must incorporate multi-scale trade-offs dynamically and quantitatively. For example, deforestation for agriculture leads to trade-offs between the provision of food in the short-term and the eventual increase of run-off and reduction of erosion regulation, which impacts flood risk and water supply and quality in the longer-term. Such trade-offs between provisioning and regulating services are well recognized [19], but they can be addressed only quantitatively in a dynamic, spatially explicit and scale-aware perspective; their quantitative accounting over variable scales of time and space is required to understand the ways that local and global changes can influence ES outcomes, and their ability to sustain acceptable levels of environmental securities.

(e) Goal 5: leave the definition of value to the decision-maker

Decisions are necessarily based on an assessment of value. Spurred by the desire of making ES a *lingua franca* for science-based policy-making, much discussion on ES has focused on their value, most often interpreted economically [20]. However, the definition of value is highly context-dependent and is ultimately a multiple objectives problem [21]. Establishing value in real life almost invariably requires negotiating difficult trade-offs. For this reason, assessment methods should not be tied to a specific notion of value, but allow for a flexible statement of the most appropriate 'objective function' to use. In spite of a long-standing emphasis on economic valuation in ES literature, no ES definition has been tied directly to economic value, and the pitfalls of the economic interpretation of ES value have been often noted [21]. An economic perspective, when necessary, should not interpret value simply as an economic ranking of the most convenient options but account for the sustainability of household livelihoods and of the larger economies within which they function.

The field of environmental securities is ultimately driven by a concern over well-being and social equity, and naturally ES science addresses only the ecological and social mechanisms behind the production and distribution of resources. We argue here that an accurate biophysical—and, when appropriate, economic—account of ES can provide a crucial foundation for a scientific approach to security as long as the above criteria are met. The rest of this contribution is a discussion of the state-of-the-art and emerging trends in ES in the light of their application to environmental securities.

3. Ecosystem services assessment methods: state-of-the-art and new perspectives

(a) State of the art

The aspects of availability, use and access have not been emphasized equally in ES literature. It has been widely recognized that, in the MEA and beyond, ES research has given more emphasis to the ecosystem side than the social [22–24]. At the same time, many have discussed the difficulties stemming from lack of consideration of the spatial connection between the ecosystems that provide benefits and the people that enjoy them [7,25–27], suggesting the lack of an adequate formalization of access dynamics.

The dominant approach for the modelling of *provision* is the application of ecological production functions [28], which has resulted in many ES assessments consisting solely of the accounting of beneficial resources or protective structures generated by ecosystems. According to Tallis *et al.* [22, p.] '... the science of ecology made huge advances when it began to consider dispersal and the importance of movement in governing the dynamics of ecological communities. However, the science of ESs has not yet made this transformation, and as a result typically depicts ESs as site-bound on static maps'. Ecological production functions [28] quantify an ecosystem's ability to supply social benefits—the necessary starting point of an ES analysis—but do not reflect the locations of beneficiaries or the spatial and temporal flow of services; as such, they quantify only *in situ* or theoretical service provision.

Recommendations to identify 'final ecosystem goods and services' [29,30] in recent literature highlight the need for a clearer identification of the *use* side, with the explicit identification of beneficiary groups for modelling and valuation [24,31]. An emphasis on beneficiaries also eliminates the frequently mentioned problem of 'double counting' [23,24] which results from independently accounting for benefits of ecological processes that belong to the same chain of provision (e.g. pollination and agricultural food supply). In the specific case of environmental securities (e.g. food, water), attention to the use side is crucial, as the focus is obviously on the actual well-being of social groups rather than on potential values of ecosystem-produced goods and services.

Attention to the modalities of *access* and uptake of ES by societies was sought by Ruhl *et al.* [25] and Fisher *et al.* [26], who classified the principal patterns of transmission of a service from provision to use areas, reflecting the understanding that ecosystems and their beneficiaries are often not co-located. Systematic quantitative methods to measure and map ES flows[1] have begun to appear [8], but have not entered mainstream practice. Failing to consistently describe, quantify and map such flows hampers the application of ES concepts to policy-making: despite the emphasis placed on value of ES [21], values are not easily understood unless potential benefits can be accounted for separately from actually accrued ones. The consequences of disregarding access are particularly important when discussing the applicability of an ES perspective to environmental security, where the supplies that actually reach social groups of interest are more important than the values resulting from the potential 'carrying capacity' of the environment—the latter often being seen as an end goal in quantification of ES.

The uneven attention to the different elements involved in ES dynamics makes it harder for current ES science to inform environmental securities. Indeed, many common ES assessment methods still avoid the complexity of addressing the spatio-temporal dynamics of ES goods and services as they are produced in nature and consumed by societies [11,32]. While incremental steps away from simple production functions [33–35] and towards more mechanistic descriptions are taken regularly [8,35–38], and efforts are being made to understand the theoretical underpinnings of ES dynamics [10,31,39], ES science does not yet provide enough consideration of the dynamic aspects of ES production and uptake to better understand the consequences of land and resource management, land cover conversion and climate change (among other factors) on the delivery of ES and their ultimate values to society [31,39–41]. The debate on environmental securities, in particular, spans spatial and temporal scales that are typically larger than the local to regional scales for which ES are commonly assessed.

(b) Improving the detail of ecosystem services assessments

The complexity of interactions among ecosystems, societies and economies does not imply that an improved, dynamic account of ES—one that remains amenable to rapid assessment in data- and resource-scarce contexts—cannot be reached. ES benefits are carried by flows of matter or information, such as water, aesthetic information or CO_2. The dynamics of these 'vectors' is not simple, but is often well understood. While a full dynamic understanding of the ecological, social and economic systems that express ES may escape us for many years to come, an understanding of the modes of flow of ES is within reach in many situations. Aspects of ES dynamics that are crucial to understanding securities and remain tractable include (i) modalities of the flow of ES from ecosystems to beneficiaries; (ii) estimation of benefits actually accrued, referenced in relation to both critical supply thresholds and the maximum potentials of provision; and (iii) spatial patterns in the distribution of accrued benefits.

The ARIES approach we describe in this section [6,42] was built around five design criteria that relate, in part, to the *desiderata* listed above (i) improving the underlying narrative to account for ES from the viewpoint of beneficiaries, distinguishing among accrued, potential and theoretical ES values; (ii) explicitly accounting for model uncertainty through probabilistic modelling of ES supply and demand; (iii) explicitly model access to benefits by incorporating the spatial and temporal dynamics of ES flow; (iv) adopting advanced ecoinformatics to enable flexible, data-driven model assembly instead of relying solely on the parametrization of fixed models, whose structural assumptions may depend greatly on context; and (v) supporting a more articulated set of results that hint not only to value, but also to efficiency and distributional equity in both ES provision and use. Of these, we briefly describe the criteria that we believe are more relevant to addressing environmental securities through ES. As will be evident later, the new contributions discussed are mostly relevant to goals (1–3); we discuss the methods versus the full list of *desiderata* in §5.

(i) Focusing on the coupled human–natural system: from services to benefits

Quantification of ES use and flows can differentiate between the benefits actually accrued by societies and the potential production capacity of an ecosystem, which can substantially improve the accuracy of ES valuation [40,43] and increase the value of an ES assessment to decision-makers. Accounting for the beneficiaries and flow of ES explicitly and spatially can also help produce policy-relevant information such as patterns of distribution (winners versus losers) that can serve as an input in addressing issues of equity [40,44].

The MEA language, which reflects an emphasis on the production side, is now ingrained in scientific dialogue to the extent that a redefinition of its key terms is impractical. For this reason, we propose a model of ES that remains compatible with the conceptual framework popularized by the MEA, but extends it via a beneficiary-oriented perspective to improve the potential for value quantification, communication of results and engagement with decision-makers. The first step for this approach is the identification and mapping of well-defined beneficiary groups, each of which is uniquely and unambiguously characterized by type of demand and criteria of value attribution for ES. A *service* in MEA parlance corresponds conceptually to a collection of *benefits*, each of which links one type of good provided by the ecosystem to one class of beneficiary through a specific type of flow. For example, 'water supply' would include one benefit for each modality of water use present in a study area, e.g. water supply for industrial, agricultural, residential and recreational use. Each benefit corresponds to a distinct model, with independent spatial and temporal scaling and value attribution methods chosen according to patterns of production and use (figure 1). A beneficiary-oriented approach also helps to systematically identify the spatial boundaries for ES quantification: each benefit can be defined in space by a supply area (*source-shed*) capable of providing a flow of benefits that intercepts locations of user demand (*benefit-shed*). As ES quantification typically proceeds from societal demand, the delineation of the benefit-shed of interest at the user side allows the source-shed to be inferred through the understanding of the dynamics of flow of each benefit.

(ii) Generation, use and depletion of ecosystem benefits

The main reason why static ES assessment approaches are commonly adopted is the difficulty of mechanistically understanding processes as diverse and complex as sediment regulation, pollination or recreation. For the same reasons, and also for continuity and comparability, ARIES also models its main elements using static functions. However, the values obtained from these functions are then used as initial conditions for dynamic *flow models*, whose algorithms simulate benefit transport and their delivery over time and space.

Figure 2 shows the central conceptual model in ARIES, where benefits produced in *source* regions flow to beneficiaries situated in *use* regions along physical or informational *flow paths*, determined by spatially distributed physical processes. Use of a benefit may be rival (each beneficiary reduces the benefit flow available to others) or non-rival (the use of a service by one beneficiary does not affect its availability for others), and the ecosystems may be supplying a valuable service to users, such as scenic views, food or drinking water (thus contributing to the MEA *provisioning* services) or mitigate the detrimental effect of a physical factor, as in the case of flood water, excess sediment or nutrients, disease or wildfire (thus contributing to *regulating* services according to the MEA; figure 3). Along flow paths, *sink* regions may absorb or deplete

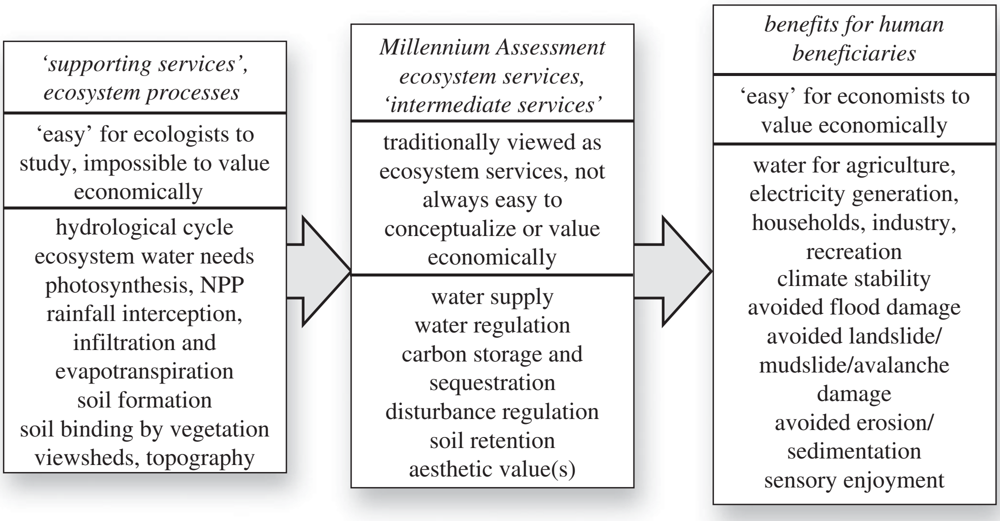

Figure 1. A beneficiaries-based conceptualization of ecosystem services (NPP, net primary productivity).

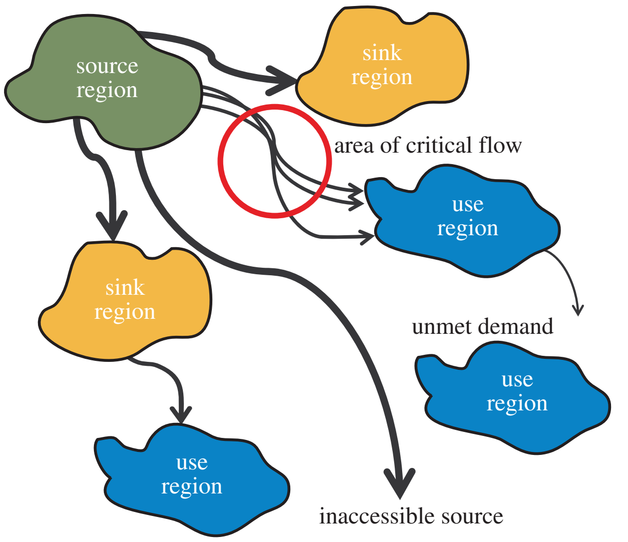

Figure 2. The ARIES conceptual model of ecosystem services provision. (Online version in colour.)

the benefit-carrying medium, preventing it from reaching beneficiaries. The role of sinks is beneficial in the case of regulating benefits and detrimental for provisioning benefits. It is worth noting that the remaining categories of ES identified in the MEA are redundant under this conceptualization: many *cultural* services [9] can be seen as provisioning services that provide benefits via informational flows, whereas *supporting* services are accounted for as part of the causal chain of provision that defines the source model, which always results in the delivery of 'final' benefits [29,30].

Spatial patterns of ES flow are determined by the nature of the medium that carries the benefit (e.g. water, CO_2, visual information), the provisioning or regulating nature of the corresponding service, the physical attributes of the landscape and the presence of natural or anthropogenic features that act as sinks. The amount of medium that actually reaches the beneficiaries (in provisioning services) or is absorbed by ecosystems on its way to them (in regulating services) is the foundation for the assessment of accrued value in ARIES. Areas where the flow trajectories of one or more benefit concentrate are critical to the delivery of the service even if they do not overlap either the source or the use regions [7,8,39,40].

(iii) The dynamic flow of ecosystem benefits in space and time

ARIES quantifies flows using a family of models collectively termed service path attribution networks (SPANs) [8,38, 39,41]. These models implement different means of propagation of the medium carrying the benefit (termed *carrier* in the following), summarized in table 1. Because explicit uncertainty is valuable for decision-making, the initial conditions of source, sink and use are computed with spatial Bayesian network models whenever appropriate [36,39]. The resulting distributions are preserved in the SPAN models, using methods such as Monte Carlo simulation and variance propagation [39], so that uncertainty information remains associated with their outputs and can be evaluated by the final user.

The dynamic modelling of benefit flow, covered in detail in Johnson *et al*. [38,39,41], is handled in a generalized way according to benefit type and flow modality by means of an agent-based approach [45] where agents represent discretized amounts of a carrier transmitted from a source to a use location. This approach offers less mechanistic accuracy compared with dedicated biophysical models such as hydrological or sediment transport models; yet, it is capable of running with probabilistic initial conditions and minimal data requirements, making it more suitable for 'first-cut' rapid assessment. In SPANs, an initial condition is 'evolved' to its final state using a carefully chosen time step, but without attempting to reference the specific time when those will be reached; the simulation stops when the entire area under investigation has been characterized with flow trajectories. This approach can produce a description of the spatial pattern of benefit distribution without requiring long data series for calibration. At the same time, the approach maintains enough temporal characterization to facilitate the investigation of scaling effects and scale-related trade-offs.

The trajectories followed by the carrier and simulated by the SPAN algorithm are used to produce different groups of mapped results. A map that is both obvious, and novel, is that of flow density, showing the amount of carrier that has travelled through the landscape to reach a specific beneficiary group during the course of the simulation. For example, in the flow density maps shown in §4*b* (seen in figure 6) higher values characterize areas that are most critical to the transmission of surface water to the beneficiary groups

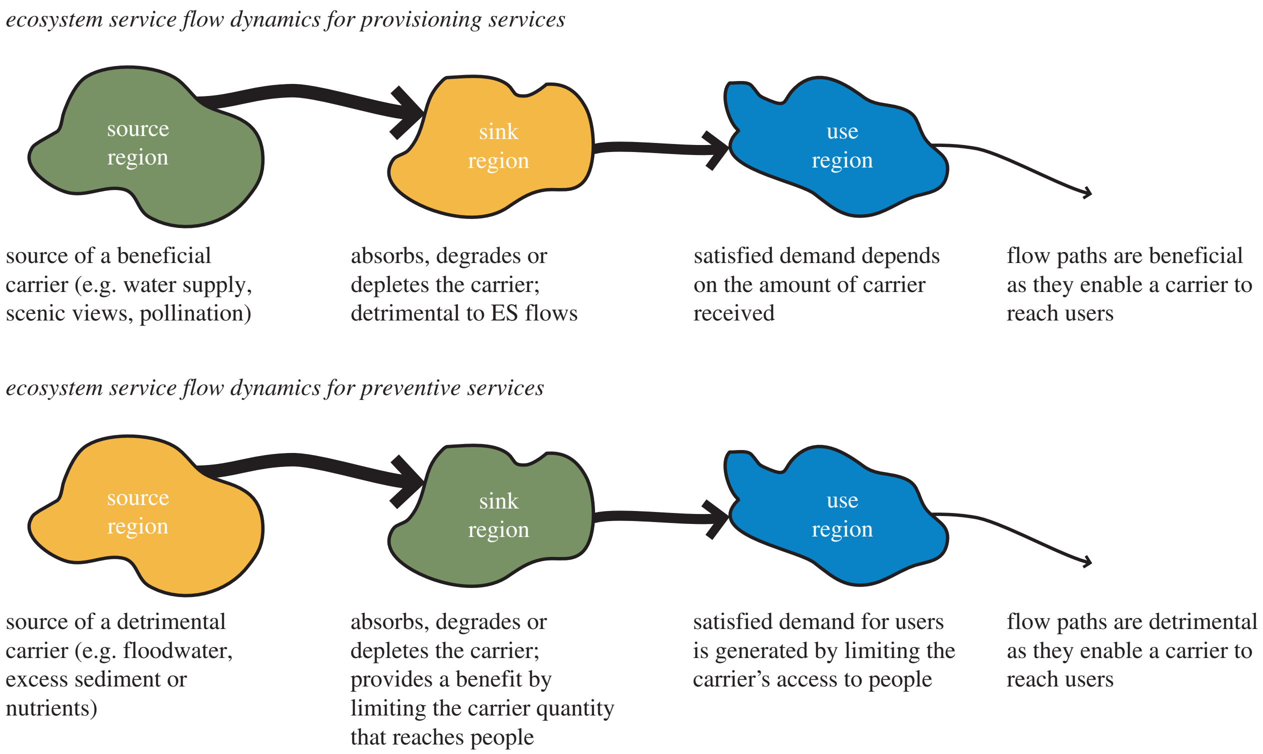

Figure 3. Ecosystem service flows for provisioning and regulating benefits. (Online version in colour.)

Table 1. Flow characteristics of benefits pertaining to selected ecosystem services.

service	type	rivalness	carrier	spatial extent	flow modality
carbon sequestration and storage	provisioning	rival	CO_2	global	atmospheric mixing
riverine flood regulation	regulating	non-rival	run-off	watershed	hydrologic processes
coastal flood regulation	regulating	non-rival	storm surge	littoral zone	wave run-up
nutrient regulation	regulating	non-rival	nutrients in water	watershed	hydrologic processes
sediment regulation	provisioning or regulating	rival	sediment	watershed	hydrologic processes
water supply	provisioning	rival	water	watershed	hydrologic processes
fisheries	provisioning	rival	fish biomass	access to fisheries + fish habitat and migration	network travel
pollination	provisioning	rival	pollen	agricultural basin	pollinator movement
aesthetic viewsheds	provisioning	non-rival	scenic quality (relative ranking)	viewshed	line of sight
open space proximity	provisioning	non-rival	open-space quality (relative ranking)	local	network travel
recreation	provisioning	non-rival	recreational enjoyment (relative ranking)	regional	network travel

under consideration. Such maps can greatly aid planning, as, in most cases, it is difficult to relate flow information to either source or use areas without modelling the flow explicitly. Because each trajectory is stored individually, it is also possible to target an in-depth study of the specific amount of benefit flowing from particular source subareas or to

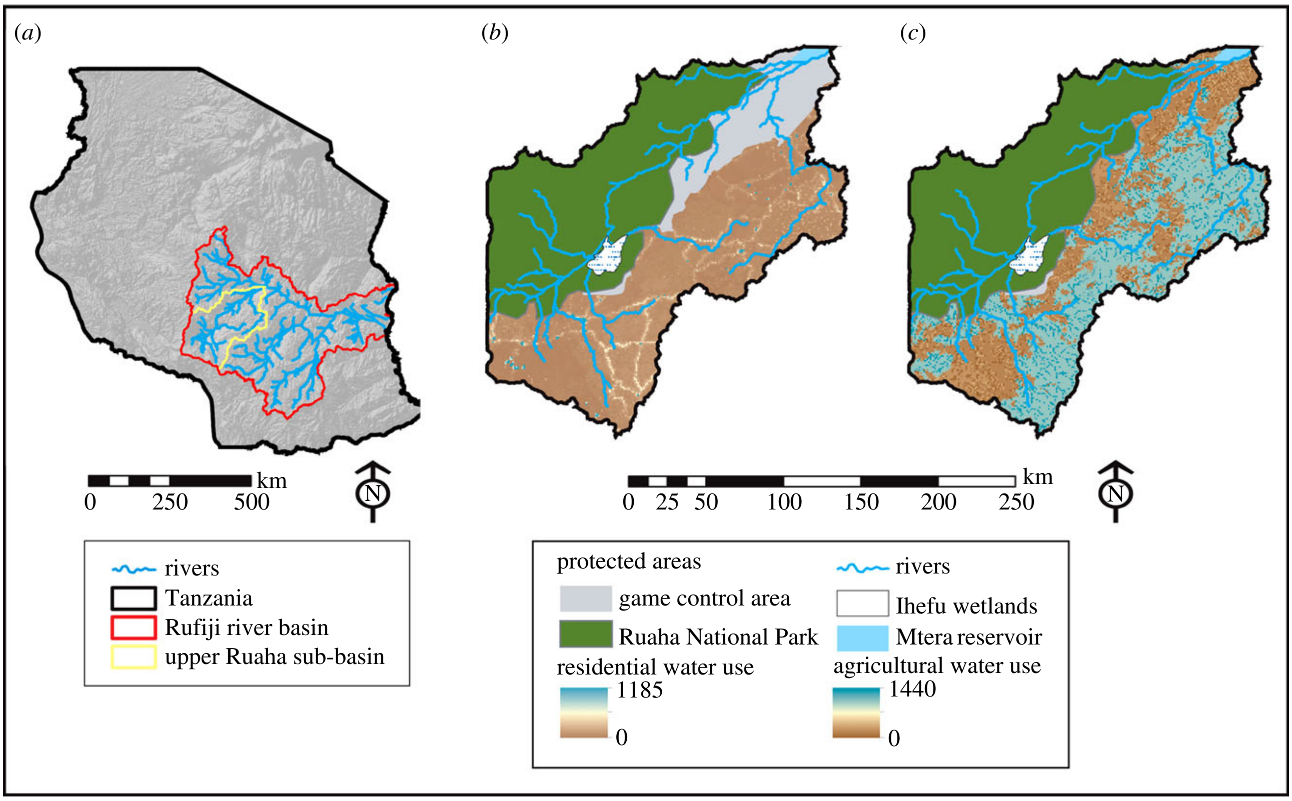

Figure 4. (*a*) Location of the Great Ruaha river watershed in Tanzania; (*b*) residential and (*c*) total agricultural water demand in the watershed.

particular subgroups of beneficiaries. This type of information can greatly aid targeted policy-making, for example, to estimate fees in polluter-pays schemata.

4. An example application: modelling livelihood impacts from disrupted hydrological services

Effectively supporting secure supplies of life-critical environmental goods should link landscape-level management to conservation interventions, aiming at long-term provision of such benefits to local, regional and national beneficiaries. More specifically, an effective integrated management approach should (i) identify possibilities for diverse user groups to benefit from an equitable water allocation scheme; (ii) ensure adequate supplies for economic development (especially as it relates to eco-tourism, and power production for downstream urban areas); and (iii) accommodate for a likely future climate where more erratic annual precipitation and increased temperatures are the norm. This section describes an ARIES study aimed to assist the management of water security in a highly threatened location.

The Great Ruaha river watershed is part of the Rufiji river basin, located in southern Tanzania (figure 4). The Rufiji basin drains an area of approximately 175 000 km^2 (nearly 20% of the land area of Tanzania). The upper reach of the Great Ruaha river feeds a perennial swamp and wetland region (the Ihefu) in the western part of the watershed, then passes through the southeast portion of the Ruaha National Park. The river serves as the primary water source for wildlife and supplies approximately 56% of the total flow for the Mtera hydroelectric power station. The watershed has been the focus of extensive hydrological study in recent decades because of the economic impact of seasonal drought on hydroelectric power production, wildlife tourism and rural livelihoods [46–48]. Irrigated agriculture, uncontrolled water diversions and livestock grazing in wetlands have all contributed to sustained dry periods of this previously perennial river [49] and call for an eco-hydrological approach to restoration [50,51].

Figure 5 is a simplified illustration of the dependencies in the Ruaha river drainage basin, outlining the context of an integrated ES analysis of the water-dependent securities in the region. Irrigated agriculture, grazing, residential consumption and commercial activities all compete for a declining water supply. Large-scale agricultural production in the upper reaches of the watershed and in the vicinity of the Ihefu wetlands limit the flow of water to the lower portion of the drainage. Lankford *et al.* [52] note the relative inflexibility of a water management scheme that fails to account for seasonal and inter-annual variations in the supply of water to the system. Water scarcity in the region threatens a system with complex interactions and trade-offs among disparate user groups and calls for an approach that can ensure equitable distribution across local to national priorities. For pastoralist communities dependent on river water, hydrological disruptions have resulted in: direct and quantifiable impacts on the provision of freshwater ES for drinking, hygiene and agriculture [53]; growing water resource conflicts between agriculturalists, pastoralists and national park interests [52,54,55]; as well as an indirect influence on disease transmission among people, livestock and wildlife [56].

The Health for Animals and Livelihood Improvement (HALI) project was established in 2006 as a multidisciplinary collaboration between Sokoine University of Agriculture, University of California at Davis, University of Vermont and the Wildlife Conservation Society. HALI recently expanded to model the effects of climate variability on livestock health and pastoralist livelihoods, with the long-term goal of identifying landscape-level interventions to adapt to the adverse human and animal health effects of climate change. An integral part of the plan has been the adoption

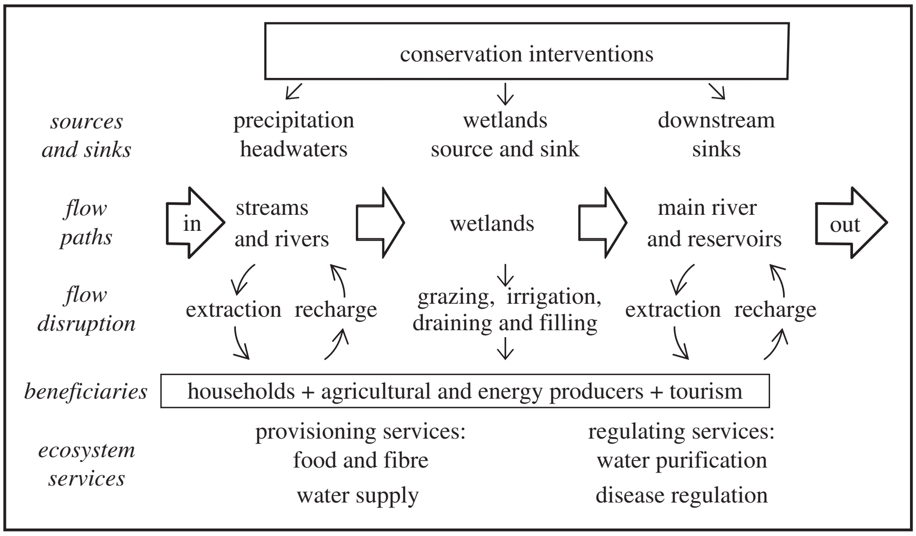

Figure 5. A simplified depiction of the social, economic and ecological dependencies in the Great Ruaha river watershed.

of an ES perspective. The ARIES modelling approach was chosen to help highlight strategies aimed at maintaining inter-sectorial water security in the area. Preliminary results of the ARIES assessment, limited to the water supply ES and obtained rapidly and with minimal data availability, are presented below to illustrate an ES approach to fostering water security that can facilitate the integration of ecological and social factors in future decision-making.

(a) The ARIES model

ARIES was applied to the watershed to summarize the interaction of climate variables and landscape-level processes with livelihood vulnerability and identify the salient traits of this relationship. As explained in §3, we address the dimensions of source, sink and use separately with static, probabilistic models, which provide initial conditions for a flow model that simulates the dynamics of the service by routing water across the landscape. Even costly and time-consuming hydrological modelling can only partially address the connections shown in figure 5. The ARIES model, while remaining an extreme oversimplification of these dependencies, has two advantages: it can be built and run quickly, using publically available, coarse resolution data, and, unique to this approach, can connect the different social groups with the provision side explicitly.

(i) Use model

Lankford *et al.* [52] identify six classes of beneficiaries in the Ruaha river watershed, including rain-fed agricultural producers, irrigated agricultural producers, pastoralist households, subsistence fisheries, eco-tourism operators (and the related wildlife that supports the sector) and power producers. These beneficiaries are arranged in a complex pattern on the landscape, where high hydrological connectivity and competition for scarce resources often leaves those at downstream positions in the watershed at a competitive disadvantage when it comes to satisfying their demand for water. Irrigated agriculture in the upper highlands of the watershed is one of the largest consumptive uses of water, especially during the dry season [49]. Among pastoralist communities in this region, livestock production is a crucial source of income, store of wealth and cornerstone of culture [57,58].

The ARIES demand model considers agricultural demand from livestock (water extracted from rivers at points of minimum distance from settlements) and irrigation. The demand is based on global data of livestock density [59], and published water usage estimates for irrigation needs of different crop types and individual livestock species in the climatic region [60]. Residential water consumption is modelled using global population maps [61] and *per capita* water usage estimates for the region. Figure 4 shows the estimated water demand for residential (*b*) and agricultural (*c*) users. The north side of the Ruaha river watershed is designated as national park, and therefore off-limits to residential and agricultural land uses. A future iteration of the model will account for the water needs of wildlife, a major determinant of the sustainability for the Tanzanian economy through eco-tourism.

(ii) Source model

The main source of water modelled is rainfall. The bulk of the annual rainfall is deposited during the rainy season from November to April. Rainfall totals are positively correlated with altitude with approximately 1600 mm per year at the highest elevation and a more modest 500–700 mm per year at lower elevations [49]. Precipitation on an annual basis was obtained from WorldClim data [62]. Monthly precipitation data will be used in later studies to account for the different seasonal distribution of water benefits. Lack of data made it impossible to account for groundwater exchange, which may be added to the model as data become available.

(iii) Sink model

Factors that diminish the availability of water before it can reach its beneficiaries (other than rival water use from beneficiaries upstream) are simplified to include only evapotranspiration and infiltration in the soil. Lacking data describing these phenomena in the region, the model uses a Bayesian approach [42] to model infiltration as a function of globally available soil, slope and vegetation cover data, whereas vegetation type and cover are used to provide an estimate of evapotranspiration. Both elements are calibrated to data from comparable ecoregions. Sink values are output as probability of occurrence of discretized values which are used directly by the water flow model. Transition to groundwater is another sink effect that may be important, but is not modelled in this case owing to a lack of data.

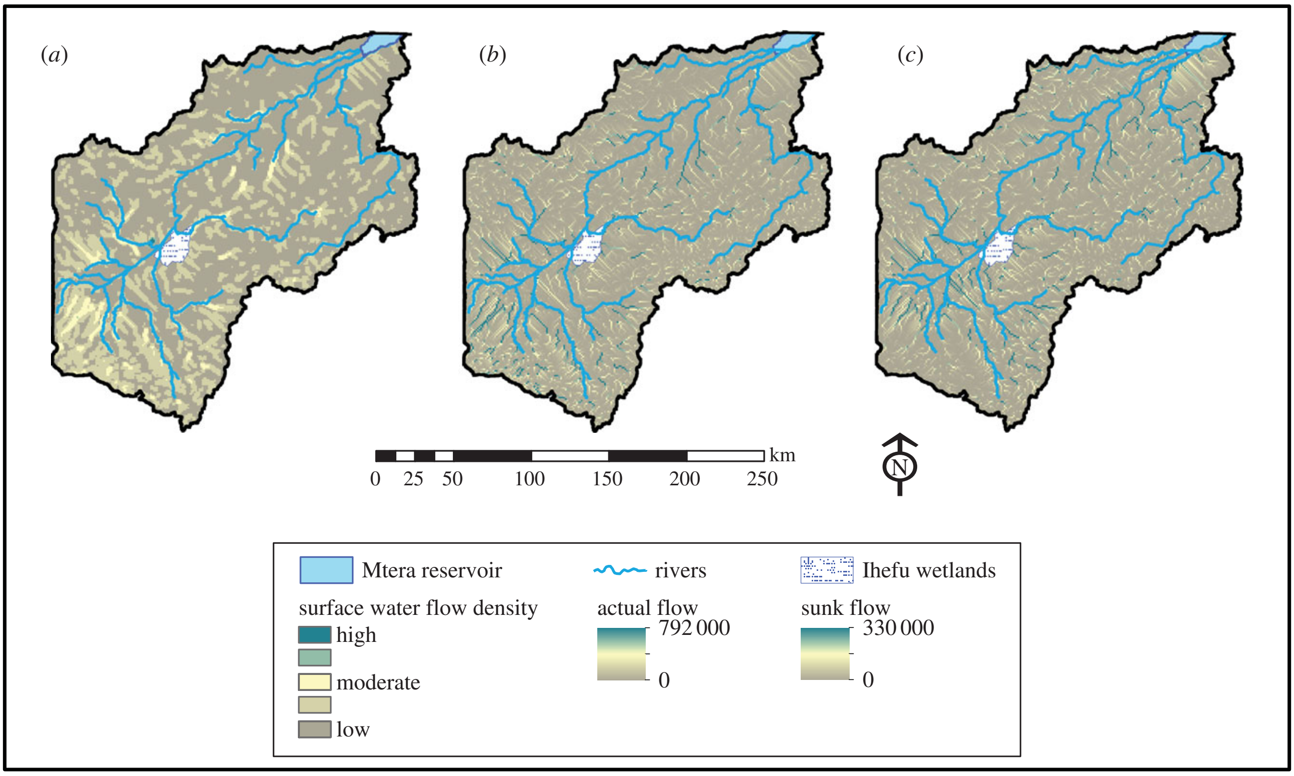

Figure 6. Cumulated annual surface water flow density in the Ruaha river watershed: (*a*) theoretical flow density, (*b*) possible flow without consideration of sink effects and (*c*) actual flow to beneficiaries. Values in mm per year.

(iv) Flow model

The ARIES surface water flow model simulates the movement of water across the landscape and its uptake by beneficiaries [38] using the source, sink and use inputs discussed above. In addition to these inputs, the model uses globally available, high-resolution slope and elevation data for the surface water routing component [63]. While the flow model is greatly simplified compared with a full hydrological model, it allows ARIES to spatially link surface water users to surface water provision. Water flows are summarized at an annual timestep, although improved data availability would allow for the comparison of seasonal flows, without modification to the models. The flow models are capable of using probabilistic information as initial conditions, and preserve the uncertainty information coming from those inputs that have been modelled using a Bayesian approach [39]. This way, the ARIES model outputs come with associated 'uncertainty maps' that show the coefficients of variation of the output distribution at each point. These maps can provide visual guidance to model reliability and offer a measure of caution during data interpretation.

(b) Results from water supply ecosystem services analysis

The ARIES flow model quantifies the connection between the provision of benefits by nature and its use by each beneficiary group. The water paths identified by the model are those that are critical to the supply for each specific configuration of beneficiaries. Each flow path is tagged with the individual value of that path to the beneficiaries it intersects, measured in terms of volume of usable water provided per year. Model outputs[2] delineate flow paths to each different class of beneficiaries, and their comparison may help identify intervention priorities in a stakeholder-specific way.

(i) Flow results

Figure 6 shows three different maps of the cumulated flows of water to all beneficiaries considered. Each flow density map shows the water paths of highest value for water supply in the region, represented as the total volume of water flow in each point over one year. The theoretical flow (figure 6*a*, reclassified into high and low categories from continuous data) is obtained by routing the available rainfall without consideration of sinks or beneficiaries, and can be compared with the *possible* (figure 6*b*) and the *actual* (figure 6*c*) flow density to show theoretical, usable and unusable water paths in the region. The possible surface water flow map (figure 6*b*) represents the maximum water delivery value if there were no sinks to diminish the overall supply, whereas the actual surface water flow (figure 6*c*) quantifies the amount of surface water that both travels across the landscape through a pixel and is used by a beneficiary. The greater the relative actual surface water flow value, the more important it is to the overall delivery of freshwater benefits to the specific beneficiaries included in the model. The southwestern section of the Ruaha river basin has a number of flow paths with relatively high value. Restricting (or limiting) water withdrawals in these locations will help increase water delivery along these same channels to downstream beneficiaries, while allowing new irrigation or pasture land may require additional infrastructure investment to ensure sufficient access to water for the beneficiaries located lower in the watershed. When the possible surface water flow exceeds the actual surface water flow, there may be opportunities to increase water delivery through land-use planning that can mediate sinks (e.g. through conservation efforts or restoration of wetland areas) and influence land-use patterns. Simulated wetland restoration scenarios can be run to obtain indications about the quantitative extent of such improvements and the beneficiary groups more likely to be affected (both positively and negatively) by each policy option.

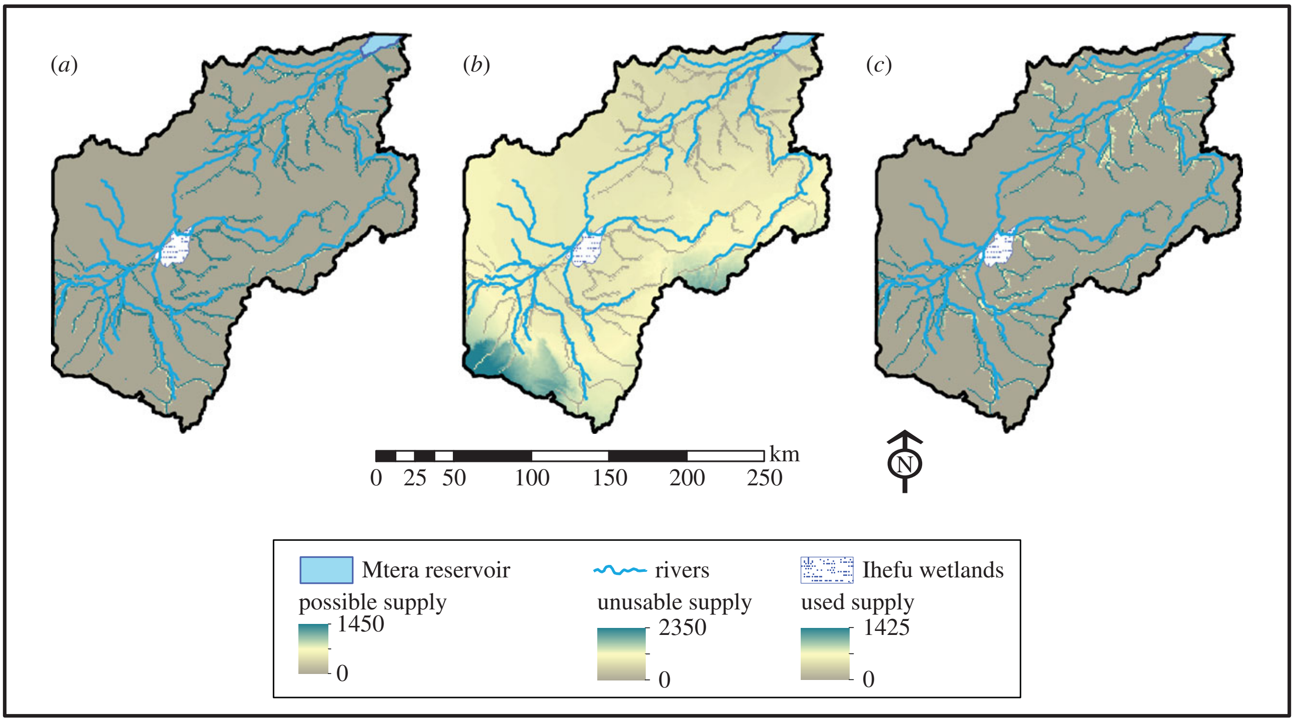

Figure 7. Source values for water supply in the Ruaha river watershed: (*a*) possible source without accounting for sink effects, (*b*) source that is unusable due to lack of flow connections to beneficiaries, and (*c*) actually used source values. Values in mm per year.

(ii) Supply results

Seeing the results from a supply point of view is useful in a policy context where the variable of interest is the accounting of natural capital, as opposed to focusing on whether the needs of stakeholders are met. The set of supply maps provided by ARIES can help understand the potential service delivery and quantify the values of natural capital in the region. In figure 7, the possible (*a*), unusable (*b*) and used (*c*) supply maps are shown, quantifying the portion of the total precipitation that has a chance to flow to beneficiaries (but may not due to the action of sinks, figure 7*a*) alongside the portion that cannot reach any beneficiary owing to the lack of pathways for the water to reach them (figure 7*b*), and the portion that is actually used by humans in the simulated scenario (figure 7*c*). While the absolute quantitative results of such a simplified model are not to be taken literally, they do hint at the scarcity of the water supply in the area: the total usable supply estimated amounts to only 4.53% of the total water balance considering rainfall and all sinks. That amount can only meet approximately 94% of the estimated need of the top 20% of water users. About 97% of the usable supply is used by the 20% top water users, leaving 80% of the users in conditions of grave scarcity over the course of one year. Seasonal unevenness in precipitation, not seen in these cumulated annual results, makes a difficult situation even direr. The substantial amount of unused precipitation (figure 7*b*) suggests that land-use planning scenarios including conservation or reforestation interventions could be investigated with the aim of improving the amounts and the evenness of distribution of usable water across different beneficiary groups.

(iii) Demand results

Figure 8 shows some results of the ARIES water model from the point of view of the beneficiaries, showing two different types of unmet need. The computed water sinks (figure 8*a*) determine the discrepancy between need and provision visible in the unmet demand (figure 8*b*) map. The inaccessible demand (figure 8*c*), by contrast, shows the beneficiaries whose need is unmet, because there is no high-value water pathway that can transport the water to the point of provision for each beneficiary class. Both maps can be useful from a policy perspective to identify trouble spots that can be handled differently. Sinks are typically more sensitive to policy choices than flow routing and amount of precipitation, both of which depend more strongly on factors, such as elevation and climate, which change more slowly and have longer response times to intervention. Comparison of the unmet versus the inaccessible maps can help highlight those beneficiaries with unmet needs whose situation is more likely sensitive to improvement through land-use interventions or infrastructure investments versus those that are likely to remain in need independent of such action.

Interesting indications for management can be derived by comparison of maps, for example computing the ratio between actual and possible values. Such a derived map expresses the relative potential in the area for improvement of water supply through action on sinks, and can offer a rough indication of where intervention may help alleviate scarcity or redistribute benefits to support equal access. Figure 9 suggests that unavailability of water in the lower part of the watershed (situated at the northeast end of the basin) is affected more strongly by sink factors than scarcity of available precipitation. Intervention in such areas is more likely to be effective in guaranteeing that more of the potential supply can be used. Such spatially explicit results can help identify target areas for further scenario investigation and possible policy action.

5. Discussion and perspectives

Several recent opinions and studies have shared our goal 1, incorporating attention to both beneficiaries [64,65] and modes of flow [66–68] in ES analysis. The approach we

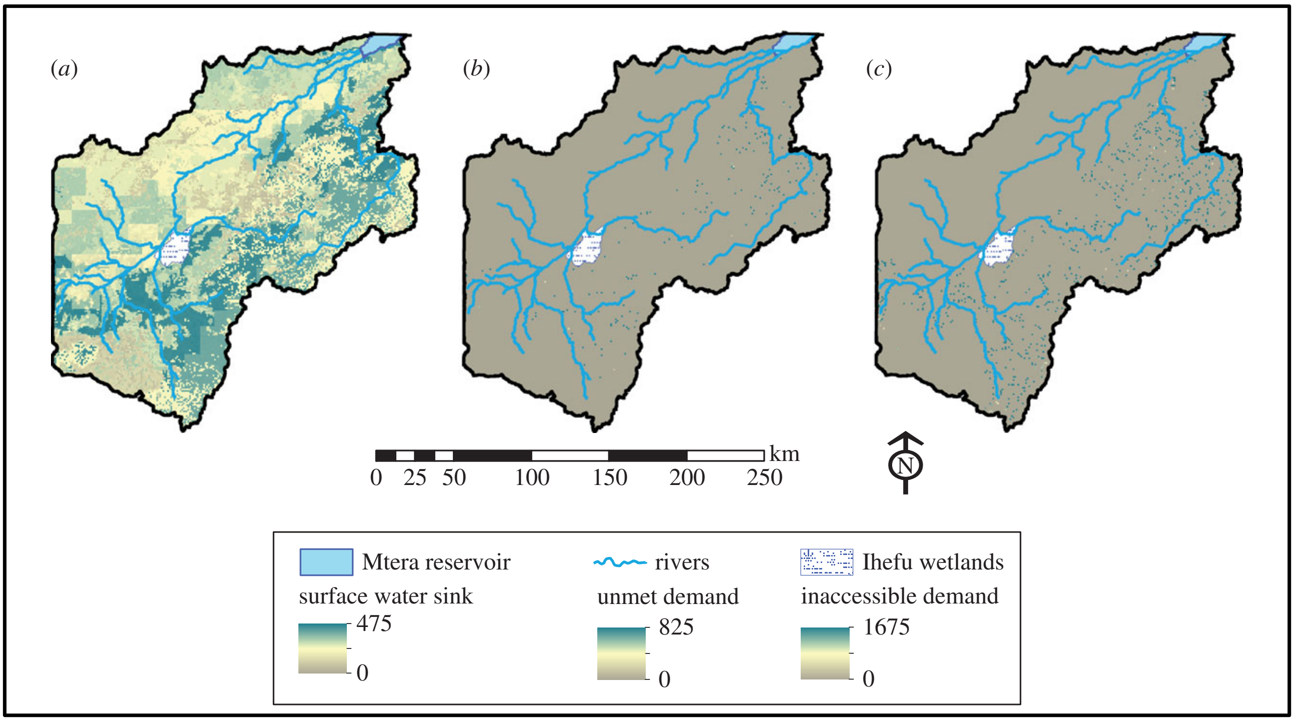

Figure 8. (*a*) Cumulated water supply lost to sink effects in a year; (*b*) yearly unmet water demand from all sectors that could be met if sink effects were reduced and (*c*) demand that cannot be met due to lack of flow paths to beneficiaries. Values in mm per year.

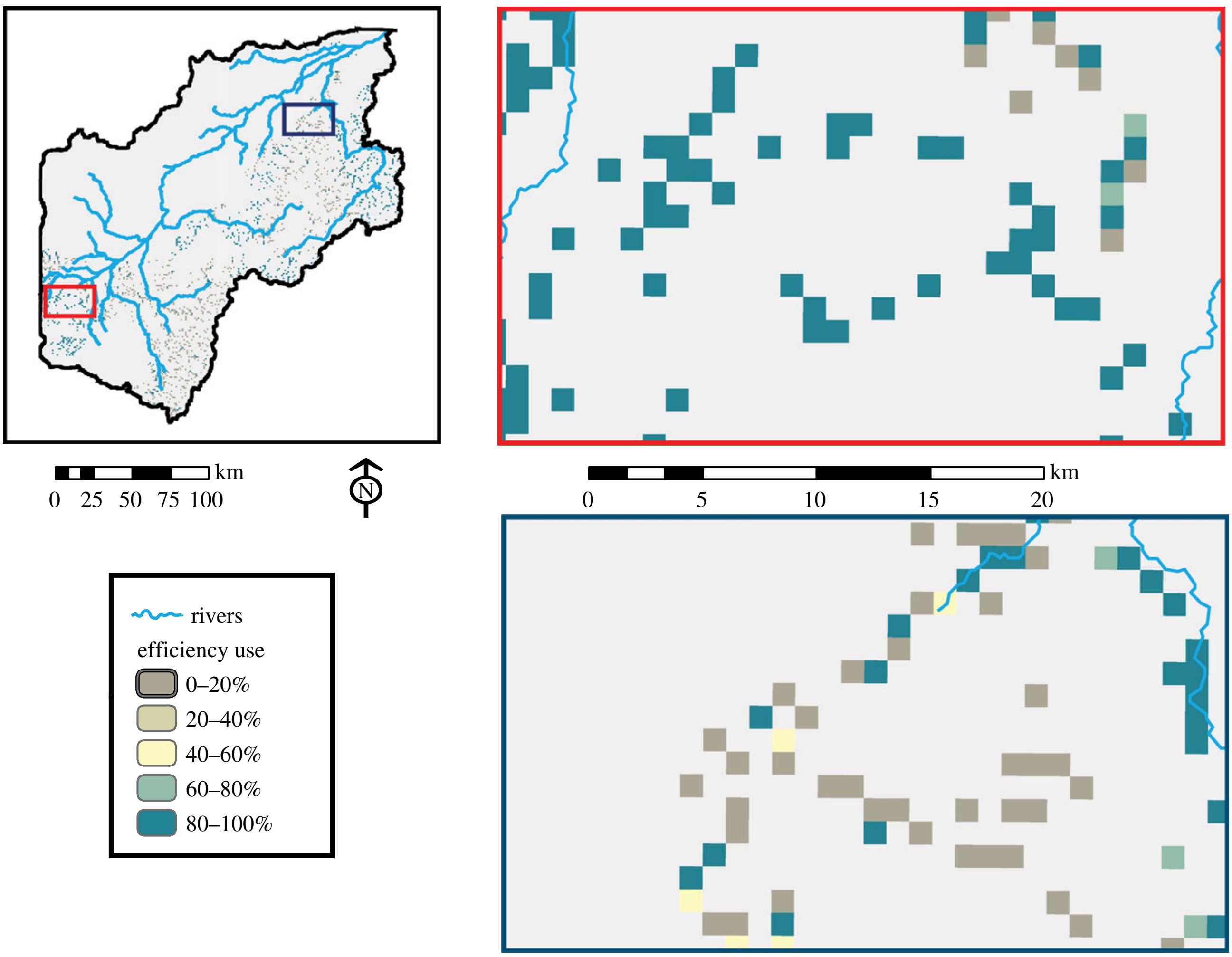

Figure 9. Ratio of actual use to possible use for the 20% top water users in the Ruaha river watershed.

have illustrated makes these elements an integral part of the ES definition and a fundamental design principle for all assessments. The results of an ES assessment incorporating flow analysis and explicit consideration of beneficiaries (summarized in table 2) can more eloquently suggest areas where demand for life-sustaining goods and services is satisfied or

Table 2. Flow model outputs generated by the SPAN algorithm.

flow model outputs	definition	estimation methods
theoretical source, sink, use maps	*in situ* provision, depletion, or use of a service	values calculated without the SPAN model, not considering service flows
possible source, use, flow maps	service dynamics when accounting for flows but not sinks	values calculated by the SPAN model considering flows but not sinks
actual source, sink, use, flow maps	service dynamics when accounting for sinks and flows	values calculated by the SPAN model considering sinks and flows
inaccessible source, sink, use maps	service flows not delivered due to a lack of flow connections	calculated by subtracting actual from theoretical values
blocked source, use, flow maps	service flows blocked by sinks	calculated by subtracting actual from possible values

not, and help target policy actions to specific regions, including those that are crucial not only to the production, but also to the transmission of benefits to societies. Goal 1 seems therefore recognized and reachable.

All-too-common conditions of data scarcity and budgetary constraint remain a limiting challenge in meeting goal 2 of improving the quantitative description of ES dynamics. Being able to decide in such conditions necessarily requires compromising on accuracy, but it remains difficult to understand how critical to the usefulness of an assessment those compromises will be. The advantages of being able to model at least some of the feedbacks and dynamic detail of flow processes are matched by potential disadvantages associated with using sophisticated methods when data are inadequate. For example, quantitative predictions that ignore factors such as groundwater exchange because of lack of data may be highly misleading, even compared with simpler conceptualizations. Studies capable of providing guidelines on what level of detail is most defensible and useful in data-limited situations are urgently needed. Lacking those, it is crucial that quantitative results from any approximate method are only compared with alternative scenarios computed with the same methods, and that the absolute outputs of such models are not used as the sole basis for decision.

An important question concerns how much of the nonlinear dynamics of coupled human–natural systems is (or can be) captured by each method. The adequacy of a method or model to the analysis of dynamic aspects such as thresholds or tipping points is very difficult to validate lacking detailed historical data, and few modelling studies exist that incorporate enough detail (e.g. using detailed and accurately calibrated hydrological assessment) to be a basis for qualitative cross-calibration. Yet, nonlinear dynamics is the source of the catastrophic behaviours of most interest to the assessment of security. Despite extensive research in both ecology and social sciences [69,70] on the dynamic behaviour of highly complex systems, the understanding of their general properties is such that the challenge of Carpenter *et al* [2] is likely to remain unmet as stated. Even in face of these limitations, we argue that the study of spatial connections between ES source and use locations should be integral to any study of ES dynamics. The spatially explicit and temporally referenced linkages provided by flow analysis can become crucial information for land management, protection and restoration. Limiting development in areas of high flow density can help maintain resource security throughout the benefit-shed. Alternatively, lands that maintain strong flow connections that are also marked for development or transformation imply a need for infrastructure development to make up for reductions in service delivery. On the other hand, flows, when accounted for, are only one of many sources of dynamic complexity. Agent-based models [45] that incorporate feedback on the ecological system from the societal side have begun to appear [71] but have not been applied to securities. The ESPA-ASSETS project [5] is committed to do so systematically through extensions of the ARIES methodology.

Goal 3 advocates an often overlooked distinction between potential and actual values. Much criticism has been directed to historical ES valuation studies for producing values that seem unrealistically high, e.g. in Costanza *et al*. [33] for specific ES. Consideration of flow dynamics and sinks, using actually accrued benefits as a base for valuation instead of theoretical provision, can help reassess such studies towards more realistic estimates. In addition to serving as a base for more correct valuation, the ability of computing spatially explicit metrics of potential versus accrued supply can form the basis of a more robust planning process aimed at maximizing service delivery and identifying solutions for delivery shortfalls. In the example presented, understanding where water use is low compared with the potential and where future development may have a disproportionate effect on downstream users, should lead decisions towards either: (i) discouraging new development in these locations because of existing or impending water shortages in a given location; or (ii) recognizing the need for infrastructure development to complement new economic development opportunities that rely on regular, consistent access to freshwater supplies. Further, the identification of areas where potential water supply is high, but actual delivery is low can help define new areas for development that can take advantage of existing, yet underused, water supplies.

The decision-maker's toolbox can be greatly enhanced by the availability of flow results (table 2). *Possible* maps show the amount of value that can be produced or used when accounting for flow connectivity between source and use regions but not for sinks, therefore representing a ceiling of benefit production under the hypothesis that the effect of sinks can be reduced through policy action. *Actual* maps show the value (for provisioning benefits) or damage (for regulating benefits) produced, sunk or used when considering both flow connectivity and sink regions as part of the overall calculation. The *blocked* maps quantify value not accrued because of sinks such as pollution or diversion, or flows of regulating benefits that are beneficially absorbed by ecosystems. Finally, *inaccessible* maps quantify value that is produced by an ecosystem but cannot be accessed by people because of a lack of flow connections on the landscape. Comparison of such maps can facilitate an improved understanding of the dynamics and efficiency of service delivery in the area. *Ex-ante* scenario analysis can be used to spot areas where intervention may help restore service delivery or to highlight those areas where service production is underused. Combinations of flow outputs may be devised to meet specific needs on a case-by-case basis. For example, combining the residential or agricultural demand with the blocked demand, can help quantify the extent of water shortage (or lack thereof) these stakeholders are facing. If agriculture is designated as a priority in a region, then the results might serve as the foundation of a plan to develop water infrastructure in alternative locations to support non-agricultural economic and residential development activities.

Goal 4 states the need of addressing trade-offs, possibly the most important aspect of ES-driven decision-making, in a dynamic way. Trade-offs can be between users of the same service, between different ES for same users, or combinations thereof, and take different meaning and relevance when considered over different horizons of space or time [15]. There is at this time no systematic methodology for addressing trade-offs, although guidelines meant for application with specific ES methods are appearing [37]. A systematic analysis of trade-offs is obviously not practical without a fully quantitative account of beneficiaries and accrued benefits, so satisfaction of goals 1–3 is a requirement for this point. But explicitly modelling the different beneficiaries of a single ES can be difficult owing to the rival nature of many services. An integrated approach where all such effects are modelled explicitly and simultaneously can help address the dual problem of access to sufficient resources and of equitable distribution of limited supplies across the landscape.

In the case study described, existing and emerging economic development in the region relies on continuous access to freshwater. The inherent trade-offs between economic development and household livelihoods translate to water shortages in semi-arid environments. Development of large-scale agricultural plots in the upper watershed limits the flow of water to the lower watershed, creating a largely inequitable situation where winners and losers in the competition for water are scattered throughout the drainage. Identifying the winners and losers under current or alternative integrated water management schemes is key to designing a mechanism to achieve (or maintain support for) an equitable distribution of water. Although we have presented only results for a single ES, flow-related metrics can also be effectively used when considering multiple ES. A relatively simple, but very useful output can be obtained by intersecting multiple flow path outputs for a 'bundle' of different ES, identifying critical landscape locations that are responsible for the transmission of a disproportionate amount of several ES within the area of interest. Such results can, however, only be obtained if multiple ES are modelled *simultaneously*, i.e. subjected uniformly to the influence of each scenario and of the mutual effects they have on each other. This is difficult in most methodologies in use today, which are typically applied separately for each service.

By virtue of its largely automated modelling infrastructure [72], ARIES can produce integrated ES models with slightly more effort than those for single services. Land cover type and other policy-controlled variables entering the models as inputs typically affect more than one service; the ARIES infrastructure ensures that a single chain of dependencies exists across the integrated model, so that simulated policy intervention inputs affect the outputs of all ES. The granularity provided by ARIES in accounting separately for each beneficiary group also allows trade-offs between different stakeholders to be represented unambiguously, as each pair of benefit and beneficiary counts in the overall simulation as a single submodel. Even with improved methodologies, important limitations remain in the face of real-life, multiple-stakeholder problems. For example, the different spatial and temporal scales that accompany each policy horizon or conflict require careful consideration of the assumptions made both when planning scenarios and when analysing results of an integrated model. While the ability to quantify flow paths and address individual beneficiaries does not solve all the difficulties inherent in modelling of trade-offs within ES assessments, techniques such as multiple criteria analysis [73] can be used to assess the concordance or discordance of a set of simulated outcomes with specific configurations of priorities, in an aggregated or spatially distributed way [74]. Such techniques, while not providing a full understanding of trade-offs in the dynamic way sought in goal 4, can help alleviate conflict and define the relative chances of successful outcomes when competing interests must be considered.

Goal 5 argues for an increased flexibility to the definition of value. In the field of environmental securities, the issue of value needs to be considered within the comprehensive framework of equity [14] rather than in the economic interpretation most common for ES literature. This article has not addressed economic value and the many implications of the need for a common currency when comparing effects for policy decisions on a diverse set of outcomes. The outputs of quantitative biophysical analysis (particularly the possible and actual estimates, table 2) can sometimes represent value in themselves, and provide a base for economic valuation [75] that can lead to improved estimates. Yet, the many facets of equity [14] and value [21] make the problem of value attribution in the comparison of results of simulated ES scenarios very specific to case studies and hard to solve in general. Biophysically based models can certainly provide a more flexible set of objective functions for evaluating different scenarios, and address some dimensions of value beyond mere quantification of supply. Of particular interest is the distributional evenness of resource access, not commonly obtainable from mainstream ES accounting methods, which can show at-a-glance whether goals of improved equity in the distribution of one or more ES are met by each scenario of intervention.

We have discussed some advantages of a beneficiary-driven, dynamic view of ES in addressing issues of importance to managing the security of supplies life-sustaining goods. While our examples did not address longer-term drivers of change such as climate, the methods discussed can be applied to scenarios incorporating such effects without modification. In all cases, it is important to remember that the methods address a problem area that has traditionally produced very simple approaches, aimed to rapid assessment and quick policy advice, and to not confuse any current ES modelling effort with an attempt to produce the full account of coupled social-natural dynamics that is only possible with in-depth and long-term scientific study. The complex and multiple-scale modelling required for such assessments is likely to remain impractical or impossible, at least on a routine basis, for some time. Yet, our examples demonstrate that significant steps, even if preliminary, can be taken to improve the state of the art; the increased availability of more sophisticated methods, remote sensing data and computing power is likely to provide refined ES-based instruments that will have a more central role in assisting decision-making aimed at addressing environmental securities, even in data- and resource-limited policy contexts.

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Endnotes

[1]We use the term *flow* here following Bagstad *et al.* [8] to refer to the transmission of a service from ecosystems to people, correspondent to the notion of access. The term has been used ambiguously in the ES literature, for example to describe the annual flow of benefits accruing to people as generated by 'stocks' of ecosystem structure [20]. Such semantic inconsistencies remain problematic across the field of ES.

[2]A dataset containing all inputs and outputs of the model can be retrieved at http://www.integratedmodelling.org/downloads/rs2013data.nc.

References

1. FAO, WFP, IFAD. 2012 *The state of food insecurity in the world 2012*. Rome, Italy: FAO.
2. Carpenter SR *et al*. 2009 Science for managing ecosystem services: beyond the millennium ecosystem assessment. *Proc. Natl Acad. Sci. USA* **106**, 1305–1312. (doi:10.1073/pnas.0808772106)
3. Daily GC. 1997 *Nature's services: societal dependence on natural ecosystems*, p. 392. Washington, DC: Island Press.
4. Daily GC, Matson PA. 2008 Ecosystem services: from theory to implementation. *Proc. Natl Acad. Sci. USA* **105**, 9455–9456. (doi:10.1073/pnas.0804960105)
5. Poppy GM *et al*. 2014 Food security in a perfect storm: using the ecosystem services framework to increase understanding. *Phil. Trans. R. Soc. B* **369**, 20120288. (doi:10.1098/rstb.2012.0288)
6. ARIES Consortium. 2012 *Artificial intelligence for ecosystem services* (*ARIES*). See http://www.ariesonline.org (accessed 1 July 2013).
7. Villa F, Bagstad K, Johnson GW, Voigt B. 2011 Scientific instruments for climate change adaptation: estimating and optimizing the efficiency of ecosystem services provision. *Econ. Agr. Rec. Nat.* **11**, 83–98.
8. Bagstad KJ, Johnson GW, Voigt B, Villa F. 2013 Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. *Ecosyst. Serv.* **4**, 117–125. (doi:10.1016/j.ecoser.2012.07.012)
9. Millennium Ecosystem Assessment. 2005 *Millennium ecosystem assessment: living beyond our means - natural assets and human well-being*. Washington, DC: Millennium Ecosystem Assessment.
10. Kareiva PM, Tallis H, Ricketts T, Daily GC, Polasky S. 2011 *Natural capital: theory and practice of mapping ecosystem services*, p. 365. New York, NY: Oxford University Press.
11. Seppelt R *et al*. 2012 Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecol. Indic.* **21**, 145–154. (doi:10.1016/j.ecolind.2011.09.003)
12. Howe C, Suich H, van Gardingen P, Rahman A, Mace GM. 2013 Elucidating the pathways between climate change, ecosystem services and poverty alleviation. *Curr. Opin. Environ. Sustain.* **5**, 102–107. (doi:10.1016/j.cosust.2013.02.004)
13. Corbera E, Pascual U. 2012 Ecosystem services: heed social goals. *Science* **335**, 655–656. (doi:10.1126/science.335.6069.655-c)
14. McDermott M, Mahanty S, Schreckenberg K. 2013 Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. *Environ. Sci. Policy* **33**, 416–427. (doi:10.1016/j.envsci.2012.10.006)
15. Rodríguez JP, Beard Jr TD, Bennett EM, Cumming GS, Cork S, Agard J, Dobson AP, Peterson GD. 2006 Trade-offs across space, time, and ecosystem services. *Ecol. Soc.* **11**, 28.
16. Farley J. 2008 The role of prices in conserving critical natural capital. *Conserv. Biol.* **22**, 1399–1408. (doi:10.1111/j.1523-1739.2008.01090.x)
17. Farley J, Schmitt F, Alvez J, Ribeiro de Freitas Jr N. 2012 How valuing nature can transform agriculture. *Solutions* **2**, 64–73.
18. Pereira H *et al*. 2005 Conditions and trends of ecosystem services and biodiversity. In *Ecosystems and human well-being: multi scale assessments.* 4 (eds D Capistrano, C Samper, MJ Lee, C Raudsepp-Hearne), pp. 171–203. Washington, DC: Island Press.
19. Foley JA *et al*. 2005 Global consequences of land use. *Science* **309**, 570–574. (doi:10.1126/science.1111772)
20. Daly HE, Farley J. 2004 *Ecological economics: principles and applications*. Washington, DC: Island Press.
21. Wegner G, Pascual U. 2011 Cost-benefit analysis in the context of ecosystem services for human well-being: a multidisciplinary critique. *Glob. Environ. Change* **21**, 492–504. (doi:10.1016/j.gloenvcha.2010.12.008)
22. Tallis H, Kareiva P, Marvier M, Chang A. 2008 An ecosystem services framework to support both practical conservation and economic development. *Proc. Natl Acad. Sci. USA* **105**, 9457–9464. (doi:10.1073/pnas.0705797105)
23. Wallace KJ. 2007 Classification of ecosystem services: problems and solutions. *Biol. Conserv.* **139**, 235–246. (doi:10.1016/j.biocon.2007.07.015)
24. Boyd J, Banzhaf S. 2007 What are ecosystem services? The need for standardized environmental accounting units. *Ecol. Econ.* **63**, 616–626. (doi:10.1016/j.ecolecon.2007.01.002)
25. Ruhl JB, Kraft SE, Lant CL. 2007 *The law and policy of ecosystem services*. Washington, DC: Island Press.
26. Fisher B, Turner RK, Morling P. 2009 Defining and classifying ecosystem services for decision making. *Ecol. Econ.* **68**, 643–653. (doi:10.1016/j.ecolecon.2008.09.014)
27. Costanza R. 2008 Ecosystem services: multiple classification systems are needed. *Biol. Conserv.* **141**, 350–352. (doi:10.1016/j.biocon.2007.12.020)
28. Daily GC *et al*. 2009 Ecosystem services in decision making: time to deliver. *Front. Ecol. Environ.* **7**, 21–28. (doi:10.1890/080025)
29. Nahlik AM, Kentula ME, Fennessy MS, Landers DH. 2012 Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. *Ecol. Econ.* **77**, 27–35. (doi:10.1016/j.ecolecon.2012.01.001)
30. Johnston RJ, Russell M. 2011 An operational structure for clarity in ecosystem service values. *Ecol. Econ.* **70**, 2243–2249. (doi:10.1016/j.ecolecon.2011.07.003)
31. Fisher B *et al*. 2008 Ecosystem services and economic theory: integration for policy-relevant research. *Ecol. Appl.* **18**, 2050–2067. (doi:10.1890/07-1537.1)
32. Seppelt R, Dormann CF, Eppink FV, Lautenbach S, Schmidt S. 2011 A quantitative review of ecosystem

service studies: approaches, shortcomings and the road ahead. *J. Appl. Ecol.* **48**, 630–636. (doi:10.1111/j.1365-2664.2010.01952.x)

33. Costanza R *et al*. 1997 The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260. (doi:10.1038/387253a0)
34. Troy A, Wilson MA. 2006 Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. *Ecol. Econ.* **60**, 435–449. (doi:10.1016/j.ecolecon.2006.04.007)
35. Tallis H, Polasky S. 2009 Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. *Ann. NY Acad. Sci.* **1162**, 265–283. (doi:10.1111/j.1749-6632.2009.04152.x)
36. Martinez-Harms MJ, Balvanera P. 2012 Methods for mapping ecosystem service supply: a review. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manage.* **8**, 17–25. (doi:10.1080/21513732.2012.663792)
37. Tallis HT *et al*. 2011 *InVEST 2.2.0 user's guide*. Stanford, CA: Stanford University.
38. Johnson GW, Bagstad K, Snapp R, Villa F. 2012 Service path attribution networks (SPANS): a network flow approach to ecosystem service assessment. *Int. J. Agric. Environ. Inf. Syst.* **3**, 54–71. (doi:10.4018/jaeis.2012070104)
39. GW Johnson, R Snapp, F Villa, K Bagstad (eds). 2012 *Modelling ecosystem services flows under uncertainty with stochastic SPAN*. Leipzig, Germany: IEMSs.
40. Syrbe RU, Walz U. 2012 Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. *Ecol. Indic.* **21**, 80–88. (doi:10.1016/j.ecolind.2012.02.013)
41. Johnson GW, Bagstad K, Snapp R, Villa F. 2010 Service path attribution networks (SPANS): spatially quantifying the flow of ecosystem services from landscapes to people. *Lect. Notes Comp. Sci.* **6016**, 238–253. (doi:10.1007/978-3-642-12156-2_18)
42. Bagstad K, Villa F, Johnson GW, Voigt B. 2011 *ARIES - artificial intelligence for ecosystem services: a guide to models and data*. ARIES report series no. 1. Burlington, VT: ARIES Consortium.
43. Burkhard B, Kroll F, Nedkov S, Muller F. 2012 Mapping ecosystem service supply, demand and budgets. *Ecol. Indic.* **21**, 17–29. (doi:10.1016/j.ecolind.2011.06.019)
44. Palomo I, Martín-López B, Potschin M, Haines-Young R, Montes C. 2013 National parks, buffer zones and surrounding lands: mapping ecosystem service flows. *Ecosyst. Serv.* **4**, 104–116. (doi:10.1016/j.ecoser.2012.09.001)
45. Bonabeau E. 2002 Agent-based modeling: methods and techniques for simulating human systems. *Proc. Natl Acad. Sci. USA* **99**, 7280–7287. (doi:10.1073/pnas.082080899)
46. SMUWC. 2002 *The sustainable management of the Usangu wetland and its catchment*. Dar Es Salaam, Tanzania: UK Department of International Development and Ministry of Water and Livestock.
47. Lankford B. 2001 Red routes on blue rivers: strategic water management for the Ruaha River Basin, Tanzania. *J. Water Resour. Dev.* **17**, 427–444. (doi:10.1080/07900620120065183)
48. Coppolillo PB, Kashaija L, Moyer DC, Knap E. 2004 *Technical report on water availability in the Ruaha River and State of Usangu Game Reserve*. New York, NY: Wildlife Conservation Society.
49. Kashaigili JJ. 2008 Impacts of land-use and land-cover changes on flow regimes of the Usangu wetland and the Great Ruaha River, Tanzania. *Phys. Chem. Earth* **33**, 640–647. (doi:10.1016/j.pce.2008.06.014)
50. Kashaigili JJ, McCartney M, Mahoo HF. 2007 Estimation of environmental flows in the Great Ruaha River catchment, Tanzania. *Phys. Chem. Earth* **32**, 1007–1014. (doi:10.1016/j.pce.2007.07.005)
51. Mtahiko MG, Gereta E, Kajuni AR, Chiombola EA, Ng'umbi GZ, Coppolillo PB, Wolanski E. 2006 Towards an ecohydrology-based restoration of the Usangu wetlands and the Great Ruaha River, Tanzania. *Wetlands Ecol. Manage.* **14**, 489–503. (doi:10.1007/s11273-006-9002-x)
52. Lankford B, Van Koppen B, Franks T, Mahoo HF. 2004 Entrenched views or insufficient science? Contested causes and solutions of water allocation; insights from the Great Ruaha River Basin, Tanzania. *Agric. Water Manage.* **69**, 135–153. (doi:10.1016/j.agwat.2004.04.005)
53. Voigt B, Gustafson C, Erickson JD. 2012 *Modeling zoonotic disease regulation under climate change scenarios in semi-arid grasslands: a scoping model of water provisioning services in the Ruaha landscape of Tanzania*. RB-08-2012. Fort Collins, CO: Adapting Livestock to Climate Change Collaborative Research Support Program.
54. Franks T, Lankford B, Mdemu M. 2004 Managing water among competing uses: the Usangu wetland in Tanzania. *Irrigation Drainage* **53**, 277–286. (doi:10.1002/ird.123)
55. Masozera M, Erickson JD, Clifford DL, Coppolillo PB, Nguvava M, Sadiki H, Mazet JK. 2010 Integrating the management of the Ruaha landscape of Tanzania with local needs and preferences. *Envirnon. Manage.* **52**, 1533–1546. (doi:10.1007/s00267-013-0175-9)
56. Mazet JAK, Clifford DL, Coppolillo PB, Deolalikar AB, Erickson JD, Kazwala K. 2009 A 'one health' approach to address emerging zoonoses: the HALI project in Tanzania. *PLoS Med.* **6,** e1000190. (doi:10.1371/journal.pmed.1000190)
57. Coppolillo PB, Dickman A. 2007 *Livelihoods and protected areas in the Ruaha landscape: a preliminary review*. New York, NY: Wildlife Conservation Society.
58. Masozera M *et al*. 2010 *Public health and rural livelihoods under water scarcity in the Ruaha landscape, Tanzania*. HALI Research Brief 10-04. Iringa, Tanzania: HALI.
59. Wint W, Robinson T. 2007 *Gridded livestock of the world*. Rome, Italy: Food and Agriculture Organizations of the United Nations.
60. Steinfield H *et al*. 2006 *Livestock's long shadow: environmental issues and options: food and agriculture organization*. Rome, Italy: FAO.
61. Bhaduri B, Bright E, Coleman P, Urban M. 2007 LandScan USA: a high-resolution geospatial and temporal modeling approach for population distribution and dynamics. *GeoJournal* **69**, 103–117. (doi:10.1007/s10708-007-9105-9)
62. Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A. 2005 Very high resolution interpolated climate surfaces for global land areas. *Int. J. Climatol.* **25**, 1965–1978. (doi:10.1002/joc.1276)
63. Farr TG *et al*. 2007 The shuttle radar topography mission. *Rev. Geophys.* **45**, 103–117. (doi:10.1029/2005RG000183)
64. Beier C, Patterson T, Chapin III FS. 2008 Ecosystem services and emergent vulnerability in managed ecosystems: a geospatial decision-support tool. *Ecosystems* **11**, 923–938. (doi:10.1007/s10021-008-9170-z)
65. Chan KMA, Shaw MR, Cameron DR, Underwood EC, Daily GC. 2006 Conservation planning for ecosystem services. *PLoS Biol.* **4**, 2138–2152. (doi:10.1371/journal.pbio.0040379)
66. Grêt-Regamey A, Bebi P, Bishop ID, Schmid WA. 2008 Linking GIS-based models to value ecosystem services in an Alpine region. *J. Environ. Manage.* **89**, 197–208. (doi:10.1016/j.jenvman.2007.05.019)
67. Wendland KJ, Honzák M, Portela R, Vitale B, Rubinoff S, Randrianarisoa J. 2010 Targeting and implementing payments for ecosystem services: opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. *Ecol. Econ.* **69**, 2093–2107. (doi:10.1016/j.ecolecon.2009.01.002)
68. Wünscher T, Engel S, Wunder S. 2008 Spatial targeting of payments for environmental services: a tool for boosting conservation benefits. *Ecol. Econ.* **65**, 822–833. (doi:10.1016/j.ecolecon.2007.11.014)
69. Scheffer M, Carpenter S, Foley JA, Folke C, Walker B. 2001 Catastrophic shifts in ecosystems. *Nature* **413**, 591–596. (doi:10.1038/35098000)
70. Holling CS. 2001 Understanding the complexity of economic, ecological, and social systems. *Ecosystems* **4**, 390–405. (doi:10.1007/s10021-001-0101-5)
71. Bolte JP, Hulse DW, Gregory SV, Smith C. 2007 Modeling biocomplexity – actors, landscapes and alternative futures. *Environ. Model. Software* **22**, 570–579. (doi:10.1016/j.envsoft.2005.12.033)
72. Villa F. 2009 Semantically-driven meta-modelling: automating model construction in an environmental decision support system for the assessment of ecosystem services flow. In *Information technology in environmental engineering* (eds IN Athanasiadis, PA Mitkas, AE Rizzoli, J Marx Gomez), pp. 23–36. New York, NY: Springer.
73. Figueira J, Greco S, Ehrgott M. 2005 *Multiple criteria decision analysis: state of the art*. New York, NY: Springer.
74. Villa F, Tunesi L, Agardy T. 2002 Zoning marine protected areas through spatial multiple-criteria analysis: the case of the Asinara Island National Marine Reserve of Italy. *Conserv. Biol.* **16**, 515–526. (doi:10.1046/j.1523-1739.2002.00425.x)
75. Portela R, Nunes PALD, Onofri L, Villa F, Shepard A, Lange GM. 2012 *Assessing and valuing ecosystem services in the Ankeniheny-Zahamena Corridor, Madagascar: a demonstration case study for the wealth accounting and the valuation of ecosystem services (WAVES) global partnership*. New York, NY: World Bank.