University of Vermont

ScholarWorks @ UVM

1-1-2014

From theoretical to actual ecosystem services: Mapping beneficiaries and spatial flows in ecosystem service assessments

Kenneth J. Bagstad
United States Geological Survey

Ferdinando Villa
Ikerbasque, the Basque Foundation for Science

David Batker

Jennifer Harrison-Cox

Brian Voigt
University of Vermont

See next page for additional authors

Recommended Citation

Bagstad KJ, Villa F, Batker D, Harrison-Cox J, Voigt B, Johnson GW. From theoretical to actual ecosystem services: mapping beneficiaries and spatial flows in ecosystem service assessments. Ecology and Society. 2014 Jun 1;19(2).

Authors

Kenneth J. Bagstad, Ferdinando Villa, David Batker, Jennifer Harrison-Cox, Brian Voigt, and Gary W. Johnson

From theoretical to actual ecosystem services: mapping beneficiaries and spatial flows in ecosystem service assessments

Kenneth J. Bagstad [1], Ferdinando Villa [2], David Batker [3], Jennifer Harrison-Cox [3], Brian Voigt [4] and Gary W. Johnson [4]

ABSTRACT. Ecosystem services mapping and modeling has focused more on supply than demand, until recently. Whereas the potential provision of economic benefits from ecosystems to people is often quantified through ecological production functions, the use of and demand for ecosystem services has received less attention, as have the spatial flows of services from ecosystems to people. However, new modeling approaches that map and quantify service-specific sources (ecosystem capacity to provide a service), sinks (biophysical or anthropogenic features that deplete or alter service flows), users (user locations and level of demand), and spatial flows can provide a more complete understanding of ecosystem services. Through a case study in Puget Sound, Washington State, USA, we quantify and differentiate between the theoretical or in situ provision of services, i.e., ecosystems' capacity to supply services, and their actual provision when accounting for the location of beneficiaries and the spatial connections that mediate service flows between people and ecosystems. Our analysis includes five ecosystem services: carbon sequestration and storage, riverine flood regulation, sediment regulation for reservoirs, open space proximity, and scenic viewsheds. Each ecosystem service is characterized by different beneficiary groups and means of service flow. Using the ARtificial Intelligence for Ecosystem Services (ARIES) methodology we map service supply, demand, and flow, extending on simpler approaches used by past studies to map service provision and use. With the exception of the carbon sequestration service, regions that actually provided services to people, i.e., connected to beneficiaries via flow paths, amounted to 16-66% of those theoretically capable of supplying services, i.e., all ecosystems across the landscape. These results offer a more complete understanding of the spatial dynamics of ecosystem services and their effects, and may provide a sounder basis for economic valuation and policy applications than studies that consider only theoretical service provision and/or use.

Key Words: *beneficiaries; benefits; demand side; mapping; provisioning areas; spatial dynamics; spatial flow*

INTRODUCTION

Recent years have witnessed substantial growth in the field and practice of mapping ecosystem services, that is, the benefits that ecosystems provide in support of human well-being (MA 2005, Schägner et al. 2013). From an early focus on developing typologies to classify ecosystem services (Costanza et al. 1997, Daily 1997, de Groot et al. 2002, MA 2005) and on mapping their values using simple value transfer (Costanza et al. 1997, Troy and Wilson 2006) or land-cover based proxies for ecosystem services (Eigenbrod et al. 2010), recent work has sought to explicitly link ecological processes to specific human beneficiaries (Boyd and Banzhaf 2007, Wallace 2007, Haines-Young and Potschin 2011, Nahlik et al. 2012) and to more rigorously model and map ecosystem services (Kareiva et al. 2011, Egoh et al. 2012, Martínez-Harms and Balvanera 2012, Villa et al. 2014).

A number of recent studies have used spatial analysis to quantify the ecological factors contributing to the provision of certain services, and in some cases to map factors related to their demand (see reviews by Egoh et al. 2012 and Martínez-Harms and Balvanera 2012). Most of these studies explore how the provision of ecosystem services varies across the landscape. From a spatial perspective, the supply side, or potential provision of ecosystem services, has been much more rigorously explored than the demand side. However, several recent papers have begun to quantify service demand through the quantification and simple overlay of service provision and use (Beier et al. 2008, Burkhard et al. 2012, Nedkov and Burkhard 2012) or to geographically conceptualize service provision, use, and flows (McDonald 2009,

Syrbe and Walz 2012, Palomo et al. 2013). However, these studies do not approach the problem in a theoretically, terminologically, or methodologically consistent manner. For example, Syrbe and Walz (2012) and Palomo et al. (2013) conceptualize and map service provisioning, service benefitting, and service connecting regions, but do not operationalize these concepts for ecosystem service flow mapping, modeling, and quantification. Overly simple approaches to quantifying service beneficiaries and flows, such as simple overlay analysis, can lead to inaccuracies in ecosystem service mapping, valuation, and trade-off analysis. This task is challenging because ecosystem services have complex flow dynamics that operate across differing spatial and temporal scales (Ruhl et al. 2007, Tallis et al. 2008, Fisher et al. 2009, Johnson et al. 2012, Bagstad et al. 2013).

The importance of spatial flows is recognized across diverse research fields related to ecosystem services, from pollination (Kremen et al. 2007, Keitt 2009), migratory species (Semmens et al. 2011), and hydrology (Reaney 2008) to pollutant fate and transport (Coulthard and Macklin 2003). To fully quantify spatial flows of ecosystem services, a new lexicon is required that can fully capture their spatially dynamic nature. In outlining the Artificial Intelligence for Ecosystem Services (ARIES) modeling system, we have proposed such a terminology for quantifying and mapping ecosystem service flows (Johnson et al. 2012, Bagstad et al. 2013, Villa et al. 2014). Other terms exist (e.g., Mitchell et al. 2013, Palomo et al. 2013); given the novelty of this work, scientific consensus has not yet been reached. We recognize that all such terms currently carry some degree of ambiguity and

[1]Geosciences & Environmental Change Science Center, U.S. Geological Survey, [2]Basque Centre for Climate Change (BC3), IKERBASQUE, Basque Foundation for Science, Spain, [3]Earth Economics, [4]Gund Institute for Ecological Economics, University of Vermont

terminological baggage, and that until the research community coalesces around a consistent set of definitions, some terminology must be chosen and applied consistently.

Systematic quantification of service flows offers an opportunity to differentiate between theoretical (in situ) and actual service provision. We define theoretical service provision as the modeled capacity of ecosystems to supply a given service; actual service provision requires the presence of beneficiaries linked by a service-specific flow path. Theoretical use entails the location and demand of all potential human beneficiaries regardless of their spatial connection to ecosystems, whereas actual use denotes demand that has been met by flow-connected ecosystems. Quantification of actual services requires the modeling of: (1) the location of ecosystems providing the service; (2) human demand for the service, which is either rival, where use of a service leaves less of it available for other users (e.g., consumptive water use), or nonrival, where its use does not prevent others from enjoying it (e.g., recreational water use or scenic views); (3) spatial flow paths for the service (e.g., hydrologic flows, lines of sight, or transportation networks); and (4) biophysical and anthropogenic landscape features that deplete or alter that spatial flow (i.e. sinks; Fig. 1). Sinks or rival use leave less of the service available for "downstream" users, signified in Fig. 1 by depleted or blocked flows of the service. Network flow propagation models or spatial analytical operations can be used to simulate the flow of services from a source area, through sink regions and on to service-specific beneficiaries (Johnson et al. 2012, Bagstad et al. 2013). Flow-based ecosystem service assessment can also enable quantification of inaccessible service provision and use, where potential beneficiaries lack a flow connection to a region providing a service, and blocked service provision, use, and flows, where sink regions block service flows between ecosystems and people.

Fig. 1. Stylized conception of regions of ecosystem service sources, sinks, uses, and flows for a given ecosystem service. Service flows are generated by source regions and depleted by sinks and rival use, but not by nonrival use.
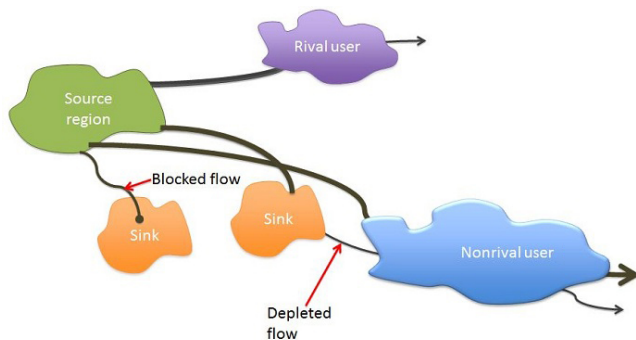
These approaches to quantifying ecosystem service flows were developed as a part of the ARIES modeling system (Villa et al. 2014; http://www.ariesonline.org), which couples probabilistic or deterministic models of ecosystem service supply and demand with network flow propagation models that quantify service flows. Bayesian networks (Cowell et al. 1999, McCann et al. 2006) or deterministic models are used, as appropriate, to map the ecological and socioeconomic factors contributing to the provision and use of ecosystem services. Ontologies (Madin et al.

2008, Villa et al. 2009) built into the ARIES system provide a formalized repository of abstract concepts and relationships that supply a semantic foundation for modeling. They also serve as a knowledge base for reasoning algorithms to assemble models that are applied to spatial data for quantifying service provision and use. This "intelligent" modeling infrastructure of ARIES (Villa 2010) can select and use basic ecosystem service models that encode ecological production functions (Nelson et al. 2009) for regions with limited data or model availability. In case study regions where higher quality data and models are available, locally calibrated models will be used that more explicitly consider regionally specific factors that influence the generation, delivery, and use of ecosystem services. Through identifying a clear chain of provision and use for each ecosystem service and using well-defined, nonoverlapping beneficiary groups (e.g., Boyd and Banzhaf 2007, Nahlik et al. 2012), this approach avoids the problem of double counting in valuation, because the base for valuation is the quantified flow of each benefit type rather than the ecological processes that brought those benefits into existence.

By definition, actual service provision, use, and flow will be less than or equal to theoretical service provision and demand. Because of differing supply, demand, and flow characteristics, we expect the ratio of actual to theoretical service provision, use, and flow will vary by service and region, carrying implications for valuation and trade-off analysis. However, quantified differences between theoretical and actual services and their subsequent implications have not yet been explored for a case study that maps ecosystem services at the regional scale. In this paper, we quantify the ratio of actual to theoretical ecosystem service provision when accounting for the location of sources, users, and the spatial connections, i.e., flows, or lack thereof between ecosystems to people. We provide examples of theoretical and actual services in the Puget Sound, Washington State, USA for five services: carbon sequestration and storage, scenic viewsheds, open space proximity, sediment regulation, and flood regulation.

METHODS

Study area
The Puget Sound, the second largest estuary in the United States, is a defining social, cultural, and economic feature of Washington State. Fed by 19 river basins, the Puget Sound is bordered by the Olympic Peninsula to the west and Cascade Mountains to the east (Fig. 2). The region is home to 4.4 million people, approximately 67% of Washington State's population, including 15 American Indian tribes and the major port cities of Seattle and Tacoma (Washington State Department of Ecology 2013).

Human population growth and economic development have altered the geological, biological, and hydrological processes in the region's riverine, nearshore, and estuarine environments (U. S. Geological Survey 2006). The past decades have seen increasing urbanization, shoreline paving and bulkhead installation, and pollution of waterways by animal and industrial waste and urban runoff. Overharvesting of fish, shellfish, and timber stocks has notably negative impacts on the Sound's ecosystems (Puget Sound Partnership 2012*a*). Between 1991 and 2001 an additional 10% of the Puget Sound Basin was paved to accommodate residential, commercial, and industrial development, roads, and other infrastructure (Puget Sound Partnership 2012*b*).

Fig. 2. Puget Sound case study region.
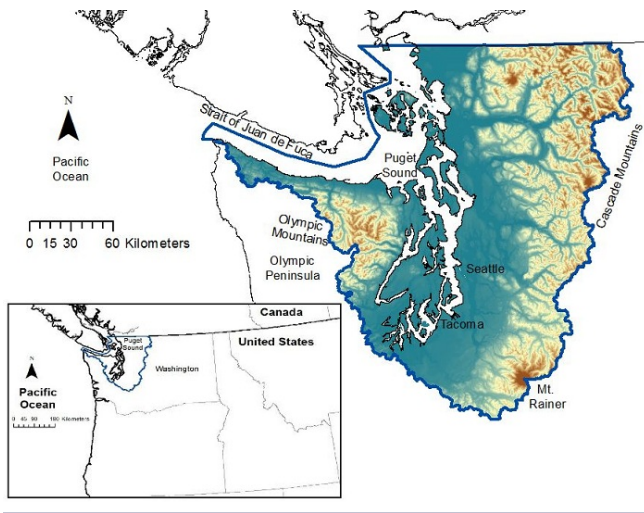
Collectively, these impacts have taken a toll on the health of the Puget Sound. Iconic species, including Chinook (*Oncorhynchus tshawytscha*) and Coho salmon (*Oncorhynchus kisutch*), steelhead (*Oncorhynchus mykiss*), and resident orca whale (*Orcinus orca*) populations, have become species of conservation concern (Gaydos and Brown 2011). More than 80% of tidal wetlands have been lost and vast areas that used to serve as floodplain wetlands are now isolated from their rivers by levees or have been filled for development (NOAA 2013). More than 70% of old-growth forests have been removed during the past 50 years while over one-third of the nearly 2500 miles of shoreline have been armored. Additionally, hundreds of thousands of gallons of oil and hazardous waste have been spilled into the Sound's rivers and marine waters (Puget Sound Partnership 2012*b*).

These challenges are compounded by climate change and sea-level rise forecasts. Potential impacts of climate change in the Puget Sound include shoreline erosion, beach and tidal flat inundation, increasing susceptibility of communities to storm surges, rising surface and water temperatures, increasing riverine flooding, and glacial retreat in the Cascade and Olympic mountains. Shellfish are being impacted by toxic algal blooms, ocean acidification, low oxygen concentrations in bottom waters due to warmer water temperatures, increased temperature stratification, and other factors (Moore et al. 2011). All of these changes combine to impact the Puget Sound ecosystem in complex, sometimes unpredictable ways, with implications for both human and nonhuman communities (Mote et al. 2005). In the face of these diverse resource management challenges, the ARIES developers have worked with Earth Economics, a Tacoma-based NGO specializing in ecological economics research and outreach, to map and value ecosystem services in the Puget Sound. From 2007 to 2012, Earth Economics hosted a series of workshops with partners in the academic, public, and NGO sectors, during which we identified ecosystem services of importance to stakeholder groups and developed, tested, and received critical feedback on a series of ecosystem service models for the Puget Sound.

Ecosystem service modeling and mapping

We modeled five ecosystem services of interest to the above-mentioned stakeholder groups in the Puget Sound region: (1) carbon sequestration and storage, (2) scenic viewsheds for homeowners, (3) open space proximity for homeowners, (4) flood regulation for developed land in the 100-year floodplain, and (5) sediment regulation for reservoirs. Data sources, model structures, and underlying assumptions are discussed in detail by Bagstad et al. (2011) and are summarized in Table 1; flow characteristics for each service are described by Bagstad et al. (2013). We present results measured in biophysical units (for carbon sequestration and storage, flood, and sediment regulation) and relative rankings (for viewsheds and open space proximity). Although it is possible to apply monetary values to some of these services, for the purposes of comparing model results monetization simply scales the model outputs by a common factor, so we do not present valuation results in this article. We modeled and compared theoretical and actual service values on a spatial grid at a 200 meter resolution.

Carbon sequestration and storage

We quantified carbon sequestration and storage in vegetation and soils using Bayesian models (Bagstad et al. 2011) calibrated with Moderate-resolution Imaging Spectroradiometer Net Primary Productivity (MODIS GPP/NPP Project, http://secure.ntsg.umt.edu/projects/index.php/ID/ca2901a0/fuseaction/projects.detail.htm), National Biomass and Carbon Dataset (http://www.whrc.org/mapping/nbcd/), and Soil Survey Geographic Database (http://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/?cid=nrcs142p2_053627) soils data, respectively. By overlaying fire boundary polygons from the Geospatial Multi-Agency Coordination Group (GeoMAC, http://www.geomac.gov/index.shtml) we estimated carbon storage losses caused by wildfire, using fuel consumption coefficients from Spracklen et al. (2009) and carbon pool data from Smith et al. (2006). By incorporating the impacts of land-cover change from urbanization (Bolte and Vache 2010) within carbon models, we quantified resultant changes in carbon storage. Our models underestimate the sequestration and storage of "blue carbon" (Laffoley and Grimsditch 2009) in the region's coastal wetlands, estuaries, and aquatic habitats. Although such estimates have been compiled for the nearby Georgia Strait (Molnar et al. 2012), they generally relied on secondary data. To avoid inconsistencies arising from use of mixed models, we did not attempt to transfer these results to the Puget Sound.

Greenhouse gas emissions provide one possible measure of the demand for carbon sequestration required to offset anthropogenic emissions. Alternatively, populations particularly susceptible to climate change impacts could be mapped as beneficiaries of climate stability, though the precise linkages between carbon sequestration and storage and mitigation of the effects of climate change is difficult to establish. Although carbon emissions can be offset anywhere on the globe, in some applications, such as ecological footprint-type analyses, understanding a region's carbon budget may be of interest. Emissions can be quantified for the study region by multiplying the region's population by per capita emissions for the state of Washington (Ramseur 2007). Mixing and removal of carbon dioxide in the atmosphere can be assumed to be instantaneous and complete; therefore no flow model is necessary for this service.

Table 1. Methods and metrics for quantifying ecosystem services in the Puget Sound Basin.

Service	Metric	Method
Carbon sequestration & storage	Carbon sequestration (source; T carbon/year)	Bayesian model of carbon sequestration calibrated using MODIS NPP data
	Vegetation & soil carbon storage (T carbon)	Bayesian models of vegetation and soil carbon calibrated using National Biomass and Carbon Dataset and Soil Survey Geographic Database data
	Loss of carbon storage from fire & urbanization (sink; T carbon/year)	Overlay carbon storage maps with fire polygons, carbon pool data, and fuel consumption coefficients and urbanization model results
	Greenhouse gas emissions (use; T carbon/year)	Per capita emissions * Population
	Ratio of actual to theoretical use	(Regional greenhouse gas emissions + Stored carbon release from fire and urbanization) / carbon sequestration
Scenic viewsheds for homeowners	Viewshed source (relative ranking, 0-100)	Bayesian model of scenic quality
	Viewshed sink (relative ranking, 0-100)	Bayesian model of visual blight
	Viewshed use	Homeowner locations based on parcel or developed land data
	Actual source	Views visible to homeowners via line-of-sight model, weighted by number of users
	Ratio of actual to theoretical source	Summed source values for landscape actually providing views / Theoretical views for entire landscape
Open space proximity for homeowners	Proximity source (relative ranking, 0-100)	Bayesian model of open space proximity quality
	Proximity sink (relative ranking, 0-100)	Highways, which reduce pedestrian access and limit visual and soundscape quality
	Proximity use	Homeowner locations based on parcel or developed land data
	Actual source	Open space accessible to homeowners via walking simulation model, weighted by number of users
	Ratio of actual to theoretical source	Summed source values for landscape actually providing proximity values / Theoretical proximity values for entire landscape
Flood regulation for developed land in 100-year floodplain	Flood source (m³ floodwater)	Mean annual precipitation
	Flood sink (m³ floodwater mitigation)	Bayesian model of landscape capacity to intercept, absorb, or detain floodwater
	Flood use	Number of developed cells in 100-year floodplain, by subwatershed
	Actual flood regulation	Percentage of floodwater mitigated * Number developed cells in 100-year floodplain, by subwatershed
	Ratio of actual to theoretical source	Flood sinks when setting values to zero in subwatersheds with no beneficiaries / Total theoretical flood sink
Sediment regulation for reservoirs	Sediment source (T sediment/year)	Bayesian model of erosion, calibrated using values from soil loss models
	Sediment sink (T/year)	Bayesian model of sediment deposition in floodplains
	Sediment users	Reservoir locations
	Ratio of actual to theoretical source and sink	Source or sink values within upstream contributing watersheds to reservoirs / Source or sink values for entire landscape

A regional carbon budget can be estimated as the balance between the supply of carbon sequestration and changes in carbon storage versus the demand for emissions offsets:

Carbon to offset anthropogenic emissions = (Carbon sequestration – Loss of carbon stored in vegetation and soils from fire and urbanization)

Actual (local) use of the carbon sequestration service would thus be greater than 100% if anthropogenic emissions exceeded the difference between carbon sequestration and stored carbon loss, and less than 100% otherwise.

Scenic viewsheds and open space proximity for homeowners
We quantified aesthetic values derived from scenic views (Bourassa et al. 2004) and proximity to open space (McConnell and Walls 2005) as two distinct ecosystem services. Both provide natural sensory experiences to their beneficiaries, e.g., views of nature; nearby open space can provide additional benefits such as natural soundscapes, privacy, and access to recreational amenities. These values typically accrue to property values and can be measured using hedonic analysis, or in this case mapped by identifying: (1) ecosystems providing high-quality views or valuable open space, (2) features that impede or degrade views or access to open space, and (3) housing locations (Bagstad et al. 2011). Within a given viewshed, our models quantified the contribution of viewshed source features such as mountains and water bodies and sinks that detract from view quality, including obstructions or visual blight such as industrial or commercial development. Source, sink, and use locations were linked by a flow model that computed visibility along lines of sight from use locations to scenic viewshed features. For open space proximity, we mapped the relative value of open space, highways that impede walking access or reduce visual and soundscape quality, and housing locations, connected by a flow model simulating physical access to desirable spaces. We used reviews of the hedonic valuation literature (Bourassa et al. 2004, McConnell and Walls 2005) to inform model development, ranking the influence of different viewshed and open space characteristics on property values to parameterize the source and sink models. Both viewshed and proximity models include distance decay functions that account for changes with distance in the value of open space and views. We then computed the ratio of actual to theoretical provision of both scenic views and open space to compare the values accruing to homeowners relative to those for the entire landscape.

Flood regulation for developed land in the 100-year floodplain
We mapped flood regulation as the ability of ecosystems to intercept, absorb, or detain floodwater prior to its reaching flood-vulnerable people, structures, or cropland. In this study, we mapped the locations of developed land within the 100-year floodplain as the beneficiary of flood regulation. Lacking basin-wide precipitation data of adequate resolution for event-based modeling, our model used mean annual precipitation records for 1971-2000 to represent floodwater sources (PRISM Climate Group 2009). We estimated flood sinks, i.e., the capacity of the landscape to intercept, absorb, or detain floodwater, using a Bayesian model of vegetation, topography, and soil influences (Bagstad et al. 2011). This green infrastructure, the ecosystem service that we used for subsequent analysis, can combine with anthropogenic gray infrastructure, such as dams and detention basins, to provide flood regulation.

Since flood regulation implies a hydrologic connection between sources, sinks, and users, we simulated its flow through a three-step process. First, we aggregated values for precipitation (sources of floodwater), flood mitigation (sinks), and users (developed land located in the 100-year floodplain) within each of the 502 12-digit Hydrologic Unit Code (HUC) watersheds within the Puget Sound region. Second, we subtracted the sink value from the source value for each subwatershed to quantify remaining floodwater and the proportion of mitigated floodwater. Third, we multiplied the proportion of mitigated floodwater for each subwatershed by the number of developed raster cells within the 100-year floodplain to yield a ranking of flood mitigation for each subwatershed. Given the difficulty in modeling flood regulation on an event-by-event basis, our approach instead yielded spatially explicit proxy information to describe flood regulation as an ecosystem service. As data availability improves (e.g., precipitation data collected through citizen science, Community Collaborative Rain, Hail, and Snow Network, http://www.cocorahs.org/), we expect to be able to improve the temporal and spatial resolution of future flood regulation modeling efforts. For this paper, our values can be interpreted as a spatially explicit, relative ranking of flood mitigation that accounts for the hydrologic colocation of flood sources, sinks, and users.

Using this metric, subwatersheds with limited ability to provide flood mitigation (i.e., small flood sink relative to source values) and few beneficiaries receive a low actual flood mitigation score; we expect this for headwater streams with greater precipitation and likelihood of rain-on-snow events, steeply sloped alpine environments with limited ability to mitigate floodwater, and few at-risk properties. Conversely, subwatersheds with large sink values and a large number of beneficiaries receive a greater flood mitigation score. Intermediate cases occur for both less developed subwatersheds with large sink values but few beneficiaries and for vulnerable subwatersheds with limited sink values but many beneficiaries. This approach will generally underestimate the flood mitigation value provided by subwatersheds upstream of a particular at-risk cell. However, ecosystems are often more effective in providing mitigation for smaller floods than major ones (Brauman et al. 2007), meaning that local-scale effects are important in provision of flood mitigation, better justifying the use of subwatersheds as units of analysis. We calculated the ratio of actual to theoretical flood sinks by dividing summed flood sink values for subwatersheds providing flood mitigation to users by summed flood sink values for the entire landscape without accounting for the presence of at-risk structures.

Sediment regulation for reservoirs
We mapped sediment regulation as the location of sediment sinks (depositional areas in floodplains), which can absorb sediment transported by hydrologic flows from upstream sources (erosion-prone areas) prior to reaching users. In this case the benefit of avoided sedimentation is provided to 29 major reservoirs. Avoided sedimentation helps maintain the ability of reservoirs to provide benefits including hydroelectric power generation, flood control, recreation, and water supply to beneficiaries through the region. Avoided reservoir sedimentation likely helps to protect each of these benefits in different ways, i.e., increased turbidity or the loss of reservoir storage capacity may have a greater impact on some provision of some benefit types than others. For our purposes we ended the modeling and mapping exercise at the reservoirs, though future work could be undertaken to map the beneficiaries of some of these specific benefits generated by each reservoir.

Reservoir sedimentation reduces their storage capacity, typically decreasing their ability to provide these benefits without costly dredging. Although the Revised Universal Soil Loss Equation (RUSLE) is commonly used to quantify erosion, it is known to perform poorly on younger, steeply sloped soils like those upstream of reservoirs in the Puget Sound (Renard et al. 1996). We thus used a probabilistic Bayesian model of soil erosion incorporating vegetation, soils, and rainfall influences and calibrated using regional data from coarser scale and/or RUSLE-derived erosion models (Bagstad et al. 2011). We probabilistically modeled sediment deposition in floodplains using data for floodplain vegetation, floodplain width, and stream gradient, which can influence rates of deposition. In future analyses, ARIES' intelligent model selection algorithms, described further in the discussion, will apply RUSLE to locations meeting the needed criteria for that model and use locally adapted probabilistic models elsewhere (Villa et al. 2014). We calculated the ratio of actual to theoretical sediment regulation using the aggregated sink values upstream of reservoirs in the Puget Sound region, divided by aggregated theoretical sink values for the entire landscape.

RESULTS

Carbon sequestration and storage
We quantified total carbon sequestration in Puget Sound at 436 kT/year and total vegetation and soil carbon storage at 28,350 kT (Fig. 3). Anthropogenic emissions were 17,359 kT/year. The loss of carbon to wildfires is relatively small in the Puget Sound Basin. GeoMAC recorded 22 wildfires over a 13-year period (2000-2012), which burned a total of 2828 ha, meaning that the average fire burned 129 ha. This translated to a potential loss of carbon storage of just 0.4 to 3.7 kT/year, depending on burn severity, as compared to losses of carbon storage from urbanization ranging from 23.7 to 42.8 kT/year under alternative land-use change scenarios (Table 2). Despite the fact that per capita emissions are lower in Washington State than the U.S. average, the sum of carbon emissions and lost carbon storage greatly exceeds carbon sequestration for the relatively populous Puget Sound region. Carbon emissions for the region thus exceed sequestration capacity by 4113 to 4351%.

Scenic viewsheds and open space proximity
Scenic viewsheds and open space proximity were both calculated as relative rankings, with theoretical source, sink, and use values ranging from 0 to 100. The most highly valued views, e.g., of tall mountains, or most valuable open space types exhibited larger values on this scale than more modestly valued views, e.g., water bodies and shorter mountains, or less valuable open space types, reflecting hedonic valuation studies (Bourassa et al. 2004, McConnell and Walls 2005; Figs. 4a, 5a). When multiple users had views of, or proximity to, a single point on the landscape, that value for these nonrival services was multiplied by the number of users, so theoretical and actual values were not directly comparable (Figs. 4b, 5b). However, we can compare theoretical and actual values by setting provision to a value of zero in areas where views or open space are inaccessible to users (Figs. 4c, 5c). Doing so shows that 15.7% of the region's theoretical viewshed value is visible to homeowners and 43.3% of the region's theoretical open space proximity value is actually accessible to homeowners.

Fig. 3. Carbon (a) sequestration and (b) storage in the Puget Sound Basin.
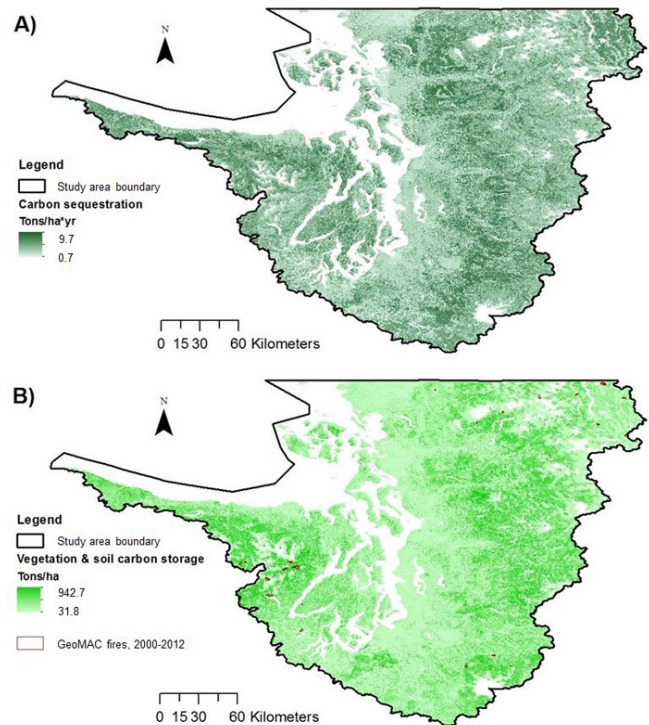
Flood regulation
We estimated that flood sinks can theoretically infiltrate, absorb, and detain 10.2 billion m³ of floodwater/year throughout the Puget Sound Basin, which is 18% of the region's average annual precipitation of 56.9 billion m³ (Figs. 6a, b). However, 56% of subwatersheds lack floodplain development. Actual flood regulation is thus concentrated in lower elevation subwatersheds that lie closer to the Puget Sound and have a greater concentration of development in floodplains (Fig. 6c). When sink values for subwatersheds without floodplain development are assigned a value of zero, the actual flood sink amounts to 65.9% of the theoretical value (Fig. 6d).

Sediment regulation
Finally, we estimated a maximum of 11,032 kT of mobilized sediment/year across the Puget Sound Basin, and more than 45.7 kT of sediment deposition/year in floodplains (Fig. 7). Of this total, erosion of 2405 kT/year of sediment occurs upstream of the 29 major reservoirs in the Puget Sound Basin, and just over 10.1 kT/year of floodplain deposition were mapped above reservoirs. Total land area of the upstream watersheds that are hydrologically linked to these reservoirs is nearly 17% of the land area in the Puget Sound Basin, and the actual source and sink values for erosion and deposition above reservoirs amount to 21.8 and 22.1%, respectively, of their aggregated theoretical values.

Table 2. Theoretical and actual service provision and use in Puget Sound Basin.

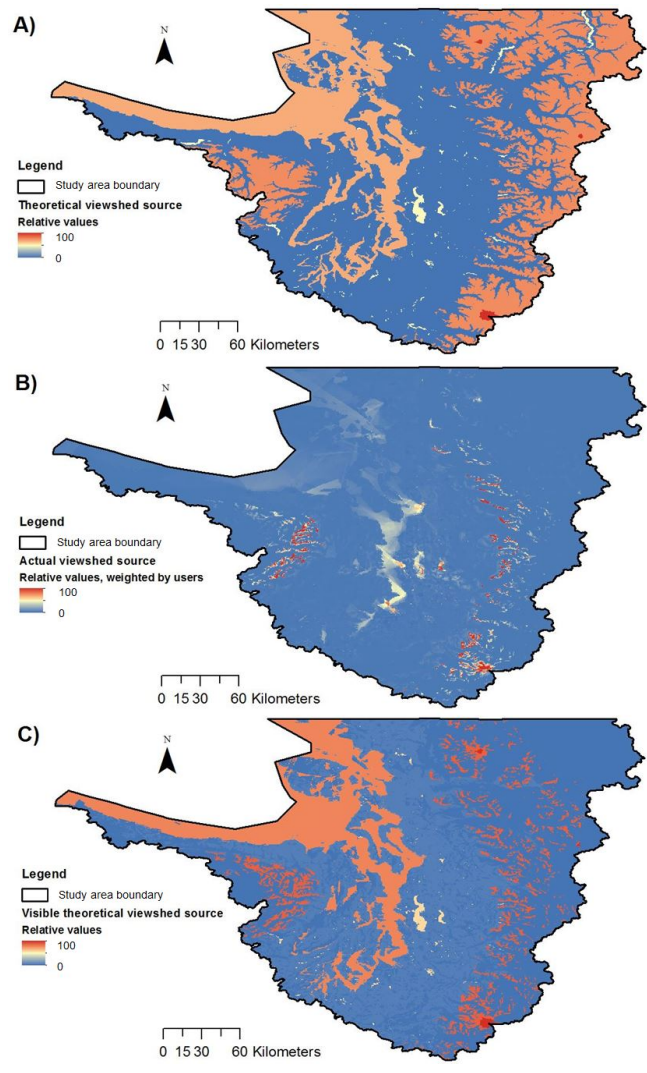| Service (units)	Metric	Explanation	Quantity
Carbon sequestration & storage (kT carbon/year)	Theoretical source	Carbon sequestration	436
	Actual use	Emissions from fire, urbanization, and anthropogenic sources less carbon sequestration	16,947 to 16,969
	Actual to Theoretical ratio		4113 to 4351%
Scenic viewsheds for homeowners (relative ranking, 0-100)	Theoretical source	Model results for viewshed quality	23,325,918
	Actual source	Viewshed quality for portion of landscape actually visible to homeowners	3,657,306
	Actual to Theoretical ratio		15.7%
Open space proximity for homeowners (relative ranking, 0-100)	Theoretical source	Model results for open space quality	25,210,195
	Actual source	Open space quality for portion of landscape actually accessible to homeowners	10,926,397
	Actual to Theoretical ratio		43.3%
Flood regulation for developed land in 100-year floodplain (m³ water/year)	Theoretical sink	Model results for floodwater interception, absorption, and detention	10,221,348,000
	Actual sink	Flood sink values within subwatersheds with flood-vulnerable property	6,735,860,000
	Actual to Theoretical ratio		65.9%
Sediment regulation for reservoirs (kT sediment/year)	Theoretical sink	Model results for deposition of eroded sediment	45.7
	Actual sink	Sediment sink values in watersheds upstream of reservoirs	10.1
	Actual to Theoretical ratio		22.1%

DISCUSSION

Ecosystem service values in the Puget Sound region

We did not attempt to exhaustively estimate ecosystem service values for the Puget Sound region, either for all relevant services or all beneficiary groups for those services that were analyzed. For instance, we could also have quantified flood regulation for farmers, the impacts of sediment delivery on drinking water quality and habitat for salmon or other fisheries, or scenic viewsheds for recreationists, particularly important given the region's natural beauty, the importance of its tourist economy, and the presence of several well-known national parks. A consideration of additional beneficiaries will of course yield different results. However, as long as beneficiary groups are distinct, the problem of double counting, often discussed in the ecosystem services literature, should successfully be avoided (Nahlik et al. 2012).

Excluding carbon sequestration and storage, a global service for which actual use greatly exceeded theoretical provision, actual service provision ranged between about 16% and 66% of corresponding theoretical values (Table 3). Because ecosystem services are by definition an anthropocentric concept, beneficiaries or users must be spatially connected to regions providing a service for that service to have value, with the exception of global services like carbon sequestration or some nonuse values. Research on spatial discounting has shown ecosystem service values to decline as distances between ecosystems and their beneficiaries increase (TEEB 2010). Whereas most such analyses have used Euclidean distance to a

Fig. 4. Theoretical and actual viewsheds in the Puget Sound Basin, showing (a) theoretical values, (b) actual values weighted by the number of homeowners, and (c) theoretical values actually visible to homeowners.

Table 3. Overall ratios of actual to theoretical service values in the Puget Sound Basin.

Service	Actual provision as a percentage of theoretical provision
Carbon sequestration & storage	4113 to 4351%
Scenic viewsheds	15.7%
Open space proximity	43.3%
Sediment regulation (sink values)	22.1%
Flood regulation	65.9%

Fig. 5. Open space proximity values in the Puget Sound Basin, showing (a) theoretical values, (b) actual values weighted by the number of homeowners, and (c) theoretical values actually accessible to homeowners.
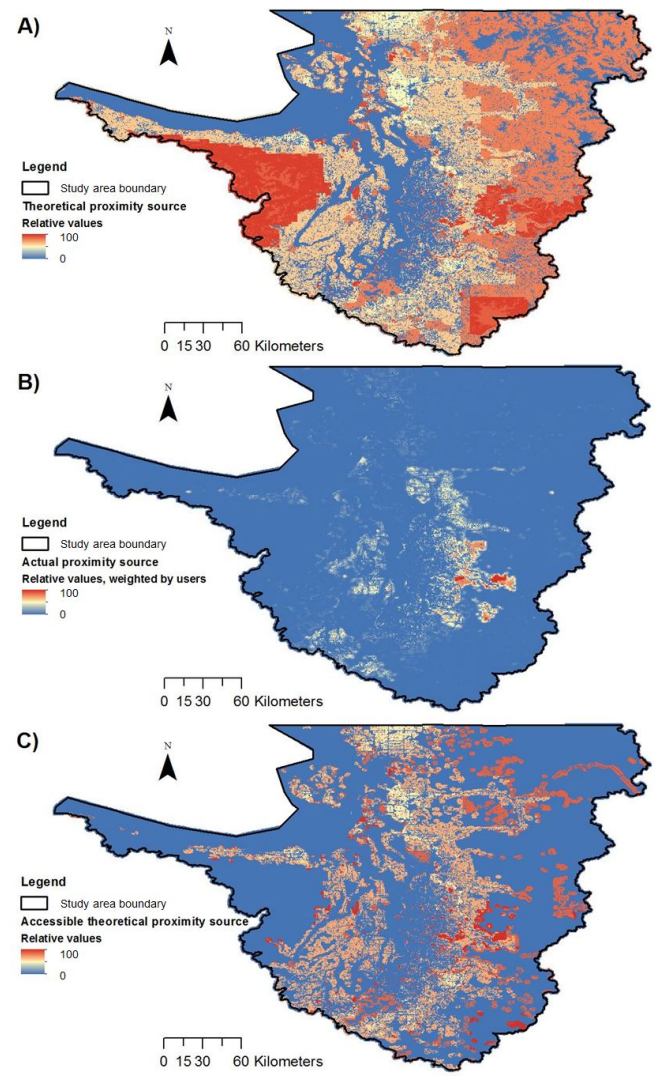

resource, a more correct approach might spatially discount ecosystem service values using service-specific flow paths. By not considering the location of beneficiaries relative to ecosystems, some ecosystem service values may be substantially overvalued (TEEB 2010).

Thus, past studies that mapped only theoretical service provision have often overstated values. Studies that model ecosystem service flows through an overlay analysis of service provisioning and benefitting regions may similarly over- or underestimate ecosystem service values by oversimplifying ecosystem service flow dynamics. When unit-based economic valuation is applied, e.g., a per-unit avoided cost per ton of sediment, or a social cost per ton of carbon, it is critical that actual values are used rather than theoretical values, to avoid overestimating the true economic value of a given service. This does not imply that all service values

Fig. 6. Theoretical and actual values for flood regulation in the Puget Sound Basin, showing (a) sources, (b) theoretical sinks and users, (c) actual values weighted by the number of users, and (d) theoretical values providing actual flood regulation.
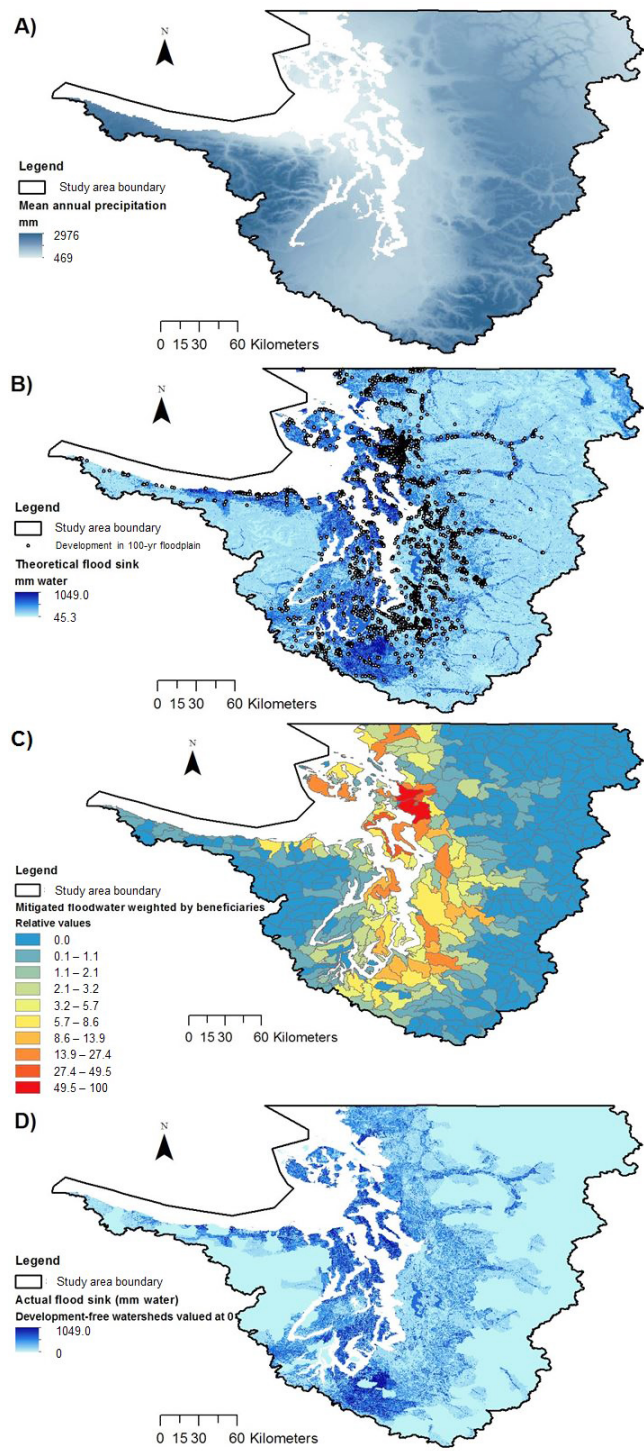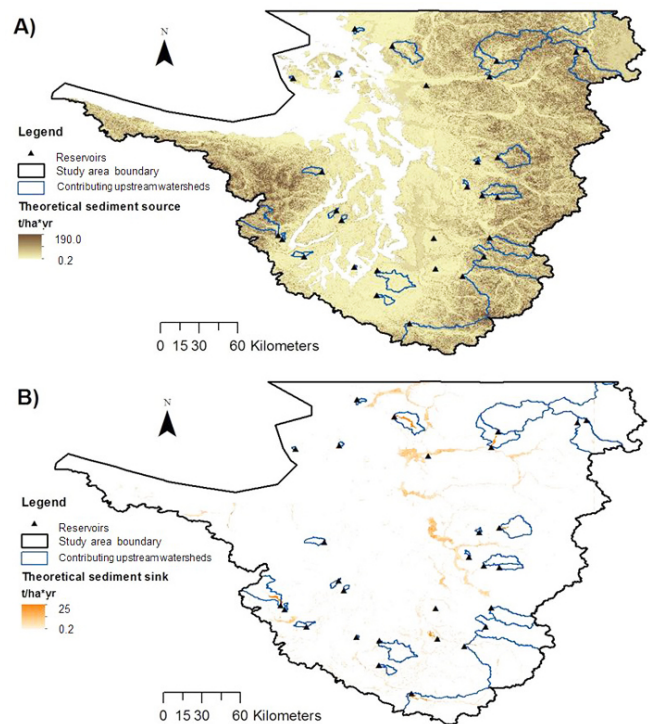

Fig. 7. Theoretical and actual (a) sources and (b) sinks for erosion and sediment deposition in the Puget Sound Basin. Actual sources and sinks are limited exclusively to sub-watersheds upstream of reservoirs

have been overestimated: for instance, those related to climate change and disaster regulation are often likely underestimated, as risk exposure and rebuilding costs have increased substantially in recent years.

Within the Puget Sound Basin, substantial parts of the Olympic and Cascade mountain ranges do not provide actual viewshed and open space proximity value to homeowners because of the lack of people with views of or proximity to these remote areas. Flood regulation value is provided only in subwatersheds with developed land in floodplains, and we only quantified sediment regulation value upstream of reservoirs. In a notable example of the importance of beneficiary location on service delivery, the removal of the Elwha and Glines Canyon dams began in 2011 with the goal of restoring salmon habitat on the Elwha River (Duda et al. 2008). The removal of these hydroelectric dams entailed the trade-off of potential hydroelectric power for the restoration of the river's salmon runs. Removal of the dams increased sediment flows to the Strait of Juan de Fuca, forming drift cells and new estuarine habitat and replenishing geomorphic features that provide coastal flood regulation benefits (Flores et al. 2013). Because we only mapped the value of sediment regulation for reservoirs in this study, we did not map the provision of sediment regulation upstream of the former Elwha River reservoirs as actual values.

Placing more beneficiaries across the landscape may have the effect of increasing ecosystem service flows and, by consequence,

actual values, but can also degrade the ecosystem's underlying ability to provide the same services, i.e., theoretical values. For instance, an expansion of the urban footprint yields an increase in beneficiaries in locations where ecosystem service flows were previously inaccessible. However, land-cover change associated with new development often reduces an ecosystem's capacity to provide services, i.e., their theoretical source values (Bagstad et al. 2012). Theoretical services may appear to be underutilized in a less-developed landscape with fewer beneficiaries, but decision makers should be aware that choices that increase actual service use through increasing access to more beneficiaries may simultaneously degrade ecosystems' capacity to provide services, i.e., theoretical source values.

Further, new beneficiaries may result in more acute trade-offs between services, as in the above-mentioned example of trade-offs between hydroelectric power, sediment transport, and salmon fisheries on the Elwha River. More beneficiaries at risk of flooding may increase the value of upstream flood regulation for minor flooding events, but at greater social cost and exposure to disaster risk when large events occur. For these reasons, four steps are important for ecosystem service quantification and decision making to protect and maintain service flows: (1) analysis of the full range of relevant ecosystem services, (2) awareness of and accounting for trade-offs between ecosystem service delivery and resource management alternatives, (3) accurate quantification and mapping of ecosystem service supply, demand, and flows, and (4) avoidance of a narrowly focused emphasis on maximizing ecosystem service values. For instance, related to the last point, higher actual ecosystem service values often imply growing social vulnerability coupled with increased scarcity and reduced resilience of natural capital, i.e., rising demand accompanied by declining theoretical service provision. In such cases, high ecosystem service values are more correctly viewed as socially undesirable.

Next steps
Two key upcoming steps will expand the applicability and accuracy of the ARIES modeling environment: the development of global models and supporting architecture for intelligent model selection, and the incorporation of external biophysical process models that more accurately represent ecosystem service production and flows. In the complex, diverse contexts that characterize a typical ecosystem service assessment, the oversimplification and structural rigidity of a "one model fits all" approach can compromise a model's utility in addressing specific values and trade-offs and informing decision needs. For this reason, the ARIES methodology aims to enable structural flexibility through an artificial intelligence (AI)-assisted modeling approach (Villa 2010) that can automatically choose model components that reflect context-specific data availability and understanding of ecosystem services. The view of ecosystem services as independent, linked source, sink, and use conditions joined through a flow process (Johnson et al. 2012, Bagstad et al. 2013, Villa et al. 2014) provides built-in modularity that fits well within an automatic model building method.

This integrated modeling approach supports the mixing of data-driven and hypothesis-driven models to select the overall approach most suited to the assessment context (Vigerstol and Aukema 2011). The capability to rank model components and choose the best available for given model contexts and data availability is a fundamental design criterion for ARIES. Data-driven approaches, such as Bayesian networks, are prioritized by ARIES when computing static components such as production functions lacking consensus methodologies. Hypothesis-driven approaches, used for flow models or trusted process-based models that have gained decision-maker confidence through years of use, are preferred when the dynamic complexity of a phenomena, e.g., sediment or water transport, are well understood and adequate data are available for parameterization. Well-known, open source models for a variety of physical processes are being integrated to extend the ARIES model base. Among these, the CAESAR-LISFLOOD flood and erosion model (T. Coulthard, http://www.coulthard.org.uk/CAESARLisflood.html) and a revised version of the ecosystem model LPJ-GUESS (Smith et al. 2011) are being integrated to improve the detail of flood, sediment, nutrient, carbon and primary production dynamics, without sacrificing usability and simplicity of the system for decision makers.

The development of a set of models that run based only on global data, extending automatically to more specialized models when the knowledge base and available data support it, is ongoing. This set of common denominator models will, over time, help meet the needs of a larger share of users, gain greater acceptance, and grow in utility and sophistication. At the same time, we expect that the modular construction of the ARIES model base will translate into easier workflows for end users, who will be able to query the system in simple ways and obtain results that automatically reflect the best available knowledge for their context. The independent extensibility of the model base is another advantage of this development paradigm. The development of the ARIES model base is increasingly benefitting from a community process that links together modelers located worldwide. An intensive modeling school is held annually by ARIES developers (Basque Centre for Climate Change, http://www.bc3research.org/springuniversity/); open source models and ontologies developed during such courses help address local resource management challenges while integrating seamlessly with the model base and extending it for the benefit of future users.

Policy implications of mapping service provision, use, and flows
The results presented in this paper have numerous practical uses for conservation and economic development planning. Notably, they identify which regions are critical to maintaining the supply and flow of benefits for specific beneficiary groups. By prioritizing conservation and restoration activities on sources and sinks for one or more ecosystem services, service flows may be maintained or increased. Conversely, focusing development or resource extraction outside these critical source and flow regions can prevent the degradation of service flows. The impacts on human well-being for specific beneficiary groups from a proposed landscape alteration can be more fully evaluated if improvements or declines in realized ecosystem services can be demonstrated. By identifying parties that benefit from access to, or whose use degrades service flows, this knowledge can also provide guidance for beneficiary-pays or polluter-pays based payments for ecosystem services programs (Salzman 2005). For a given service, maps can be generated (1) for an ecosystem, showing the beneficiary groups receiving benefits from that region of interest or (2) for a beneficiary group, identifying the locations on the landscape from which that user's benefits are derived (Johnson et al. 2012). Finally, basing economic valuation on maps of actual

rather than theoretical service provision should improve the accuracy and credibility of valuation for use in decision making. Although the results we presented in this paper are aggregated across the entire Puget Sound Basin, further quantitative analysis of results, i.e., clustering, hotspots, or other analyses, would be instructive, particularly for decision making. Further quantitative spatial analysis of these results is underway and will be presented in a future paper.

Understanding the flow paths of benefits from ecosystems to people is a problem that has eluded past work in ecosystem services (Ruhl et al. 2007, Tallis et al. 2008). For many researchers, the flow problem has been expressed as a spatial mismatch between ecosystem service provision and use (Costanza 2008, Fisher et al. 2009). More recently, concepts of service provisioning, service benefitting, and service connecting regions have advanced the science of ecosystem service mapping (Syrbe and Walz 2012, Palomo et al. 2013); although these concepts still fall short of a methodology for consistently quantifying ecosystem service flows (Bagstad et al. 2013). By explicitly demonstrating spatial links from ecosystems to people and the difference between theoretical and actual services, we can better illustrate how specific beneficiary groups gain value from ecosystem services. This can instruct policy, providing new information about the winners and losers in management actions that impact ecosystem services. Mapping theoretical and actual ecosystem services through spatially explicit modeling of beneficiaries and spatial flows is an important step in raising awareness of the value of ecosystem services. This can lead to both better appreciation of their value by the groups that benefit most from nature's services, and a stronger body of knowledge to support sound resource management.

Responses to this article can be read online at:
http://www.ecologyandsociety.org/issues/responses.php/6523

LITERATURE CITED

Bagstad, K. J., G. W. Johnson, B. Voigt, and F. Villa. 2013. Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. *Ecosystem Services* 4:117-125. http://dx.doi.org/10.1016/j.ecoser.2012.07.012

Bagstad, K. J., D. Semmens, R. Winthrop, D. Jaworski, and J. Larson. 2012. *Ecosystem services valuation to support decision making on public lands: a case study for the San Pedro River, Arizona.* Scientific Investigations Report 2012-5251, U.S. Geological Survey, Reston, Virginia, USA.

Bagstad, K. J., F. Villa, G. W. Johnson, and B. Voigt. 2011. *ARIES – Artificial Intelligence for Ecosystem Services: a guide to models and data, version 1.0 Beta.* The ARIES Consortium, Bilbao, Spain. [online] URL: http://www.ariesonline.org/docs/ ARIESModelingGuide1.0.pdf

Beier, C. M., T. M. Patterson, and F. S. Chapin III. 2008. Ecosystem services and emergent vulnerability in managed ecosystems: a geospatial decision-support tool. *Ecosystems* 11 (6):923-938. http://dx.doi.org/10.1007/s10021-008-9170-z

Bolte, J. P., and K. Vache 2010. *Envisioning Puget Sound alternative futures: PSNERP Final Report.* Puget Sound Nearshore Ecosystem Restoration Project, Olympia, Washington, USA. [online] URL: http://www.pugetsoundnearshore.org/ supporting_documents/FRAP%20final%20report.pdf

Bourassa, S. C., M. Hoesli, and J. Sun. 2004. What's in a view? *Environment and Planning A* 36:1427-1450. http://dx.doi. org/10.1068/a36103

Boyd, J., and S. Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63:616-626. http://dx.doi.org/10.1016/j.ecolecon.2007.01.002

Brauman, K. A., G. C. Daily, T. K. Duarte, and H. A. Mooney. 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. *Annual Review of Environment and Resources* 32:67-98. http://dx.doi.org/10.1146/annurev. energy.32.031306.102758

Burkhard, B., F. Kroll, S. Nedkov, and F. Muller. 2012. Mapping ecosystem service supply, demand and budgets. *Ecological Indicators* 21:17-29. http://dx.doi.org/10.1016/j.ecolind.2011.06.019

Costanza, R. 2008. Ecosystem services: multiple classification systems are needed. *Biological Conservation* 141:350-352. http:// dx.doi.org/10.1016/j.biocon.2007.12.020

Costanza, R, R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253-260.

Coulthard, T. J., and M. G. Macklin. 2003. Modeling long-term contamination in river systems from historical metal mining. *Geology* 31(5):451-454. http://dx.doi.org/10.1130/0091-7613 (2003)031<0451:MLCIRS>2.0.CO;2

Cowell, R. G., A. P. Dawid, S. L. Lauritzen, and D. J. Spiegelhalter. 1999. *Probabilistic networks and expert systems.* Springer, New York, New York, USA.

Daily, G. C., editor. 1997. *Nature's services: societal dependence on natural ecosystems.* Island, Washington, D.C., USA.

de Groot, R. S., M. A. Wilson, and R. M. J. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41:393-408. http://dx.doi.org/10.1016/S0921-8009(02)00089-7

Duda, J. J., J. E. Freilich, and E. G. Schreiner. 2008. Baseline studies in the Elwha River ecosystem prior to dam removal: introduction to the special issue. *Northwest Science* 82:1-12. http://dx.doi.org/10.3955/0029-344X-82.S.I.1

Egoh, B., E. G. Drakou, M. B. Dunbar, J. Maes, and L. Willemen. 2012. *Indicators for mapping ecosystem services: a review*. Report EUR 25456 EN. Publications Office of the European Union, Luxembourg.

Eigenbrod, F., P. R. Armsworth, B. J. Anderson, A. Heinemeyer, S. Gillings, D. B. Roy, C. D. Thomas, and K. J. Gaston. 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology* 47(2):377-385. http://dx.doi.org/10.1111/j.1365-2664.2010.01777.x

Fisher, B., R. K. Turner, and P. Morling. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics* 68:643-653. http://dx.doi.org/10.1016/j.ecolecon.2008.09.014

Flores, L., J. Harrison-Cox, S. Wilson, and D. Batker. 2013. *Nature's value in Clallam County: the economic benefits of feeder bluffs and 12 other ecosystems*. Earth Economics, Tacoma, Washington, USA.

Gaydos, J. K., and N. A. Brown. 2011. Species of concern within the Salish Sea: changes from 2002 to 2011. Proceedings of the 2011 *Salish Sea Ecosystem Conference,* October 25-27, 2011, Vancouver, British Columbia, Canada. The SeaDoc Society, Eastsound, Washington, USA. [online] URL: http://www.seadocsociety.org/files/pdfs/Salish%20Sea%20Species%20of%20Concern%202011%20Final.pdf

Haines-Young, R., and M. Potschin. 2011. *Common international classification of ecosystem services (CICES): 2011 Update*. Contract No. EEA/BSS/07/007, European Environment Agency, Copenhagen, Denmark.

Johnson, G. W., K. J. Bagstad, R. R. Snapp, and F. Villa. 2012. Service path attribution networks (SPANs): a network flow approach to ecosystem service assessment. *International Journal of Agricultural and Environmental Information Systems* 3 (2):54-71. http://dx.doi.org/10.4018/jaeis.2012070104

Kareiva, P., H. Tallis, T. H., Ricketts, G. C., Daily, and S. Polasky, editors. 2011. *Natural capital: theory and practice of mapping ecosystem services*. Oxford University Press, Oxford, UK. http://dx.doi.org/10.1093/acprof:oso/9780199588992.001.0001

Keitt, T. H. 2009. Habitat conversion, extinction thresholds, and pollination services in agroecosystems. *Ecological Applications* 19 (6):1561-1573. http://dx.doi.org/10.1890/08-0117.1

Kremen, C., N. M. Williams, M. A. Aizen, B. Gemmill-Harren, G. LeBuhn, R. Minckley, L. Packer, S. G. Potts, T. Roulston, I. Steffan-Dewenter, D. P. Vázquez, R. Winfree, L. Adams, E. E. Crone, S. S. Greenlead, T. H. Keitt, A.-M. Klein, J. Regetz, and T. H. Ricketts. 2007. Pollination and other ecosystem services

produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecology Letters* 10:299-314. http://dx.doi.org/10.1111/j.1461-0248.2007.01018.x

Laffoley, D., and G. Grimsditch. 2009. *The management of natural coastal carbon sinks.* International Union for Conservation of Nature and Natural Resources (IUCN), Gland, Switzerland.

Madin, J. S., S. Bowers, M. P. Schildhauer, and M. B. Jones. 2008. Advancing ecological research with ontologies. *Trends in Ecology and Evolution* 23(3):159-168. http://dx.doi.org/10.1016/j.tree.2007.11.007

Martínez-Harms, M. J., and P. Balvanera. 2012. Methods for mapping ecosystem service supply: a review. *International Journal of Biodiversity Science, Ecosystem Services & Management* 8 (1-2):17-25. http://dx.doi.org/10.1080/21513732.2012.663792

McCann, R. K., B. G. Marcot, and R. Ellis. 2006. Bayesian belief networks: applications in ecology and natural resource management. *Canadian Journal of Forest Research* 36:3053-3062. http://dx.doi.org/10.1139/x06-238

McConnell, V., and M. Walls. 2005. *The value of open space: evidence from studies of nonmarket benefits.* Resources for the Future, Washington, D.C., USA

McDonald, R. I. 2009. Ecosystem service supply and demand along the urban-to-rural gradient. *Journal of Conservation Planning* 5:1-14.

Millennium Ecosystem Assessment (MA). 2005. *Millennium Ecosystem Assessment: living beyond our means-natural assets and human well-being*. World Resources Institute, Washington, D.C., USA.

Mitchell, M. G. E., E. M. Bennett, and A. Gonzalez. 2013. Linking landscape connectivity and ecosystem service provision: current knowledge and research gaps. *Ecosystems* 16:894-908. http://dx.doi.org/10.1007/s10021-013-9647-2

Molnar, M., M. Kocian, and D. Batker. 2012. *Valuing the aquatic benefits of British Columbia's Lower Mainland: nearshore natural capital valuation.* David Suzuki Foundation and Earth Economics, Vancouver, British Columbia, Canada.

Moore, S. K., N. J. Mantua, and E. P. Salathé Jr. 2011. Past trends and future scenarios for environmental conditions favoring the accumulation of paralytic shellfish toxins in Puget Sound shellfish. *Harmful Algae* 10:521-529. http://dx.doi.org/10.1016/j.hal.2011.04.004

Mote, P. W., A. K. Snover, L. Whitely Binder, A. F. Hamlet, and N. J. Mantua. 2005. *Uncertain future: climate change and its effects on Puget Sound - Foundation Document*. Climate Impacts Group, Center for Science in the Earth System, Joint Institute for the Study of the Atmosphere and Oceans, University of Washington, Seattle, Washington, USA.

Nahlik, A. M., M. E. Kentula, M. S. Fennessy, and D. H. Landers. 2012. Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. *Ecological Economics* 77:27-35. http://dx.doi.org/10.1016/j.ecolecon.2012.01.001

National Oceanic and Atmospheric Administration (NOAA). 2013. *The Northwest Region*. NOAA, Washington, D.C., USA. [online] URL: http://www.habitat.noaa.gov/restoration/regional/northwest.html

Nedkov, S., and B. Burkhard. 2012. Flood regulating ecosystem services – mapping supply and demand, in the Etropole municipality, Bulgaria. *Ecological Indicators* 21:67-79. http://dx.doi.org/10.1016/j.ecolind.2011.06.022

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K. M. A. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts, and M. R. Shaw. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* 7(1):4-11. http://dx.doi.org/10.1890/080023

Palomo, I., B. Martín-López, M. Potschin, R. Haines-Young, and C. Montes. 2013. National parks, buffer zones and surrounding lands: mapping ecosystem service flows. *Ecosystem Services* 4:104-116. http://dx.doi.org/10.1016/j.ecoser.2012.09.001

PRISM Climate Group. 2009. *PRISM climate data*. PRISM Climate Group, Northwest Alliance for Computational Science & Engineering, Oregon State University, Corvallis, Oregon, USA. [online] URL: http://prism.oregonstate.edu

Puget Sound Partnership. 2012*b*. *State of the Sound report*. Puget Sound Partnership, Tacoma, Washington, USA.

Puget Sound Partnership. 2012*a*. *The 2012/2013 action agenda for Puget Sound*. Puget Sound Partnership, Tacoma, Washington, USA.

Ramseur, J. L. 2007. *CRS Report for Congress: state greenhouse gas emissions: comparison and analysis*. Congressional Research Service, Washington, D.C., USA.

Reaney, S. M. 2008. The use of agent based modelling techniques in hydrology: determining the spatial and temporal origin of channel flow in semi-arid catchments. *Earth Surface Processes and Landforms* 33:317-327. http://dx.doi.org/10.1002/esp.1540

Renard, K. G., G. R. Foster, G. A. Weesies, D. K. McCool, and D. C. Yoder. 1996. *Predicting soil erosion by water: a guide to conservation planning with the revised universal soil loss equation (RUSLE)*. Handbook 703, U.S. Department of Agriculture, Washington, D.C., USA.

Ruhl, J. B., S. E. Kraft, and C. L. Lant. 2007. *The law and policy of ecosystem services*. Island, Washington, D.C., USA.

Salzman, J. 2005. Creating markets for ecosystem services: notes from the field. *New York University Law Review* 80:870-961.

Schägner, J. P., L. Brander, J. Maes, and V. Hartje. 2013. Mapping ecosystem services' values: current practice and future prospects. *Ecosystem Services* 4:33-46. http://dx.doi.org/10.1016/j.ecoser.2013.02.003

Semmens, D. J., J. E. Diffendorfer, L. López-Hoffman, and C. D. Shapiro. 2011. Accounting for the ecosystem services of migratory species: quantifying migration support and spatial subsidies. *Ecological Economics* 70:2236-2242. http://dx.doi.org/10.1016/j.ecolecon.2011.07.002

Smith, B., P. Samuelsson, A. Wramneby, and M. Rummukainen. 2011. A model of the coupled dynamics of climate, vegetation and terrestrial ecosystem biogeochemistry for regional applications. *Tellus A* 63(1):87-106. http://dx.doi.org/10.1111/j.1600-0870.2010.00477.x

Smith, J. E., L. S. Heath, K. E. Skog, and R. A. Birdsey. 2006. *Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States*. Northeastern Research Station General Technical Report NE-343, United States Department of Agriculture Forest Service, Newton Square, Pennsylvania, USA.

Spracklen, D. V., L. J. Mickley, J. A. Logan, R. C. Hudman, R. Yevich, M. D. Flannigan, and A. L. Westerling. 2009. Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States. *Journal of Geophysical Research* 114:D20301. http://dx.doi.org/10.1029/2008JD010966

Syrbe, R.-U., and U. Walz. 2012. Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. *Ecological Indicators* 21:80-88. http://dx.doi.org/10.1016/j.ecolind.2012.02.013

Tallis, H., P. Kareiva, M. Marvier, and A. Chang. 2008. An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences* 105(28):9457-9464. http://dx.doi.org/10.1073/pnas.0705797105

The Economics of Ecosystems and Biodiversity (TEEB). 2010. The economics of valuing ecosystem services and biodiversity. Chapter 5 *in* P. Kumar, editor. *The economics of ecosystems and biodiversity: Ecological and economic foundations*. Earthscan, London, UK.

Troy, A., and M. A. Wilson. 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics* 60:435-449. http://dx.doi.org/10.1016/j.ecolecon.2006.04.007

U.S. Geological Survey. 2006. *USGS Science for Puget Sound Restoration and Preservation: Coastal Habitats in Puget Sound (CHIPS)*. U.S. Geological Survey, U.S. Department of the Interior, Washington, D.C., USA. [online] URL: http://pubs.usgs.gov/fs/2006/3081/pdf/fs20063081.pdf

Vigerstol, K. L., and J. E. Aukema. 2011. A comparison of tools for modeling freshwater ecosystem services. *Journal of Environmental Management* 92:2403-2409. http://dx.doi.org/10.1016/j.jenvman.2011.06.040

Villa, F. 2010. Bridging scales and paradigms in natural systems modelling. Metadata and semantic research. *Communications in Computer and Information Science* 108:1-7. http://dx.doi.org/10.1007/978-3-642-16552-8_1

Villa, F., I. N. Athanasiadis, and A. E. Rizzoli. 2009. Modelling with knowledge: a review of emerging semantic approaches to environmental modelling. *Environmental Modelling and Software* 24:577-587. http://dx.doi.org/10.1016/j.envsoft.2008.09.009

Villa, F., K. J. Bagstad, B. Voigt, G. W. Johnson, R. Portela, M. Honzák, and D. Batker. 2014. A methodology for adaptable and robust ecosystem services assessment. *PLOS One* 9(3):e91001. http://dx.doi.org/10.1371/journal.pone.0091001

Wallace, K. J. 2007. Classification of ecosystem services: problems and solutions. *Biological Conservation* 139:235-246. http://dx.doi.org/10.1016/j.biocon.2007.07.015

Washington State Department of Ecology. 2013. *Saving the Sound*. Washington State Department of Ecology, Olympia, Washington, USA. [online] URL: http://www.ecy.wa.gov/puget_sound/Overview.html