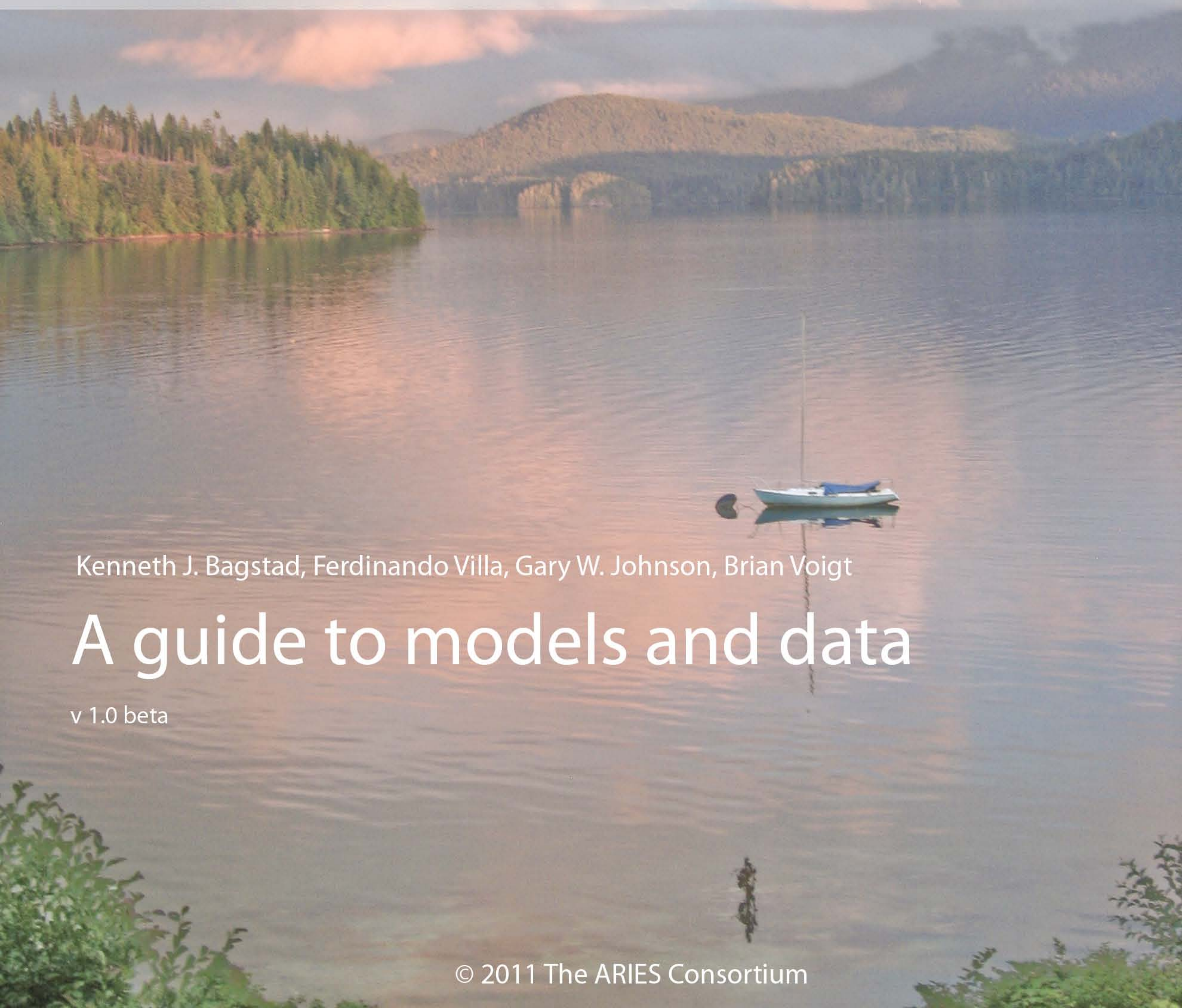
ARIES

Artificial Intelligence for Ecosystem Services



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A guide to models and data

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List of Acronyms

AGIC – Arizona Geographic Information Council
BLM – United States Department of Interior Bureau of Land Management
CDIAC – Carbon Dioxide Information Analysis Center
CIESIN - Center for International Earth Science Information Network
EPA – United States Environmental Protection Agency
FAO – Food and Agriculture Organization of the United Nations
FEMA – Federal Emergency Management Agency (United States)
FTM – Foiben-Taosarintanin'i Madagasikara (Madagascar National Mapping Agency)
GLCF/UMD – University of Maryland Global Land Cover Facility
INECOL – Instituto de Ecologia (Mexico)
NASA JPL – National Aeronautics and Space Administration, Jet Propulsion Laboratory
NBII/MEA – National Biological Information Infrastructure (United States) & Millennium Ecosystem Assessment
NLCD – National Land Cover Dataset (United States)
NOAA-NGDC – National Oceanic & Atmospheric Administration (United States) – National Geophysical Data Center
Oregon DOF – Oregon Department of Forestry
ORNL – Oak Ridge National Laboratory
PRISM/OSU – Oregon State University, PRISM Climate Group
SAGE/UW – Mad – University of Wisconsin, Madison Center for Sustainability and the Global Environment
SPRNCA - San Pedro Riparian National Conservation Area
SRTM – Shuttle Radar Topography Mission
SSURGO – Soil Survey Geographic Database (United States Department of Agriculture-Natural Resources Conservation Service)
STATSGO – United States General Soil Map (United States Department of Agriculture-Natural Resources Conservation Service)
SWReGAP – Southwest Regional GAP analysis
TIGER – Topologically Integrated Geographic Encoding and Reference System (United States Census Bureau)
TNC – The Nature Conservancy
UNEP-WCMC – United Nations Environment Program – World Conservation Monitoring Centre
USFS – United States Department of Agriculture Forest Service
USGS – United States Geological Survey
VCGI – Vermont Center for Geographic Information
VT AOT – Vermont Agency of Transportation
Washington DNR – Washington State Department of Natural Resources
Washington DOE – Washington State Department of Ecology

1. The ARIES Approach to Modeling Ecosystem Services

1.1 Introduction

Ecosystem services are the many economic benefits provided by nature to humans. With the growing interest in incorporating ecosystem services into decision making about how human economies and nature interact across multiple scales, there is a growing need for quantitative methods and tools to model the complex relationships between ecosystems and human activities and values. ARIES is an innovative and unique tool for decision-makers and researchers to answer precisely these types of questions.

To date, numerous researchers have used spatial data to map and value ecosystem services for particular case studies (Eade and Moran 1996, Chan et al. 2006, Raudsepp-Hearne et al. 2010), and in recent years, ecosystem service-specific computing tools have emerged to systematize the mapping and valuation process (Tallis et al. 2011). ARIES (ARtificial Intelligence for Ecosystem Services) is a novel methodology and software platform that differs in four key ways from previous approaches to ecosystem services quantification and valuation (Villa et al. 2009).

First, rather than delivering simply a single model or collection of models for ecosystem services assessment, ARIES provides an intelligent modeling platform capable of composing complex ecosystem services models from a collection of models specified by the user. These component models can be defined within ARIES using its native modeling language or developed independently in another language or architecture and used by ARIES via its model-wrapping mechanism. Once properly wrapped, ARIES is capable of automatically negotiating the differences in input data, units, modeling paradigms and applicable scales between component models.

Additionally, model composition can be defined conditionally within ARIES, enabling different component models to be replaced dynamically based on the spatial, temporal, cultural, or other contexts of the ecosystem service assessment. Once these conditions are defined, the user can simply select the contexts of their case study and run the models of interest to them. Without any further assistance, ARIES can incorporate existing ecological process models where appropriate and turn to internally-defined models where the known process models are inadequate for local contexts.

The second difference between ARIES and previous approaches is that, once defined and composed, ARIES models may be accessed and run remotely through any web browser, with all calculations handled by a separate model server and results returned to the user via a web interface. The advantage is that data and model storage, data processing, model runs, and results reporting are managed by the server without requiring the user to purchase or gain proficiency with modeling or geographic information system (GIS) software.

As a third distinction, the top-level ecosystem service models in ARIES are designed to propagate uncertainty throughout all their calculations. Since many models written in the ARIES modeling language are based on a probabilistic, Bayesian approach, they are able to explicitly convey uncertainty about their inputs to their outputs and are capable of operating even in data-scarce conditions where deterministic models cannot run. Bayesian models in ARIES can be either parameterized directly by the modeler or automatically trained to extract the quantitative relationships between their inputs using machine learning techniques (Pearl 1988).

Fourth and finally, ARIES explicitly accounts for the complex spatial dynamics of ecosystem services. Many researchers have noted that provision and use of ecosystem services take place at different temporal and spatial scales (Ruhl et al. 2007, Fisher et al. 2008, Tallis et al. 2008). Yet aside from hydrologic ecosystem service models, researchers have too often ignored the fact that the point of origin of ecosystem services, the location(s) of human beneficiaries, and the spatio-temporal nature of ecosystem service flows[1] affect how much of a service is available for use by people.

ARIES maps the locations and quantity of <u>potential provision</u> of ecosystem services (*sources*), their human beneficiaries (*users*), and any biophysical features that can deplete service flows (*sinks*)[2]. ARIES then uses a family of agent-based algorithms to map the movement of services (*flows*) between source and use locations (Johnson et al. 2010). These algorithms move an ecosystem service carrier across the landscape using the appropriate service-specific flow path (e.g., hydrologic networks for flood, sediment, and nutrient regulation and water supply, lines of sight for scenic views, transportation networks for recreation or ecosystem goods, distance decay for open space proximity and certain non-use values). Flow model characteristics are summarized by ecosystem service in Table 1.

Several other key characteristics enable ecosystem services flow mapping in ARIES. Source, sink, and use values are represented in either concrete units (e.g., tons of CO_2, mm of water, kg of fish) or abstract units (e.g., aesthetic value or recreation site quality,

[1] Others in the ecosystem services literature have referred to "flows" of ecosystem services, typically in terms of stock-flow dynamics in modeling (e.g., Daly and Farley 2004), where services are conceptualized as an annual flow of benefits from nature to people. This approach does not consider the spatial dynamics of services, where a particular biophysical or information carrier must be transmitted from ecosystems to people in order for a service to exist. This flow concept, described by Ruhl et al. (2007) has often been described but rarely if ever quantified.

[2] These sources, sinks, and users may be modeled using deterministic process models, spatial datasets, and/or probabilistic, Bayesian models as appropriate. Initial model development in ARIES has largely used Bayesian models, which were developed using GeNIe, an open source freely available Bayesian network modeling program (Decision Systems Laboratory 2010, http://genie.sis.pitt.edu).

measured from 0-100). Each ecosystem service is further categorized as either a *provisioning* or *preventive* and a *rival* or *non-rival* service, as explained below.

A provisioning service is one in which the matter, energy, or information generated by an ecosystem source is of direct value to human users, such as drinking water, fish, or scenic views. A preventive service is one in which a benefit is provided to people by an ecosystem reducing the flow of something dangerous to them (e.g., excess sediment, nutrients, or flood water). For provisioning services, the source locations provide the ecosystem service benefit and sinks limit the amount of service received, while for preventive services, sink locations (e.g., areas that absorb flood water, sediment, or nutrients) provide protection from detrimental sources. The effects of some service flows like sediment transport may be either beneficial or detrimental, depending on the human user – in some cases excess sediment or excessively turbid waters damage human well-being, while in other cases naturally delivered sediment provides benefits, such as in maintaining soil fertility or natural coastal processes. Finally, understanding whether the benefit is rival or non-rival indicates whether the use of that service by one beneficiary depletes the quantity available to other beneficiaries elsewhere on the landscape.

Table 1: Source, sink, use, and flow characteristics for ARIES modules.

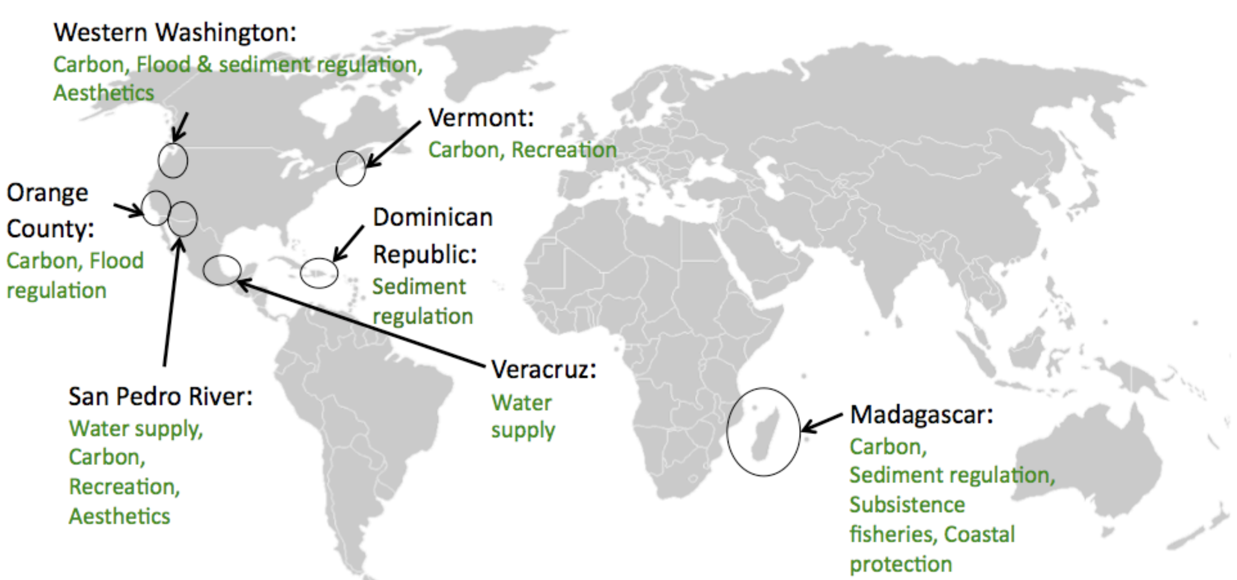
Service	**Carbon sequestration & storage**	**Open space proximity**
Benefit type	Provisioning	Provisioning
Medium/units	Tons CO_2 absorbed/emitted	Open space (abstract units, 0-100)
Scale	Global	Walking distance
Movement	Atmospheric mixing	Walking simulation
Decay	None	Gaussian
Rival?	Rival	Nonrival
Source	Vegetation & soil C sequestration	Open spaces esp. in urban areas
Sink	Stored C release (fire, land use change)	Obstructions (e.g., highways)
Use	CO_2 emitters	Property/housing value
Service	**Aesthetic viewsheds**	**Flood regulation**
Benefit type	Provisioning	Preventive
Medium/units	Scenic beauty (abstract units, 0-100)	Water (runoff, mm/yr)
Scale	Viewshed	Watershed
Movement	Line of sight	Hydrologic flow
Decay	Inverse square	None
Rival?	Nonrival	Nonrival
Source	Mountains, water bodies, etc.	Rainfall & snowmelt
Sink	Visual blight	Water absorbed by soil & vegetation
Use	Property/housing value	Economic assets in floodplains

Service	**Sediment regulation**	**Water supply**
Benefit type	Provisioning or Preventive	Provisioning
Medium/units	Sediment (tons)	Surface or groundwater (mm/yr)
Scale	Watershed	Watershed
Movement	Hydrologic flow	Hydrologic flow, surface & groundwater
Decay	None	None
Rival?	Rival	Rival
Source	Landscapes along waterways	Precipitation, infiltration, and others
Sink	Riparian zones where deposition occurs	Infiltration, evapotranspiration, others
Use	Multiple	Surface water withdrawals or wells
Service	**Coastal flood regulation**	**Subsistence fisheries**
Benefit type	Preventive	Provisioning
Medium/units	Storm surge (m)	Fish (kg)
Scale	Coastal zones	Walking distance
Movement	Wave runup	Walking simulation
Decay	As a function of sinks	Gaussian
Rival?	Nonrival	Rival
Source	Coastal zones prone to storms	Fishing grounds
Sink	Vegetation & topographic features	None
Use	Economic assets in coastal flood zones	Subsistence communities near fisheries
Service	**Recreation**	**Nutrient regulation**
Benefit type	Provisioning	Preventive
Medium/units	Recreational enjoyment (abstract units, 0-100)	Nutrients in water (kg N/P)
Scale	Travel distance	Watershed
Movement	Travel simulation	Hydrologic flow
Decay	Gaussian	None
Rival?	Nonrival but congestible	Nonrival
Source	Recreational areas suitable for a given activity	Landscapes along waterways
Sink	None	Filters in landscape & along waterways
Use	Recreationists interested in a given activity	Multiple

ARIES is capable of switching models for an individual service based on data availability and a broad range of social and ecological contexts. For example, when modeling a particular service across a steep environmental gradient, such as aridity, certain factors might be more important influences on ecological processes in arid versus humid environments. When equipped with appropriate decision rules, ARIES can determine which areas of the landscape to apply which data and models toward, better accounting for key contextual factors in ecosystem service provision. Alternatively a model such as the Revised Universal Soil Loss Equation (Renard et al. 1996), which is known to work poorly in hilly to mountainous terrain, could be applied side-by-side with a data-driven model better capable of accounting for soil erosion on steeper slopes.

ARIES will eventually include a generalized global model that can quantify provision, use, and spatial dynamics of each ecosystem service anywhere on Earth, using global datasets (typically ranging from 1 degree^2 to 1 km^2 in spatial resolution). The ARIES GeoServer currently stores several hundred spatial datasets that can be incorporated into ecosystem service models at global through local scales. ARIES' model library has initially been populated with a set of local case studies chosen to reflect a diverse set of ecological conditions influencing ecosystem service provision and socioeconomic characteristics influencing use of and demand for particular ecosystem services. Local case studies also benefit from finer-scale spatial data, which are often also more descriptive of local conditions than global datasets. In this way, users will be able to coarsely map ecosystem services anywhere in the world, map at a greater level of detail in the areas covered by case studies, or provide data and knowledge to develop more locally accurate and applicable case studies in other areas. The initial ARIES beta release includes eight ecosystem services modeled for seven case study regions, with locally important ecosystem services modeled for each case study (Figure 1). The chapters in this modeling guide illustrate local models and data for each case study currently active. Future modeling guide releases will describe global models and ARIES intelligent "model switching" process for each ecosystem service.

Figure 1: ARIES case studies and ecosystem services models included in version 1.0 beta release.



Users interested in developing new case studies will eventually be able to create or edit models and submit local spatial data for their case study directly to the system. Until these features become available, users interested in developing local case studies should contact the ARIES Consortium to discuss the fit of existing models to their context and the partnership or training opportunities to develop new case studies.

While we stress that ARIES is capable of using a variety of input models, including deterministic models, the system's use of probabilistic models is unique in the field of ecosystem services modeling. Since many users may be unfamiliar with probabilistic approaches to modeling, we provide a basic overview of these concepts below.

1.2 Probabilistic modeling in ARIES

Bayesian statistical approaches have been used to address a variety of issues in environmental valuation and value transfer (Brundson and Willis 2002), including determination of which independent variables to include in regression models (Moeltner and Rosenberger 2007, Leon-Gonzalez and Scarpa 2008), accounting for the "n vs. k" problem in function transfer, and in handling the effects of methodological independent variables when using a transfer function (Moeltner et al. 2007, Moeltner and Woodward 2009). Despite the use of Bayesian approaches to meta-analysis and function transfer, the ARIES project is the first to systematically use Bayesian models to map ecosystem services provision, use, and spatial dynamics.

Basic principles of Bayesian models in ecology are described well by McCann et al. (2006), part of a special issue of the Canadian Journal of Forest Research dealing with Bayesian modeling in ecology. Marcot et al. (2006) provide basic principles for Bayesian modeling that we have followed in our work regarding model construction, development of prior and conditional probabilities, testing, and review of the models. Per Marcot et al.'s recommendations on keeping conditional probability tables (CPTs) tractable and transparent, we generally use no more than 3-5 discrete states for each variable (often classified as "high-moderate-low" or "very high-high-moderate-low-very low"), we make each variable a function of no more than 3-5 other variables, and we use intermediate variables where appropriate. We direct readers with further interest in Bayesian networks and stronger backgrounds in probability theory to Pearl (1988).

Some readers who are unfamiliar with Bayesian approaches may feel uncomfortable with the perceived subjectivity of assigning prior and conditional probabilities in our models. We feel that the assignment of such probabilities, which are only used in the absence of training data, is a better way to incorporate expert opinion than asserting the rigid and non-transparent structure and parameterization of deterministic equations. Brundson and Willis (2002) address this point quite well:

> *"Some would argue that incorporating beliefs about models other than those implied by empirical measurement is a subjective, or unscientific, approach. In response, it could be stated that, certainly, Bayesianism has the potential for this problem to arise, and so one must have a strict 'code of conduct' for prior*

> *distribution specification. For example, making use of the outcomes of previous studies to provide prior beliefs is a reasonable scientific standpoint. Indeed, it could be argued that it is unscientific to ignore these prior results! Another way of avoiding subjectivity is to use non-informative priors in cases where prior information is unavailable or unobserved. Of course, one could argue that even a non-informative prior gives us some form of information about the distribution of an unknown parameter: after all, a specific distribution is being supplied rather than the information that any distribution might apply. However, in many cases non-informative priors do make reasonable models for a state of no subjective knowledge. In several 'text-book' examples of Bayesian analysis, for example multiple linear regression analysis assuming normal error terms, the adoption of non-informative priors results in tests algebraically identical to classical inferential procedures. In most cases, analysts are reasonably satisfied with regarding such classical approaches as 'objective'."*

As described for each model, we incorporate model elements from the literature and conversations with local experts on each ecosystem service, facilitated by our case study partners – Conservation International in Madagascar, CI and INECOL in Veracruz, Mexico, Earth Economics in Western Washington, the U.S. Geological Survey and Bureau of Land Management for the San Pedro River Watershed, and the U.N. Environmental Program-World Conservation Monitoring Center (UNEP-WCMC) for marine ecosystem services. Case studies in the Dominican Republic, Orange County, California, and Vermont were developed as part of a graduate-level course on ecosystem services modeling held at the University of Vermont in the spring of 2010.

Users should note that Bayesian models are not always appropriate or necessary in the ARIES system. Where well-accepted, peer-reviewed ecological process models can provide input data or values for the source, sink, or use components of an ecosystem service assessment, these models can be incorporated in the model chain instead of probabilistic models. The ARIES system is then instructed as to which cases it should use probabilistic versus deterministic models (e.g., in a particular part of the world, at a particular spatial scale, or where the results of another 'context model' match a specified output). Also, in some cases (particularly for the beneficiary or use models), a single spatial data layer or a simple GIS operation on two or more layers may suffice to map beneficiaries, rather than requiring a Bayesian or deterministic model.

1.3 Next Steps for the ARIES Modeling Platform

Like all models, ARIES remains a work in progress as the Consortium continues to add the latest ecological knowledge and datasets, incorporate existing process-based models where these can better inform ecosystem services modeling, build probabilistic models for additional ecosystem services, and customize existing probabilistic models to provide more accurate estimates for new case study regions. Further testing of these models and ongoing review with local experts and decision makers for the case studies is a critical next step for the ARIES system in order to move it from a position of demonstrating the spatial dynamics of ecosystem services to being able to quantitatively inform conservation, restoration, land use, development, and resource extraction choices. Next steps and future research needs specific to each set of ecosystem service models are described in the following chapters. Future releases of the modeling guide will describe periodic improvements made to the ARIES models, data sources, and interface.

The economic valuation system in ARIES is currently limited in scope. Services are expressed in biophysical units (e.g., for carbon, sediment, or flood water) or abstract units (e.g., for ranking view quality or recreational value), as appropriate. Medium-term plans include support for non-monetary expressions of preferences for alternative bundles of ecosystem services, based on a spatial extension of Multiple Criteria Analysis (Villa et al. 2002a). Support for monetary valuation is planned via integration of ARIES with the Ecosystem Services Database (ESD, Villa et al. 2002b, 2007, McComb et al. 2006). This will enable users to view primary economic valuation studies from their region of interest. Because ARIES is capable of mapping ecosystem service flows (the connection between ecosystems and their human beneficiaries, spatial determinants of supply and demand) and of handling data probabilistically, future additions to the system are being planned to enable more sophisticated forms of value transfer that incorporate benefit flows and Bayesian approaches.

ARIES is an open-source project built on the principles of environmental sustainability, socially just access to resources, and a more efficient allocation between market and non-market goods and services. The ARIES Consortium believes in the need for transparency in the use of data and models. We encourage all ARIES users to submit suggestions, edits, and additions to our library of models and spatial data, and to this modeling guide, as we continue working to improve the system.

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2. Carbon Sequestration and Storage

2.1 Introduction

The importance of ecosystems in storing and sequestering carbon is increasingly recognized given the threat of climate change and the rapid human-induced rise in atmospheric CO_2 concentrations (Portela et al. 2008). Carbon sequestration and storage help to provide a more stable global climate by taking up greenhouse gases and keeping them out of the atmosphere. Different portions of the natural landscape store and release carbon at different capacities and rates. In the ARIES system, the areas of carbon sequestered in vegetation and soils are designated as *sources* of the ecosystem service while the areas of potential stored carbon release due to fire, land use change, deforestation, or other vegetation and soil disturbances are *sinks*[3]. By subtracting the potential stored carbon release from carbon sequestration in a region of interest, we can compute the carbon available to offset anthropogenic emissions. Greenhouse gas emitters can be conceptualized as the *beneficiaries* of carbon sequestration and storage. Since greenhouse gas emitters benefit from the waste absorption capacity of the biosphere, carbon sequestration and storage can be divided among emitters. Existing and proposed systems to cap and assign property rights to atmospheric greenhouse gas emissions use this framework. In other words:

(1) Carbon sequestration (source) – stored carbon release (sink) = carbon available to offset emissions (use)

By mapping levels of carbon sequestration, stored carbon release, and anthropogenic emissions in a common unit (tonnes C/yr), we can fully describe regional carbon balances – the level of a region's net release or uptake of atmospheric CO_2. This will become increasingly important as local, state, and national governments continue to inventory greenhouse gas emissions and implement strategies to address climate change. The use of carbon sequestration is rival and can be assumed to occur globally due to the relatively fast atmospheric mixing of CO_2. The process to map regional carbon balance is fully described in the section on carbon flows (Table 2.1, Section 2.5).

In carbon modeling, sequestration is seen as a rate or flow (e.g., tons C/ha-yr) while storage is commonly computed as a stock (e.g., tons C/ha). The ARIES carbon models estimate the difference between vegetation and soil carbon sequestration and stored carbon release (i.e., due to deforestation, land use change, or fire) – both of which are

[3] Our use of the term *sinks* is different from the typically used term in the carbon literature; our use follows the generalized ARIES ecosystem services terminology used to identify the spatial dynamics of services as described in the introduction chapter to this modeling guide. The designation of areas of carbon sequestration as *sources* of the service and areas of potential stored carbon release as sinks reflects the designation of carbon sequestration and storage as a *provisioning service* (Table 2.1).

rates or flows[4]. Existing datasets for net primary productivity (i.e., carbon sequestration) and vegetation and carbon soil storage are typically modeled based on various biotic, physical, and climatic factors; such datasets are used to train Bayesian networks in cases where NPP or carbon storage datasets are incomplete or have inadequate spatial resolution.

Table 2.1: Summary characteristics of the ARIES carbon models.

Service	Carbon sequestration & storage
Benefit type	Provisioning
Medium/units	Tonnes C absorbed/emitted
Scale	Global
Movement	Atmospheric mixing
Decay	None
Rival?	Rival
Source	Stored C release (fire, land use change, other disturbance)
Sink	Vegetation & soil C sequestration
Use	CO_2 emitters

Although the outputs of carbon models do not require quantification of beneficiaries to establish value, it is also possible to map beneficiaries of climate stability, particularly in regions most vulnerable to climate change. Regions and human populations vulnerable to climate change are described in the ecosystem services and climate change literature (MA 2005, Schröter et al. 2005, Stern 2006, Parry et al. 2007). These groups include coastal populations at risk of sea level rise and storm intensification, populations dependent on glaciers and snowpack for water supplies, populations in arid regions at risk of drought, and populations using infrastructure built on permafrost, among others. Future ARIES models will be capable of mapping vulnerable populations as additional beneficiaries of a stable climate, and to quantify the impact of climate change scenarios on specified social groups.

Because the ecological processes influencing landscape scale carbon dynamics differ by region, local carbon models have been developed in ARIES for **Madagascar** as well as four ecologically distinct areas in the United States: **Orange County, California, the San Pedro River Watershed (Arizona and Northern Sonora, Mexico), Vermont,** and **Western Washington State**. These models are intended to be representative of carbon dynamics in wider regions. For instance, the Orange County models are designed to be applicable for urbanized areas within California coastal sage, chaparral, montane chaparral and woodland ecosystems, as well as other Mediterranean climatic regions; the San Pedro models to Chihuahuan and Sonoran deserts, Southwestern Sky Island and Sierra Madre Occidental pine-oak forests; the Vermont models to the Northern Forest region ranging from the Adirondacks to Maine; and the Western Washington models to Oregon, Washington, and British Columbia coastal forests, including the Cascade and

[4] This framework is analogous to proposed forest-based carbon credit programs, where credits could be issued for sequestration plus avoided deforestation (e.g., REDD, Gibbs et al. 2007).

Coast Ranges. In addition to these regionally targeted models, a generalized global model of carbon sequestration and storage is planned for a future release of ARIES. This model will use global datasets and provide coarser quality model outputs in the absence of locally validated ARIES models.

Other authors who modeled and mapped carbon sequestration and storage have taken into account different sets of drivers including: land use-land cover (Tallis et al. 2011); timber harvest or deforestation probabilities (Tallis et al. 2011, Wundscher et al. 2008); carbon pools and decay rates (Eade and Moran 1996, Chan et al. 2006, Egoh et al. 2008, Tallis et al. 2011, Wendland et al. 2010); biotic life zones (Wundscher et al. 2008); tree height, DBH, and stem density by forest type (Naidoo and Ricketts 2006); population density, slope, elevation, mean annual precipitation, soil texture and depth, and climatic indices (Iverson et al. 1994, Gaston et al. 1998), including the difference between mean summer high and winter low temperatures (Auch 2010). Agricultural practices can also greatly impact the sequestration or release of the soil carbon pool in agricultural settings (Lal 2004, Tilman et al. 2006). We drew on these studies in developing our carbon sequestration and stored carbon release models.

2.2 Carbon source models

Although carbon sequestration data are available globally at 1 km^2 resolution (Table 2.2), we developed simple Bayesian network models that include the influences on carbon sequestration (e.g., vegetation, soils, climate). Existing datasets can be used in ARIES to provide mean values to use in training finer-grained models, allowing estimation of carbon sequestration changes in scenarios or for up-scaled modeling of carbon sequestration when higher resolution input data are available.

Based on the literature and discussions with regional experts, we set carbon sequestration as a function of vegetation density and sequestration rate, two intermediate variables created to keep conditional probability tables tractable (Marcot et al. 2006). We set sequestration rate as a function of soil C:N ratio and the difference between mean summer high and winter low (in Madagascar and Western Washington), and as a function of land cover, vegetation type, and actual evapotranspiration (in Orange County). We set vegetation density as a function of hardwood:softwood ratio, percent tree canopy cover, and successional stage (in Western Washington), and percent tree canopy cover and forest degradation status (in Madagascar). For the San Pedro, Orange County, and Vermont agricultural carbon models, we used a collapsed number of variables, removing the intermediate nodes for vegetation density and sequestration rate. For the San Pedro model, we estimate sequestration as a function of vegetation type, percent tree canopy cover, and mean annual precipitation. For the Orange County model, we used the above noted variables as input nodes to sequestration rate, then combined sequestration rate with percent tree canopy cover to estimate annual vegetation and soil carbon sequestration. Actual evapotranspiration (AET) has been found to have a strong relationship with primary productivity, and therefore carbon sequestration (Lieth and Box 1972, Elegene et al. 1989, Metherell et al. 1993). This is especially true in water-limited regions such as semi-arid biomes, as with the Orange County case study (Claudio et al. 2006, Fuentes et al., 2006). Vegetation type can help to predict the quantity of vegetation sequestration and storage capacities from expected biomass for certain plant species (Kirby and Potvin 2007). In the Vermont agricultural carbon model, we estimated sequestration as a function of vegetation carbon storage (itself a function of mean annual precipitation, vegetation type, and the difference between mean summer high and winter low) and soil C:N ratio (Liu et al. 2010, Figure 2.1)[5]. We used Jenks Natural Breaks to discretize summer high-winter low, soil C:N ratio, and actual evapotranspiration. We used equal intervals to discretize vegetation and soil carbon sequestration, hardwood:softwood ratio, and percent tree canopy cover.

We based prior probabilities for the models on either the actual distribution of regional data (where we have these datasets), expert opinion (where consensus by experts was possible), or uninformed priors (where there was true uncertainty and a lack of consensus by experts). We filled out conditional probability tables by setting extremes set at both ends (i.e., "pegging the corners," Marcot et al. 2006) and interpolating intermediate values. Where possible we used expert opinion about which variables are most influential, and which should have the greatest influence on the contingent probability tables, and what the general level of uncertainty was for that system (i.e., how wide to set the distribution of values across discrete states). All else being equal, we set vegetation density at its highest values at greater percent tree canopy cover, later successional stages, more softwoods, and no forest degradation (where applicable). We set sequestration rate with its highest values at higher C:N ratios, higher actual evapotranspiration, lower differences between mean summer high and winter low temperatures, and land cover and vegetation types with greater biomass (where applicable). We set sequestration to its greatest values at high levels of vegetation density and sequestration rate.

[5] Bayesian network models for carbon source and sink models can be downloaded from http://ariesonline.org/modules/carbonspecs.html.

Figures 2.1: Bayesian network models for carbon sources (sequestration).

Figure 2.1.1: Carbon sequestration for Madagascar.

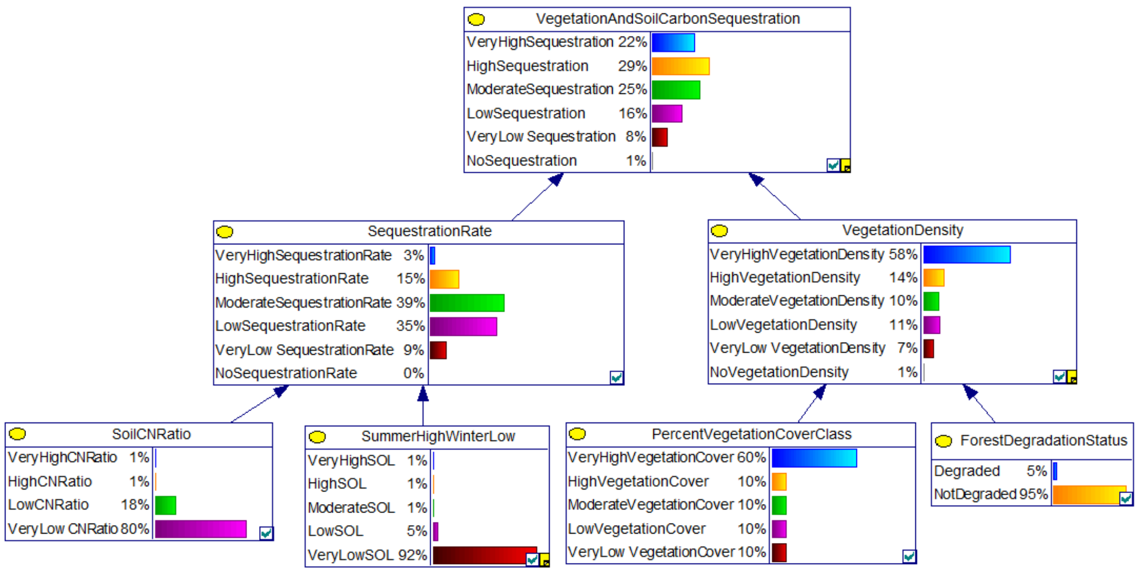

Figure 2.1.2: Carbon sequestration for Orange County, California.

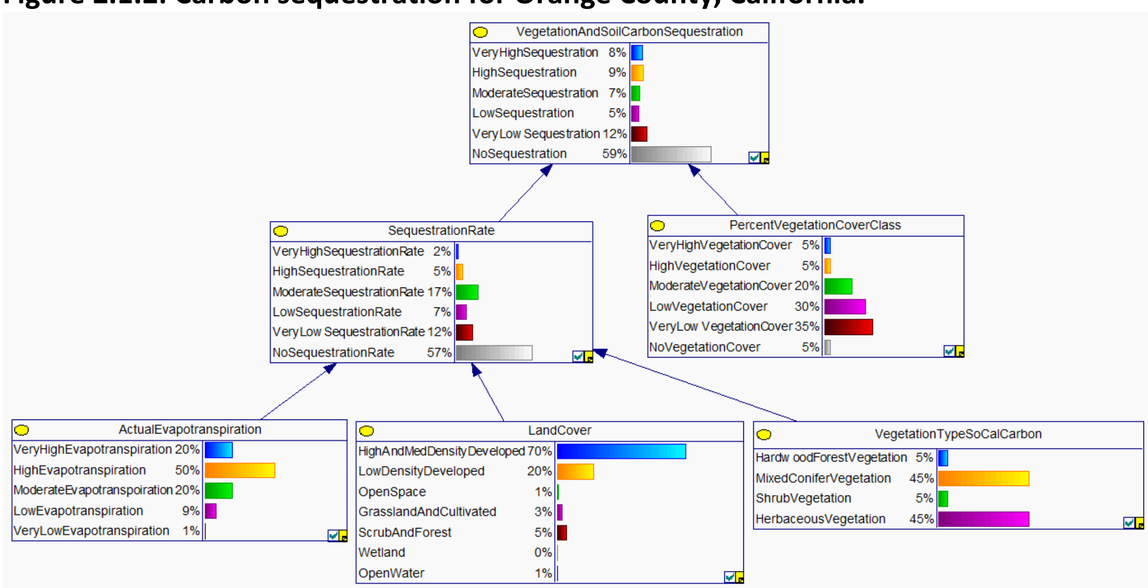

Figure 2.1.3: Carbon sequestration for the San Pedro River Watershed.

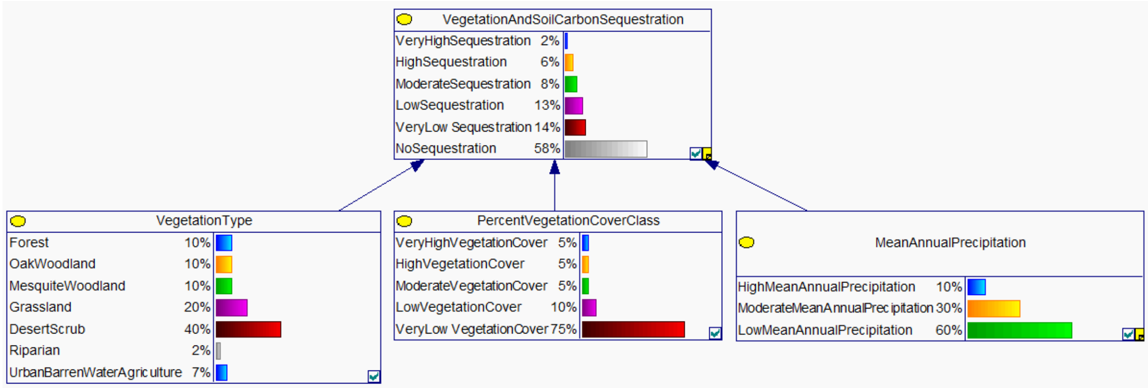

Figure 2.1.4: Carbon sequestration for Vermont agricultural land.

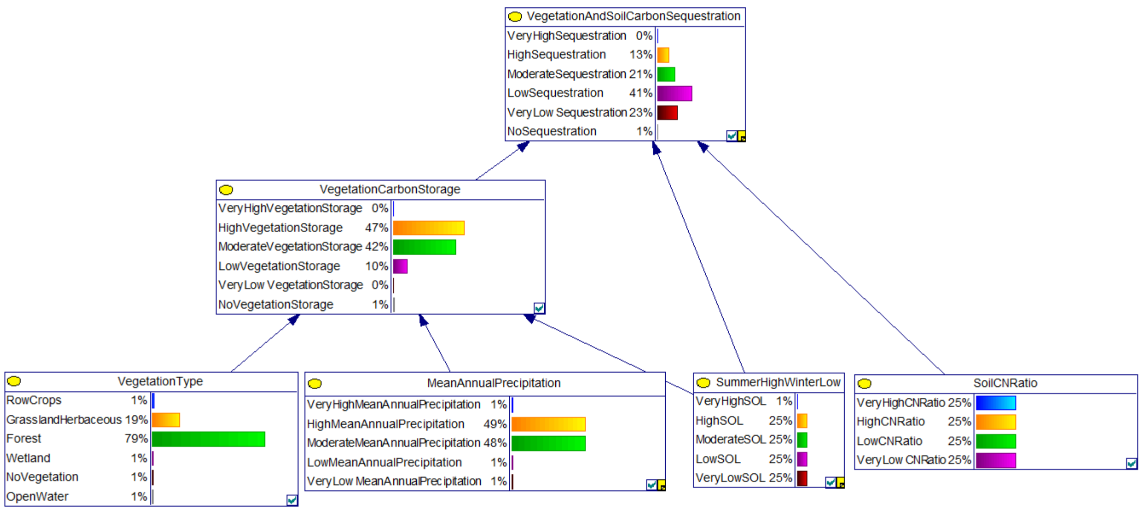

Figure 2.1.5: Carbon sequestration for Western Washington.

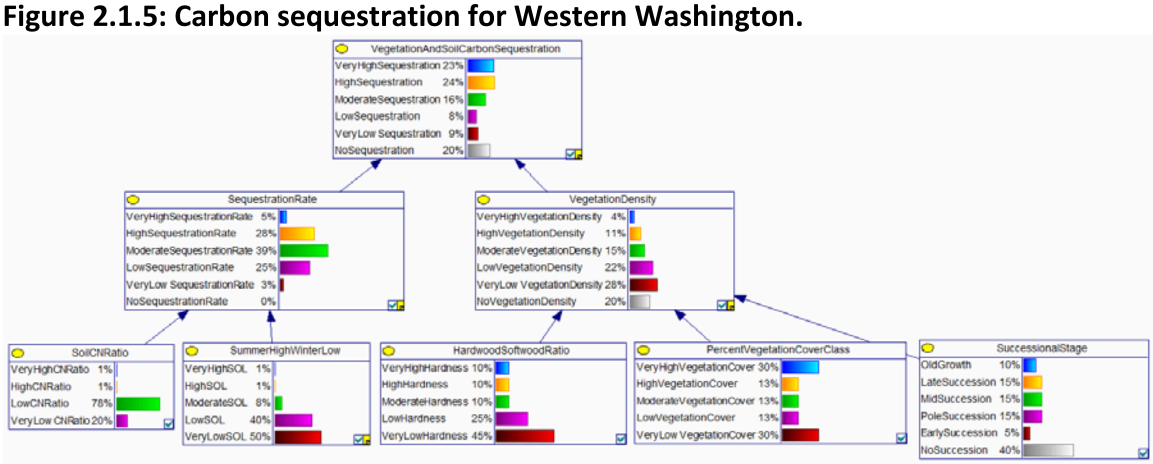

Table 2.2: Datasets used for carbon source models (sequestration).

Layer	Case studies	Source	Spatial extent	Spatial resolution	Year
Average annual actual evapotranspiration	Orange County	SAGE / UW-Mad	Global	$0.5^o \times 0.5^o$	1950-1999
Carbon sequestration	All	NBII/MEA	Global	1 km x 1 km	2000
Forest successional stage	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Hardwood:softwood ratio	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Land cover	Orange County	NLCD 2001	United States	30 m x 30 m	2001
Mean annual precipitation	San Pedro, Vermont	PRISM / OSU	United States	800 m x 800 m	1971-2000
Percent tree canopy cover	Orange County, San Pedro, Western Washington	NLCD 2001	United States	30 m x 30 m	2001
	Madagascar	GLCF/UMD	Global	1 km x 1 km	2000
Soil C:N ratio	Madagascar, Western Washington	FAO soils	Global	0.0833 min x 0.0833 min	1970-1978
Summer high – winter low	Vermont, Western Washington	PRISM / OSU	United States	800 m x 800 m	1971-2000
	Madagascar	WorldClim	Global	30 arc-seconds^2	1950-2000
Vegetation type	Orange County, Vermont	NLCD 2001	United States	30 m x 30 m	2001
	San Pedro	SWReGAP	AZ, CO, NM, NV, UT	30 m x 30 m	1999-2001

2.3 Carbon sink models

Sinks of atmospheric carbon are conceptualized as stored carbon potentially released due to fire, deforestation, or other land use change. Stored carbon release leaves less carbon absorption through sequestration (carbon sources) available to offset other anthropogenic emissions (mapped as carbon use).

We set stored carbon release as a function of vegetation and soil carbon storage (the sum of vegetation carbon storage and soil carbon storage) and the risk of deforestation and/or fire, with greater stored carbon release at higher risk and carbon storage levels. Soil carbon storage is influenced by slope, soil pH, soil oxygen conditions (i.e., greater storage in wetlands where anaerobic conditions inhibit respiration), vegetation density (an intermediate variable incorporating tree canopy cover and degradation status in

Madagascar, tree canopy cover, and vegetation type in the San Pedro, and successional stage, tree canopy cover, and hardwood:softwood ratio in Western Washington, noted as important determinants of carbon sequestration in the Pacific Northwest by Nelson et al. 2008), and soil carbon:nitrogen ratio. The importance of these variables in influencing soil carbon dynamics has been noted by previous authors (including those listed in Section 2.1). We set vegetation carbon storage as a function of the difference between mean summer high and winter low temperature (Auch 2010) and vegetation density, with population density added as an influence in Madagascar.[6] For the San Pedro, we set vegetation carbon storage as a function of mean annual precipitation and vegetation density. For the Orange County model, deforestation was not considered as an influence on stored carbon release (though it would be included in non-urban areas within the same biome), slope was dropped as an influence on soil carbon storage (since slope/aspect influence AET and other water balance measurements in chaparral and scrub ecosystems, Miller 1947, Parsons 1973, Ng and Miller 1980), and actual evapotranspiration and percent tree canopy cover were added as influences on soil carbon storage. We set vegetation carbon storage as a function of land cover, vegetation type, percent tree canopy cover, and AET for the Orange County model. The Vermont model used soil tillage and biomass removal rate as influences on agricultural stored carbon release (Gollany et al. 2010, Gonzalez-Chavez et al. 2010). This model considered soil C:N ratio, biomass residue input (Hai et al. 2010), and vegetation type as influences on soil carbon storage and vegetation type, mean annual precipitation, and the difference between mean summer high and winter low temperature (Figure 2.2).

Iverson et al. (1994) and Gaston et al. (1998) provide discretization of continuous variables for slope and population density. Bosworth and Tricou (1999) and Darby et al. (2009) provide discretization for vegetation carbon storage in the Vermont carbon model. We used Jenks Natural Breaks to discretize soil carbon storage, summer high-winter low, vegetation and soil carbon storage, soil C:N ratio, vegetation carbon storage, fire frequency, and actual evapotranspiration. We used equal intervals to discretize hardwood:softwood ratio and percent tree canopy cover.

Prior probabilities for the models are either based on the actual distribution of regional data (where we have these datasets), expert opinion (where consensus by experts was possible), or uninformed priors (where there was true uncertainty and a lack of consensus by experts). We filled out conditional probability tables by setting extremes set at both ends (i.e., “pegging the corners”) and interpolating intermediate values. Where possible we used expert opinion about which variables are most influential and hence have the greatest influence on the contingent probability tables, and about the general level of uncertainty for that system (i.e., how wide to set the distribution of values across discrete states). All else being equal, we set soil carbon storage at its

[6] Iverson et al. (1994) and Gaston et al. (1998) note the importance of this variable in measuring carbon storage in developing tropical nations, where subsistence firewood collection is an economically important activity.

highest values at low or high pH, high C:N ratio, level slopes, high vegetation density, and on anoxic (i.e., wetland) soils, and vice versa. We set vegetation carbon storage at its highest values with low differences between mean summer high and winter low temperature, high vegetation density, and low population density (in Madagascar). We set stored carbon release at its highest with greater vegetation and soil carbon storage and greater deforestation and fire risk.

The output of the carbon sink model is the potential stored carbon release. To better estimate actual carbon release in a given year, the user would need to overlay areas of fire or land use change. Actual carbon loss could then be estimated for that year. This feature will be included in carbon flow models within a future ARIES release.

Figures 2.2: Bayesian network models for carbon sinks (stored carbon release).

Figure 2.2.1: Potential stored carbon release for Madagascar.

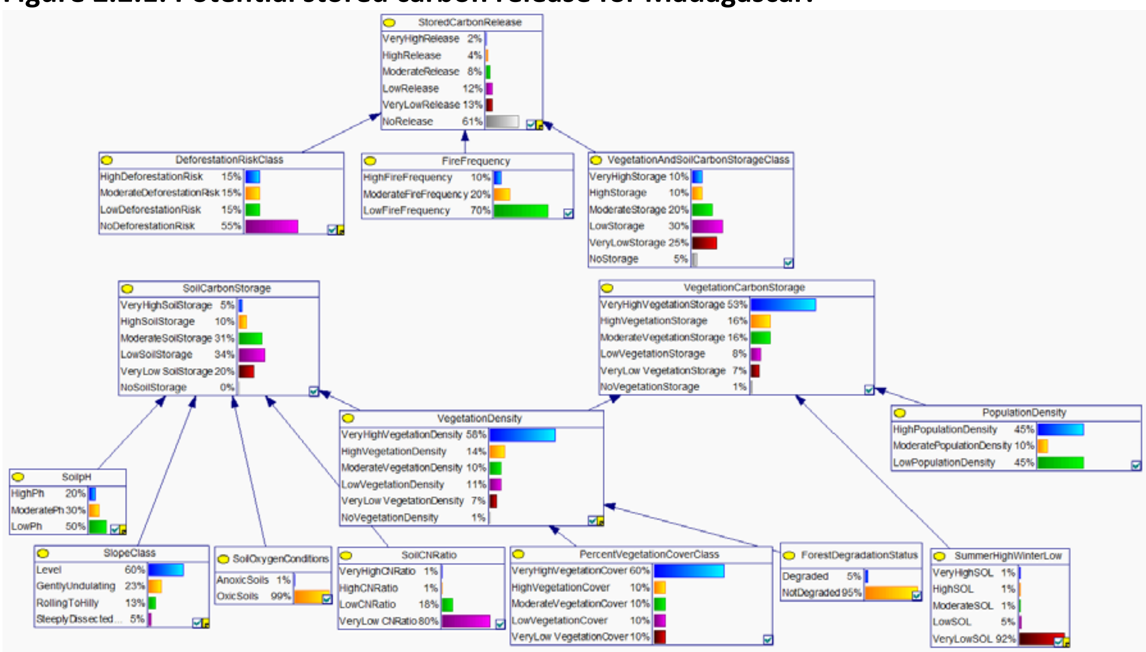

Figure 2.2.2: Potential stored carbon release for Orange County, California.

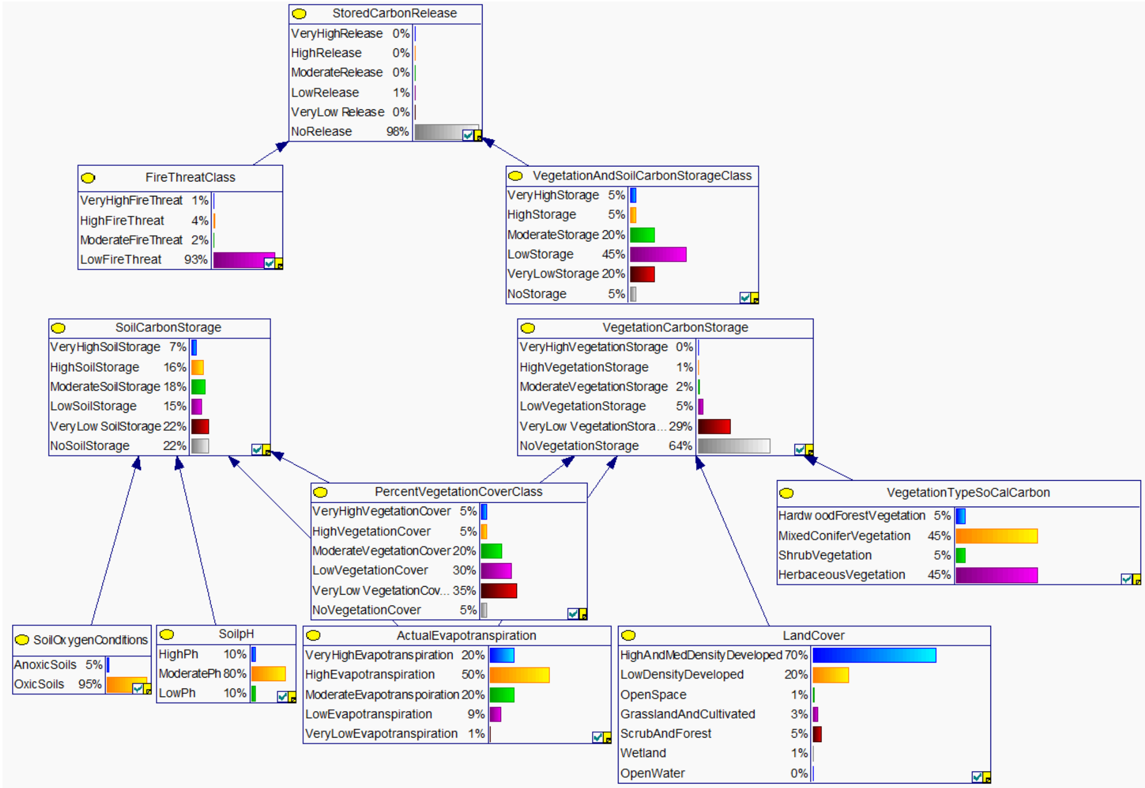

Figure 2.2.3: Potential stored carbon release for the San Pedro River Watershed.

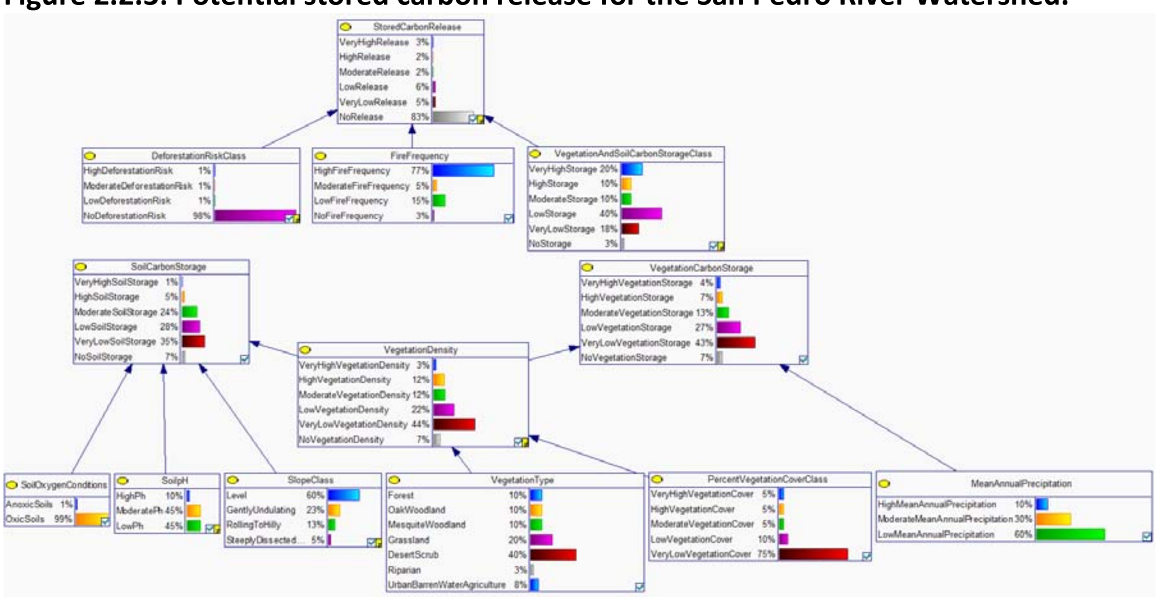

Figure 2.2.4: Potential stored carbon release for Vermont agricultural land.

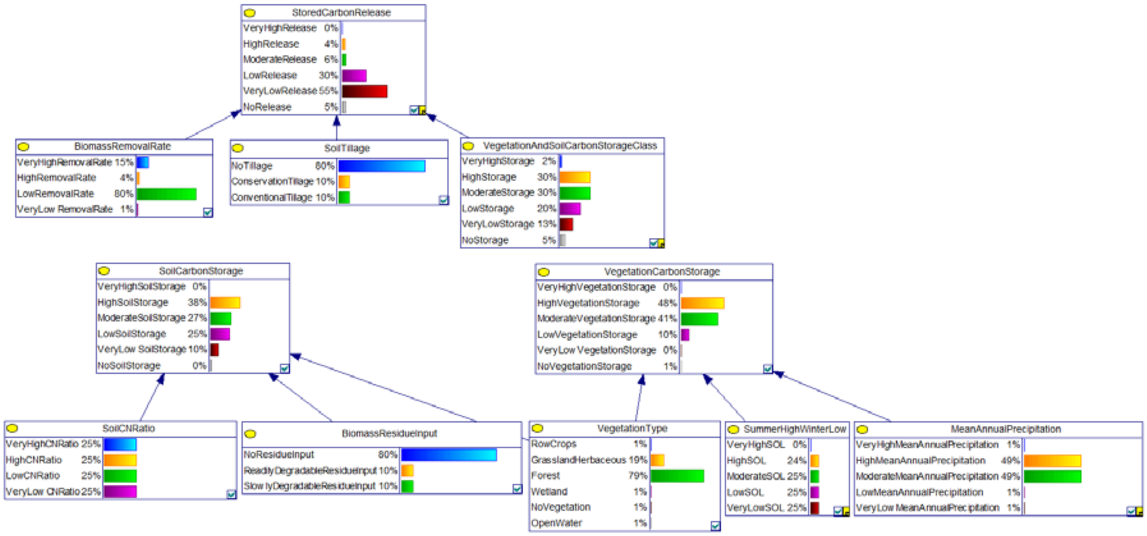

Figure 2.2.5: Potential stored carbon release for Western Washington.

Table 2.3: Datasets used for carbon sink models (potential stored carbon release).

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Average annual actual evapotranspiration	Orange County	SAGE / UW - Mad	Global	0.5^o x 0.5^o	1950-1999
Deforestation risk	Madagascar	GLCF / UMD	Global	250 m x 250 m	2001-2005

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Fire frequency	Orange County	California Fire & Resource Assessment Program	California	100 m x 100 m	2003
	San Pedro	SWReGAP	AZ, CO, NM, NV, UT	30 m x 30 m	2000
	Western Washington	WA DNR & OR DOF	Washington & Oregon	1.5 km x 1.5 km	1970-2007
Forest successional stage	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Hardwood : softwood ratio	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Land cover	Orange County	NLCD 2001	United States	30 mx 30 m	2001
Mean annual precipitation	San Pedro, Vermont	PRISM / OSU	United States	800 m x 800 m	1971-2000
Percent tree canopy cover	Orange Co., San Pedro, Western Washington	NLCD 2001	United States	30 m x 30 m	2001
	Madagascar	GLCF / UMD	Global	1 km x 1 km	2000
Population density	Madagascar	LandScan / ORNL	Global	30 arc-second[2]	2006
Slope	Madagascar, San Pedro, Western Washington	Derived from global SRTM data	Global	90 m x 90 m	2000
Soil C:N ratio	Madagascar, Western Washington	FAO soils	Global	0.0833 min[2]	1970-1978
Soil carbon storage	All	FAO soils	Global	0.0833 min[2]	1970-1978
Soil oxygen conditions (i.e., wetlands)	Orange County, San Pedro, Western Washington	NLCD 2001	United States	30 x 30 m	2001
	Madagascar	Kew Gardens vegetation map	Madagascar	30 x 30 m	1999-2003
Soil pH	Orange Co., San Pedro, Western Washington	SSURGO soils data	United States	30 x 30 m	n/a
	Madagascar	FAO soils	Global	0.5 min[2]	1970-1978
Summer high – winter low	Vermont, Western Washington	PRISM / OSU	United States	800 m x 800 m	1971-2000
	Madagascar	WorldClim	Global	30 arc-seconds[2]	1950-2000

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Vegetation carbon storage	Orange County, San Pedro, Vermont, Western Washington	National Biomass and Carbon Dataset	United States	30 m x 30 m	2000
	Madagascar	ORNL / CDIAC, Ruesch & Gibbs	Global	1 km x 1 km	2000
Vegetation type	Orange County	USFS	Northern Orange & Southern LA Counties	Unknown	2003
	San Pedro	SWReGAP	AZ, CO, NM, NV, UT	30 m x 30 m	1999-2001
	Vermont	NLCD 2001	United States	30 x 30 m	2001

2.4 Carbon use models

The beneficiaries of carbon sequestration and storage are greenhouse gas emitters who release CO_2 into the atmosphere, relying on ecosystems to absorb and store carbon in order to avoid even larger rises in atmospheric CO_2 than are currently seen. Our carbon use models thus display greenhouse gas emitters. Spatially explicit data on greenhouse gas emissions exist for the United States. Globally, we use population density data multiplied by per capita emissions for the country or sub-national region of interest (Table 2.4).

Table 2.4: Datasets used for carbon use models (anthropogenic carbon emissions).

Layer	Case studies used	Source	Spatial extent	Data type/spatial resolution	Year
GHG emissions	Orange Co., San Pedro, Western Washington, Vermont	Vulcan Project, Arizona State Univ.	United States	10 km x 10 km	2002
Per capita emissions	Global, Madagascar	Energy Information Administration: International Energy Annual	Global	Non-spatial	2006
Population density	Global, Madagascar	LandScan / ORNL	Global	30 arc-second[2]	2006

2.5 Carbon flow models

Since carbon dioxide is relatively quickly mixed in the atmosphere, the benefits of carbon sequestration and storage can be enjoyed by any human beneficiary on Earth, regardless of location. As such, no flow model is necessary for carbon sequestration and storage. However, for a given region, we can calculate the differential between carbon uptake by ecosystems (sequestration minus release of stored carbon) and anthropogenic carbon release. This information can be used in a flow model to show whether that region has a negative or positive carbon balance, i.e., whether its emissions are greater or less than the amount of carbon sequestered. Linking the source, sink, and use data with the flow model, we estimate the following indicators for carbon sequestration and storage flows[7]:

1. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Carbon sequestration: The amount of carbon taken up by vegetation and soils and added to biotic and soil carbon stocks. This is the quantity available for mitigating anthropogenic carbon emissions.
    b. Stored carbon release: The total emissions of carbon from the landscape, i.e., due to fire, deforestation, or other land use-land cover change.
    c. Greenhouse gas emissions: The total anthropogenic greenhouse gas emissions in an area, i.e., the demand for carbon mitigation, excluding emissions from deforestation, fire, or other land use change (which is accounted for by stored carbon release).
2. Possible use. These values are calculated by running flow models *without accounting for sink values* (potential stored carbon release) – i.e., benefits in the absence of stored carbon release. The possible values represent the maximum achievable service delivery based on the theoretical source value.
    a. Potential carbon mitigation use: The amount of carbon mitigation used by people to offset emissions if stored carbon release via fire or deforestation did not occur. In aggregate, this value will equal the minimum total value of either the source (carbon sequestration) or use (greenhouse gas emissions) values. If total carbon sequestration is less than total greenhouse gas emissions, then the potential carbon mitigation use values will all be less than the greenhouse gas emissions values at each point on the map. Otherwise, the values will equal greenhouse gas emissions everywhere.
3. Actual source, sink, and use. Actual carbon sequestration benefits provided, received, and degraded *with a full accounting for source, sink, and use values and flows*.
    a. Detrimental carbon source: The portion of the region's stored carbon release that cancels out carbon sequestration by ecosystems. If stored carbon release is less than carbon sequestration, the entire sink is used, cancelling out that much corresponding sequestration.
    b. Used carbon sink: The quantity of carbon mitigation actually used by the area's human population.

[7] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

   c. Satisfied carbon mitigation demand: The amount of emissions that are mitigated by local net carbon uptake (minimum of source minus sink and the use).
4. Inaccessible source and use. Theoretical values minus possible values; accounts for local surplus or deficit of carbon mitigation.
   a. Carbon mitigation surplus: Calculated when local sequestration exceeds emissions plus atmospheric carbon sources.
   b. Carbon mitigation deficit: Calculated when local emissions exceed net carbon uptake (sequestration minus stored carbon release).
5. Blocked sink and use. Source or use values degraded by sinks.
   a. Depleted carbon mitigation: Carbon sequestration capacity “cancelled out” by fire, deforestation, or other land use-land cover change, hence unavailable to offset other anthropogenic emissions.
   b. Depleted carbon mitigation demand: Carbon mitigation that would have been used by people but which is unavailable as it has already been claimed by the action of sources.

The biophysical unit for this ecosystem service is in tonnes of carbon. Economic value can be assigned to the biophysical quantity by using a market price or social cost of carbon (see Nordhaus 2010, Stern 2006, Tol 2008). Both of these approaches have limitations, and are fraught with ethical and philosophical implications about how to spread the costs of climate change across present and future generations (Ackerman and Stanton 2010).

2.6 Caveats and directions for future research

All models would benefit from further expert review to improve the overall model structure and conditional probability tables. The models would benefit from further comparison to existing carbon inventories, where available (e.g., Turnblom et al. 2002 for King County, Washington). Where possible, deterministic models (e.g., CENTURY, BIOME-GBC) could be used to estimate carbon sequestration, storage, and stored carbon release, with their results compared to probabilistic Bayesian models.

2.7 Acknowledgements and additional contributors

Ted Auch and Serguei Krivov provided input on the initial ARIES carbon models. Mark Casias developed the case study for Orange County. Sam Gorton developed the agricultural case study for Vermont. Dave Batker, Jim Pittman, and Paula Swedeen provided data and model review for the Western Washington case study. Miroslav Honzák provided data and model review for the Madagascar case study. The USGS and BLM provided funding for development of the San Pedro ecosystem services models in ARIES. An expert review panel including individuals from the U.S. Geological Survey, University of Arizona, Bureau of Land Management, and other organizations provided data and model review for the San Pedro case study.

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3. Aesthetic viewsheds and proximity

3.1 Introduction

In the ecosystem services literature, aesthetic values are often defined as the value derived from viewsheds (Bourassa et al. 2004) or proximity to open space (Fausold and Lileholm 1999, McConnell and Walls 2005, Brander and Koetse 2007). These values often accrue to housing or property values and are frequently measured using the shadow price of viewshed quality or access to open space from a hedonic price equation. Homeowner benefits are typically derived from sensory enjoyment of views, proximity to open space for recreation, or a sense of increased privacy in low density residential areas (recreational benefits for park visitors (e.g., hikers) and scenic road drivers (Walsh et al. 1990) are discussed in Chapter 9). Both viewsheds and open space proximity are considered non-rival, provisioning services whose value is measured in abstract units. The scale of analysis ranges from walking distance (aesthetic proximity) to entire viewsheds (aesthetic viewsheds). Table 3.1 summarizes the characteristics of the aesthetic viewshed and aesthetic proximity models.

Table 3.1: Summary characteristics of the ARIES aesthetics models.

Service	Aesthetic viewsheds	Aesthetic proximity
Benefit type	Provisioning	Provisioning
Medium/units	Scenic beauty (abstract units, 0-100)	Open space (abstract units, 0-100)
Scale	Viewshed	Walking distance
Movement	Line of sight (ray casting)	Walking simulation
Decay	Inverse square	Gaussian
Rival?	Nonrival	Nonrival
Source	Mountains, water bodies, etc.	Open spaces, esp. in urban areas
Sink	Visual blight	Obstructions (e.g., highways)
Use	Property/housing value	Property/housing value

Mapping the presence of housing units and the location(s) of open space is a relatively straightforward exercise. However, the flow of benefits from open space and scenic viewsheds to households differs. Proximity to open space values decline with increasing distance to open space, while viewshed values are affected by natural interruptions (e.g. topography) and anthropogenic features (e.g. development) which limit the line of sight to aesthetic assets such as mountains or water bodies. Cultural values favoring proximity to (or views of) significant landscape features are likely to differ by region. Within developed nations these preferences have been relatively well explored in the hedonic pricing and contingent valuation literature. However, such preference information is much sparser in developing nations.

We developed prototype aesthetic viewshed and proximity models for the **San Pedro River Watershed (Southeast Arizona and Northern Sonora, Mexico)** and **Western Washington State**. These models are intended to be representative of aesthetic preferences over broad geographic regions. For instance, the San Pedro models are applicable to the Chihuahuan and Sonoran deserts, Southwestern Sky Island and Sierra Madre Occidental pine-oak forests while the Western Washington models could reasonably be applied to the Cascade and Coast Ranges of Oregon, Washington, and British Columbia.

Cultural values favoring proximity to or views of particular landscape features may differ by region, however. Thus, we do not currently apply the San Pedro aesthetics models to Northern Sonora, where preferences may differ. In addition, these models rely on parcel or housing data, which in the U.S. are provided by county assessors' offices. These datasets use widely varying formats and require *ad hoc* annotation prior to use in models. Because of the problems of spatially varying preferences and uneven availability of parcel and housing data, it is not currently feasible to develop global models for aesthetic proximity and viewsheds. A generalized model for the United States that relied on nationwide datasets (e.g., NLCD developed land as a proxy for residential land use) and generalized preferences for landscape features could be developed for a future release of ARIES.

In Western Washington, sources of aesthetically valuable views include large mountains like Mount Rainier and other high peaks in the Cascade and Olympic Mountains, and water bodies such as the Pacific Ocean, Puget Sound, or inland lakes. Users of aesthetic views are found in residential areas. As a view travels from source to user, it may be physically blocked by buildings, trees, or topography. Its quality may be diminished by air pollution or visual blight, such as highways, forest clearcuts, or visually unappealing land uses including commercial, industrial, or transportation. Such sinks, or locations of visual blight, can also be mapped. Higher concentrations of residential users or visual blight lead to higher levels of ecosystem service use or sinks, respectively. The proximity source model for Western Washington accounts for the type of open space – either natural ecosystems (e.g., forests, wetlands, beaches, riverfront, lakefront) or managed open space (e.g., cemeteries, farmland, golf courses) and its quality (e.g., area, protected status, water quality, potential for crime). Sink models simply account for the location of highways that limit easy access, reduce privacy, and increase noise for nearby residents. Use models account for the location and value of housing and urban proximity, which acts as a proxy for open space scarcity and congestion and has been shown to be a key variable influencing the value of open space (Brander and Koetse 2007).

The San Pedro models account for different preferences for open space and viewsheds and different land cover types present in southwestern deserts. The San Pedro proximity model accounts for the presence of desert scrub and grassland as dominant land cover types and relatively higher values for water features, given their rarity. Proximity values in Arizona have been well described for riparian areas in Tucson (Colby and Wishart 2002, Bark-Hodgins and Colby 2006, Bark et al. 2009) and for rural ranchettes (Sengupta and Osgood 2003). Water quality and crime (often associated with urban parks) are not included in the model while fire threat becomes an important variable in determining how open space is valued. Viewshed preferences have been well described for the San Pedro (Steinitz et al. 2003). Key local adaptations of the viewshed model include the presence of mines and transmission lines as visual blight. These features are much more likely to visually stand out against the desert landscape versus the forested landscapes of Western Washington.

In this chapter, we first describe proximity models and data sources for the San Pedro River Watershed and Western Washington. Descriptions of viewshed models and data sources follow.

3.2 Aesthetic proximity source models

Aesthetic proximity values depend foremost on having some form of open space. For the San Pedro, major open space types include desert scrub, grassland, farmland, parks, forests and woodlands, and riparian and wetlands (Figure 3.1.1)[8]. For Western Washington these include forests, wetlands, beach, riverfront, lakefront, golf courses, cemeteries, farmland, or parks (Figure 3.1.2). We use the intermediate variable "Open Space Resource" to aggregate these open space types. Along with the type of open space, its quality matters in determining proximity value. We aggregated several independent measures of open space quality – open space area and formal protection (for both models), water quality and crime (Western Washington), and fire threat (San Pedro) – into a single intermediate variable, "Resource Quality," in order to maintain tractability of the conditional probability tables. Anderson and West (2002) found park value to increase with size, though Brander and Koetse (2007) note that open space value on a per hectare basis declines as its size grows. All else being equal, we would generally expect lower per-area value in the vast open landscapes of the rural Southwest than around urban areas. A series of Maryland studies noted that homeowners more highly value land that is permanently protected over land that may be developed in the future (Irwin and Bockstael 2001, Irwin 2002, Geoghegan 2002, Geoghegan et al. 2003). Troy and Grove (2008) found crime to reduce the value of parks in urban areas in Baltimore, a result that could also potentially apply to older, economically distressed suburbs. We do not include crime in the San Pedro model since the region lacks large urban centers with higher crime rates. Finally, poor water quality could reduce the value of open space due to odors, public health concerns, or reduced recreational opportunities. We did not include water quality in the San Pedro model since, given its rarity in this region, we assume the presence of water to indicate higher quality open space. We added the variable "fire threat" to the San Pedro model. In fire-prone regions of the west, living near fire-prone ecosystems is a risk that may be understood by landowners, leading to lower perceived open space proximity value (Loomis 2004). Finally, we set the top node, "Theoretical Proximity Source" as a function of Open Space Resource and Resource Quality.

[8] Bayesian network models for aesthetics sources, sinks, and use models can be downloaded from http://ariesonline.org/modules/aesthspecs.html.

We derived prior probability distributions for the presence/absence of open space types based on 2001 NLCD and local land use data (i.e., for parks, lakefront, or riverfront) in the San Pedro and Western Washington. The Open Space Resource node uses a NoisyMax node[9], based on the simplifying assumption that the most highly valued land use type will be representative of the total value (i.e., there are no synergistic effects among value components, so a high probability of presence of the most valued nearby landscape component can be taken as the likelihood of a high total potential value). We set the highest values in the contingent probability table for Open Space Resource for beach, parks, riparian, lakefront, and riverfront (which frequently feature public access and open water), the lowest values for farmland (known to provide disamenities like noise and odors), cemeteries and golf courses (which may have limited public access), desert scrub and grasslands (extremely abundant vegetation types in the Southwest) and intermediate values for wetlands and forests. In a global meta-analysis of proximity studies, Brander and Koetse (2007) found parks to be more highly valued than forests, which were more highly valued than farmland. However, these relative values could be adjusted for different parts of the world based on local preferences and hedonic studies indicating relative values of different types.

We assumed that 25% of the landscape is protected in Western Washington and 60% was protected in Southeast Arizona. We assumed that 10% of parks are located in urban areas in Western Washington where crime may be problematic. We discretized park size by Jenks Natural Breaks. For Western Washington, we assumed that smaller open space parcels are most abundant, with the abundance of parks in a particular size class declining as park size grows. These assumptions were reversed for Southeast Arizona, since in the rural landscape few small open space parcels and many large open space parcels would be found. We assumed that 85% of open space has no open water, and assume that equal areas meet water quality standards, are waters of concern, or require a TMDL (indicating poor water quality). We assumed that 75% of the landscape is at a high fire threat. We assumed that the highest “Resource Quality” will occur in large, formally protected open space with no water quality or crime problems and low fire risk, and that the lowest value will occur in small, unprotected parcels of open space with crime, water quality problems and/or higher fire frequency. We thus “pegged the corners” of the contingent probability table for Resource Quality and filled in intermediate values (Marcot et al. 2006). We assumed that Formal Protection will have the greatest influence on Resource Quality, since unprotected land is potentially much less valuable for its open space quality. Finally, in defining the contingent probability table for Theoretical Proximity Source, we assumed that high value and high quality

[9] A NoisyMax node is a deterministic computation used instead of a large conditional probability table, making the simplified model computable when the assumption of independence of the causal influences can be defended (Pearl 1988).

locations produced the highest theoretical proximity source value and vice versa, pegged the corners, and interpolated intermediate values.

Figures 3.1: Bayesian network models for aesthetic proximity sources.

Figure 3.1.1: Aesthetic proximity sources for San Pedro River Watershed.

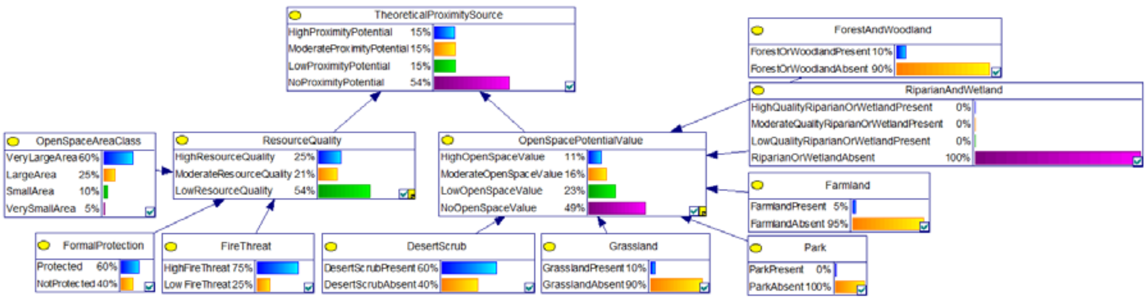


Figure 3.1.2: Aesthetic proximity sources for Western Washington.

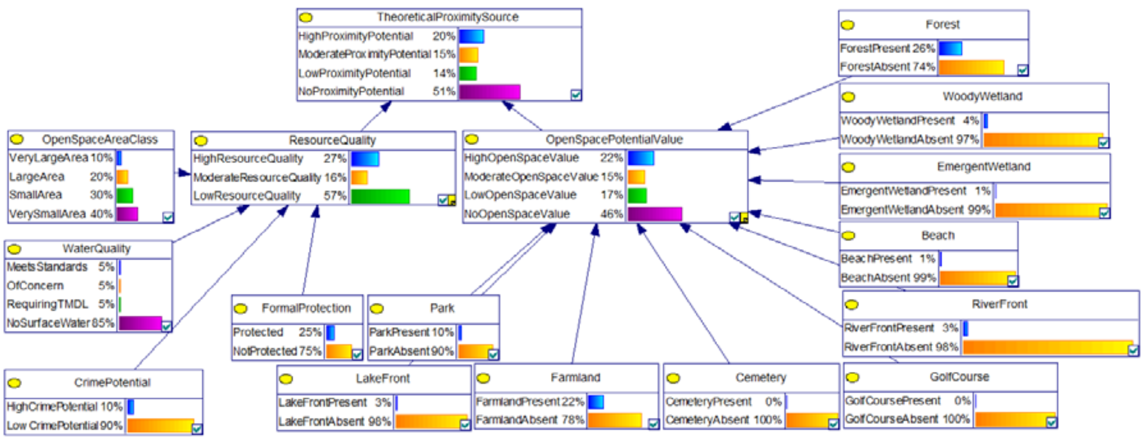


Table 3.2: Datasets used for the proximity source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Area	All	Calculated areas of NLCD open space types	San Pedro, Western Washington	30 m x 30 m	Varies
Beach	Western Washington	WA DOH	Washington State	Unknown	2006
Crime (urban areas)	Western Washington	U.S. Census Bureau	Washington State	Unknown	2006
Desert scrub	San Pedro	SWReGAP	Southwest U.S.	30 m x 30 m	2000
Emergent wetland	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Farmland	San Pedro	SWReGAP	Southwest U.S.	30 m x 30 m	2000
	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Fire threat	San Pedro	SWReGAP & TNC fire data	Southwest U.S.	30 m x 30 m	2000

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Forests	San Pedro	SWReGAP	Southwest U.S.	30 m x 30 m	2000
	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Formal protection	All	World Database on Protected Areas	Global	Unknown	2009
Grassland	San Pedro	SWReGAP	Southwest U.S.	30 m x 30 m	2000
Lakefront	Western Washington	Washington DNR (50 m buffer around lakes layer)	Washington State	Unknown	Unknown
Park	San Pedro	AGIC	Arizona	Unknown	2010
	Western Washington	Federal, state, and county park layers combined	Western Washington	Unknown	Varies; generally 2000-present
Riparian & wetland quality	San Pedro	Southwest Regional Gap Analysis LULC & Stromberg et al. (2006) riparian quality	SPRNCA	30 m x 30 m	2000
Riverfront	Western Washington	Washington DNR (100 m buffer around rivers layer)	Washington State	Unknown	Unknown
Woody wetland	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Water quality	Western Washington	Washington DOE	Washington State	Unknown	2004

3.3 Aesthetic proximity sink models

Transportation infrastructures that limit access to or diminishes the aesthetic quality of open space comprise the sink model. We assume that infrastructure located between a user and potentially valued open space depletes the value of the open space by 50%. While highways may increase accessibility to open spaces for distant users who use them to travel to recreation sites (as represented in the ARIES recreation models), they reduce access and enjoyment of open space at the neighborhood scale. We use a highways data layer, so no model is required (Table 3.3).

Table 3.3: Datasets used for the proximity sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Highways	All	TIGER/Line files	United States	Unknown	2000

3.4 Aesthetic proximity use models

The use model accounts for the location and value of housing and urban proximity (e.g., urban, suburban, or rural setting). For aesthetic proximity use to occur, housing must be located near open space. Determinants of the proximity use model include housing, its value, and urban proximity (see Figure 3.2). Numerous authors have found open space to be more valuable in urban settings where user populations and access scarcity are greater and less valuable in rural settings (Bin and Polasky 2005, Doss and Taff 1996, Boyer and Polasky 2004, Brander et al. 2006, Mahan et al. 2000, Reynolds and Regalado 2002, and Schultz and Taff 2004 for wetlands, Anderson and West 2002 for parks, Brander and Koetse 2007 for all open space).

We discretized housing value and population density, a proxy for urban proximity, using Jenks Natural Breaks. Based on relevant spatial data, we set urban proximity priors to reflect 5, 25, and 70% of the landscape in Western Washington and 2.5, 7.5, and 90% in the San Pedro as urban, suburban, and rural settings, respectively. We assume that 75% of all housing is valued at moderate to low levels, 15% at very low levels, and 10% at high or very high levels. Finally, we assumed that 10% of the landscape has housing in Western Washington and that 2% of the landscape has housing in the San Pedro. The top node for "Homeowner proximity use" requires housing to be present in order to have value. We then set value to decline more quickly moving from urban to rural and less quickly moving from high to lower classes of housing values. Brander and Koetse (2007) found per capita income to be a positive but non-significant independent variable in a meta-analysis of proximity values. We thus include housing value in our models, but make its prior influence on proximity use value weaker than the presence of housing or urban proximity. Training the models to real data can be used to reveal the actual weight of the variable in each use case.

Figures 3.2: Bayesian network models for aesthetic proximity use.

Figure 3.2.1: Aesthetic proximity use for San Pedro River Watershed.

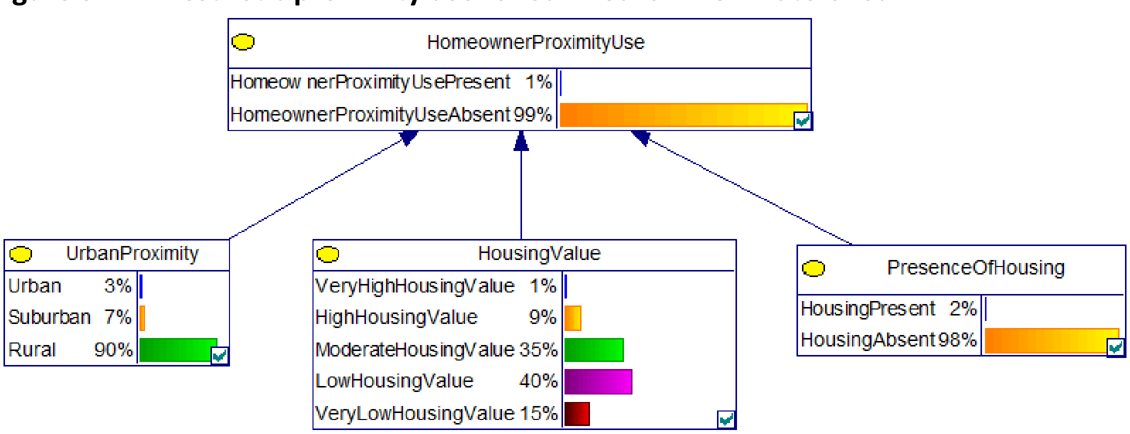


Figure 3.2.2: Aesthetic proximity use for Western Washington.

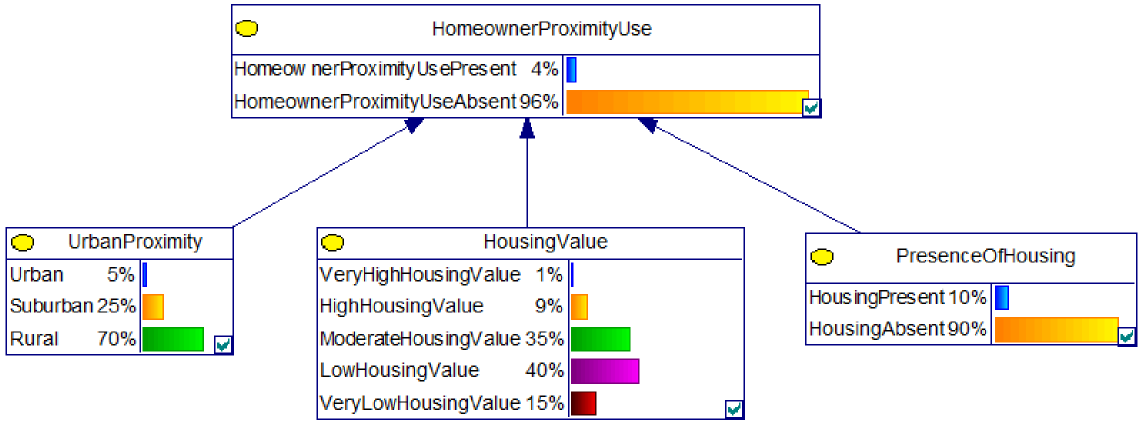


Table 3.4: Datasets used for the proximity use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Housing values	All	County assessors' offices	Pima & Pinal Cos., AZ; Grays Harbor, King, Kitsap, Mason, Snohomish, & Thurston Cos., WA	Parcel	2004 (Kitsap Co.), 2006 (King Co.); 2010 (Pinal & Pima Cos.); uncertain for others
Presence of housing	All	County assessors' offices	Pima & Pinal Cos., AZ; Clallam, Grays Harbor, Jefferson, King, Kitsap, Mason, Snohomish, & Thurston Cos., WA	Parcel	2004 (Kitsap Co.), 2006 (King Co.); 2010 (Pinal & Pima Cos.); uncertain for others
Urban proximity (population density by block group)	San Pedro	U.S. Census Bureau	United States	Census block groups	2000
	Western Washington	Washington Dept. of Financial Management	Washington State	Census block groups	2000-2007

3.5 Aesthetic proximity flow models

Most studies have found proximity value to decline with distance from open space. McConnell and Walls (2005) reviewed studies of housing values within a 0.8 to 1.6-km radius of open space, and note that open space-related amenity values drop rapidly past that distance. Brander and Koetse's (2007) meta-analysis used 100 m change in the distance to open space as a dependent variable in their analysis to show how proximity value changes with distance to open space.

We used a walking simulation model to represent aesthetic proximity flow. The value of open space proximity is highest at the edge of accessible open space and rapidly decreases with increasing linear distance up to 0.8 km. A slower decay rate occurs from 0.8 to 1.6 km, and the value is assumed to be zero beyond a distance of 1.6 km from the open space parcel. Although Sengupta and Osgood (2003) describe the value of proximity to rivers in the arid Southwest as having a less steep distance decay function, we currently use a uniform distance decay function in the proximity flow model.

We quantify the aesthetic value of open space in abstract units, from 0-100. Using the base source, sink, and use data, we estimate the following indicators for open space proximity flows[10]:

1. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Potential proximate open space: All possible areas capable of supplying open space of varying quality.
    b. Potential proximity sink: All highways that could separate residences from open space.
    c. Homeowners with open space demand: All possible residences, as anyone is capable of gaining value from living near open space.
2. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* – i.e., benefits provided in the absence of highways that block direct access to open space. These values represent the maximum achievable service delivery based on the theoretical source value.
    a. Possible proximate open space: The density of service flow along each walking path between an open space and user, before accounting for highways that limit local access.
    b. Accessible open space: Open space providing value when accounting for proximity and the location of homeowners but not highways that limit local access.
    c. Open space proximate homeowners: Homeowners benefiting from proximity after accounting for sources of open space and their flow paths, but before accounting for highways that limit local access.
3. Actual flow, source, sink, and use. Actual proximity benefits provided, degraded, and received *with a full accounting for source, sink, and use values and flows*.
    a. Accessible proximity: The density of service flow along each walking path between an open space and user, when accounting for sinks and flow paths.
    b. Enjoyed open space: Open space providing proximity when accounting for flow paths, sinks, and the location of beneficiaries.

[10] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

   c. Blocking proximity sink: Highways that actually separate residences from open space.
   d. Homeowners with proximate open space: Homeowners benefiting from proximity after accounting for sources of open space, sinks, and flow paths.
4. Inaccessible source, sink, and use. Theoretical values minus possible values; accounts for sources that do not provide, sinks that do not degrade, and beneficiaries that cannot use an ecosystem service due to a lack of flow connections.
   a. Unaccessed open space: Potential sources of aesthetic enjoyment that are not accessible to homeowners since no homeowners live in the area.
   b. Inaccessible proximity sink: Highways that do not restrict access to open space because they are not proximate to both homeowners and open space.
   c. Homeowners without proximate open space: Homeowners lacking any proximity to open space (typically in urban areas).
5. Blocked flow, source, and use. Flows, source, or use values degraded by sinks.
   a. Blocked proximity: The density of service flow along each walking path between an open space and user that is blocked by highways.
   b. Blocked open space: Open space that is blocked by the action of sinks.
   c. Homeowners with blocked proximity: Homeowners who do not receive benefits from open space proximity because their access is blocked by highways.

3.6 Aesthetic viewshed source models

Viewshed models should account for local preferences to the degree possible as these preferences are unlikely to be uniform everywhere (Bourassa et al. 2004). For the San Pedro, mountains and certain visually significant landscape types (e.g., riparian, diverse natural vegetation) were the preferred elements in viewsheds (Steinitz et al. 2003 based on local viewshed surveys and using the USFS 1995 framework). Mountains and open water are commonly valued natural objects in viewsheds in Western Washington (Benson et al. 1998, Bourassa et al. 2004). We set "Theoretical Natural Beauty," the source value for viewsheds, as dependent on the presence of these locally significant visual features.

We estimated priors for the San Pedro from appropriate LULC data: 1.3% of the landscape was alpine and cliff, 2.1% forest, 6.4% woodland, 1.4% riparian and water, and 88.8% visually neutral or negative landscape features. We estimated that 5% of the landscape was large mountains (>1,800 m), 40% small mountains (1,400-1,800 m), and 55% no mountains (<1,400 m). In the contingent probability table for Theoretical Natural Beauty, we set instances of alpine and cliff and riparian as the highest potential value (especially when combined with mountain views), woodland and forests as intermediate values, and other vegetation types as the lowest values (Figure 3.3.1).

For Western Washington, we set priors assuming that 10% of the landscape is ocean, 2% is inland lakes, 2% large mountains (>2,750 m), and 10% small mountains (2,000-2,750 m). For Western Washington, we aggregated these values as Theoretical Natural Beauty in a contingent probability table by noting that higher values were ascribed to ocean views, lowest values were ascribed to mountain views, and intermediate values were ascribed to lake views for the region (Benson et al. 1998, Bourassa et al. 2004, Figure 3.3.2). Although skyline views may be valuable, we do not include them in our analysis since skylines are man-made features and thus do not provide an ecosystem service.

Table 3.5: Datasets used for the viewshed source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Lake	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Mountain	All	SRTM	Global	90 m x 90 m	2000
Ocean	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Scenic vegetation	San Pedro	SWReGAP	Southwest US	30 m x 30 m	2000

Figures 3.3: Bayesian network models for aesthetic viewshed sources.

Figure 3.3.1: Aesthetic viewshed sources for San Pedro River Watershed.

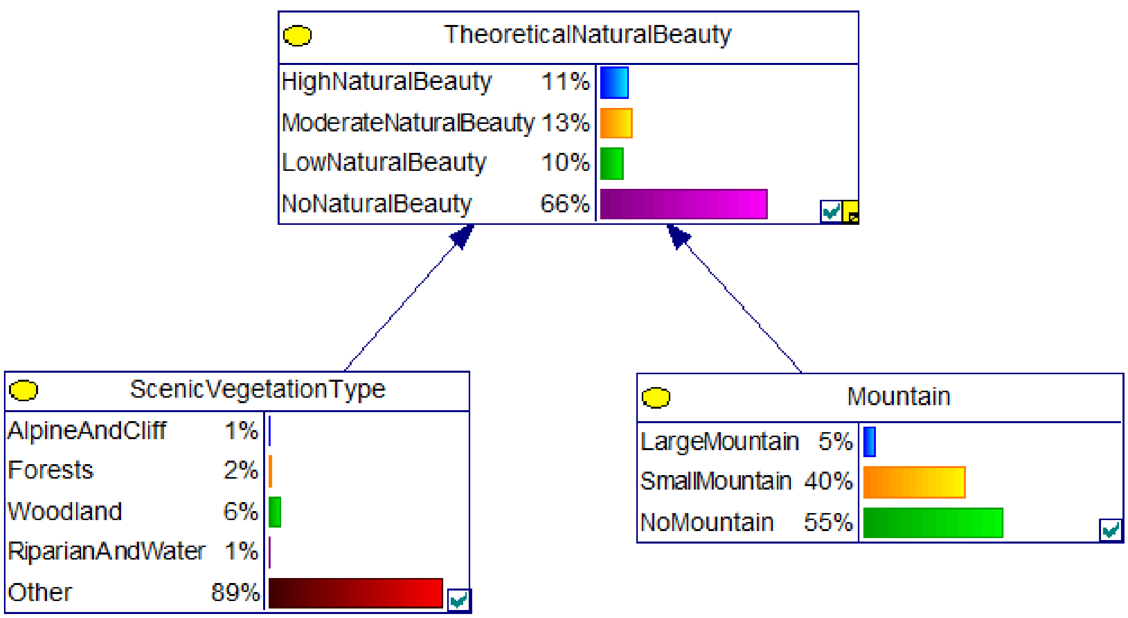
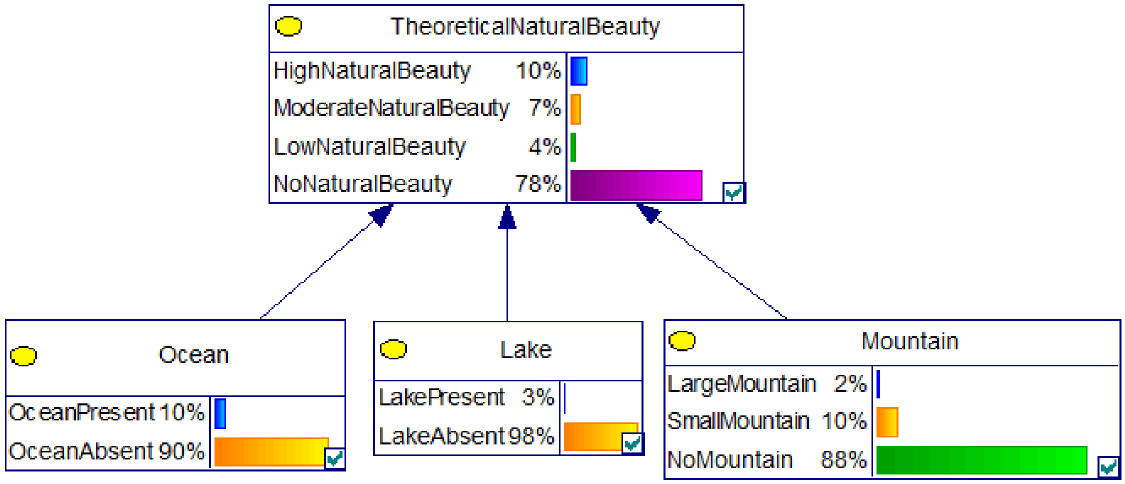


Figure 3.3.2: Aesthetic viewshed sources for Western Washington.



3.7 Aesthetic viewshed sink models

Undesirable visual features, or visual blight, can reduce the quality of views (Benson et al. 1998, Bourassa et al. 2004, Gret-Regamey et al. 2008). In the San Pedro such undesirable features include highways, mines, developed land, and transmission lines (Steinitz et al. 2003, Figure 3.4.1). These features are each present on less than 1% of the landscape. In Western Washington, views of lost forest cover, including clearcuts, may also act as a sink, reducing view quality (Wundscher et al. 2008). We assumed that highways or other major roads occupy 2.5% of the landscape, commercial, industrial, or transportation land uses occupy 15% of the landscape, and clearcuts occupy 2.5% of the landscape in Western Washington (Figure 3.4.2).

We aggregated the types of "Visual Blight" using a NoisyMax node, assuming that the greatest source of blight will override lesser sources of blight. For the San Pedro, we assume that mines have the greatest visual impact, followed by transmission lines and developed land, with highways having the least visual impact. For Western Washington, we assume that clearcuts reduce view quality less than highways, commercial, industrial, or transportation land uses.

Although not currently included in the models, dust, air pollution, or persistent cloudy or foggy conditions also reduce views, and could act as sinks. These conditions can be simulated in the flow models by changing the decay rates for views.

Figures 3.4: Bayesian network models for aesthetic viewshed sinks.

Figure 3.4.1: Aesthetic viewshed sinks for San Pedro River Watershed.

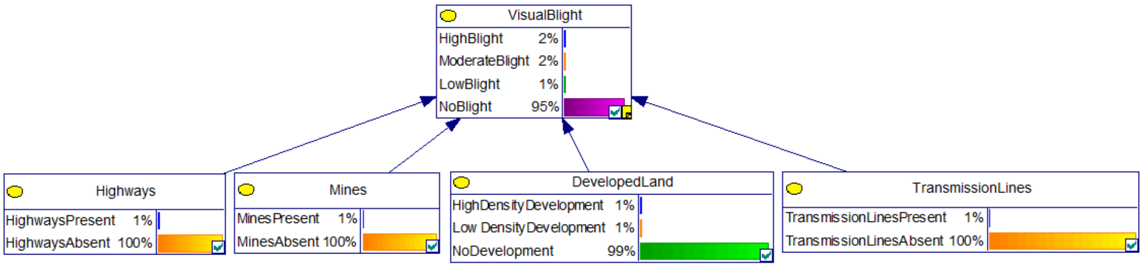


Figure 3.4.2: Aesthetic viewshed sinks for Western Washington.

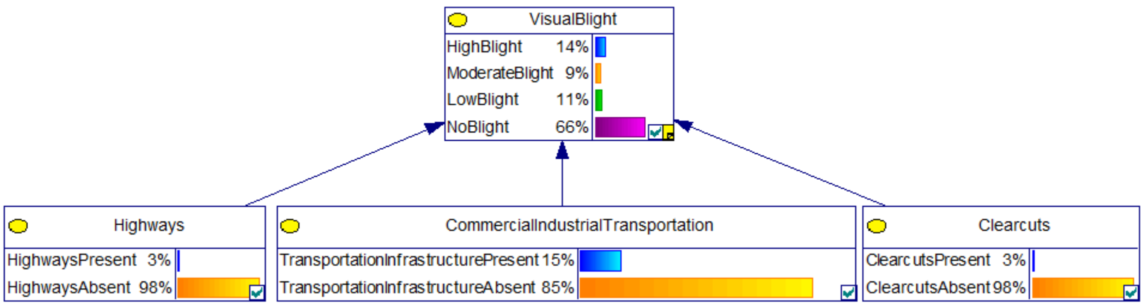


Table 3.6: Datasets used for the viewshed sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Clearcuts	Western Washington	WA DNR	Washington State	Unknown	2006
Commercial, Industrial, Transportation	Western Washington	NLCD 1992	Western Washington	30 m x 30 m	1992
Developed land	San Pedro	SWReGAP	Southwest US	30 m x 30 m	2000
Highways	All	TIGER/Line files	United States	Unknown	2000
Mines	San Pedro	SWReGAP	Southwest US	30 m x 30 m	2000
Transmission lines	San Pedro	TIGER/Line files	Arizona	Unknown	2000

3.8 Aesthetic viewshed use models

The use model for aesthetic viewsheds is quite similar to that for proximity, with the exception that we do not use the “Urban Proximity” node. This is because views are potentially equally valuable in urban, suburban, or rural settings. The use model thus simply identifies housing and its value as determinants of use. We assumed the same priors as for the aesthetic proximity use model. The contingent probability table for “View Use” simply states that in order to have value, housing must be present, and that the added value from aesthetic viewsheds is greater for higher-value housing (Figure 3.5).

Figures 3.5: Bayesian network models for aesthetic viewshed use.

Figure 3.5.1: Aesthetic viewshed use for San Pedro River Watershed.

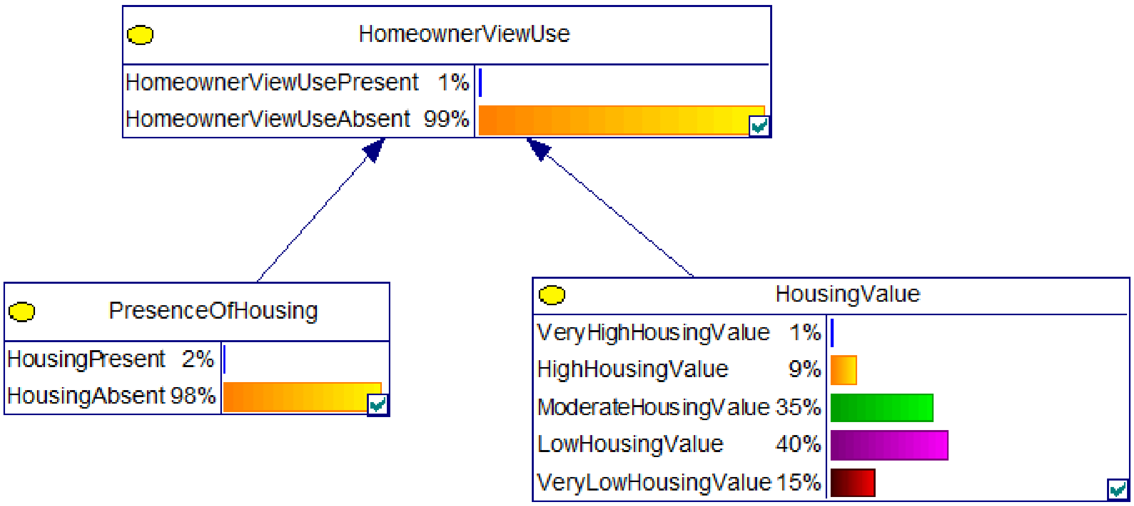


Figure 3.5.2: Aesthetic viewshed use for Western Washington.

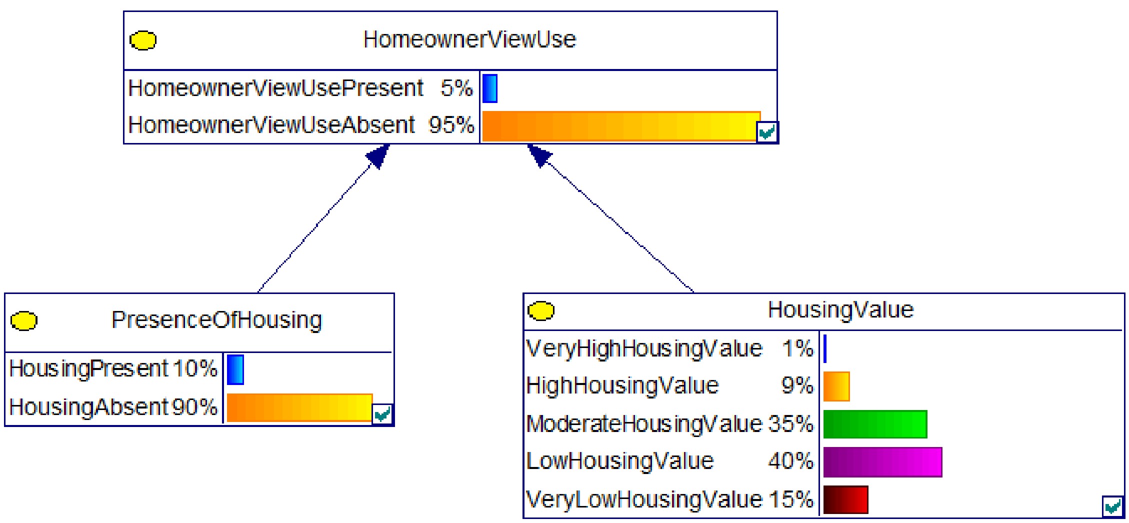


Table 3.7: Datasets used for the viewshed use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Housing values	All	County assessors' offices	Pima & Pinal Cos., AZ; Grays Harbor, King, Kitsap, Mason, Snohomish, & Thurston Cos., WA	Parcel	2004 (Kitsap Co.), 2006 (King Co.); 2010 (Pinal & Pima Cos.); uncertain for others
Presence of housing	All	County assessors' offices	Pima & Pinal Cos., AZ; Clallam, Grays Harbor, Jefferson, King, Kitsap, Mason, Snohomish, & Thurston Cos., WA	Parcel	2004 (Kitsap Co.), 2006 (King Co.); 2010 (Pinal & Pima Cos.); uncertain for others
Scenic highways	All	TIGER/Line files and Rand McNally Road Atlas scenic drives	Southeast Arizona, Western Washington	Unknown	
View use	Western Washington	County assessors' offices	King County, WA	Parcel	2006 (King Co.)

3.9 Aesthetic viewshed flow models

View flows are accounted for through a line-of sight (ray casting) model (Johnson et al. 2010). The model relies on a digital elevation model (DEM, Table 3.8) to identify locations where topography blocks views. Using top surface LIDAR data instead of elevation would account for the presence of trees and buildings and could more accurately represent obstructions to viewsheds. However, LIDAR data are not always available and are often at very high spatial resolution (slowing processing time). The relative view quality of objects in the landscape (desirable and undesirable) is projected toward potential viewers. When a view from a residential location includes visual blight, a sink, view quality is depleted.

Steinitz et al. (2003) note that for southeast Arizona, the view of another residential property depletes view quality only within a 0.8-km radius of the viewer's perspective (i.e., the effect drops off relatively quickly). Thus, we use a steep decay function to model the effects of sinks.

We quantify view quality in abstract units, from 0-100. Linking the source, sink, and use data with the flow model, we estimate the following indicators for aesthetic viewshed flows [11]:

1. Theoretical source, sink, and use. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Potential views: All possible areas capable of supplying high-quality natural views (e.g., mountains, water bodies, locally significant vegetation types).
    b. Potential visual blight: All possible areas of visual blight that could degrade view quality.
    c. Homeowners with view demand: All possible residences, as anyone is capable of gaining value from having high quality views.
2. Possible flow, source, and use. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* (visual blight) – i.e., benefits in the absence of visual blight. The possible values represent the maximum achievable service delivery based on the theoretical source value.
    a. Possible views: The flow of aesthetic information (views) from natural areas toward homeowners, when not accounting for sinks.
    b. Visible natural beauty: Open space providing views when accounting for lines of sight and the location of homeowners but not visual blight.
    c. Homeowners with possible views: Homeowners benefiting from views when sources of high-quality views and their flow paths are accounted for, but visual blight is not.
3. Actual flow, source, sink, and use. Actual viewshed benefits provided, received, and degraded *with a full accounting for source, sink, and use values and flows*.
    a. Actual views: The actual flow of aesthetic information (views) from natural areas toward homeowners, when accounting for sinks and flow paths.
    b. Enjoyed views: Open space providing views when accounting for flow paths, sinks, and the location of beneficiaries.
    c. Relevant visual blight: Areas of visual blight located between visually valuable views and beneficiaries that actually degrade high quality views.
    d. Homeowners with views: Homeowners benefiting from views when accounting for sources of views, sinks, and flow paths.
4. Inaccessible source, sink, and use. Theoretical values minus possible values; accounts for sources that do not provide, sinks that do not degrade, and beneficiaries that cannot use an ecosystem service due to a lack of flow connections.
    a. Unseen views: Potential sources of aesthetic enjoyment that are not accessible to homeowners since they are not connected by flow paths or no homeowners live in the area.

[11] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

   b. Inaccessible visual blight: Sinks that do not degrade views because they are not within view of homeowners and sources of views.
   c. Homeowners without views: Homeowners lacking any views due to their lack of flow connections (i.e., living in areas too flat or distant from high quality views).
5. Blocked flow, source, and use. Flows, source or use values degraded by sinks.
   a. Blocked views: Flows of aesthetic information (views) toward homeowners that are blocked by visual blight.
   b. Degraded natural beauty: Sources of views that are blocked by the presence of visual blight.
   c. Homeowners with degraded views: Homeowners who would receive benefits from views but have their views degraded by visual blight.

Table 3.8: Datasets used for the viewshed flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Elevation	All	SRTM	Global	90 m x 90 m	2000

3.10 Caveats and directions for future research

As discussed in the introduction to this chapter, cultural preferences for or against views or proximity to particular landscape features are likely to vary, though some preferences, such as those for views of mountains and water bodies seem to be consistent across the literature. Because of this, models should be regionally distinct when possible by relying on information about local preferences. Fortunately these preferences have been assessed in our two case study regions, and meta-analyses can provide more general guidance for how people value aesthetic views and proximity.

Another key limitation of these models is the uneven availability of parcel and housing data. These data are generally only available in more populous and wealthy counties. For instance, housing presence data were only available for 2 of 5 counties in Southeast Arizona and 8 of 15 counties in Western Washington. Housing value data were available for 2 of 5 counties in Southeast Arizona and 6 of 15 counties in Western Washington. Where data are unavailable, Bayesian prior probabilities are currently used in the models. While proxy data like NLCD developed land or zoning data can be used to identify residential locations, these data can include other types of developed land (lumping commercial and industrial developed land uses with residential land use) or can include areas zoned for residential use but not currently developed. Overlaying zoning data, where available, over NLCD data may be a way around this problem. However, the high degree of local variation in parcel, housing, and zoning datasets means that, for the time being, there will continue to be greater up front data processing needs for aesthetics models than other ARIES models as new case studies are developed.

3.11 Acknowledgements and additional contributors

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4. Flood regulation

4.1 Introduction

Modeling disturbance regulation (de Groot et al. 2002, MA 2005) as a discrete set of ecosystem service benefits in ARIES requires us to identify specific benefits and beneficiaries, so that corresponding ecosystem values can be unambiguously quantified. We have so far conceptualized the benefits of disturbance regulation into three distinct groups of benefits: 1) protection of economically valued assets from flooding along rivers and lakes, 2) protection of lives and assets from storm-related flooding in coastal zones, and 3) prevention of landslides, mudslides, or avalanches. This chapter describes ecosystem service models for flood regulation along rivers while Chapter 6 covers coastal flood regulation. ARIES models to address landslide, mudslide, and avalanche protection have not yet been developed.

The ARIES flood models start by mapping *sources* of precipitation and snowmelt, which can cause floods, *sinks* that absorb, detain, or promote infiltration of floodwater, and *beneficiaries* that may receive flood mitigation services. Sink models incorporate vegetation and soil data that describe how well different areas can promote infiltration and evapotranspiration. Vegetation and soils provide what is collectively termed *green infrastructure*, which acts along with *gray infrastructure* such as dams and detention basins that detain flood waters. Flood flow models spatially link sources of floodwater, beneficial sinks, and beneficiaries in the landscape (Table 4.1). These models account for the location and width of floodplains and the effects of levees, which protect assets at risk behind levees but at the same time increase the energy of the water conveyed downstream, potentially increasing downstream damage. Different beneficiary groups may be protected from flooding – crops, privately owned housing or other buildings, publicly owned infrastructure, and human life. Spatial data showing riverine flood zones and maps of population, agriculture, and structures allow beneficiaries to be mapped.

Table 4.1: Summary characteristics of the ARIES flood regulation models.

Attribute	Flood regulation
Benefit type	Preventive
Medium/units	Water (runoff, mm/yr)
Scale	Watershed
Movement	Hydrologic flow (rising into floodplains)
Decay	None
Rival?	Nonrival
Source	Rainfall & snowmelt
Sink	Water absorbed by soil & vegetation
Use	Lives and economic assets in floodplains

The ARIES flood regulation model operates on an annual time step moving water according to hydrologic flow properties. While modeling at finer time scales (i.e., modeling specific flood events) would be desirable, the data needed to populate these models at such time scales are rarely available. For example, although precipitation data may be present on daily time scales for the United States, spatially explicit data for soil moisture, snowpack, frozen soils, and storage levels in dams and detention basins (where relevant) are generally unavailable. Model results for specific flood events by external flood models can be incorporated the ARIES system; full support for such models within ARIES will be explored in the future. The current ARIES flood models can be used to explore general flood vulnerability, the beneficiaries of flood regulation, and the effects of flood mitigation alternatives (e.g., forest management, levee setbacks) on provision of flood regulation plus other ecosystem services co-benefits.

We developed flood regulation models for **Orange County, California** and **Western Washington State**. These models are intended to be representative of flood regulation dynamics in wider regions. For instance, the Orange County models are designed to be applicable for developed landscapes within California coastal sage, chaparral, montane chaparral and woodland ecosystems and the Western Washington models can be applied to Oregon, Washington, and British Columbia coastal forests, including the Cascade and Coast Ranges. In addition to these regionally specific models, a generalized global model of flood regulation is planned for a future release of ARIES. This model will use global datasets and provide coarser resolution model outputs; it will automatically be used in the absence of regionally-specific ARIES case studies.

Ecological restoration focused on improving flood regulation (e.g., reconnecting rivers to floodplains through levee setbacks) can provide a wide range of economic benefits from ecosystem services (Opperman et al. 2009). This strategy is a key component of the flood mitigation plan in King County, WA (King County 2006). Such scenarios can be simulated using the ARIES scenario editor in the flood model and other relevant ecosystem services models. While estimates of the economic value of ecosystem services derived from forestry and flood control projects has been well explored in Puget Sound (Leschine et al. 1997, American Forests 1998, Swedeen and Pittman 2007), ARIES provides previously missing spatial planning opportunities to explore tradeoffs between multiple ecosystem services associated with forest management and other flood mitigation projects.

In the Orange County case study, we quantified and mapped flood protection provided throughout the 223 km^2 San Gabriel River/Coyote Creek watershed in northwestern Orange County, California. A higher-resolution analysis of flood regulation services provided by an undeveloped 600-acre site being considered for development was also conducted (Casias 2010). Southern California is characterized by flashy flood dynamics due to runoff-prone topography (short distance from steep mountains to ocean), long dry periods followed by bursts of short wet periods, and a large proportion of highly urbanized, impervious surfaces, where floods can peak in a matter of minutes. These factors make the identification and measurement of natural flood regulating regions vital to sustainable and socially optimal urban development.

Past ecosystem services studies have essentially mapped flood sinks using spatial data; we drew on these approaches in developing our sink models. Eade and Moran (1996) mapped flood regulation based on soil drainage classifications, while Chan et al. (2006) did so by estimating percent natural land cover, percent natural land cover within riparian zones, distance to the 100-year floodplain, percent agricultural land, and housing units in the 100-year floodplain. Boyd and Wainger (2003) mapped flood regulation using spatial data including floodplain locations, housing and commercial units and value, percent floodplain as impervious and wetland. Boyd and Wainger also included an environmental justice component to their measures, by mapping median income and percent black or Hispanic populations within their impacted area. At the continental scale, Bradshaw et al. (2007) quantified the influence of forest cover on flood frequency and severity when controlling for rainfall, slope, and landscape degradation. We drew on these studies in developing our flood sink models, then extended these approaches by explicitly accounting for the spatial dynamics of flood regulation.

4.2 Flood regulation source models

We use annual precipitation as the source of floodwater in the ARIES flood model (Table 4.2). Flood regulation is a preventive service, where sinks increase the flow of benefits by reducing the flow of a threat that originates in source regions. For event-based flood modeling, snowmelt is an extremely important variable in seasonally cold-weather climates, such as Western Washington. As discussed above, data limitations prevent event-based flood modeling in ARIES, so snow presence and snowmelt are not currently included in the flood source model.

Table 4.2: Datasets used for the floodwater source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Annual precipitation	All	PRISM / OSU	United States	800 m x 800 m	1971-2000

4.3 Flood regulation sink models

We defined flood sink value, the top-level output of the sink model, as the sum of green infrastructure storage (the sum of infiltration, absorption, detention, or evapotranspiration of potential flood waters by vegetation, soils, and floodplains) and gray infrastructure storage (the sum of storage in detention basins and reservoirs) (Figure 4.1)[12]. Both gray and green infrastructure can be "saturated" when their individual components are at full capacity. Because of this, we added the mean days of precipitation per year as an influence to green and gray infrastructure storage in the Western Washington model. This accounts for the fact that green and gray

[12] Bayesian network models for flood sink models can be downloaded from http://ariesonline.org/modules/floodspecs.html.

infrastructure are likely to be saturated for more of the year in regions where precipitation is more evenly distributed over the course of a year, allowing soil moisture to remain more temporally uniform (Table 4.3). We did not include this variable in the Orange County model, since we assume the system to be "unsaturated" for most of the year, since Southern California experiences low annual rainfall and flood events are extremely flashy.

By computing the difference between precipitation and runoff (which accounts for vegetation and soil characteristics), we can estimate the contribution of green infrastructure to flood mitigation. We can thus use the difference between precipitation and runoff as training data for the Bayesian network. Models such as the Curve Number method (CN, SCS 1972), which incorporates data on precipitation, hydrologic soils group, and land use-land cover, can also be used to calculate runoff.

We set soil infiltration as a function of impervious surface cover, slope, and hydrologic soils group. These variables have been routinely recognized as predictive variables for potential soil infiltration (i.e., USACE 1998, Tetra Tech, Inc. 2005, Laton et al. 2006, BOR 2007). We considered adding water table depth (available from SSURGO/STATSGO data) as an influence on infiltration but ultimately decided not to include it to maintain tractability in the contingent probability table. Evapotranspiration reduces soil moisture, thereby allowing increased infiltration. In addition, it serves as a proxy for other flood mitigation processes due to the presence of vegetation. We set evapotranspiration as a function of percent tree canopy cover and vegetation type (in both models) and added influences for successional stage and vegetation height for the Western Washington model, as Jones and Post (2004) and Moore and Wondzell (2005) note the importance of forest cover and successional stage as drivers of hydrologic processes in Pacific Northwest forests.

We discretized mean days of precipitation per year using Jenks Natural Breaks and estimated its priors on a review of the data for Western Washington. We reviewed spatial data for Orange County and Western Washington to derive priors for impervious surface cover, slope, and hydrologic soils group. We discretized impervious surface cover to account for ecological thresholds typically present when impervious surface exceeds 10% (Booth and Jackson 1997). We used equal intervals to discretize percent tree canopy cover and Jenks Natural Breaks to discretize vegetation height for Western Washington. We estimated priors based on spatial data for percent tree canopy cover, successional stage, vegetation height, and vegetation type.

For the soil infiltration contingent probability table, we set the highest values of infiltration at low impervious surface cover and slope and hydrologic soils groups A and B. We set the lowest values for infiltration under opposite conditions and interpolated intermediate values. We set the evapotranspiration contingent probability table to its greatest values in cases of greater percent tree canopy cover, later successional stage, tall vegetation (where applicable), and wetlands, and vice versa, and interpolated intermediate values. We set evapotranspiration as slightly lower than wetlands for forests, grassland, and shrubland, and substantially lower for developed and cultivated land use types.

We set evapotranspiration and soil infiltration as equivalent influences on the green infrastructure storage contingent probability table. In the Western Washington model, we set mean days of precipitation per year as a strong influence on the contingent probability tables for both gray and green infrastructure storage (i.e., much greater storage when there were very low or low mean days of precipitation per year, and vice versa). We summed values for dam and detention basin storage to quantify gray infrastructure storage, and added this to the value of green infrastructure storage to estimate the total flood sink.

Figures 4.1: Bayesian network models for floodwater sinks.

Figure 4.1.1: Floodwater sinks for Orange County.

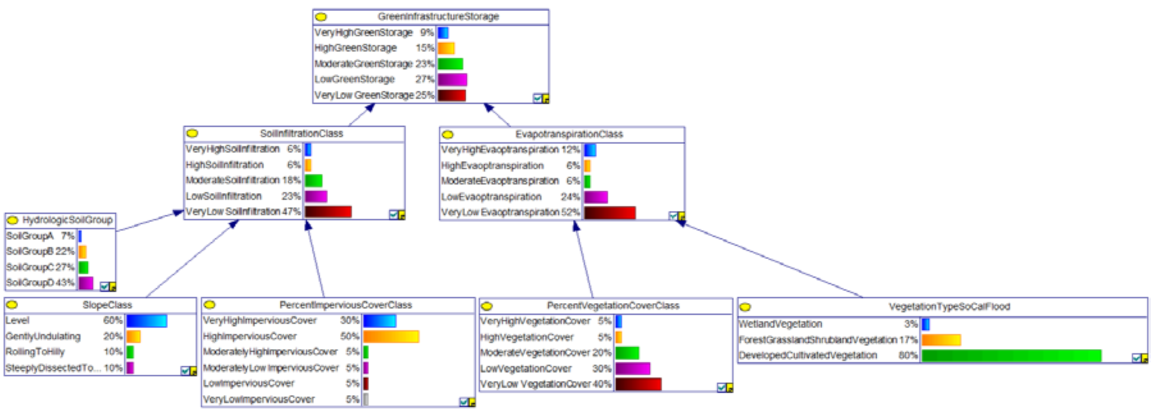

Figure 4.1.2: Floodwater sinks for Western Washington.

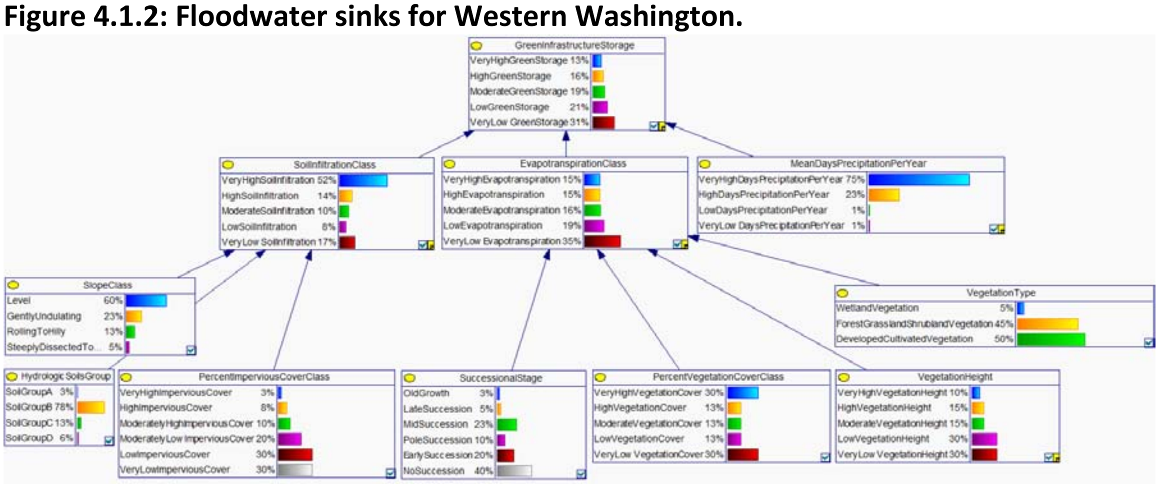

Table 4.3: Datasets used for the floodwater sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Average annual actual evapotranspiration	All	SAGE / UW Mad	Global	$0.5^{o} \times 0.5^{o}$	1950-1999
Average annual runoff	All	SAGE / UW Mad	Global	$0.5^{o} \times 0.5^{o}$	1955-1990
Average annual soil infiltration	Orange County (no data for Western Washington)	LA Basin Groundwater Augmentation Model (GWAM)	West Coyote Hills site	Unknown	1951-2002
Detention basins	Western Washington	County GIS offices	King, Pierce, San Juan Counties	Unknown	Varies
Dam storage	All	National Atlas of the United States	United States	Unknown	2006
Hydrologic soils group	All	SSURGO & STATSGO soil data	Orange County, Western Washington	Unknown	n/a
Impervious surface cover	All	NLCD 2001	United States	30 m x 30 m	2001
Mean days of precipitation per year	Western Washington	PRISM / OSU	Continental United States	Unknown	1971-2000
Slope	All	Derived from SRTM	Global	90 m x 90 m	2000
Successional stage	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Tree canopy cover	All	NLCD 2001	United States	30 m x 30 m	2001
Vegetation height	Western Washington	Puget Sound LIDAR Consortium	Western Washington	30 m x 30 m	2000-2006
Vegetation type	All	NLCD 2001	United States	30 m x 30 m	2001
	Orange County	USFS	Northern Orange & Southern LA Counties	Unknown	2003

4.4 Flood regulation use models

Beneficiaries of flood regulation can be mapped using spatial data and simple GIS overlay operations, eliminating the need for more complex approaches. In these case studies, we identified different beneficiary classes, including farmers, residents, and municipalities with public infrastructure located within the floodplain boundaries (Table 4.4). We mapped beneficiaries in both the 100-year and 500-year floodplains in order to differentiate between levels of risk from catastrophic floods of different sizes.

Table 4.4: Datasets used for the flood use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Farmland	All	NLCD 2001	United States	30 m x 30 m	2001
Floodplain extents	All	FEMA Q3 Flood Data	United States	Unknown	Varies
Highways	All	TIGER/Line files	United States	Unknown	2000
Presence of housing	Western Washington	County assessors' offices	Clallam, Grays Harbor, Jefferson, King, Kitsap, Mason, Snohomish, Thurston cos., WA	Parcel	2004 (Kitsap Co.), 2006 (King Co.); uncertain for others
Railways	All	TIGER/Line files	United States	Unknown	2000

4.5 Flood regulation flow models

The source and sink models determine the quantity (in mm/yr) of precipitation falling on the landscape and absorbed or detained by the landscape, while the use model defines the location of potential flood regulation beneficiaries. The flow model routes water from its source locations through the watershed based on the topography of the location (Table 4.5). Once the flow of water moving across a landscape intersects a stream, its movement is no longer determined by topography and instead follows the direction of the streambed. Once floodwater is in a stream, it can overtop the streambanks, depending on the amount of floodwater, floodplain width, and the presence of levees. If the downstream flow reaches a dam, floodwater is temporarily detained unless excess water in an already-full reservoir must be released downstream. If floodwater reaches a user, it causes damage. This damage can be attributed to upstream flood sources, and mitigated damage can be attributed to upstream flood sinks, which provide the ecosystem service of flood regulation.

While this is an admittedly simplistic way to move water and water-related ecosystem service carriers (e.g., drinking water, flood water, suspended sediment, dissolved nutrients), this approach has the benefit of being applicable at relatively coarse spatial scales and at any location on Earth. Future work on ARIES will seek to incorporate locally tested hydrologic models to route water and water-related ecosystem service carriers across the landscape at variable spatial scales and under variable environmental conditions (e.g., using different models at large vs. small spatial scales and in arid versus humid ecological systems).

Linking the source, sink, and use data with the flow model, we estimate the following indicators for flood regulation flows [13]:

1. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
   a. Runoff: The quantity of runoff produced by each portion of the landscape.
   b. Potential runoff mitigation: All areas capable of absorbing or detaining flood water.
   c. Potentially vulnerable populations: Any areas where people or economically valuable assets are located in flood zones.
2. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* (areas that allow detention, infiltration, or slowing of flood water) – i.e., benefits in the absence of flood regulation. The possible values represent the maximum delivery of floodwater based on the theoretical source value.
   a. Potentially damaging flood flow: The flow route of floodwater across the landscape in the absence of sinks.
   b. Potentially damaging runoff: Runoff capable of harming people or damaging property when accounting for flow paths but not sinks.
   c. Potential flood damage received: People and property receiving damage when accounting for sources of floodwater and its flow path but not accounting for the action of sinks that reduce potential damage from floodwater.
3. Actual flow, source, sink, and use. Actual flood regulation benefits provided, received, and mitigated *with a full accounting for source, sink, and use values and flows*.
   a. Actual flood flow: The flow route of floodwater across the landscape in the presence of sinks.
   b. Flood damaging runoff: Runoff that actually harms people or damages property when accounting for flow paths and sinks.
   c. Utilized runoff mitigation: Sinks that actively reduce floodwater, providing the benefit of reduced flood damage for people.
   d. Flood damage received: Actual damage received by people and property when accounting for sources of floodwater, flow paths, and sinks encountered.
4. Inaccessible source and use. Theoretical values minus possible values; accounts for sources that do not provide floodwater and beneficiaries that cannot use an ecosystem service due to a lack of flow connections.
   a. Benign runoff: Runoff that does not have people or economically valuable assets lying in its path.

[13] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

   b. Unutilized runoff mitigation: Sinks capable of reducing flood flows but lacking associated human beneficiaries who value this protection, or lacking flood water to mitigate.
5. Blocked flow, source, and use. Flows, source, or use values mitigated by sinks.
   a. Absorbed flood flow: Flood flows that are absorbed by sinks prior to reaching vulnerable human beneficiaries.
   b. Flood mitigated runoff: The portion of the total runoff that is absorbed, detained, or slowed by the action of flood sinks.
   c. Flood mitigation benefits accrued: People or economically valuable assets who are spared from flood damage due to the flood regulation activity of sinks.

Table 4.5 Datasets used for the flood flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Dams	All	National Atlas of the United States	United States	Unknown	2006
Elevation	All	SRTM	Global	90 m x 90 m	2000
Floodplain extents	All	FEMA Q3 Flood Data	United States	Unknown	Varies
Hydrography (stream networks)	Orange County	Cal-Atlas Geospatial Clearinghouse	California	Unknown	Unknown
	Western Washington	WA DNR	Washington State	Unknown	Unknown
Levees	Western Washington	County GIS offices	King, Lewis, Pierce Cos	Unknown	Varies

The limitations of modeling flood regulation at such a coarse time scale have also been discussed above. Our model outputs currently map the spatial linkages between sources of precipitation, green and gray infrastructure influences on flood mitigation, and the location of beneficiaries. While we are not yet capable of mapping flood regulation on an event-by-event basis, these models can better account for the spatial dynamics of flooding and allow users to overlay flood control benefits with provision and use maps for other ecosystem services. Yet moving toward a finer temporal scale flood model that can account for individual flood events remains an end goal in order to maximize the utility of the ARIES flood regulation models. Event-based flood modeling is made more challenging by limitations in the spatial resolution of event-based rainfall data. This limitation, which is discussed in more detail in Section 8.6 in the water supply chapter, is more serious in arid and semiarid environments where rainfall patterns are more uneven.

4.7 Acknowledgements and additional contributors

Mark Casias developed the Orange County case study. Dave Batker, Jim Pittman, and Paula Swedeen provided data and model review for the Western Washington case study.

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5. Subsistence fisheries

5.1 Introduction

Subsistence harvesting of ecosystem goods – food, fuel, fiber, and other basic resources – is a critical contributor to livelihoods in much of the developing world as well as in parts of developed nations (MA 2005). These ecosystem goods have been termed the "GDP of the poor" (TEEB 2008), since they provide employment and livelihoods for so many of the world's poor while not being monetized as part of traditional national economic accounts like GDP. Societal dependence on subsistence fisheries, combined with the recognition that the world's oceans are in crisis due to overfishing, pollution, and climate change (Diaz and Rosenberg 2008, Cooley and Doney 2009, Worm et al. 2009, Hoegh-Guldberg and Bruno 2010) strongly argues the case for more sustainable management of aquatic and marine resources. By mapping societal dependence on subsistence fisheries, we can demonstrate direct linkages between ecosystems and human well-being. In addition, by linking flows of sediment, nutrients, and fresh water from land into the coastal zone, the complex tradeoffs between land management choices and provision of coastal and marine ecosystem services can be illuminated. This understanding is critical given the growing recognition of complex spatial flows of ecosystem processes and services between terrestrial, coastal, and marine environments (McCulloch et al. 2003, Fabricus 2005, Silvestri and Kershaw 2010).

We developed our initial subsistence fisheries case study for **Madagascar**, due to its high rate of poverty, dependence on fisheries as a key protein source in rural communities, and linkages between deforestation, erosion, and sedimentation that take place on land but strongly affect its coastal and marine systems (see Chapter 7 for a discussion of ARIES sedimentation models for Madagascar). The Madagascar models rely on global spatial datasets for fisheries, population density, and poverty, combined with non-spatial data on national fisheries use from the FAO (2008) and The Sea Around Us Project (2010). Since such data are available for all nations, it is quite feasible to extend coverage of this subsistence fisheries model to other countries. However, since key harvested fish species and dependence on subsistence fisheries differ by nation, models should be developed on a nation-by-nation basis rather than as part of a generalized global model.

For Madagascar, FishBase (Froese and Pauly 2010) lists eight commercially important fish species, in some cases providing descriptions of their habitat, commercial importance, and typical means of harvest. From this list, The Sea Around Us project (Close et al. 2006) provided relative abundance maps for four species: southern meagre or cob (*Argyrosomus hololepidotus)*, sky emperor (*Lethrinus mahsena)*, slender emperor (*Lethrinus variegatus)*, and mangrove red snapper (*Lutjanus argentimaculatus)*. Our initial fisheries models focus on subsistence use of fisheries as the only class of beneficiaries, ignoring for the time being both recreational and commercial fisheries. Subsistence fisheries require an understanding of the species and quantity of fish harvested and used and the ability of users to access the resource (Table 5.1), and are thus a logical starting point for modeling how beneficiaries use and value fisheries. Commercial and recreational fisheries have different beneficiary groups and flow characteristics (i.e., means for their beneficiaries to reach and use the fishery resource). Although recreational and commercial fisheries are of great importance, their flow models are likely to be more complex, requiring accounting for trade networks and recreational choices (described in more detail for recreation in Chapter 9).

Table 5.1: Summary characteristics of the ARIES subsistence fisheries models.

Service	Subsistence fisheries
Benefit type	Provisioning
Medium/units	Fish (kg)
Scale	Walking distance
Movement	Walking simulation
Decay	Gaussian
Rival?	Rival
Source	Fishing grounds
Sink	None
Use	Subsistence communities near fisheries

5.2 Subsistence fisheries source models

The data available to establish the quantity of fish supplied and demanded in subsistence fisheries are generally quite sparse. We obtained global relative abundance maps for four species of commercial importance in Madagascar. We also obtained historical catch data for Madagascar from the FAO (2008). These catch data do not explicitly describe subsistence catch, however, nor do they contain catch records specific to any of the four species for which we have relative abundance data.

Despite these limitations, we can use these data, along with the assumptions described below, to make a first estimation of the catch for each species (i.e., its supply or source value). These assumptions can be adjusted should more refined fisheries information become available. Historical catch data from The Sea Around Us Project (2010) show that 71.6% of the total catch for Madagascar from 1950-2004 is "non-identified marine pelagic fishes." In 2005, Madagascar produced 138,477 tonnes of fish from capture fisheries, of which 77,636 tonnes were for commodity trade and production (FAO 2008). Another 72,300 tonnes were taken by traditional fishermen – however, this catch is not necessarily mutually exclusive of the commodity sector. Subtracting the commodity catch from the total catch leaves 60,841 tonnes of fish, which we assume goes toward local consumption. Assuming 71.6% of the locally consumed fish are "non-identified marine pelagic fishes," these would account for 43,562 tonnes in 2005. Assuming that each of the three marine pelagic fish species for which we have distribution data (*L. mahsena*, *L. variegatus*, *L. argentimaculatus)* accounted for 20% of that total (leaving 40% of the catch from other species), this would give 8,712 tonnes of fish caught for each of these species. We omit southern meagre (*A. hololepidotus)* from the model because it is a demersal species and we lack data on the catch of non-identified marine demersal fish species. We then divide the total catch for each species according to their relative abundance along the Madagascar coast in order to produce a map of subsistence fish production for each species under the above assumptions (Equation 1). A similar equation holds for subsistence fisheries use (Equation 2).

$$(1)\quad S = a_1S_1 + a_2S_2 + a_3S_3 + \ldots + a_nS_n$$
$$(2)\quad D = a_1S_1 + a_2S_2 + a_3S_3 + \ldots + a_nS_n$$

Where S = the total source or supply of subsistence fisheries (kg)
D = the total demand for subsistence fisheries (kg)
a_n = the percentage of the total demand met with species n
S_n = the total mass of species n caught (kg)

Given better data on the relative use of different species, we need only adjust the model coefficients in order to more accurately quantify human dependence on Madagascar's fisheries. The unit of measure for the fisheries source/supply and use/demand models is kg of fish biomass. This allows us to use a common unit in the flow models to determine the accessibility between catch areas and the places where fish are consumed.

Table 5.2: Datasets used for the subsistence fishery source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
L. borbonicus, L. mahsena, L. argentimaculatus relative abundance	Madagascar	Sea Around Us Project	Global	$0.5^o \times 0.5^o$	1950-2003

5.3 Subsistence fisheries sink models

As an ecosystem good, subsistence fisheries do not have a sink that depletes the quantity of the good as it moves along flow paths from ecosystems to people. In other words, there are no anthropogenic or biophysical features on the landscape that deplete the quantity of caught fish as they are moved from the point of extraction to the point of consumption. Thus, fish harvest implies use of the resource.

5.4 Subsistence fisheries use models

People are more likely to access fisheries for subsistence purposes when they lack other sources of nutrition (e.g., in poorer regions) and when they have access to a fishery. We used distance to the coast, poverty, and population density as determinants of the degree of subsistence fisheries use (Figure 5.1)[14]. We assigned prior probabilities for

[14] Bayesian network models for subsistence fisheries use models can be downloaded from http://ariesonline.org/modules/fishspecs.html.

distance to the coast, poverty, and population density based on reviews of available spatial data for Madagascar. We discretized population density by Jenks Natural Breaks, percent poverty by equal intervals, and distance to the coast based on easily walkable distances (<1 km, 1-5 km, >5 km). In the use model's contingent probability table, we expect greater subsistence fishery use to occur at greater levels of poverty, closer proximity to the coast, and moderate to low population densities (i.e., versus large cities there is too much crowding, water pollution, and lack of access to fisheries to enable widespread subsistence fishing).

This model only addresses coastal and marine subsistence fishing. We currently do not account for subsistence fisheries from rivers and other inland water bodies. Fish could also be traded to groups located farther from the coast, with the benefits thus extending further inland. However, modeling of such trade networks is beyond the scope of this assessment and may be handled by a separate family of models to be developed.

The Bayesian network (Figure 5.1) returns likelihood values for four classes of subsistence marine fisheries use. To convert this to total demand, we note that the average Malagasy consumes 6.8 kg of fish per year (FAO 2008). In areas with high subsistence use, we assume 100% of this per capita use to come from the oceanic subsistence fishery. In areas with moderate subsistence use, we assume 67% of this total to come from the oceanic subsistence fishery (4.6 kg/capita), and in areas with low subsistence use we assume 33% of this total to come from the oceanic subsistence fishery (2.3 kg/capita). We assume that the remaining unsatisfied demand from moderate and low subsistence users comes from either trade or inland fisheries in rivers and lakes. Similarly to mapping the supply or source of subsistence fisheries, we assume that each of the three valuable species supplies 20% of the fish use for subsistence users (Equation 2).

Figure 5.1: Bayesian network models for subsistence fishery use in Madagascar.

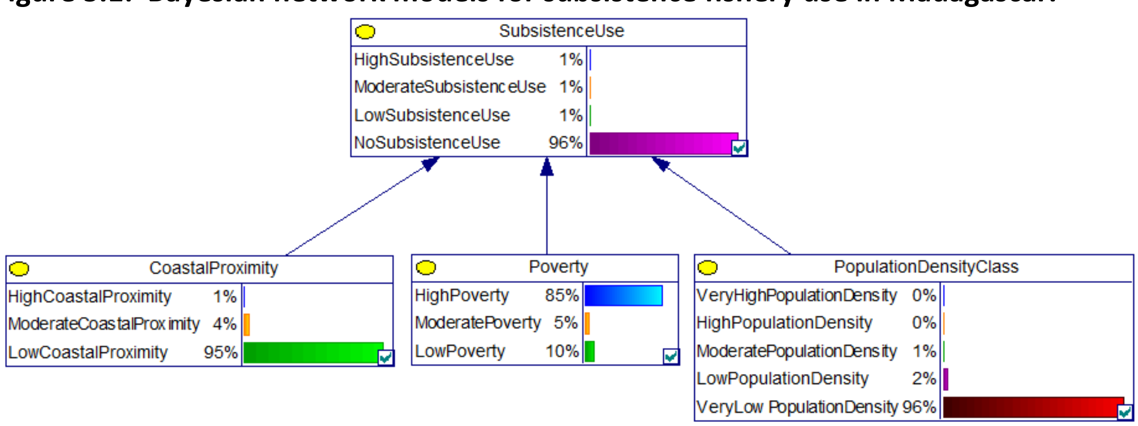


Table 5.3: Datasets used for the subsistence fishery use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Distance to coast	Madagascar	FTM	Madagascar (Ring buffer around coastline)	Unknown	n/a
Population density	Madagascar	LandScan / ORNL	Global	30 arc-second^2	2006
Poverty	Madagascar	Elvidge et al. (2009)	Global	30 arc-second^2	2004

5.5 Subsistence fisheries flow models

Subsistence fisheries flow models are designed to show the expected spatial dependence of specific fisheries users on particular fisheries areas. We assume that subsistence fisheries users move from a point of origin (their homes) to an oceanic fishery, harvest fish, and return to their homes, where fish are consumed, via road or path networks (Table 5.4). Thus the flow models move a given quantity of fish (in kg) from the ocean to areas of demand along roads or paths. Each coastal ocean pixel has an estimated potential source of fish, as described above for the source model. Each pixel on land has an estimated potential use for oceanic fish, as described above for the use model.

We model the flow of subsistence fisheries using a distance decay function, assuming that flow is greatest close to the coast and declines relatively quickly moving inland. This takes the shape of a Gaussian curve with high subsistence use within 1 km of the coast, steep decline at distances of 2-3 km of the coast, and slowly declining subsistence use out to a distance of 5 km of the coast. Along with proximity, users must have some form of access (e.g., roads, paths) to access the coastal fisheries.

Linking the source and use data with the flow model, we estimate the following indicators for subsistence fisheries flows[15]:

1. Theoretical source and use. These are the values initially estimated by the source and use models *without accounting for flows*.
    a. Subsistence fish supply: The supply of harvestable fish.
    b. Subsistence fish demand: The demand for fish from subsistence users.
2. Actual flow, source, and use. Actual subsistence fisheries benefits provided and received *with a full accounting for source and use values and flows*.
    a. Subsistence fish flow: The movement of fish from areas where they are caught to communities where they are consumed, based on transportation networks, supply, demand, and distance decay.

[15] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts. When there are no sinks, possible and actual values are identical, and blocked flows, blocked sources, and blocked uses do not exist.

   b. Utilized subsistence fish: The quantity of fish harvested for subsistence use.
   c. Satisfied subsistence fish demand: The portion of demand for fish satisfied by flows of fish to people from a fishery.
3. Inaccessible source and use. The difference between theoretical and actual values due to a lack of connections between source and use locations.
   a. Unutilized subsistence fish: Fisheries that are not used due to lack of proximity or pathways to users. This value may also be positive if some of the fish supply remains after all demand is satisfied.
   b. Unsatisfied fish demand: The portion of demand for fish not satisfied due to inadequate size of the fishery or a lack of proximity to a fishery. If this is zero, some unutilized fish may be a result of a true surplus and not just a lack of proximity. If this is greater than zero, there is no surplus within range.

Table 5.4: Datasets used for the subsistence fishery flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Paths	Madagascar	FTM	Madagascar	Unknown	Unknown

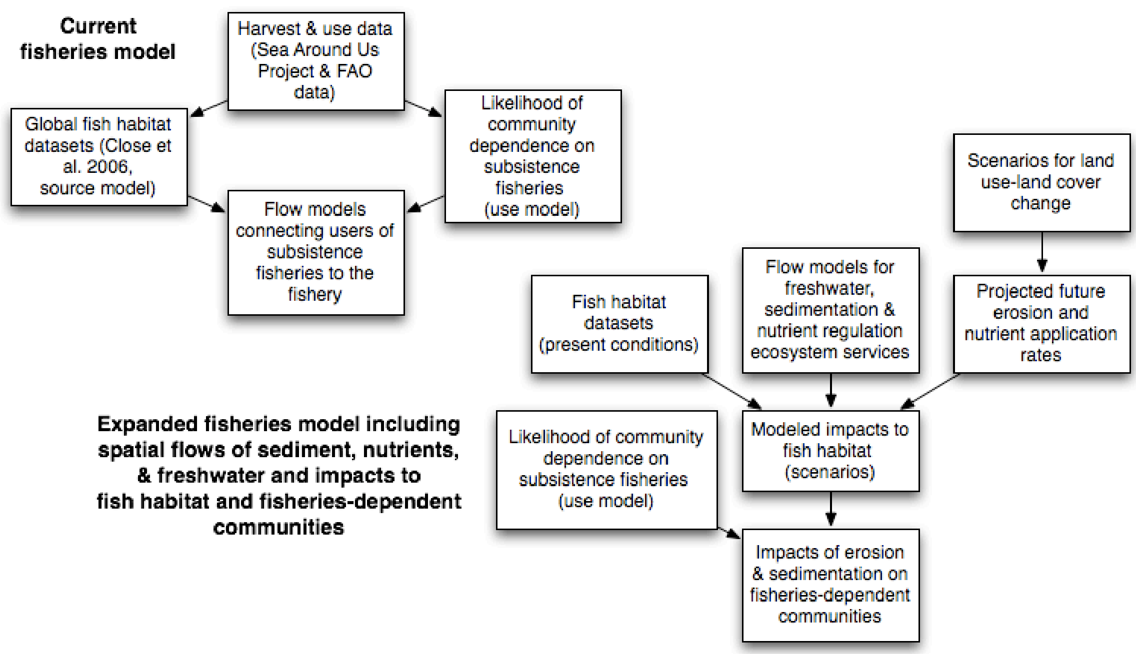
5.6 Caveats and directions for future research

Several significant assumptions are built into these models due to the limited amount of information about fish populations, their locations, and levels of subsistence fisheries use. While the species distribution maps from The Sea Around Us Project show the relative distribution of each species based on habitat suitability (Close et al. 2006), they do not show populations or total abundance of each species. We thus make assumptions that the three modeled species are caught in quantities related to their local habitat suitability, that they fill a uniform amount of local demand for subsistence fisheries, and that the species make up a specific amount of the local catch, in this case 20% each of the "non-identified marine pelagic catch." These parameters could easily be refined based on improved local data.

While we initially model only three species, additional species could easily be added to the model by including their distribution maps in the source model and adjusting the assumptions about the quantity of local catch and use for all species. To extend these models to other countries, we would review local fisheries data to develop locally appropriate assumptions about subsistence fish catch and use, and run source, use, and flow models based on the revised data.

Another future research direction for the ARIES fisheries models would link spatial flow models for sediment (Chapter 7), freshwater (Chapter 8) and nutrients (ARIES models in development, using sources including Potter et al. 2010) between terrestrial and marine environments with the goal of better understanding the impact of terrestrial land management on coastal and marine ecosystem services. We could model how these flows affect fish habitat quality (e.g., mangrove, seagrass, coral, and coastal wetland ecosystems) and fisheries-dependent communities (Figure 5.2). Although the importance of such flows are recognized (McCulloch et al. 2003, Fabricus 2005, Diaz and Rosenberg 2008, Silvestri and Kershaw 2010), models to spatially link ecosystem services supply and demand across linked terrestrial and marine systems have thus far been lacking. The ARIES flow modeling framework is a feasible way to address this gap in modeling, mapping, and valuing marine ecosystem services.

Figure 5.2: Current subsistence fisheries models and models accounting for spatial flows of sediment, nutrients, and freshwater from terrestrial to coastal and nearshore marine environments.



5.7 Acknowledgements and additional contributors

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6. Coastal flood regulation

6.1 Introduction

Modeling disturbance regulation (de Groot et al. 2002, MA 2005) as a discrete set of ecosystem service benefits in ARIES requires us to identify specific benefits and beneficiaries, so that corresponding ecosystem values can be unambiguously quantified. We have so far conceptualized the benefits of disturbance regulation into three distinct groups of benefits: 1) protection of economically valued assets from flooding along rivers, 2) protection of lives and assets from storm-related flooding in coastal zones, and 3) prevention of landslides, mudslides, or avalanches. This chapter describes ecosystem service models for coastal flood regulation while Chapter 4 covers flood regulation along rivers. ARIES models to address landslide, mudslide, and avalanche protection have not yet been developed.

High-profile tropical storm and tsunami events including the Indian Ocean tsunami in 2004 and Hurricane Katrina in 2005 spurred great interest in understanding the role of coastal ecosystems in mitigating storm surge damage (Chatenoux and Peduzzi 2007, Day et al. 2007, Cochard et al. 2008). Beyond these well-publicized disasters, tropical storms and tsunamis continue to cause loss of life and property in coastal zones around the world. In recent decades, population growth and coastal migration have placed more people and property at risk of coastal flooding, and have also damaged or caused outright loss of coastal ecosystems capable of mitigating flood damage. In addition, climate change threatens to warm the oceans and melt polar ice caps. The resulting sea level rise and increased number and size of tropical storms can further increase coastal flood risk (Emanuel 2005, Vermeer and Rahmstorf 2009). These intersecting demographic, land use, and climate drivers mean that accurately valuing coastal ecosystem services and proactively protecting and managing these resources is important today and will be even more so in the future.

We developed the initial ARIES coastal flood regulation case study in **Madagascar**. This island's coastal ecosystems include dunes, coral reefs, seagrass, and mangroves, all of which play a role in dissipating wind and wave energy generated by coastal storms. Unlike more northerly nations bordering the Indian Ocean, Madagascar is rarely subject to earthquake-generated tsunami waves. However, eastern Madagascar is highly susceptible to tropical storms coming off the southwestern Indian Ocean. Its storms are relatively poorly studied, with limited infrastructure for measuring and modeling storm magnitude and impact versus tropical storm prone regions in more developed parts of the world (Naeraa and Jury 1998).

The coastal storm protection models described in this chapter are intended as a proof-of-concept framework to spatially link: 1) storm wave *sources*, 2) ecosystem *sinks* that mitigate wave damage, and 3) social groups or assets – the *beneficiaries* that are susceptible to harm by coastal flooding, via flow models that account for wave movement (Table 6.1). These models use historical storm track, wind speed, and atmospheric pressure data from tropical storms. Tropical Storms Daisy, Geralda, and Litanne, three severe storms that made landfall near the port of Toamasina in 1994, are used in the first version of this model, based on availability of information from Naeraa and Jury (1998). Naerra and Jury modeled aspects of these storms and their attendant storm surges, providing data for calibration of model results. Our model can easily be applied to other historical storm tracks in Madagascar, to simulated future storm tracks, or to other simulated large wave events, such as tsunamis. These models can also be relatively easily generalized to other parts of the world, as they rely on global datasets for historical storms, coastal ecosystems capable of mitigating storm surges, and lives and property at risk of flood damage. However, the models do not yet incorporate external process models to describe storm surge, wave mitigation, or wave run-up. The interface support to enable the above features would make the model harder to run for non-specialists and is instead planned for a specialized version of the online interface for the ARIES coastal flood regulation models.

While we do not directly use external models, the Bayesian models described below do incorporate important elements of coastal process models to inform their structure. Incorporation of external coastal process models will greatly improve the quality and reliability of the results and will be considered for inclusion in future releases of the ARIES coastal regulation model. Section 6.7 discusses future research directions for modeling coastal flood regulation.

Table 6.1: Summary characteristics of the ARIES coastal flood regulation models.

Service	Coastal flood regulation
Benefit type	Preventive
Medium/units	Storm surge (m)
Scale	Coastal zones
Movement	Wave runup
Decay	Function of sink presence and strength
Rival?	Nonrival
Source	Coastal zones prone to storms
Sink	Vegetation, coral reefs, and topographic features
Use	Lives and economic assets in coastal flood zones

6.2 Coastal flood source models

Developing a generalizable model to predict storm surge height from wind speed data is a difficult process, as local coastal geomorphology has a strong influence on surge height (NOAA-NHC 2010). Given the limited spatial data availability for tropical storms in Madagascar, we developed a probabilistic Bayesian model to predict storm surge height (available in Naeraa and Jury 1998 for Tropical Storms Daisy, Geralda, and Litanne) based on wind speed, atmospheric pressure, and ocean depth (Figure 6.1, Table

6.2)[16]. We discretized wind speed using Meteo-France's tropical storm rating system for the Southwest Indian Ocean (World Meteorological Organization 2006). We discretized values for atmospheric pressure using break points of 970 and 990 mb and values for ocean depth using break points of 200, 50, 20, and 0 m below sea level. We set prior probabilities based on corresponding spatial data for Madagascar storms and bathymetry. We set values in the contingent probability table for storm surge size to be greatest at lowest pressure, highest winds, and over the shallowest water depths. We set ocean depth as the strongest influence on values in the contingent probability table, followed by wind speed, then atmospheric pressure.

Figure 6.1: Bayesian network models for coastal flood sources.

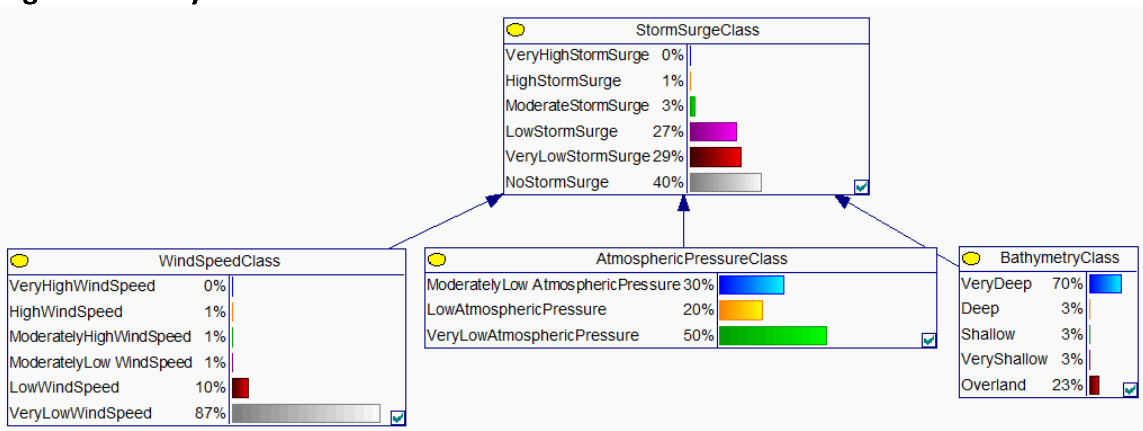


Table 6.2: Datasets used for the coastal flood source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Bathymetry	Madagascar	NASA ETOPO1	Global	1 arc-minute[2]	n/a
Tropical storm tracks (incl. wind speed and atmospheric pressure)	Madagascar	UNEP-WCMC	Global	Unknown	1980-2005

6.3 Coastal flood wave sink models

We consulted the literature on coastal protection to determine which variables to incorporate into a Bayesian network to model coastal protection (Trigo-Teixeira et al. 2000, Danielsen et al. 2005, Katiresan and Rajendran 2005, Chatenoux and Peduzzi 2007, Chen et al. 2007, Day et al. 2007, Barbier et al. 2008, Iverson and Prasad 2008, Koch et al. 2009). Substantial further work remains to improve the models presented below, including but not limited to the inclusion of existing process models to replace Bayesian models, where appropriate. We assume that these coastal ecosystems

[16] Bayesian network models for coastal flood regulation source and sink models can be downloaded from http://ariesonline.org/modules/coastspecs.html.

provide wave protection in water depths of less than 50 m and overland, so only activate the model at shallow water depths.

We set total coastal flood protection as a function of engineered shoreline protection (artificial coastal protection), natural coastal and terrestrial ecosystems, and the influence of tides (with less protection provided at high tide than low tide). We set coastal ecosystem protection as a function of the effects of coral reefs (using coral presence and bleaching as a surrogate for their ecological function) and the presence of seagrass beds and mangroves. We set prior probabilities for these variables based on reviews of the corresponding spatial data. We discretized terrestrial vegetation types based on the growth form of the dominant vegetation (trees, shrubs, herbaceous, wetland, other), which influences the degree to which that ecosystem can dissipate wave energy.

We assume that coastal regions with greater area, ecosystem health, and density of mangroves, seagrass beds, and coral reefs would benefit from greater wave protection. Seagrasses provide seasonal coastal protection during the growing season (Chen et al. 2007), with substantially lower protection outside the peak growing season. Fortunately, the majority of tropical storms occur during the warm season. The combined effects of seagrass, coral, and mangrove coastal protection are nonlinear, which is reflected in the contingent probability table for “Coastal ecosystem protection.” In other words, the presence of higher quality habitats of more than one ecosystem type is demonstrably more beneficial than the presence of a single high-quality habitat alone, so the ecosystem services provided by the coastal seascape are often more than the sum of their parts (Barbier et al. 2008, Koch et al. 2009).

In the contingent probability table for “Total coastal flood protection” (aggregating the effects of natural and artificial coastal flood regulation), we currently assumed that all else being equal, higher-quality natural coastal protection is of greater value than artificial coastal flood protection. This assumption can be revised in the contingent probability table where appropriate, but given the extremely limited artificial coastal flood protection in Madagascar (near major port cities only), its influence is relatively minor regardless. We set the influence of tides in the contingent probability table as relatively small. ODINAFRICA (2010) notes that the average tides height in eastern Madagascar is only 0.3 m, as compared to 3.2 m on the west coast of Madagascar. Naeraa and Jury (1998), however, report an average tide height of around 1.0 m for eastern Madagascar. Still, this is a relatively small influence relative to other coastal regions of the world. Tidal influence could thus be increased in the contingent probability table when the model is applied to regions with a greater tidal range.

Figure 6.2: Bayesian network models for coastal flood wave sinks.

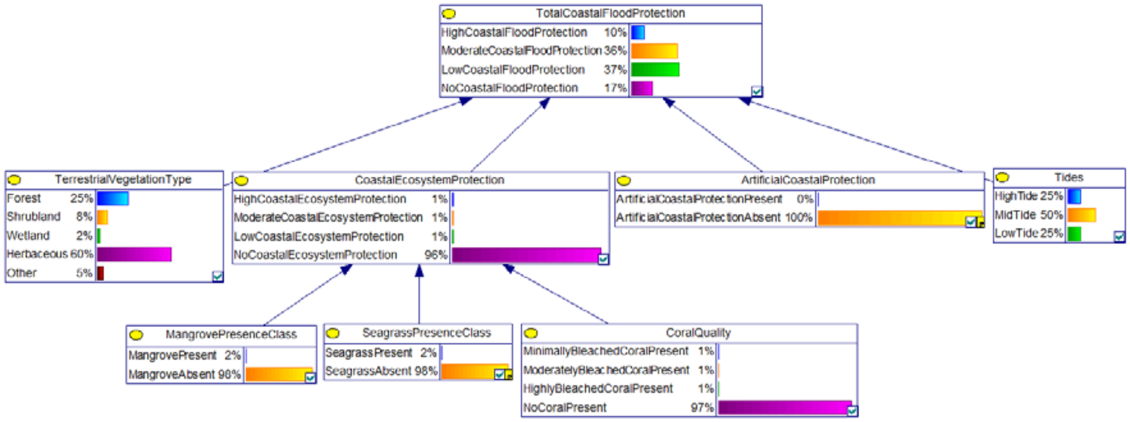


Table 6.3: Datasets used for the coastal flood wave sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Artificial coastal protection	Madagascar	FTM	Madagascar	Unknown	n/a
Coral reef bleaching	Madagascar	UNEP-WCMC & ReefBase	Global	Unknown	2003
Mangroves	Madagascar	FTM	Madagascar	Unknown	Mid-1990s
Seagrass	Madagascar	UNEP-WCMC	Global	Unknown	2005
Terrestrial vegetation type	Madagascar	FTM	Madagascar	Unknown	Mid-1990s

6.4 Incorporating coastal flood wind sink models

As a force capable of damage to life and property, wind acts differently than waves, with wind typically slowed by tall and/or dense vegetation (Raupach and Thom 1981, Day et al. 2007). While we do not currently include wind movement and mitigation in the coastal storm protection model, the value of vegetation for wind mitigation could be accommodated using a Bayesian network or other process model. In developing a Bayesian network model that could be used in future versions of ARIES, we assumed vegetation height and percent tree canopy cover to be determinants of a coastal wind sink model. We further assume that vegetation height acts as a stronger influence than percent tree canopy cover on overall wind mitigation (Figure 6.3). In the absence of data, prior probabilities for vegetation height are currently uninformed, while priors for percent tree canopy cover were derived from a review of the corresponding dataset. We discretized vegetation cover using equal intervals and have not yet quantitatively discretized vegetation height, owing to a lack of data. We set the highest and lowest sink values in the contingent probability table under conditions of tall, dense tree canopies and short, sparse tree canopies, respectively.

Figure 6.3: Bayesian network models for coastal flood wind sinks.

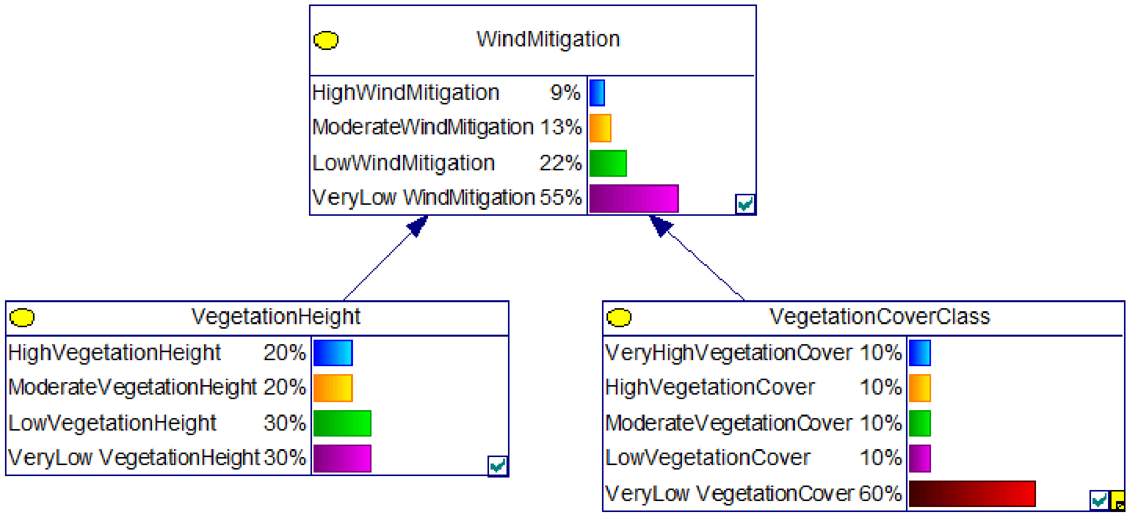


Table 6.4: Datasets used for the coastal flood wind sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Tree canopy cover	Madagascar	GLCF / UMD	Global	1 km x 1 km	2000
Vegetation height	Madagascar	NASA-JPL	Not yet obtained		

6.5 Coastal flood use models

To map coastal flood use, we identify areas with human life and economic value (e.g., housing and other infrastructure) at risk. Since spatial data exist for population and economic assets at risk (Table 6.5), no model is needed. In parts of the world where coastal flood zones have been mapped, these data could be combined with layers of population density, agricultural zones, and housing, infrastructure, or other built capital to more precisely map the beneficiaries of coastal flood regulation.

Table 6.5: Datasets used for the coastal flood use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Economic value at risk	Madagascar	CIESIN / Columbia Univ.	Global	2.5 minute^2	1981-2000
Lives at risk	Madagascar	CIESIN / Columbia Univ.	Global	2.5 minute^2	1981-2000

6.6 Coastal flood flow models

To map the flow of coastal flood waters in ARIES, we begin by mapping source values for a storm surge located 100 km offshore, using wind speed, atmospheric pressure, and ocean depth data for each historical storm of interest as described in Section 6.2. We start the storm 100 km offshore in order to fully attribute storm surge mitigation value to coastal ecosystems (coral reefs, seagrass, and mangroves) while minimizing the distance the model must move each storm across open ocean. We set a storm swath width of 100 km, centered on the historical storm track line, meaning that we project a storm surge extending perpendicular to the storm track for 50 km on each side. Costanza et al. (2008) used an analogous but somewhat simpler approach to modeling the value of storm protection by coastal wetlands on the U.S. Gulf and Atlantic coasts. In their study, they also used a 100 km wide storm swath, and moved the storm 100 km inland to estimate impacts to people and the ecosystem services provided by coastal wetlands that can reduce the storm surge. While the width of the swath might be expected to be related to the storm wind speed, among other variables, Tropical Storms Daisy, Geralda, and Litanne had roughly the same wind speed. We thus use an equal width swath for all three storms, but can easily tune this model parameter to be smaller or larger in cases where different storm tracks are used and the relationship between storm size and surge width is well understood.

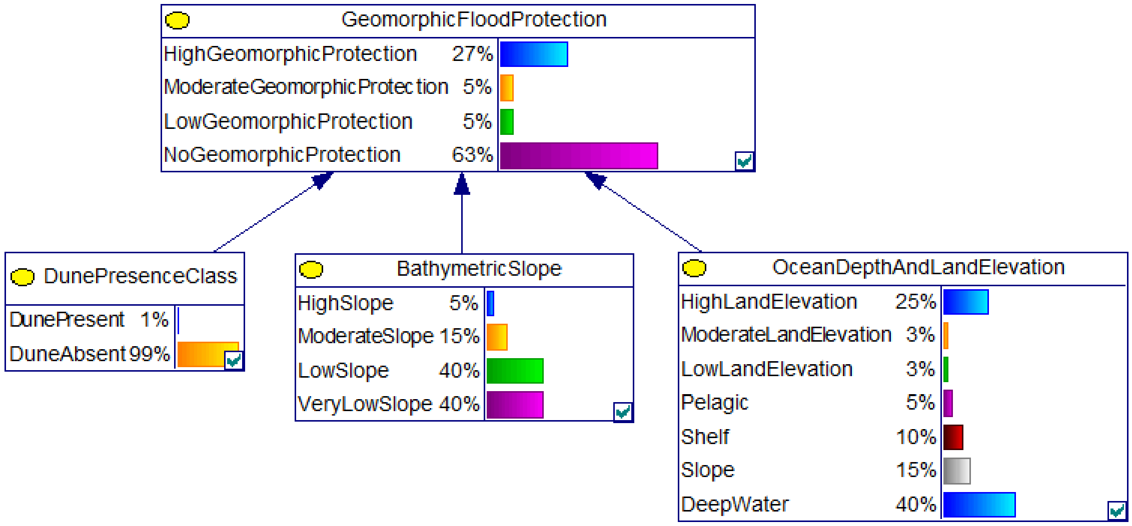
We next move the surge toward land along the historical storm track[17]. The surge is amplified (over water) or diminished (over land and water) based on the area's geomorphology. The ecosystem service is calculated as the mitigation of the storm surge by coastal and terrestrial ecosystems (as described in Section 6.3), as opposed to geomorphic features. The influence of geomorphic features on the storm surge is modeled using the Bayesian network and data described below (Figure 6.4, Table 6.6). We set geomorphic flood protection as a function of slope (above or below-water), sea floor depth or terrestrial elevation, and the presence of dunes. We discretized slope using category breakpoints from other ARIES models (with breakpoints at 1.2, 4.6, and 16.7 degrees) and depth using breakpoints of 2000, 200, 60, and 0 meters below sea level and 5 and 10 meters above sea level. We set priors based on reviews of each corresponding dataset. We set values in the contingent probability table for geomorphic flood protection as very low in all deepwater and slope regions, protection as higher in shallowly sloped pelagic and shelf areas (as steep slopes cause more wave build-up), and set the greatest level of protection in regions above sea level with high slope (particularly in the presence of dunes).

Table 6.6: Datasets used for the coastal flood flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Bathymetry and elevation	Madagascar	NASA ETOPO1	Global	1 arc-minute^2	Unknown
Dune presence	Madagascar	FTM	Madagascar	Unknown	Unknown
Slope (incl. bathymetric slope)	Madagascar	Derived from NASA ETOPO1	Global	1 arc-minute^2	Unknown

[17] Future improvements to the coastal flood flow model will include use of a wave run-up model, which will greatly improve the realism of the flow model results.

Figure 6.4: Bayesian network model for coastal flood flows.



Linking the source, sink, and use data with the flow model, we estimate the following indicators for coastal flood regulation flows. Related flow concepts are defined for sources, sinks, beneficiaries, and flows of waves and wind, which have different underlying factors that influence their movement, mitigation, and damage received or avoided[18]:

1. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Coastal wave source & Coastal wind source: The height of coastal waves or wind generated at a certain part of the land or seascape.
    b. Potential wave mitigation & Potential wind mitigation: All areas capable of reducing wind and wave energy.
    c. Potentially wave vulnerable populations & Potentially wind vulnerable populations: Any areas where people or economically valuable assets are located in flood zones.
2. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* (areas that mitigate wind or waves) – i.e., benefits in the absence of coastal flood regulation. The possible values represent the maximum delivery of coastal wind or wave damage based on the theoretical source value.
    a. Potentially damaging wave flow & Potentially damaging wind flow: The flow route of wind or waves across the land or seascape in the absence of sinks.
    b. Potentially damaging wave source & Potentially damaging wind source: Wind or waves capable of harming people or damaging property when accounting for flow paths but not sinks.

[18] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

   c. Potential flood damage received & Potential wind damage received: People and property receiving damage when accounting for sources of wind or waves and their flow paths but not accounting for the action of sinks that reduce potential damage from wind or waves.
3. Actual flow, source, sink, and use. Actual wind or wave mitigation benefits provided, received, and mitigated *with a full accounting for source, sink, and use values and flows*.
   a. Actual wave flow & Actual wind flow: Wind or wave flows when accounting for their routing and sinks that deplete their magnitude.
   b. Flood damaging wave source & Flood damaging wind source: Wind or waves that actually harm people or damage property when accounting for flow paths and sinks.
   c. Utilized wave mitigation & Utilized wind mitigation: Sinks that actively reduce wind or waves, providing the benefit of reduced flood or wind damage for people.
   d. Flood damage received & Wind damage received: Actual damage received by people and property when accounting for sources of wind or waves and their sinks and flow paths.
4. Inaccessible source and sink. Theoretical values minus possible values; accounts for sources that do not cause wind or wave damage or sinks that do not mitigate wind or waves due to a lack of flow connections.
   a. Benign wave source & Benign wind source: Wind or waves that do not have people or economically valuable assets lying in their path.
   b. Unutilized wave mitigation & Unutilized wind mitigation: Sinks capable of reducing wind or wave flows but lacking associated human beneficiaries who value this protection, or lacking wind or waves to mitigate.
5. Blocked flow, source, and use. Flows, source, or use values mitigated by sinks. .
   a. Absorbed wave flow & Absorbed wind flow: Flood or wind flows that are absorbed by sinks prior to reaching human beneficiaries of wind or wave regulation.
   b. Mitigated wave source & Mitigated wind source: The portion of coastal wind and waves that are reduced by the action of wind and wave sinks.
   c. Flood mitigation benefits accrued & Wind mitigation benefits accrued: People or economically valuable assets who are spared from coastal flood or wind damage due to the flood or wind regulation activity of sinks.

6.7 Caveats and directions for future research

The current ARIES coastal flood regulation module is intended as a proof-of-concept model to link ecosystems, their human beneficiaries, and ecosystem service flows through space. Substantial future work could improve these models' scientific quality and value for decision making. While the initial models demonstrate storm mitigation and impacts from three well-documented historical storms in Madagascar, the models could easily be extended to model historical storm impacts elsewhere in Madagascar, in other parts of the world, or to impacts from tsunamis or other waves using wave information in the coastal flood regulation source model. However, the additional interface elements needed to correctly parameterize and use such extended models would necessarily make the resulting toolkit harder to understand and use, making it suitable only to specialist users. Thereby, we expect additional work on this and other configuration-intensive models (such as flood regulation) to happen in the context of more specialized tools directed more specifically to governmental agencies.

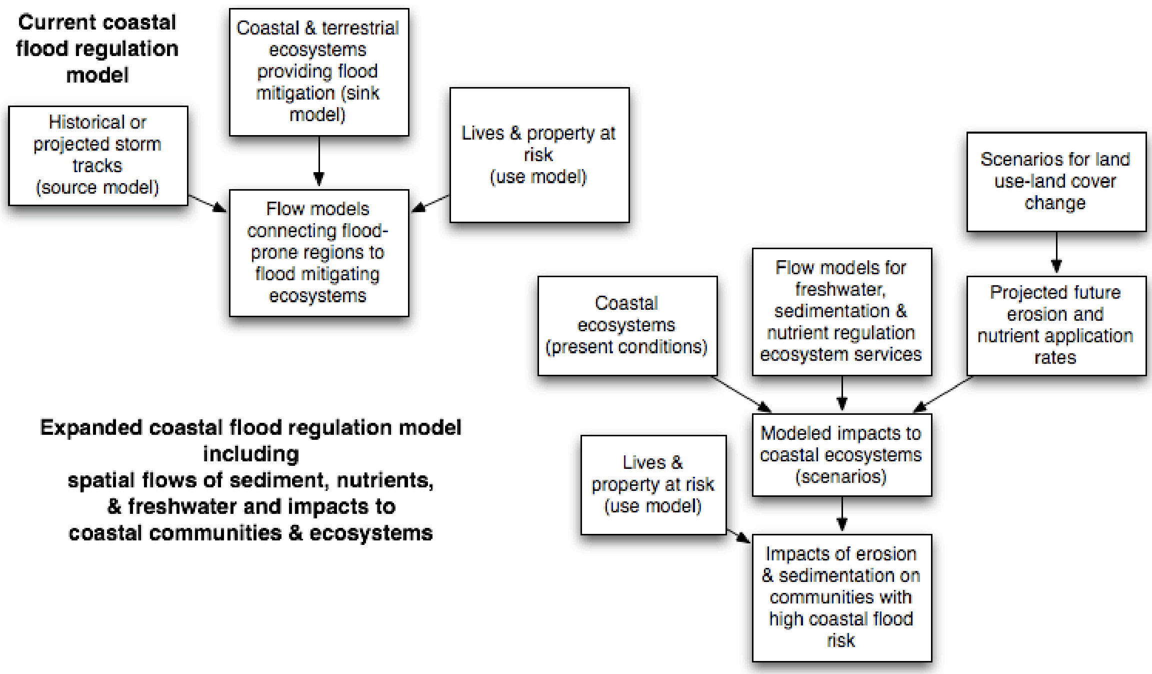
In developing such extended models, care should be taken (when modeling historical storms) not to use data on coastal ecosystems and development levels (generally from around the year 2000) with historical storms where coastal ecosystem and development conditions were quite different (i.e., modeling a storm from 1980 using other spatial data from the year 2000).

A key constraint of our models is that we do not include natural effects of water temperature or other factors in strengthening or weakening a storm. The weakening of a storm that passes over colder water should not be mistakenly attributed to an ecosystem service provided by coastal systems, and the effects of the two interacting factors on reducing storm surge height could be better explored using models of specific storm events.

A key step in improving the models' validity and value in decision-making lies in incorporating existing, accepted coastal process models where appropriate. Such models could be better used to predict storm surges, wave run-up, and the effects of geomorphology and ecosystems on wave size and inflicted damage. Cochard et al. (2008) provide a recent review of coastal storm modeling in the context of the 2004 Indian Ocean tsunami. The models in this review and elsewhere vary widely in their spatial and temporal scales of operation, the types of coastal ecosystems modeled, and in being derived from laboratory experiments, field experiments, or spatial data. Experimental evidence to quantify the mitigation value is often lacking, as is an understanding of the spatial scales and contexts under which vegetation provides wave mitigation value, which has important implications for coastal policy (Feagin et al. 2010). Proper model integration in ARIES will involve determining which models can operate at relevant spatial scales while using available spatial data. Once process models that are compatible with the ARIES modeling framework have been identified, we will specify the geographic and spatial context under which to select each model, so that the system can intelligently apply the correct model under the proper circumstances.

Like the ARIES fisheries models, another future research direction for the coastal flood regulation models would link spatial flow models for sediment (Chapter 7), freshwater (Chapter 8) and nutrients (ARIES models in development, using sources including Potter et al. 2010) between terrestrial and marine environments with the goal of better understanding the impact of terrestrial land management on coastal and marine ecosystem services. In conjunction with local experts, we could model how these flows affect the ability of mangrove, seagrass, coral, and coastal wetland ecosystems to provide coastal flood regulation (Figure 6.5). Although the importance of such flows are understood (McCulloch et al. 2003, Fabricus 2005, Diaz and Rosenberg 2008, Silvestri and Kershaw 2010), models to spatially link ecosystem services supply and demand across linked terrestrial and marine systems have thus far been lacking. The ARIES flow modeling framework is a feasible way to address this gap in modeling, mapping, and valuing marine ecosystem services.

Figure 6.5: Current coastal flood regulation models and models accounting for spatial flows of sediment, nutrients, and freshwater from terrestrial to coastal and nearshore marine environments.



6.8 Additional contributors

UNEP-WCMC provided funding for development of marine ecosystem services models in ARIES and supplied other marine biotic datasets. Liz Selig and Carmen Lacambra provided helpful guidance in obtaining additional global marine datasets.

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7. Sediment regulation

7.1 Introduction

Erosion and sedimentation are major global problems that impose a high cost on the functioning of ecosystems and ecosystem service delivery (Yang et al. 2003). Excessive erosion and sedimentation have negative impacts on agriculture, water supply, electric power generation, and navigation, as well as on coastal and nearshore marine ecosystems and the services they provide (see Chapters 5 and 6). At the same time, natural sediment delivery can be a beneficial process in some contexts, and the disruption of natural sediment delivery can also have negative impacts. For example, reduced sediment delivery to deltas can lead to loss of coastal wetlands and the critical services they provide (Costanza et al. 2006, Day et al. 2007).

We model *sources* of waterborne sediment, *sink* regions where sediment deposition occurs, and *users* who either value or are harmed by the delivery of sediment or the presence of excessively turbid waterways (Table 7.1). Sediment regulation can thus be classified as either a provisioning or preventive service whose benefits are rival and are measured (in tons of sediment) at the watershed scale. Running the sediment flow model allows the user to map spatial connections between sources of sediment, areas that promote sediment deposition, and users that benefit from or are harmed by sediment delivery. Additionally, the sediment models can quantify the erosion control benefits that vegetation provides *in situ* for farmers or other beneficiaries who want to avoid soil loss from their lands. This benefit can be estimated by simply running the erosion source model in the present state and comparing results with either no vegetation or a different vegetation type, without needing to run a flow model.

Table 7.1: Summary characteristics of the ARIES sediment regulation models.

Service	Sediment regulation
Benefit type	Provisioning or Preventive
Medium/units	Sediment (tons)
Scale	Watershed
Movement	Hydrologic flow
Decay	None
Rival?	Rival
Source	Landscapes along waterways
Sink	Riparian zones where deposition occurs
Use	Areas where sedimentation is desirable, areas where sedimentation is undesirable, areas where excessively turbid water is undesirable

Spatial modeling of sedimentation has frequently relied on models such as the Universal Soil Loss Equation (USLE, Wischmeier and Smith 1978), Revised Universal Soil Loss Equation (RUSLE, Renard et al. 1996), or Spatially Explicit Delivery Model (SEDMOD, Fraser 1999). USLE and RUSLE multiply five factors – rainfall runoff erosivity, soil erodibility, slope steepness and length, cover management, and conservation practice –

to estimate soil loss over a given spatial and temporal extent (e.g., tons sediment/ha-yr). Spatial data for soil loss and RUSLE factors are available globally as a 0.5x0.5 degree raster dataset (Yang et al. 2003) and for the western United States as a vector dataset by 8-digit watershed Hydrologic Unit Code (HUC) (USEPA 2010). The use of these deterministic models, where appropriate, can make probabilistic modeling of sedimentation unnecessary. However, USLE and RUSLE have several well-known limitations (Roose 1996): 1) they apply only to sheet erosion versus linear or mass erosion; 2) they have only been tested in regions with 1-20% slopes and are inappropriate for areas with steeper slopes or young mountains where greater erosion is possible; 3) energy-rainfall relationships have been best tested for the U.S. Great Plains, meaning that locally appropriate rainfall runoff erosivity factors must be carefully developed and applied; 4) data are not valid for individual storms but only for averages; a Modified USLE (Williams 1975) must be used to model sediment loads produced by a single storm; and 5) the equations simplify interactions between factors to attempt to isolate the relative effects of each. Thus USLE/RUSLE are likely most appropriate in level to moderately hilly landscapes with similar physical and ecological characteristics as the central United States; their application is harder to justify in other parts of the world. By using the ARIES internal rule base to select the appropriate models, we can apply the RUSLE model on relatively level landscapes and regionally appropriate ad hoc erosion models for steeper slopes or areas where RUSLE is known to be inadequate.

Prototype sediment regulation models have been developed for the **Dominican Republic, Madagascar,** and **Western Washington**. These models are intended to be representative of sediment regulation dynamics over broad areas with similar biophysical contexts. For instance, the Western Washington models are designed to be applicable to Oregon, Washington, and British Columbia coastal forests, including the Cascade and Coast Ranges. Likewise, the Dominican Republic models are designed to be applicable to all of Hispaniola and potentially the Greater and Lesser Antilles. In addition to these regionally specific models, a generalized global sediment regulation model is planned for a future release of ARIES. This model will use global datasets and provide coarser resolution model outputs; it will automatically be used in the absence of regionally-specific ARIES case studies.

Sedimentation is particularly problematic in Madagascar, where high rates of deforestation (Harper et al. 2007) and low natural rates of succession have led to high levels of erosion (Wendland et al. 2010). Excess sedimentation can be particularly damaging to rice fields (Carret and Loyer 2003). In Western Washington, sedimentation is important both for providing beneficial coarse sediments (e.g., gravels) for salmon spawning and for avoided habitat siltation for salmon and other economically beneficial species (WRIA 9 Steering Committee 2005, Steel et al. 2008). Sediment delivery can also have negative impacts on drinking water intakes, recreation areas, and hydroelectric power generation. In the Dominican Republic, sediment loss from intensive agricultural practices is impacting hydroelectric production in parts of the country, impeding development efforts and reducing human well-being (Siegel and Alwang 2004, IDIAF

2006). Because of the strong connections, both demonstrated and hypothesized, between areas of high biodiversity, carbon storage, and erosion control, researchers in many regions are exploring the overlap between areas that provide these services. A major goal of this work is to support development of economic incentives for forestry and agricultural practices that reduce erosion while improving carbon sequestration and storage, biodiversity, and other ecosystem services.

Other ecosystem services researchers have also attempted to map sedimentation values. While Eade and Moran (1996) and Tallis et al. (2011) modeled sedimentation using the USLE, Egoh et al. (2008) and Wendland et al. (2010) used other proxy data. Egoh et al. combined local estimates of soil erosion potential with expert rankings of the ability of tree canopy cover to prevent erosion. Drawing on Quinton et al.'s (1997) work in semiarid Spain, Egoh et al. note that soil erosion is slightly reduced at about 30% tree canopy cover and significantly reduced at about 70% tree canopy cover. By combining areas of high erosion potential and <30, >30, or >70% tree canopy cover, they estimate spatially explicit values of vegetation for erosion control. Wendland et al., noting the established link between forest cover and sedimentation for Madagascar (Albietz 2007), map upstream forest cover from population centers, irrigated rice fields, and mangroves – areas that benefit from sediment-free water. We drew on these studies in developing our sediment regulation models, then extended these approaches by explicitly accounting for the spatial dynamics of sediment regulation.

7.2 Sediment regulation source models

Based on the published contributions and models mentioned, along with stakeholder input from case study partners, we designed probabilistic models of sedimentation to complement deterministic (RUSLE) models. We set annual sediment loss as a function of runoff, "Vegetative maturity" (i.e., vegetation characteristics that affect runoff), and soil erodibility (Figure 7.1). We set vegetative maturity as a function of vegetation type and percent tree canopy cover for all case studies, and included successional stage as a further determinant of vegetative maturity for the Dominican Republic and Western Washington. We set runoff as a function of annual precipitation and tropical storm probability for the Dominican Republic and Madagascar. For Western Washington, we used annual precipitation as a direct influence on annual sediment source value.[19] Finally, we set soil erodibility in all models as a function of hydrologic soils group, soil texture, and slope. In Western Washington, we added slope stability as a fourth influence on soil erodiblity, since these data were available for the region. We discretized percent tree canopy cover using Quinton et al.'s (1997) breakpoints (30% and 70% canopy cover) and used Jenks Natural Breaks to discretize all other continuous variables. We estimated priors based on the individual datasets for the Dominican Republic, Madagascar, and Western Washington.

[19] Runoff could also be calculated as the output of the Curve Number method (SCS 1972) or a more complex deterministic runoff model.

For Madagascar, we completed the contingent probability table for vegetative maturity by "pegging the corners" (Marcot et al. 2006) for highest vegetative maturity under conditions of very high tree canopy cover and forest/wetland vegetation type, the lowest vegetative maturity for very low tree canopy cover and cropland/developed vegetation type, and interpolated intermediate values. We set forests and wetlands as having the highest maturity, followed by degraded forests, savannas, and cropland/developed land. For the Dominican Republic and Western Washington, we set conditional priors for vegetative maturity by ranking the order of importance of child nodes, with vegetation type as the most important and successional stage and percent tree canopy cover as progressively less important. We set the greatest vegetative maturity for Western Washington for forests and wetlands, followed by shrubland and grasslands, followed by cropland, barren, and developed. For the Dominican Republic, we set the greatest vegetative maturity for water, wetlands, and mangroves, followed by forests and shrubland, followed by shade coffee and cocoa, followed by intensive cropland and pasture, and finally by urban and roads. We set the contingent probability table for runoff in Madagascar and the Dominican Republic by pegging the corners for high precipitation and/or high tropical storm probability leading to the greatest runoff and vice versa, and interpolated intermediate values. We set values in the contingent probability table for erodibility as greatest on steep, coarse soils with high infiltration potential (hydrologic soils group A) and erodibility (for Western Washington), and vice versa, and interpolated intermediate values. We set the top node, the annual sediment erosion source value, at zero for all soils with very low erodibility, set it at its highest on very erodible soils with very high runoff and no vegetative maturity, and interpolated intermediate values.

Figures 7.1: Bayesian network models for sediment sources.

Figure 7.1.1: Sediment sources for the Dominican Republic.

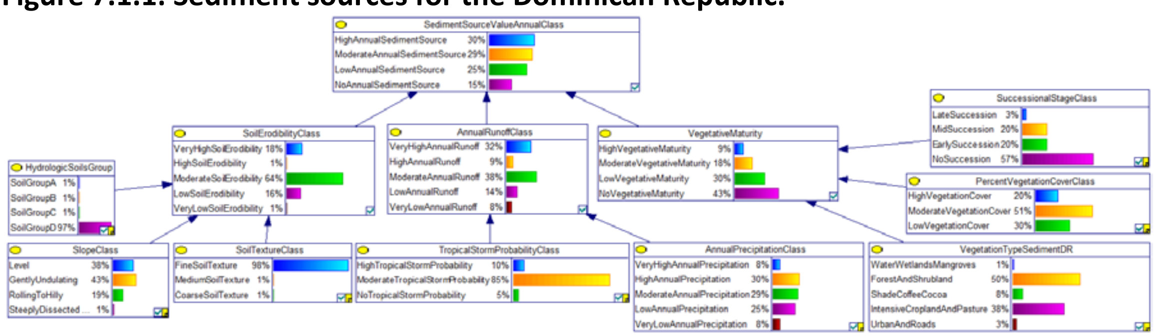

Figure 7.1.2: Sediment sources for Madagascar.

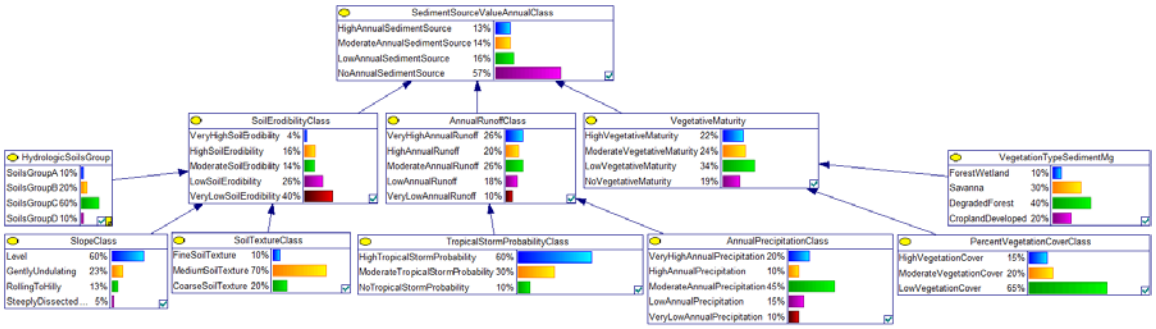

Figure 7.1.3: Sediment sources for Western Washington.

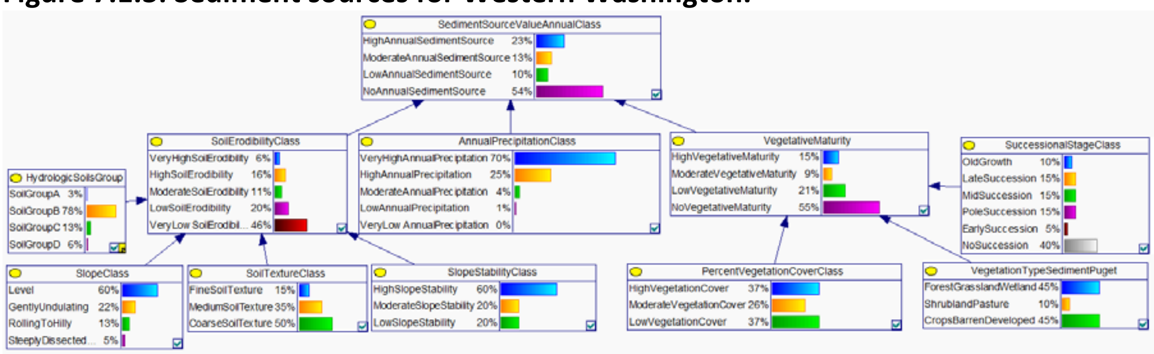

Table 7.2: Datasets used for the sediment source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Average annual precipitation	Dominican Republic, Madagascar	WorldClim	Global	30 arc-seconds[2]	1950-2000
	Western Washington	PRISM / OSU	United States	800 m x 800 m	1971-2000
Average annual runoff	Dominican Republic, Madagascar	SAGE / UW Mad	Global	0.5° x 0.5°	1955-1990
Average annual soil loss (RUSLE)	Dominican Republic, Madagascar	Yang et al. (2003)	Global	0.5° x 0.5°	2000
	Western Washington	U.S. EPA (2010)	Western U.S.	Unknown	Not available
Hydrologic soils group	Dominican Republic, Madagascar	Gately (2008)	Global	0.083° x 0.083°	n/a
	Western Washington	SSURGO soil data	Western Washington	Unknown	n/a
RUSLE factors	Dominican Republic, Madagascar	Yang et al. (2003)	Global	0.5° x 0.5°	2000

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
	Western Washington	U.S. EPA (2010)	Western U.S.	Unknown	Not available
Slope	All	Derived from SRTM	Global	90 m x 90 m	n/a
Slope stability	Western Washington	WA DNR	Washington State	30 m x 30 m	n/a
Soil texture	Dominican Republic, Madagascar	FAO Soils	Global	0.083° x 0.083°	n/a
	Western Washington	SSURGO soil data	Western Washington	Unknown	n/a
Successional stage	Western Washington	BLM/Interagency Vegetation Mapping Project	Western Washington & Oregon	25 m x 25 m	1996
Tree canopy cover	Dominican Republic, Madagascar	GLCF / UMD	Global	1 km x 1 km	2000
	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Tropical storm probability	Dominican Republic, Madagascar	CIESIN / Columbia Univ.	Global	2.5 minute^2	1981-2000
Vegetation type	Dominican Republic	Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ)	Northwest Dominican Republic	30 m x 30 m	Not available
	Madagascar	FTM	Madagascar	Unknown	Mid-1990s
	Western Washington	NLCD 2001	United States	30 m x 30 m	2001

7.3 Sediment regulation sink models

Erosion sinks are areas where sediment accumulates as it flows through a watershed. We only consider the deposition of sediment in floodplains and reservoirs, as opposed to sediment carried and then deposited by overland flow before reaching a stream. We define sediment deposition ("Annual sediment sink", measured in tons of sediment per year) to be a function of three stream and floodplain variables – stream gradient, floodplain tree canopy cover, and floodplain width – plus dams that cause sediment deposition in reservoirs (Figure 7.2)[20].

The erosion sink models for the three case studies are identical, except that we set prior probabilities for the presence of infrastructure much lower for the Dominican Republic and Madagascar than in more developed settings (e.g., Western Washington). We

[20] Bayesian network models for sediment source and sink models can be downloaded from http://ariesonline.org/modules/soilspecs.html.

discretized floodplain tree canopy cover using Jenks Natural Breaks and stream gradient using breakpoints of 0-2% for low gradient streams, 2-5% for moderate gradient streams, and >5% for high gradient streams. We based priors for all nodes on relevant spatial data for each case study. We set the contingent probability table for annual sediment sink by assuming deposition to be greatest in low-gradient streams with wide floodplains and high levels of tree canopy cover, and lowest under the opposite conditions, with intermediate values interpolated. The presence of reservoirs, which create slack water flow conditions, leads to high deposition levels in all circumstances.

Figures 7.2: Bayesian network models for sediment sinks.

Figure 7.2.1: Sediment sinks for the Dominican Republic.

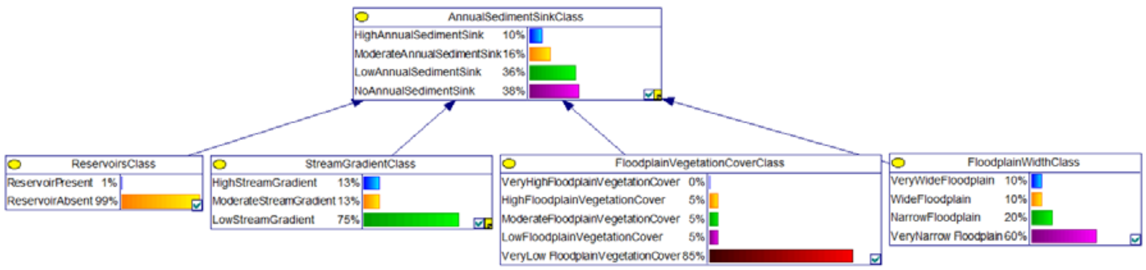

Figure 7.2.2: Sediment sinks for Madagascar.

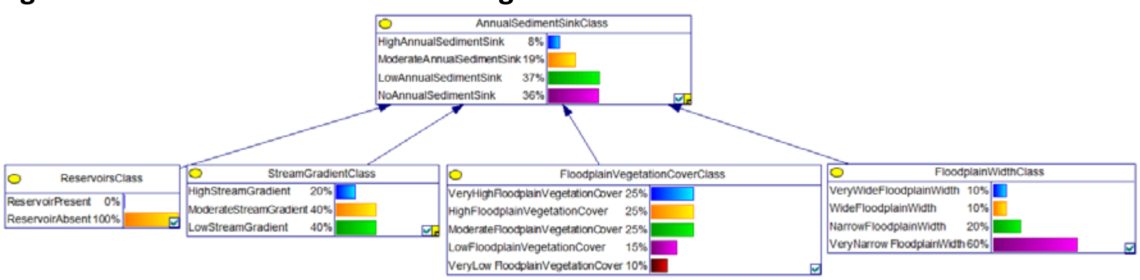

Figure 7.2.3: Sediment sinks for Western Washington.

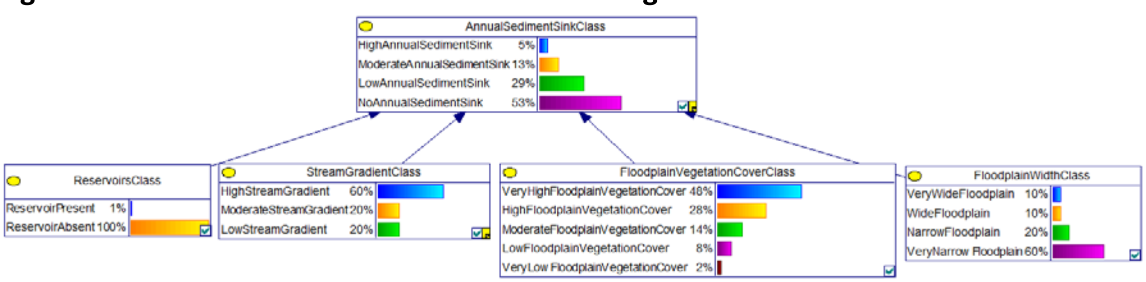

Table 7.3: Datasets used for the sediment sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Floodplain tree canopy cover	Dominican Republic,	GLCF / UMD; Dartmouth Flood	Global	1 km x 1 km	2000

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
	Madagascar	Observatory			
	Western Washington	NLCD 2001, FEMA Q3 flood data	United States	30 m x 30 m	2001
Floodplain width	Western Washington	FEMA & WA DOE	Western Washington	Unknown	Unknown
Reservoirs	Dominican Republic	Digitized based on Global Database of Dams	Dominican Republic	Unknown	2010
	Madagascar	FTM	Madagascar	Unknown	Unknown
	Western Washington	ORNL	United States	Unknown	2005
Stream gradient	Dominican Republic	Digital Chart of the World hydrography + SRTM slope	Dominican Republic	90 m x 90 m	2000
	Madagascar	FTM + SRTM slope	Madagascar	90 m x 90 m	2000
	Western Washington	WA DNR + SRTM slope	Western Washington	Unknown	2000

7.4 Sediment regulation use models

While not explicitly presenting ecosystem service flow model results, both Tallis et al. (2011) and Wendland et al. (2010) incorporate beneficiaries in their sedimentation models. Tallis et al. map the locations of reservoirs where avoided sedimentation is a benefit, while Wendland et al. map human population density (for drinking water), mangroves (for avoided sedimentation of fish habitat), and rice fields (for avoided crop damage). Mapping these beneficiaries can often be done with a single spatial data layer or simple GIS operations rather than Bayesian networks. For instance, we can map: 1) the location of reservoirs, drinking water intakes, and navigation infrastructure (where high turbidity or excess sedimentation are undesirable), 2) floodplain farmers (where sedimentation may be beneficial or undesirable, using a simple overlay of floodplains and farmland), or 3) erosion-prone farmers (where erosion is undesirable, by simply intersecting erosion sources and farmland).

Table 7.4: Datasets used for the sediment use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Farmland	Dominican Republic	Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ)	Northwest Dominican Republic	30 m x 30 m	Unknown
	Madagascar	FTM	Madagascar	Unknown	Mid-1990s
	Western Washington	NLCD 2001	United States	30 m x 30 m	2001
Floodplains	Dominican Republic, Madagascar	Dartmouth Flood Observatory	Greater Antilles, Madagascar	250 m x 250 m	2003-2010
	Western Washington	FEMA & WA DOE	Western Washington	Unknown	Unknown
Reservoirs	Dominican Republic	Digitized from Global Database of Dams	Northwest Dominican Republic	Unknown	2010
	Madagascar	FTM	Madagascar	Unknown	Unknown
	Western Washington	ORNL	United States	Unknown	2005

7.5 Sediment regulation flow models

The source and sink models estimate the annual quantity (in tons or kg of sediment per hectare) of sediment that erode from one part of the landscape (in the source model) and are deposited elsewhere (in the sink model). For the preventive service case, use models map the location of potential beneficiaries of avoided 1) detrimental sedimentation, 2) detrimental erosion, or 3) excessively turbid surface water. In the provisioning service case, use models identify regions that could benefit from sediment deposition. Flow models are not necessary to calculate the benefit of avoided erosion: we simply estimate the erosion source value with and without vegetation in order to determine the effects of vegetation on reduced erosion. For the other beneficiary classes, the flow models describe the amount of beneficial or detrimental sediment delivered or the amount of sediment carried in flowing water (i.e., turbidity). Since we do not model wind-based erosion, sediment flows are modeled using a relatively simple hydrologic model. We use hydrography and SRTM elevation data to derive flow direction to route water across the landscape and through waterways (Table 7.5). During flood events, sediment can be deposited in floodplains, thus floodplain extents and the presence of levees are used in the water and sediment routing models. Finally, dams are included in the flow models, because essentially all sediment will be deposited into a reservoir as the speed of flowing water slows dramatically when a stream empties into a reservoir. Whenever sediment is deposited on the landscape, its effect, whether beneficial or detrimental, is assigned to any users in the same spatial location as the sink. If no human users (people or assets) are present at the sink site, then no service is accrued by sediment deposition in that location.

While our approach is an admittedly simplistic way to move water and water-related ecosystem service carriers (e.g., drinking water, flood water, suspended sediment, dissolved nutrients), it has the benefit of being applicable at relatively coarse spatial scales and at any location on Earth. Future work on ARIES will seek to incorporate appropriate existing hydrologic models to route water and water-related ecosystem service carriers across the landscape at variable spatial scales and under variable environmental conditions (e.g., using different models at large vs. small spatial scales and in arid versus humid ecological systems).

Linking the source, sink, and use data with the flow model, we estimate the following indicators for sediment regulation flows. Related flow concepts are defined for the different beneficiary classes of sediment regulation – those benefitting from sediment deposition, avoided detrimental sedimentation, avoided detrimental erosion, and avoided excessively turbid surface water[21]:

1. Theoretical source, sink, and use. These are the values initially estimated by the models *without accounting for flows*.
    a. Maximum sediment source: Locations of areas capable of providing sediment (i.e., areas of erosion or sources of sediment) to downstream areas.
    b. Maximum potential deposition: Areas capable of accumulating waterborne sediment (e.g., flat depositional areas, reservoirs).
    c. Potential sediment deposition beneficiaries, Potential reduced sediment deposition beneficiaries & Potential reduced turbidity beneficiaries: Beneficiaries who could receive beneficial sedimentation or reduced detrimental sedimentation.
2. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* (areas that allow sediment deposition) – i.e., benefits in the absence of sediment regulation. The possible values represent the maximum achievable sediment delivery based on the theoretical source value.
    a. Possible sediment flow: The downstream movement of sediment when not accounting for sediment sinks.
    b. Possible sediment source: Areas which, based on the flow pattern of sediment but disregarding the effects of sinks, provide sediment which reaches downstream users who either benefit from or are damaged by sediment delivery.
    c. Possible sediment deposition beneficiaries, Possible reduced sediment deposition beneficiaries & Possible reduced turbidity beneficiaries: Beneficiaries who receive beneficial or detrimental sedimentation when accounting for flow paths but not sinks.

[21] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

3. Actual flow, source, sink, and use. Actual sediment regulation benefits provided, received, and captured *with a full accounting for source, sink, and use values and flows*.
    a. Actual sediment flow: The downstream movement of sediment that accounts for sediment sinks.
    b. Actual sediment source: Areas which, based on the flow pattern of sediment and accounting for sinks, provide sediment to downstream users who either benefit from or are damaged by sediment delivery.
    c. Utilized deposition: Depositional areas that undergo sedimentation, receiving upstream sediment and actively performing a sediment trapping function.
    d. Actual sediment deposition beneficiaries, Actual reduced sediment deposition beneficiaries & Actual reduced turbidity beneficiaries: Beneficiaries who receive beneficial or detrimental sedimentation when accounting for flow paths and sinks.
4. Inaccessible source, sink, and use. Theoretical values minus possible values; accounts for sources that do not provide sediment, sinks that do not capture sediment, and beneficiaries that cannot use sediment regulation due to a lack of flow connections.
    a. Unutilized sediment source: Areas which, based on the flow pattern of sediment, do not provide sediment to downstream users who either benefit from or are damaged by sediment delivery.
    b. Unutilized deposition: Potential depositional areas that do not undergo sedimentation, as they lack a connection to an upstream sediment source of sufficient size to undergo deposition at the sink.
    c. Inaccessible sediment deposition beneficiaries: Potential beneficiaries who lack connection to an upstream source of sediment.
5. Blocked flow, source, and use. Source, flow, or use values mitigated or degraded by sinks.
    a. Absorbed sediment flow: The flow of sediment that does not reach downstream beneficiaries who benefit from either avoided detrimental sedimentation or beneficial sediment delivery.
    b. Negated sediment source: Areas which, due to deposition occurring in downstream sink areas, do not provide sediment to downstream users who either benefit from or are damaged by sediment delivery.
    c. Lost valuable sediment & Blocked harmful sediment; Reduced turbidity: Beneficiaries who either receive less beneficial or detrimental sediment as a result of sinks.

Table 7.5: Datasets used for the sediment flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Dams	Dominican Republic	Digitized from Global Database of Dams	Northwest Dominican Republic	Unknown	2010
	Madagascar	FTM	Madagascar	Unknown	Unknown
	Western Washington	National Atlas of the United States	United States	Unknown	2006
Elevation	All	SRTM	Global	90 m x 90 m	2000
Floodplain extents	Dominican Republic, Madagascar	Dartmouth Flood Observatory	Dominican Republic, Madagascar	250 m x 250 m	Based on 2003-2010 flood data
	Western Washington	FEMA Q3 Flood Data	United States	Unknown	Varies
Hydrography (stream networks)	Dominican Republic	Digital Chart of the World	Dominican Republic	Unknown	Unknown
	Madagascar	FTM	Madagascar	Unknown	Unknown
	Western Washington	WA DNR	Washington State	Unknown	Unknown
Levees	Western Washington	County GIS offices	King, Lewis, Pierce Cos	Unknown	Varies

7.6 Caveats and directions for future research

Global and Western U.S. estimates of soil loss provided by the RUSLE data (Yang et al. 2003, USEPA 2010) represent an upper bound on sediment loss, since “RUSLE does not estimate the amount of sediment leaving a field or watershed, instead estimating soil movement at a particular site (Yang et al. 2003).”

We currently use probabilistic models of sediment deposition and transport. Where process-based models of sediment transport and deposition exist, they should be incorporated into the ARIES system if they are capable of running at corresponding spatial and temporal scales while using publicly available spatial data. We currently model annual sediment loss, deposition, and transport. Future ARIES models could also potentially incorporate event-based erosion and sedimentation models. Event-based models could better account for the fact that even in wide floodplains with high tree canopy cover and low stream gradient, there will be periods of both high and low sedimentation based on streamflow conditions and sediment loads. The timing of sediment flows may matter for certain beneficiaries, such as farmers, where sedimentation may be a benign or beneficial process if it occurs outside the growing season, while it may be a highly damaging process if it occurs in the growing season. However, event-based erosion modeling is made more challenging by limitations in the spatial resolution of event-based rainfall data. This limitation, which is discussed in more detail in Section 8.6 in the water supply chapter, is more serious in arid and semiarid environments where rainfall patterns are more uneven.

Our beneficiary maps currently include only erosion and sedimentation on farmland and reservoir sedimentation. Inclusion of spatial data for drinking water sources and navigation infrastructure would enable these beneficiaries to be mapped.

Finally, sedimentation has important impacts on aquatic habitats, both beneficial and detrimental. Coarse sediment delivery can be beneficial for salmon breeding, while excess fine sediment can be highly detrimental (WRIA 9 Steering Committee 2005, Steel et al. 2008). Sediment delivery to deltas is critical to formation and maintenance of beaches and interdunal wetlands, which provide a variety of ecosystem services (Costanza et al. 2006, Day et al. 2007). Sedimentation can affect coastal fisheries (Chapter 5) and the nearshore ecosystems capable of providing coastal flood regulation (Chapter 6). Such benefits can be mapped by using data to aggregate sites where fishing could take place (lakes, rivers, coast) with key habitats for fish (coastal wetlands, salmon spawning grounds, coral reefs, seagrass beds, and mangroves) while accounting for socioeconomic variables including population density and public access to fishing. However, since sediment regulation is an "intermediate service" affecting the quality of the fishery in conjunction with other factors, we do not include fisheries as direct beneficiaries of sediment regulation. The effects of sedimentation on other ecosystem services are best modeled by using sedimentation scenarios, where current sedimentation conditions are compared to reduced or increased levels of sedimentation to illustrate these effects on delivery of other ecosystem services. In a future ARIES release, we will connect the sediment regulation models to riverine, lacustrine, coastal, and marine fisheries models. We will thus be able to map environmental flows between terrestrial and aquatic systems. Similarly, the multiple benefits provided by sediment delivery to deltas will be able to be mapped as inputs to coastal flood regulation, fisheries, and other relevant ecosystem service models.

7.7 Acknowledgements and additional contributors

Nathaly Agosto Filion and Lee Gross developed the case study for the Dominican Republic. Dave Batker, Jim Pittman, and Paula Swedeen provided data and reviewed models for the Western Washington case study. Miroslav Honzák provided data and reviewed models for the Madagascar case study.

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8. Water supply

8.1 Introduction

Freshwater supply in the form of both surface and groundwater plays a critical role in supporting human well-being. With the recognition of the role that ecosystems play in protecting the quantity, quality, and timing of freshwater flows (Sedell et al. 2000), payments for ecosystem services programs have been rapidly emerging, particularly in the developing world, to safeguard these systems for the benefit of downstream water users (Echavarria 2002, Munoz-Pina et al. 2008, Goldman 2009). To better quantify the values generated by ecosystem water supply and regulation, ARIES simulates surface and groundwater flow to connect human beneficiaries (e.g. agriculture, industry, domestic use) with their upstream water sources and sinks. Once these ecosystem dependencies are identified, the various landscape effects on water quality and quantity along these flow paths can be estimated, and the final water value to users can be assessed under different land management scenarios. In ARIES, water quantity is largely a function of topographically-based hydrologic simulation. Water quality can be modeled by accounting for flows of sediment (Chapter 7), nutrients, and pathogens (to be considered in future releases of ARIES). In this chapter, we describe models for assessing the spatial distribution of quantities of surface and groundwater.

Water supply is a complex ecosystem service to spatially model, given that groundwater and surface water are closely connected but move based on different controlling factors, that these influences on hydrology and hydrologic models differ greatly based on the spatial and temporal scale and the region of the world considered, and that available spatial data can rarely support modeling at both high spatial and temporal resolutions. Given these limitations, our initial water supply models operate at an annual scale (which is matched by available spatial data for important variables including precipitation, infiltration, snowmelt, and evapotranspiration). We currently consider only flows of surface water, though we do model the infiltration of surface water into groundwater and groundwater extraction from wells. We also use a set of generalized models to represent *sources* of surface water (precipitation, snowmelt, springs, baseflow to rivers, and incoming inter-basin water transfers), *sinks* of surface water (evapotranspiration, infiltration), *beneficiaries* or users of surface water, and the flow of surface water across the landscape (routed using SRTM elevation data, Table 8.1, Figure 8.1). The long-term intent of our modeling process is to incorporate existing hydrologic models wherever appropriate that will more realistically account for hydrologic processes. Over time this includes modeling groundwater and its movement and use. As spatial data continue to improve, it may also be possible to model water supply at finer temporal scales.

We developed local water supply models in ARIES for the **San Pedro River Watershed (Arizona and Northern Sonora, Mexico)** and **La Antigua River Watershed (Veracruz, Mexico)** due to the particularly critical nature of water in arid and semiarid environments, and in tropical environments undergoing population growth and deforestation. These models are intended to be representative of water supply dynamics in wider regions. For instance, the La Antigua models are designed to be applicable for Oaxacan-Veracruz volcanic forests and Gulf-Caribbean mangrove ecosystems, while the San Pedro models are designed for Chihuahuan and Sonoran deserts, Southwestern Sky Island and Sierra Madre Occidental pine-oak forests. In addition to these regionally specific models, a generalized global model of water supply is planned for a future release of ARIES. This model will use global datasets and provide coarser quality model outputs in the absence of regionally-specific ARIES models.

Table 8.1: Summary characteristics of the ARIES water supply models.

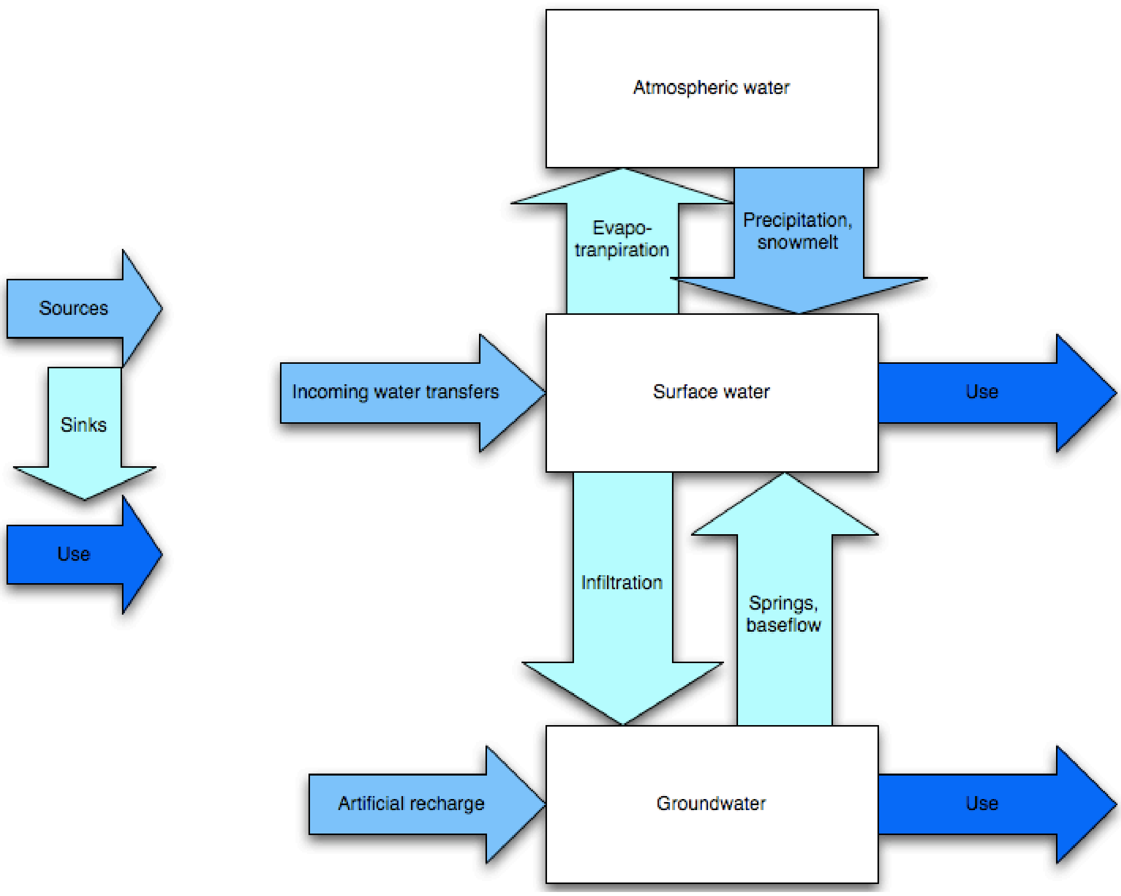
Service	Water supply
Benefit type	Provisioning
Medium/units	Surface or groundwater (mm^3/yr)
Scale	Watershed
Movement	Hydrologic flow, surface & groundwater
Decay	None
Rival?	Rival
Source	Precipitation, snowmelt, springs, baseflow, and incoming water transfers (surface water); artificial recharge and infiltration (groundwater)
Sink	Infiltration and evapotranspiration (surface water); springs and baseflow (groundwater)
Use	Surface water withdrawals or wells

On the San Pedro, tradeoffs between human water use and the groundwater and surface flow that support the San Pedro River's riparian ecosystems are at the forefront of debates about population, water use, and sustainability (Steinitz et al. 2003, Kepner et al. 2004, Stromberg and Tellman 2009). In the La Antigua watershed, rural landowners in the upper watershed derive their livelihoods from small-scale agricultural production, forestry, ranching, and extraction of forest products. The conversion of upstream forest to agriculture and pasture land impacts biodiversity and hydrologic benefits for downstream beneficiaries. While Mexico's Payment for Hydrological Environmental Services program was designed to combat both water scarcity and deforestation, the program has suffered from a lack of spatial targeting of payments to areas with low opportunity costs and high service provision and deforestation threats (Munoz-Pina et al. 2008). Tools like ARIES can assist in spatial planning for such programs.

Past studies have used a variety of spatial data to map water supply and regulation services on the landscape. These have typically included overlays of supply and demand (Boyd and Wainger 2003, Wundscher et al. 2008), estimates of water stored in soils and aquifers using infiltration data (Egoh et al. 2008), precipitation and evapotranspiration data (Chan et al. 2006), or the SCS curve number (SCS 1972, Gately 2008) or Budyko Curve method to account for precipitation and evapotranspiration across the landscape

(Tallis et al. 2011). Given the difficulty in developing a generalized model of hydrologic processes that is applicable at multiple spatial scales and in different ecological contexts, the initial ARIES water supply models include direct data or Bayesian models (for surface water sinks) that are applicable to our case study regions but that incorporate many of the influences on hydrologic processes that were used by the above authors. In cases where vegetation-hydrology relationships are poorly understood, such as in tropical forests (Bruijnzeel 2004), ARIES' data-driven modeling approach may be more appropriate than using process-based approaches. In many other cases, future generation ARIES models will link existing hydrologic models, improving model quality and credibility.

Figure 8.1: Conceptual diagram of the linkages between sources, sinks, and uses of surface and groundwater.



8.2 Water supply source models

Spatial data or calibrated hydrologic model outputs can generally be used as the source value for surface and groundwater supply, with no Bayesian model needed. There are at least five potential sources of surface water, which can be summed to obtain the total annual surface water source value: precipitation, snowmelt, springs, baseflow to rivers, and incoming inter-basin water transfers where water is discharged into surface water bodies. If we run the model using annual average values, snowmelt only becomes important in locations with glaciers (i.e., annual snowmelt in all other locations is included in annual precipitation totals). Sources of groundwater include areas of infiltration and deep percolation that lead to aquifer recharge, along with artificial groundwater recharge.

For the San Pedro, we use annual precipitation as the source value for surface water. We show initial results for a representative dry (2002) and wet year (2007), since the 30-year average data from PRISM is less meaningful in arid environments where annual precipitation is highly variable. If desired, a user could also input precipitation data from other years to use as the surface water source value. For groundwater, we can compare spatial data for soil infiltration, infiltration results from the surface water sink Bayesian network model, and the results of hydrologic models (once incorporated) as possible source values. In the future, we could also incorporate data on the location of groundwater recharge facilities in the Sierra Vista area, assuming the data were available. We do not include snowmelt in the source model, as there is no persistent snowpack in the mountains within the San Pedro River Watershed. Until detailed surface and groundwater models are incorporated, we lack data on baseflow. Although incoming interbasin water transfers are proposed (Bureau of Reclamation 2007), there are currently no incoming water transfers from outside the basin. Finally, while we have data on the location of springs in the San Pedro, we do not use these data in the source model as we do not know their discharge volume, and most spring discharge quickly infiltrates back into the soil via ephemeral stream channels (Dave Goodrich, Tom Meixner, personal communication).

For La Antigua, we also use annual precipitation as the source value for surface water. Infiltration results from the surface water sink Bayesian network model can serve as the source value for groundwater. Like the San Pedro, there is no persistent snowpack in the mountains within the La Antigua watershed. We also lack detailed hydrologic data or models on springs, baseflow, incoming interbasin water transfers, and groundwater recharge, so we do not include these as sources of surface or groundwater.

Table 8.2: Datasets used for the water supply source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Precipitation	All	WorldClim	Global	30 arc-seconds[2]	1950-2000
	San Pedro	PRISM / OSU	United States	800 x 800 m (1); 4 x 4 km (2)	(1) 1971-2000; (2) dry year (2002), wet year (2007)
Snowmelt	n/a for current case studies	Univ of Delaware Global Water Balance Archive	Global	0.5^{o} x 0.5^{o}	1950-1999

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Soil infiltration	San Pedro	USGS	Continental United States	1 km x 1 km	Derived from 1951-1980 runoff data
Springs	San Pedro, not used for source value	AGIC	Arizona	Unknown	Unknown

8.3 Water supply sink models

Surface water sinks include areas of evapotranspiration and infiltration. Conversely, groundwater sinks include springs and baseflow to rivers. Lacking an external groundwater model, we currently do not include springs or baseflow as groundwater sinks for the San Pedro. For both the San Pedro and La Antigua, we set the total surface water sink as the sum of evapotranspiration and deep soil infiltration (Figure 8.2)[22]. Runoff data will play a role in training of the Bayesian network models to help account for the difference between precipitation and sinks.

For the San Pedro, nationwide data are available for deep soil infiltration and global data for actual evapotranspiration. While these data sources can be used as training data for Bayesian network models, both datasets are problematic. The evapotranspiration dataset has low spatial resolution (0.5 x 0.5 degree) and does not capture local variation in vegetation type, tree canopy cover, and temperature, all of which are key influences on evapotranspiration. The infiltration data, having been developed at the national level, are unlikely to account for the limited area over which infiltration actually occurs in the semiarid Basin and Range region of the southwestern United States. We therefore use a Bayesian network that considers vegetation type, percent tree canopy cover, and annual maximum temperature as influences on evapotranspiration. We set the locations of stream channels and limestone bedrock and the intersection of valley fill alluvium and the mountain fronts to account for the two key locations of deep percolation and groundwater recharge: the mountain fronts, stream banks, and ephemeral stream channels (Pool and Dickinson 2007). In the future, we could also incorporate data on the location of groundwater recharge facilities in the Sierra Vista area, assuming the data are available. We set priors for these nodes based on a review of the corresponding spatial data. We set the highest values for the evapotranspiration conditional probability table under greater percent tree canopy cover and higher temperatures, all else being equal. We set the highest evapotranspiration rates for vegetation type to riparian, followed by forests, then mesquite woodland, oak woodland, agriculture, urban, and grassland, with the lowest values set for desert scrub. We set the conditional probability for infiltration as highest

[22] Bayesian network models for water sink models can be downloaded from http://ariesonline.org/modules/waterspecs.html.

at the mountain fronts and as slightly lower in stream channels, and set it as extremely low elsewhere.

For La Antigua, we set the evapotranspiration as a function of vegetation type and percent tree canopy cover, and set deep infiltration as a function of hydrologic soils group, slope, and percent impervious surface cover. We set the conditional probability for evapotranspiration to be highest with greater tree canopy cover and for riparian vegetation and forests and lowest for agriculture, urban, and grassland. We assume infiltration to be greatest on shallow slopes, low levels of impervious surface cover, and hydrologic soils groups A and B.

Figures 8.2: Bayesian network models for water supply sinks.

Figure 8.2.1: Water supply sinks for La Antigua River Watershed.

Figure 8.2.2: Water supply sinks for San Pedro River Watershed.

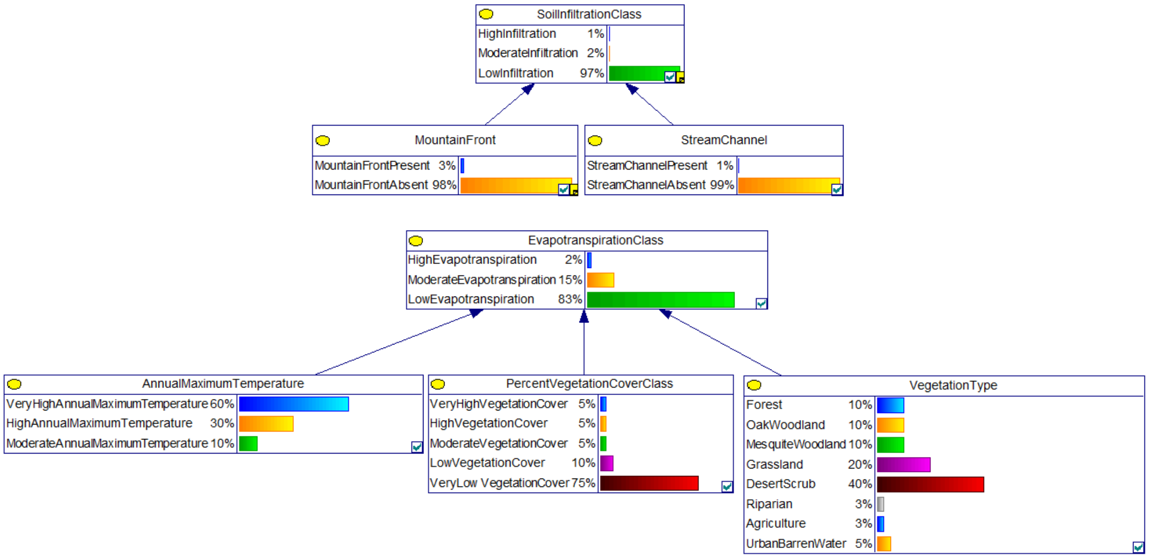


Table 8.3: Datasets used for the water supply sink models

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Actual evapotranspiration	All	SAGE / UW Mad	Global	$0.5^o \times 0.5^o$	1950-1999
Annual maximum temperature	San Pedro	WorldClim	Global	30 arc-seconds[2]	1950-2000
		PRISM / OSU	United States	800 m x 800 m	1971-2000
Hydrologic soils group	La Antigua	Gately (2008)	Global	$0.083^o \times 0.083^o$	Unknown
Hydrography	La Antigua	INECOL	La Antigua	Unknown	Unknown
	San Pedro	National Hydrography Dataset	Arizona	Unknown	Unknown
		U.S. EPA San Pedro Data Browser	Upper San Pedro in Sonora, MX	Unknown	Unknown
Impervious surface cover	La Antigua	NOAA-NGDC	Global	1 km x 1 km	2000-2001
Mountainfront recharge zones	San Pedro	AGIC	Arizona	Unknown	Unknown
Runoff	All	SAGE/Univ. of Wisconsin	Global	$0.5^o \times 0.5^o$	1955-1990
Slope	La Antigua	Derived from SRTM	Global	90 m x 90 m	2000
Soil infiltration	San Pedro	USGS	Continental United States	1 km x 1 km	1951-1980
Springs	San Pedro	AGIC	Arizona	Unknown	Unknown
Tree canopy cover	All	GLCF / UMD	Global	1 km x 1 km	2000
	San Pedro	NLCD 2001	United States	30 m x 30 m	2001
Vegetation type	La Antigua	INECOL	La Antigua	Unknown	Unknown
	San Pedro	SWReGAP	Southwest US	30 m x 30 m	2000

8.4 Water supply use models

Users access groundwater through wells and surface water through surface diversions, direct pumping from water bodies, and outgoing inter-basin water transfers. Users can be split by use type (e.g., agriculture, domestic, industrial use) if deemed relevant to the case study of interest.

For the San Pedro, we mapped the location and volume of the two surface water diversions on the river at St. David and Pomerene (Table 8.4). We use well data and capacity to identify groundwater use. Although the state database of wells identifies users, they are not explicitly grouped by use, so at this time we do not separate out agricultural, mining, military, or domestic water uses. Also, since we do not currently have an integrated groundwater flow model, we do not explicitly connect sources, sinks, and users for groundwater.

For La Antigua, we used spatial and tabular data to map the location and volume of surface water diversions. Well data and well capacity could be used to identify groundwater use. In either case, legally binding water rights would also be informative for further identifying beneficiaries and use. We mapped four distinct beneficiary classes, including agriculture, aquaculture, industrial, and residential water use.

Table 8.4: Datasets used for the water supply use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Surface diversions	San Pedro	Digitized locations of St. David and Pomerene Diversions	San Pedro	Unknown	2010
Water extraction amounts and user types	La Antigua	INECOL	La Antigua	Unknown	Unknown
Well capacity	San Pedro	Arizona Dept. Water Resources Wells 55 Database	Arizona	Unknown	2010
Well depth	San Pedro	Arizona Dept. Water Resources Wells 55 Database	Arizona	Unknown	2010
Well locations	All	INECOL	Puebla, Sonora, Veracruz	Unknown	Unknown
Well user type	San Pedro	Arizona Dept. Water Resources Wells 55 Database	Arizona	Unknown	2010

8.5 Water supply flow models

The source models determine the annual quantity (in mm^3/yr) of precipitation and other surface water sources (in the surface water models) or infiltration to groundwater (in the groundwater source models). The sink models estimate the annual quantity of water transitioning between surface and groundwater, and vice versa, and the use models estimate the quantity of water used by beneficiaries in each location. Surface and groundwater flows must be modeled separately, as they move at different rates, with flows governed by different factors.

Currently, we map surface water flow using a simple water routing model. This model relies on the SRTM elevation data to identify flow directions for water (Table 8.5). Water is moved across the landscape using this derived flow direction layer until it encounters a stream (represented using a hydrography layer), at which point it moves downstream through the stream network. Users or sinks encountered in transit reduce the quantity of water carried across the landscape.

While our approach is an admittedly simplistic way to move water and water-related ecosystem service carriers (e.g., drinking water, flood water, suspended sediment, dissolved nutrients), this approach has the benefit of being applicable at relatively coarse spatial scales and at any location on Earth. Future work on ARIES will seek to incorporate appropriate existing hydrologic models to route water and water-related ecosystem service carriers across the landscape at variable spatial scales and under variable environmental conditions (e.g., using different models at large vs. small spatial scales and in arid versus humid ecological systems).

Subsurface water flows are considerably more complex, and are governed by factors including geology (i.e., porosity of rock layers) and groundwater elevations. Subsurface flows are commonly modeled using the MODFLOW model (Harbaugh et al. 2000). Future releases of ARIES will investigate the feasibility of linking groundwater models to source, sink, and use models to fully and more accurately represent water flows using accepted hydrologic models.

Linking the source, sink, and use data with the flow model, we estimate the following indicators for water supply. Surface and groundwater concepts are both listed[23]:

1. Theoretical source, sink, and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Surface water supply & Groundwater recharge: All atmospheric, ground, or surface water transitions (i.e., precipitation, snowmelt, baseflow, incoming water transfers, and springs for surface water, infiltration and artificial

[23] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts.

recharge for groundwater) that result in an initial source quantity of surface or groundwater.

   b. Maximum surface water sink & Maximum groundwater sink: Available absorption capacity at any location where surface water can transition into groundwater (via percolation/recharge) or atmospheric water (via evapotranspiration), or where groundwater can transition into surface water (via springs or baseflow).
   c. Surface water demand & Groundwater demand: Total demand for water, which can be treated as separate surface and groundwater demand or combined into total water demand.
2. Possible flow, source, and use. These values are calculated by running flow models *without accounting for sink values* (areas of surface and groundwater transitions). The possible values represent the maximum delivery of surface water based on the theoretical source value.
   a. Possible surface water flow & Possible groundwater flow: The movement of surface water via topography and stream networks, and groundwater via appropriate groundwater flow paths while disregarding sinks.
   b. Possible surface water supply & Possible groundwater recharge: Atmospheric, ground, or surface water transitions providing an initial source quantity of surface or groundwater, that are capable of providing water to human beneficiaries when accounting for surface or groundwater flow paths but not sinks.
   c. Possible surface water use & Possible groundwater use: Water actually reaching a user, but not accounting for the activity of sinks.
3. Actual flow, source, sink, and use. Actual water supply benefits provided, received, and degraded *with a full accounting for source, sink, and use values and flows*.
   a. Actual surface water flow & Actual groundwater flow: The movement of surface and groundwater, accounting for flow topology and sinks.
   b. Used surface water supply & Used groundwater recharge: Atmospheric, ground, or surface water transitions that result in an initial source quantity surface or groundwater, that are capable of providing water to human beneficiaries when accounting for surface or groundwater flow paths and sinks (i.e., locations actually providing water to human beneficiaries).
   c. Actual surface water sink & Actual groundwater sink: Locations where surface water transitions into groundwater (via infiltration) or atmospheric water (via evapotranspiration), or where groundwater transitions into surface water (via springs or baseflow).
   d. Satisfied surface water demand & Satisfied groundwater demand: The portion of demand for water satisfied by extraction of surface or groundwater.
4. Inaccessible source and use. Theoretical values minus possible values; accounts for sources that do not provide, sinks that do not degrade, and beneficiaries that cannot use an ecosystem service due to a lack of flow connections.

   a. Unusable surface water supply & Unusable groundwater recharge: The source locations of surface or groundwater that are not available to human users because their flow paths do not move them in a direction where they can be accessed by human beneficiaries.
   b. Inaccessible surface water demand & Inaccessible groundwater demand: Demand for surface or groundwater that goes unsatisfied due to a lack of surface or groundwater flow connections between sources of water and the location of water users.
5. Blocked flow, source, and use. Source, flow, or use values degraded by sinks.
   a. Sunk surface water flow & Sunk groundwater flow: Water flow that fails to reach a user because it encountered a sink and transitioned from surface or groundwater into the atmosphere, surface, or groundwater.
   b. Sunk surface water supply & Sunk groundwater demand: Source locations of surface or groundwater that fail to reach a user due to their encountering a sink and that transitions water to the atmospheric, surface, or groundwater.
   c. Blocked surface water demand & Blocked groundwater demand: Demand for surface or groundwater that goes unsatisfied due to the action of sinks that transition water between surface, atmospheric, and groundwater.

Table 8.5: Datasets used for the water supply flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Elevation	All	SRTM	Global	90 m x 90 m	2000
Hydrography (stream networks)	La Antigua	INECOL	Rio La Antigua	Unknown	Unknown
	San Pedro	National Hydrography Dataset	Arizona	Unknown	Unknown
		U.S. EPA San Pedro Data Browser	Upper San Pedro in Sonora, Mexico	Unknown	Unknown

8.6 Caveats and directions for future research

Several key next steps will improve the quality of next-generation ARIES water supply models. First, existing hydrologic models should be incorporated, including models to handle groundwater flow and surface water-groundwater interactions. Second, fully implemented scenarios, especially for climate change impacts on rainfall quantity, seasonality, and snowpack would provide improved support for water supply planning. Third, more explicit identification of user groups would help to identify tradeoffs in water use in support of economic equity (i.e., identifying access to water for the poor, particularly in the developing world). Finally, improving the temporal resolution of the models would enable users to better explore the risks of seasonal water shortages.

Ideally, ARIES would be capable of calling on several hydrologic models, using automated reasoning to select the correct model for the spatial and temporal scale of analysis and for the relevant ecosystem types. For instance, SWAT, a well-accepted daily time step hydrologic model (Neitsch et al. 2005) could be run in data-rich regions while FIESTA (Mulligan and Burke 2005) could be called on for water supply assessment in tropical mountain environments with poorer data quality. FIESTA is in the process of being linked to the WEAP model (Yates et al. 2005), which itself interfaces with the MODFLOW groundwater model (Harbaugh et al. 2000). Locally calibrated MODFLOW and SWAT models are available for the Upper San Pedro River Watershed (Pool and Dickinson 2007), but have not yet been connected to ARIES. Incorporation of MODFLOW would enable representation of groundwater flows and baseflow contributions to stream flow. For the San Pedro and other arid land rivers these are key parameters to model in order to better understand tradeoffs in water availability for people, riparian ecosystems, and the services supported by these riparian ecosystems. Ongoing work with hydrologists in the La Antigua watershed will improve the accuracy and policy relevance of the ARIES system in this case study.

While we currently model water supply at an annual time step, seasonal water availability is critical for people and ecosystem water needs. Thus, modeling water availability for finer temporal scales would be desirable for many applications. Modeling wet and dry season interaction between surface water, groundwater, and soil water is critical not only in arid and semiarid environments but increasingly in humid but seasonally dry regions such as the Pacific Northwest, particularly as population and water demand growth interact with climate change to increase water stress (Goodstein and Matson 2007). However, modeling the spatial linkages between sources, sinks, and users of water becomes increasingly challenging at finer temporal scales. Gridded average annual precipitation datasets are available at spatial resolutions of 800x800 m and 1 km^2 for the United States and globally, respectively. Annual precipitation data for the U.S. are available, but at 4 x 4 km spatial resolution. Event-specific interpolation of rainfall in sparsely gaged areas becomes even more challenging, particularly in arid environments where precipitation is highly patchy even within small watersheds. Because of this limitation, past SWAT modeling in the San Pedro has used the subwatershed as the unit of analysis (Hernandez et al. 2003). While others have also used agent based models to map water movement (Reaney 2008), resolving the tradeoff between high spatial and temporal resolution in an agent based modeling framework may be difficult and will require further exploration.

8.7 Acknowledgements and additional contributors

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9. Recreation

9.1 Introduction

Ecosystems provide settings for a diverse array of recreational activities. Recreational values are among the best recognized of all ecosystem services by the public, and human preferences for recreation have been well studied by economists and other social scientists. From a spatial perspective, we can map *sources* of recreational value (areas capable of providing the natural setting needed for a particular activity), *sinks* of recreational value (landscape features that reduce those source values, if applicable), and the *users* of a particular recreation area for a given activity. Users may simultaneously value a bundle of recreational attributes (e.g., the quality of an area for hunting or fishing plus the quality of scenic views), built infrastructure (i.e., trails, other facilities), relative congestion, and the management policies that facilitate a particular recreational experience (Lawson and Manning 2002, Arnberger and Haider 2007, Boyd and Banzhaf 2007, Bullock and Lawson 2008). The ARIES recreation models map an ecosystem's capacity to support a particular recreational activity, as opposed to the other attributes that contribute to the overall quality of a recreational experience. We assume that only the ecosystem attributes supporting recreation represent actual natural capital and are thus the ecosystem service (Boyd and Banzhaf 2007). By mapping the ecosystem's contribution toward different recreational attributes, we can explore tradeoffs between different types of recreational use, tradeoffs between recreation and other ecosystem services, and relative preferences for certain recreational attributes.

A recreation flow model accounts for travel from a person's home to a particular location suitable for a given recreational activity. Travel cost and recreational site choice studies use basic transportation routing to connect recreationists to recreation sites, but typically do not facilitate comparisons with other ecosystem services (Clawson and Knetsch 1966, Hunt et al. 2005, Hunt 2008). A recreation flow map illustrates where a particular recreation area draws its users from or to which (substitute) areas a specific recreational user group in a region gravitates toward. All recreation models thus contain a transportation network-based flow model to move people toward recreational opportunities. While flows of most other ecosystem services are defined physically (i.e., through movement of water, nutrients, sediment, or atmospheric gases), biologically (i.e., through migration and movement of key species), or through trade networks (i.e., for ecosystem goods), recreational flows are based on human preferences for a particular activity and perceptions about places capable of providing a desirable setting for that activity. This adds substantial complexity to understanding recreational flows, as preferences are shaped by past experiences and place attachment (Hunt et al. 2005, Hunt 2008), as well as distance and subjective measures of a site's suitability for a given recreational activity.

The spatial scale of a recreation model is defined as a reasonable travel distance whose value exhibits a Gaussian decay with distance from the source. Recreational services are nonrival but congestible and are measured in abstract units. Table 9.1 summarizes the characteristics of the recreation models.

Table 9.1: Summary characteristics of the ARIES recreation models.

Service	Recreation
Benefit type	Provisioning
Medium/units	Recreational enjoyment (abstract units, 0-100)
Scale	Travel distance
Movement	Travel simulation
Decay	Gaussian
Rival?	Nonrival but congestible
Source	Recreational areas suitable for a given activity
Sink	None
Use	Recreationists interested in a given activity

To calculate the recreation source value to feed into the transportation flow model, an optional initial physical or biological flow model may also be needed (e.g., to identify spatial dependencies for areas having high quality viewsheds, providing sources of water for water-based recreation, or providing habitat for recreationally valued species). For instance, a physical or biological flow model could show where high-quality views are provided to key vantage points (along with visual blight that reduces view quality), where runoff is provided to a watershed valued for rafting or fishing, or where critical habitat outside a protected area supports populations of recreationally valued species. These models can then show off-site areas outside of protected areas that are critical toward maintaining the quality of that resource. The transportation flow model is then used to link source (recreational use) points to the location of recreational users. For instance, we can use a viewshed model (fully described in Chapter 3) to identify mountain summits that can provide scenic views, with view quality estimated using a line of sight flow model. We then apply a transportation flow model to link potential users to these scenic trails themselves.

We developed recreation case study models for **Vermont** and the **San Pedro River Watershed (Arizona and Northern Sonora, Mexico).** These models are intended to be representative of site-level recreation conditions across wider regions. For instance, the Vermont viewshed model could be applied more broadly to the Northern Forest region of northeastern United States, stretching from upstate New York's Adirondack Mountains to Maine. The model could also be adapted to other regions by adjusting the values associated with viewshed features to reflect local land cover types, topography, and view preferences. The San Pedro birding, hunting, and wildlife viewing models should be generally applicable throughout the Southwest and beyond, provided that locally desirable species are included in the models for areas beyond Southeast Arizona. Given the heterogeneity of recreational preferences in different parts of the world, we do not currently envision creating a generalized global ARIES recreation model, but instead plan to develop local case studies such as those described in this chapter. These future case studies could allow us to expand the geographic coverage of ARIES recreation models while enabling us to better understand when and where development of national or global scale recreation models might be more appropriate.

In our Vermont case study, we map the value of scenic viewsheds to hikers of the region's mountain summits. Scenic features act as sources of high quality views, including large mountains like the Green and Adirondack Mountains, water bodies, especially large lakes like Lake Champlain, and a heterogeneous pattern of land cover representing both forested and agricultural lands. Recreational users of aesthetic views are those people that access view points along trails or at scenic vistas. Their enjoyment may depend on the relative elevation of the vista (Zube et al. 1975). As a view travels from source to user, it may be physically blocked by buildings, trees, or topography. Its quality may be depleted by air pollution or visual blight, such as highways, forest clearcuts, or visually unappealing land use types like commercial, industrial, or transportation uses. Such sources of visual blight can also be mapped.

Managers at the Bureau of Land Management's San Pedro Riparian National Conservation Area (SPRNCA) identified birding, hiking, mountain biking, equestrian, hunting, nature photography, and visitation of historic and prehistoric sites as key recreational activities in the SPRNCA. We developed models for birding, hunting, and wildlife viewing, three biodiversity-based recreation ecosystem services provided by the Southeast Arizona's San Pedro River Watershed. The San Pedro is internationally known as a birding area, and draws visitors from across the United States and world (Colby and Orr 2005, Southwest Wings Festival 2010). Birders are generally interested in viewing a large number of species in a given area, as well as rare or unique species. As a key water source in an arid region, the San Pedro is also habitat for several valued game species, including white-tailed and mule deer, javelina, white-winged and mourning dove, and Mearns', Gambel's, and scaled quail. Although wildlife viewing was not explicitly given as a recreational activity in the SPRNCA, the opportunity to view a high diversity of mammal, bird, reptile, and butterfly species are valued by many visitors to the area.

Other ecosystem services researchers have mapped potential recreational value across the landscape by overlaying factors including viewsheds or visibility (Eade and Moran 1996, Chen et al. 2009), proximity or access to roads, population centers, or recreation infrastructure (Eade and Moran 1996, Boyd and Wainger 2003, Chan et al. 2006, Beier et al. 2008), and land ownership and cover characteristics (Boyd and Wainger 2003, Chan et al. 2006). Most of these authors, however, develop a general model of recreation site quality, rather than looking at sites' suitability for a specific recreational activity. We thus drew selectively from these studies when developing our recreation models.

9.2 Recreation source models

9.2.1 Recreation source models: birding, hunting, and wildlife viewing

Potentially valuable birding areas can be identified using spatial data for bird species richness and the presence of rare birds. We map the presence of rare and charismatic birds by noting the number of bird species' habitats present, based on a list of ten rare or charismatic birds for the San Pedro River Watershed and surrounding mountains (Southwest Wings Festival 2010)[24]. Hunting potential can be identified based on habitat maps for the above-listed game species – javelina plus two species each of deer and doves and three species of quail. We map wildlife viewing potential by averaging decile values for diversity of amphibian, bird, mammal, and reptile species. We set the source value for birding, hunting, and wildlife viewing as a function of public access, potential presence of surface water (springs or streams) and riparian habitat quality (where known), along with the appropriate bird richness and rarity, harvestable species habitat, or overall biodiversity value (Figure 9.1.1)[25]. We set priors for each variable based on reviews of the corresponding spatial data.

High diversity of birds and wildlife or habitat for rare or game species are clear prerequisites for supporting related recreational activities, as is public access, particularly in states like Arizona where access on private lands is likely to be controlled. We set these factors as the strongest influences on recreation source values in their respective contingent probability tables. We set the presence of perennial or intermittent surface water, including streams and springs, as an important but slightly lesser influence on source values in the contingent probability tables, since the presence of surface water is highly important for attracting wildlife in arid environments. Where riparian condition is known, we assigned higher values for birding, hunting, and wildlife viewing quality to higher-quality riparian areas.

9.2.2 Recreation source models: viewsheds

Mountains, open water, forested and open lands are commonly valued objects in viewsheds (USFS 1974, Zube et. al. 1975, USFS 1995, Chhetri and Arrowsmith 2003, Manning et. al. 2006, Goonan et al. 2007). We define several types of open space, including agricultural, forested, and other types of land cover. Agricultural lands include pasture land, crop land, and orchards. Other open land includes barren lands, brush and transitional lands, and wetlands. Forested lands include broadleaf, coniferous, and

[24] The ten rare and charismatic bird species included for this model include: elegant trogon (*Trogon elegans*), red-faced warbler (*Cardelina rubrifrons*), sulphur-bellied flycatcher (*Myiodynastes luteiventris*), painted redstart (*Myioborus pictus*), five-striped sparrow (*Aimophila quinquestriata*), Cassin's sparrow (*Aimophila cassinii*), Botteri's sparrow (*Aimophila botterii*), vermillion flycatcher (*Pyrocephalus rubinus*), tropical kingbird (*Tyrannus melancholicus*), and yellow-billed cuckoo (*Coccyzus americanus*).

[25] Bayesian network models for recreation source and sink models can be downloaded from http://ariesonline.org/modules/recspecs.html.

mixed forests. We estimated prior probability distributions for elevation, land cover, and the presence of open water based on relevant spatial data for Vermont.

We set overall view quality, or "Theoretical natural beauty" as a function of water views, open space views, and the presence of mountains (Figure 9.1.2). We set the highest values for large mountains, lakes, and forested open space, the lowest values for developed land, and no mountains or water views, and intermediate values for other open space and agriculture. These relative values could be adjusted for different parts of the world based on local view preferences. The highest and lowest quality view combinations allowed us to "peg the corners" of the contingent probability table and interpolate intermediate values for the remaining values of the theoretical natural beauty contingent probability table (Marcot et al. 2006).

People value highly scenic landscapes (those with high natural beauty as defined in the source model) but also landscapes with a diversity of landforms, water characteristics, and vegetation patterns (USFS 1995, Chhetri and Arrowsmith 2003). The precise relationship between this variability and value may vary regionally. Research is lacking for the eastern United States, but data from Switzerland show that in a reforesting landscape people prefer heterogeneous patches ranging from slightly to mostly reforested (Hunziker and Kienast 1999). The diversity of landscape types present in views will be incorporated into future recreational viewshed flow models, building on the viewshed flow model described in Chapter 3.

Table 9.2: Datasets used for the recreation source models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Amphibian, bird, mammal, reptile species richness	San Pedro	USGS Southwestern Biological Center Sonoran Desert Research Station	AZ, CO, NM, NV, UT	Unknown	1999-2001
Elevation	Vermont	SRTM	Global	90 m x 90 m	Unknown
Habitat for game species	San Pedro	SWReGAP	AZ, CO, NM, NV, UT	240 m x 240 m	1999-2001
Hydrography	San Pedro	National Hydrography Dataset	Arizona	Unknown	Unknown
	Vermont	National Hydrography dataset	Vermont	Unknown	Unknown
Lakes and ponds	Vermont	National Hydrography dataset	Vermont	Unknown	Unknown
Public lands	San Pedro	AGIC	Arizona	Unknown	2010
Rare & charismatic bird habitat presence	San Pedro	SWReGAP	AZ, CO, NM, NV, UT	240 m x 240 m	1999-2001

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Riparian condition class	San Pedro	Stromberg et al. (2006)	SPRNCA	Unknown	2001-2004
Springs	San Pedro	AGIC	Arizona	Unknown	Unknown

Figures 9.1: Bayesian network models for recreation source values.

Figure 9.1.1: Recreation sources for San Pedro River Watershed: Birding, hunting, and wildlife viewing.

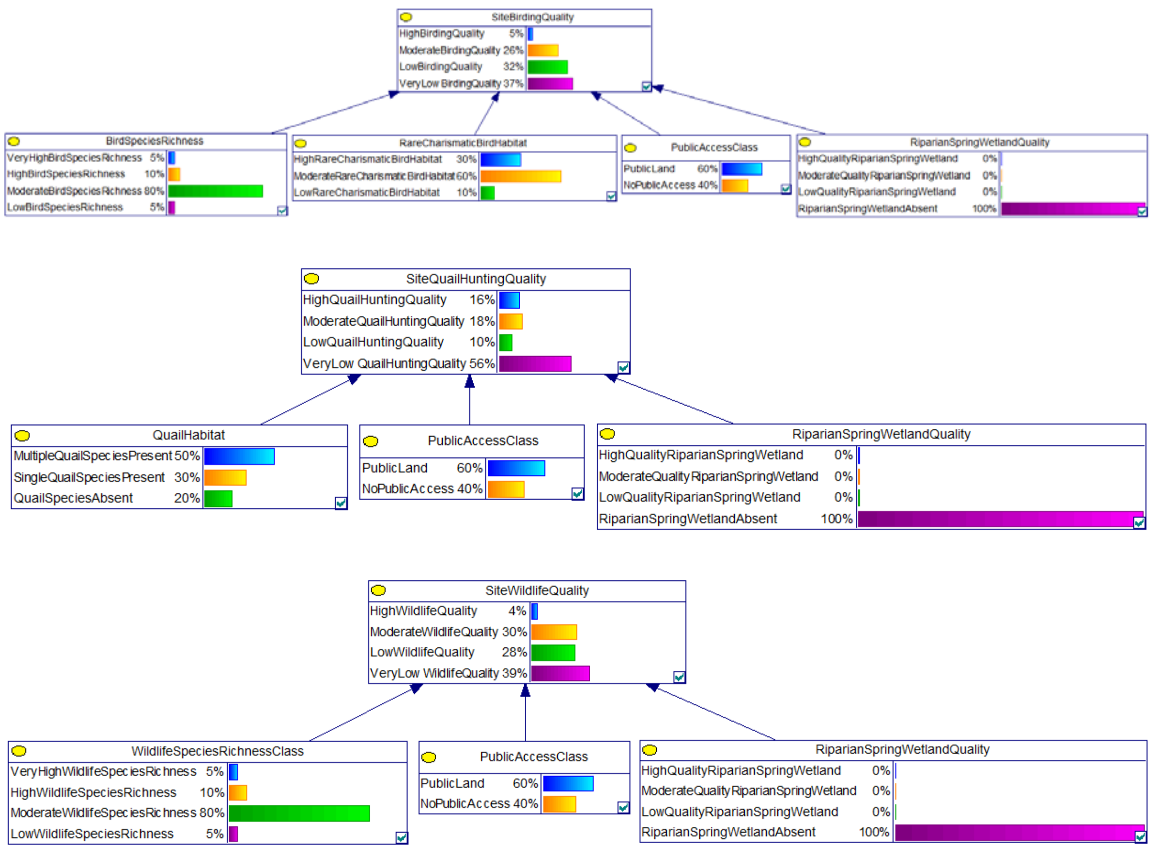
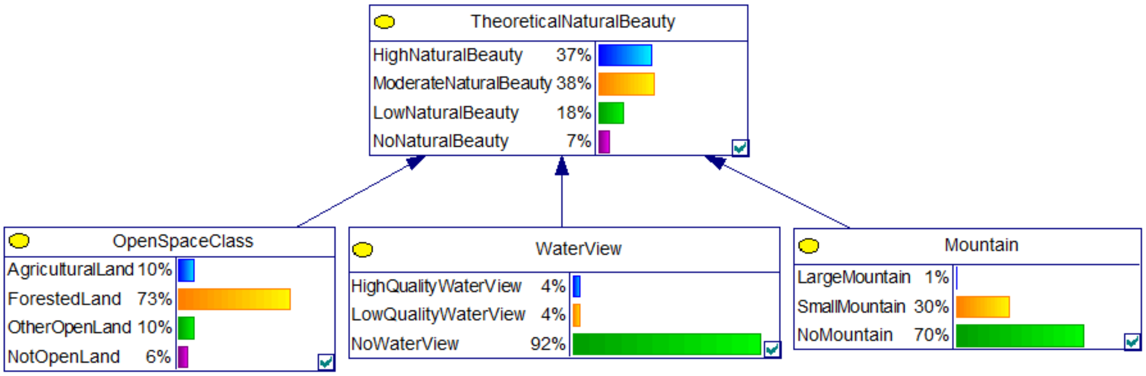


Figure 9.1.2: Recreation sources for Vermont: Scenic viewsheds.



9.3 Recreation sink models

Sinks will be present for some, but not all types of recreation. For most types of recreation where the source value can be assessed *in situ*, no sink model is necessary. This is true for the birding, hunting, and wildlife viewing models for the San Pedro, where we do not specify the dependence of a particular species on additional habitat outside its currently mapped habitat. However, habitat-based flow models could eventually be developed or existing ones incorporated to account for spatial dependencies in wildlife habitat (e.g., Semmens et al. in press).

For viewsheds, the sink model identifies areas of visual blight that reduce view quality, similar to the viewshed model described in Chapter 3. We assumed that obstructions (e.g., buildings, topography, or vegetation) or undesirable visual features (blight associated with development, energy infrastructure, or roads) reduce view quality (Benson et al. 1998, Bourassa et al. 2004, Gret-Regamey et al. 2008). Views of lost forest cover, including clearcuts, could also reduce view quality (Palmer 2008, Wundscher et al. 2008). We set prior probability distributions using corresponding spatial datasets. Based on a 1998 Vermont Agency of Natural Resources assessment, approximately 2% of the Vermont landscape was heavily cut or clear cut. We assume this value as our prior probability for clearcuts. We set values in the contingent probability table for visual blight as high when individual or combined blight features were observed in the landscape and as zero when these features were absent, and interpolated intermediate values.

Figures 9.2: Bayesian network models for recreation sinks.

Figure 9.2.1: Recreation sinks for Vermont: Scenic viewsheds.

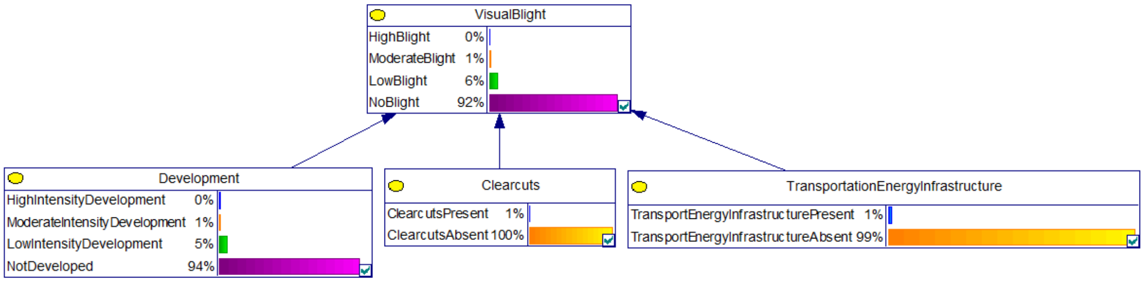


Table 9.3: Datasets used for the recreation sink models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Developed land	Vermont	NLCD 2001	United States	30 m x 30 m	2001
Energy infrastructure	Vermont	VCGI	Vermont	Unknown	Unknown
Transportation infrastructure	Vermont	VT AOT	Vermont	Unknown	2010

9.4 Recreation use models

Initial mapping of recreational use relies on population or housing density data, combined with visitor ZIP Code data for site use, where available. For some activities, it may be possible to estimate the percentage of the population taking part in that recreational activity (i.e., the number of licensed hunters or anglers in a state relative to its total population). Representing users as a uniform percentage of the population engaging in a particular activity makes the admittedly naïve assumption that the same percentage of recreational users across all communities engage in a particular activity. It also assumes that different user groups for the same activity have similar preferences, which is not always a realistic assumption. For example, Hunt et al. (2005) found urban and rural hunters to prefer different types of hunting experiences.

A more realistic model would account for the fact that different types of communities are likely to prefer different recreational activities and to value attributes of a particular recreational experience differently. Indeed, in some cases individuals will choose their location of residence to provide proximity to an especially valued recreational amenity.

9.4.1 Recreation use models: birding, hunting, and wildlife viewing

Our initial use models for the San Pedro start with a population density map and anecdotal information on total visitation and the distance groups typically travel to reach the SPRNCA (Mark Rekshynskyj, Jim Mahoney, Gordon Lewis, personal communication). We then assign the home locations of visitors to the SPRNCA based on population density for the estimated number of visitors coming from within the watershed, from the Tucson area, and from more distant locations. This is an admittedly simplistic way to map visitors, but in the absence of better data (e.g., surveys where visitors identify their ZIP Code of origin), it at least enables mapping of the spatial dependencies between recreation areas and recreationists. These simplistic assumptions about visitation can be easily replaced with actual data in locations where better survey data are available.

9.4.2 Recreation use models: viewsheds

The use model for aesthetic views is based on locations where hikers have access to views (i.e. trails, vistas, outcroppings). Since only Vermont's tallest mountains have elevations above the treeline, many of these viewpoints are rock outcrops. These use points are relatively small in number, and are generally well known. We digitized these points to create a use layer. After running the view flow model to measure the view quality at each viewpoint, a transportation flow model can then be used to connect use of each recreational site to its potential user population. Lacking data about the annual number of Vermonters and out of state visitors who hike to places with scenic views, we assumed that 33% of the state's population will hike in a given year. This assumption, which can easily be adjusted to reflect actual data, can be used to identify the location of users on the landscape and to map spatial flows of visitors (Section 9.5).

Table 9.4: Datasets used for the recreation use models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Scenic viewpoints	Vermont	Digitized locations of peaks in Vermont with scenic views	Vermont	n/a	2010
Population density	All	U.S. Census Bureau	Arizona	Census block groups	2000-2007

9.5 Recreation flow models

With information on how many visitors participate in a given activity at a particular recreation site and how far they travel, we can implement a simple flow model by distributing the potential visitor population across the landscape based on population density, road network, and recreation site data. This allows the model to estimate travel times for people traveling from their residences to recreational sites. ZIP Code recreational travel cost expenditures, recreational preference surveys, or visitor/hunting permit data can show how far people travel to a particular site. For a given recreational area, this will result in a map showing the spatial extent and density of its user population. ZIP Code data are often available from state park systems and for the National Park Service through the University of Idaho Park Studies Unit's Visitor Services Project (http://psu.uidaho.edu/vsp.htm), however data collection methods are not uniform, and data are rarely available for local parks and other Federal public lands (e.g., U.S. Forest Service).

Birders from around the nation and world visit the San Pedro. Hunters and other recreational users (e.g., hikers, mountain bikers, equestrians, viewers of historical sites) are less likely to travel great distances. These visitors are more likely to come from "local" areas such as the San Pedro Valley itself or from Tucson, while recreationists from Phoenix are more likely to choose closer sites for their activities, and more rarely travel to the San Pedro.

For the Vermont case study, we lack data about the distance that hikers typically travel to reach their preferred hiking destination. We assume that they will hike summits within a 1-hour driving distance from home 70% of the time, a 1-2 hour driving distance from home 20% of the time, and a greater than 2 hour driving distance from home 10% of the time. These assumptions can easily be adjusted to reflect actual data or alternate patterns of behavior.

It is more difficult to map the converse link between people and ecosystems – where a given user population travels for a given recreational activity (i.e., linking one user population to a variety of recreation sites, rather than one recreation site to its user population). Doing so requires a model of recreational choice (i.e., a random utility model) – that quantifies how users choose between and travel to different recreational sites. The distance potential recreational users are willing to travel to a given site differs based on the quality of the site and its substitutes, congestion, management policies, and place attachment, which may be specific to a given recreational activity. At this point, lacking a recreational choice model (Hunt et al. 2005, Hunt 2008), we aggregated user data from multiple sites to produce a preliminary map of visitor choices.

Linking the source, sink, and use data with flow models, we estimate the following indicators for recreation flows [26]:

1. Theoretical source and use. These are the values initially estimated by the source, sink, and use models *without accounting for flows*.
    a. Recreational attractiveness: The potential suitability, desirability, or capacity of a site for a particular recreational activity (this can be reflected as a relative "magnetism" for which a certain area can draw visitors).
    b. Potential recreational users: The residential location of users of a particular type of recreational activity.
2. Actual flow, source, and use. Actual recreation benefits provided and received *with a full accounting for source and use values and flows*.
    a. Recreational user flow: The movement of people toward recreation areas, based on transportation networks, recreational preferences, site quality, and a distance decay function.

[26] See Johnson et al. (2010) for a description of theoretical, possible, actual, inaccessible, and blocked source, sink, use, and flow concepts. When there are no sinks, possible and actual values are identical, and blocked flows, blocked sources, and blocked uses do not exist.

    b. Recreational use: The amount of recreational use actually seen at a recreation area when accounting for demand and spatial flows of visitors.
    c. Actual recreational users: The residential location of users who actually travel to sites via recreation flows to engage in recreational activities at source areas.
3. Inaccessible source and use. Theoretical values minus possible values; accounts for sources that do not provide and beneficiaries that cannot use an ecosystem service due to a lack of flow connections.
    a. Transportation restricted recreational use: Recreational areas whose current accessibility via transportation networks makes their use level more limited than their attractiveness alone would dictate.
    b. Transportation restricted recreational users: Users too distant from a recreational resource to benefit from it.

Table 9.5: Datasets used for the recreation flow models.

Layer	Case studies used	Source	Spatial extent	Spatial resolution	Year
Road speed limits & travel capacity	All	TIGER/Line files	United States	Unknown	2000
	San Pedro	AGIC	Arizona	Unknown	2009
Trails	San Pedro	BLM	SPRNCA	Unknown	Unknown

9.6 Caveats and directions for future research

Full implementation and testing of the recreation flow model has not yet been completed. Further work on testing our travel models against other approaches from the recreation literature (e.g., Lawson and Manning 2003) will likely improve the quality of the flow models and their relevance for management. Since recreation flows encompass human choice and transportation networks, a great deal of care needs to be placed on the underlying assumptions about total visitation, distance travelled, and visitor choice between substitute sites. This is especially important because underlying datasets about recreation choices and tradeoffs are rare, and modeling in the recreation literature has traditionally centered on statistical modeling rather than spatial mapping and comparison of ecosystem service tradeoffs.

Past spatial recreation modeling has often taken place at the site level, examining visitor movement within a single park, rather than at the landscape level, comparing visitor choice between parks (Lawson and Manning 2003). Random utility models (RUMs) have a long history of use in econometrics, recreation, and land use modeling research. A model is estimated based on survey data or observed behavior in which agents are assumed to select the recreation site that maximizes their utility for a desired recreation activity (Grijalva et al. 2002). An actor in a RUM evaluates a set of choices (i.e., parks) each with their own set of recreational activities, site characteristics, and park amenities to decide which location is best suited to provide the recreational benefits that a user is seeking. Further review of these statistical models is needed to determine their potential value in modeling recreation flows. RUMs explore the tradeoffs between site quality, travel cost, and other relevant factors in selecting recreation sites. This modeling approach has not been applied in broader mapping ecosystem service flows and tradeoffs. However, such models may offer an approach for modeling visitor choices between alternative recreation sites, which is not addressed by the current generation of ARIES recreation models.

The results of the recreation models, which map site quality for a given recreational activity, offer promise in being used to better explore ecosystem services tradeoffs. Forthcoming generations of the ARIES system will provide support for non-monetary comparisons of tradeoffs between multiple ecosystem services based on the concordance between stakeholder preferences and expressed land values (Villa et al. 2002). By offering different groups of users a way to compute and compare the overall utility coming from alternative levels of recreation, water supply, carbon sequestration, and flood control benefits, for example, managers can explore public preferences for ecosystem service provision under a range of realistic provision and use scenarios, since merely maximizing provision for all ecosystem services is rarely a realistic goal. Additionally, park managers often focus on recreation management tradeoffs for visitors, and their jurisdiction is limited by park boundaries. Yet because parks often end up providing a variety of other ecosystem services that flow across park boundaries to other beneficiaries, visualizing these benefits, beneficiaries, flows, and tradeoffs offers new means of improving ecosystem services-based management for protected areas and surrounding private lands alike.

9.7 Acknowledgements and additional contributors

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