Chapter 3

Forest Ecosystem Services: Water Resources

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INTRODUCTION

Since the publication of the Millennium Ecosystem Assessment (MEA 2005), awareness has steadily grown regarding the importance of maintaining natural capital. Forest vegetation is a valuable source of natural capital, and the regulation of water quantity and quality is among the most important forest ecosystem services in many regions around the world. Changes in forest cover alter the provision of water resource ecosystem services via influences on precipitation regimes, drinking water supply and purification, flood control, maintenance of streamflows, provision of recreational opportunities, and cooling water for thermoelectric power production. In this chapter, we describe how the ecosystem service values of water resources from forest landscapes can be estimated. Although much of the literature we reference is focused on the Southern United States, the concepts and methods described here are broadly applicable to other regions.

Information regarding the economic values of ecosystem services complements cultural and moral sentiments regarding the value of nature. This information can help governments, corporations, traditional cultures, and individuals make more informed decisions regarding the conservation of natural capital (Daily and others 2011).[3] In general, ecosystem service valuation studies seek to integrate ecosystem service assessments with economic analyses, and several stages of analyses need to be integrated in the valuation process (see fig. 3.1). Recommendations on the stages to be followed when conducting ecosystem service valuation studies, with special emphasis on scoping and defining ecosystem services, is presented in a guidebook especially designed for Federal resource managers (Olander and others 2015). Good summaries of the concepts and methods used to integrate economic analyses with ecosystem service assessments are also available (e.g., Bateman and others 2011). A great deal of information focusing on the economic analyses of ecosystem services can be found in the literature on non-market valuation (e.g., Champ and others 2003, Freeman and others 2014), and specific applications of ecosystem service valuation methods to water resources have been published (Birol and others 2006, Young and Loomis 2014).

Forests, Water Resource Ecosystem Services, and Human Well-Being

In this chapter, we refer to people who benefit from water resource ecosystem services as *beneficiaries* who may be divided into two groups.[4] First, people who use water resources are referred to as ecosystem service *users*. The benefits derived from water use may require capital and labor inputs, such as in the provision of drinking water supplies, and maintaining forest vegetation can help reduce the cost of inputs in the production of these services (e.g., Holmes 1988). The value to beneficiaries who use water resources for activities such as fishing or boating may also be augmented by enhanced water quality provided by forested landscapes (e.g., Johnston and Wainger 2015). Forest cover can also protect economic use values by reducing the risk of damages from floods and droughts. For example, mangrove forests help protect people and structures from flooding associated with storm surges (Barbier and others 2013, Das and Vincent 2009), and knowledge is increasing regarding the

[3] Although some forested watersheds in the Southern United States are protected by local, State, and Federal forest conservation areas (Caldwell and others 2014), most forest land is privately owned and subject to market forces that can create economic incentives to convert forests to alternate uses (Alig and Plantinga 2004). Where these pressures exist, compensating private landowners for the social values their forests provide to other water users may be an effective way to limit land use change and promote forest conservation (Holman and others 2007). Within the Southern United States, efforts are underway by national, State, and local conservation organizations to protect source waters, stream flows, and reservoir capacity in forested watersheds such as those located along the Mills River (NC), Upper Neuse River (NC), Catawba-Wateree River (NC, SC), Savannah River (SC, GA), and their tributaries.

[4] Many of the water related ecosystem services provided by forest landscapes are cultural ecosystem services, which are defined and described in more detail in chapter 2.

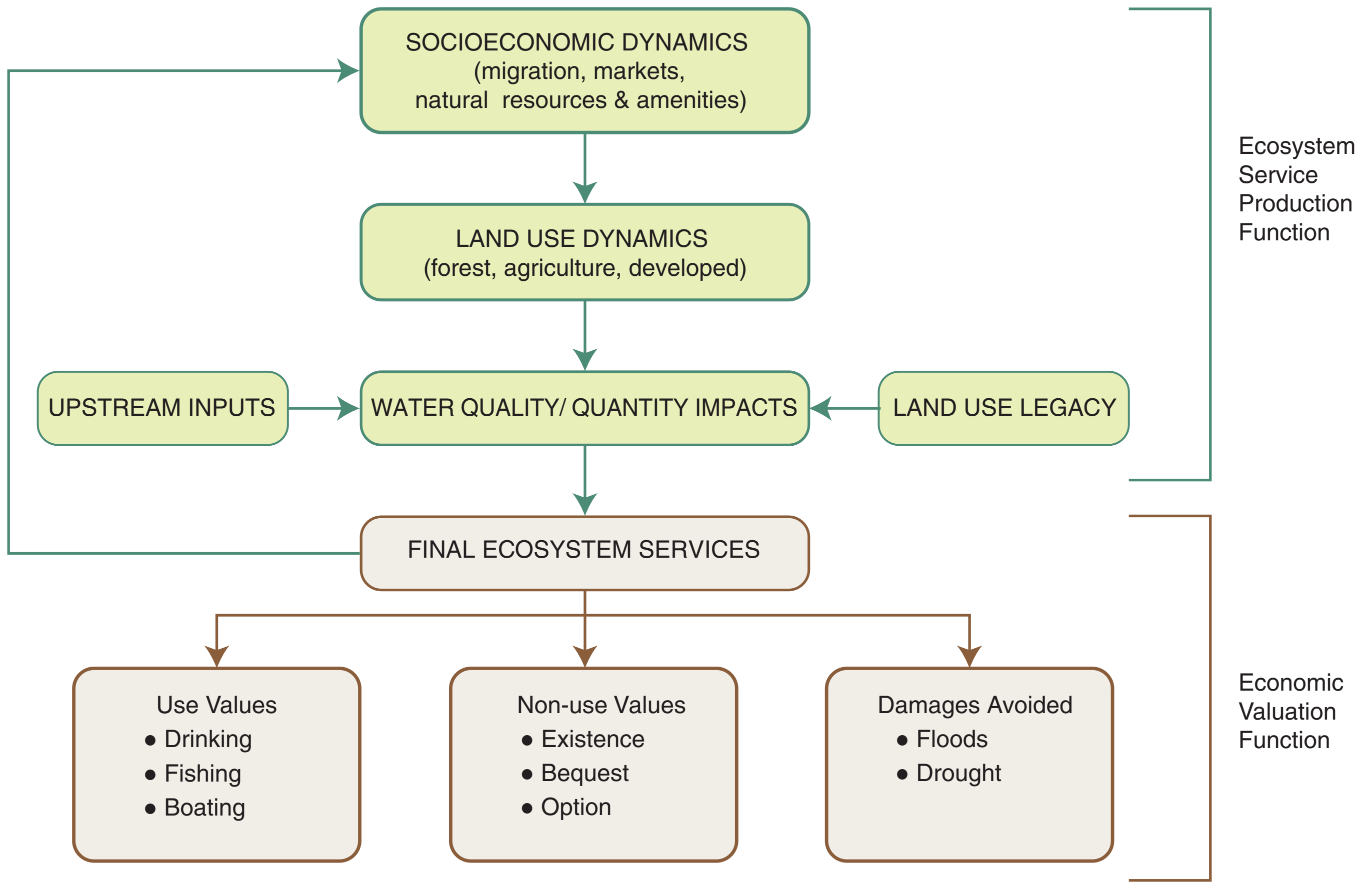

Figure 3.1— Ecosystem service assessments can be linked with economic valuation functions to measure how changes in forest landscapes affect the economic value of water resources.

impacts of large-scale deforestation and tree die-off on local and remote precipitation patterns (Devaraju and others 2015, Stark and others 2016).

The second group of beneficiaries from hydrological systems includes people who value water resources that they do not actually use; this group is referred to as *non-users*. Non-use benefits include the values associated with future water use as well as knowledge that specific water resource ecosystem services exist. A good example of non-use values of water resources is found in a study of the value of wetlands restoration in the greater Everglades ecosystem (Milon and Scrogin 2006). That study provided estimates of Florida residents' willingness to pay to conserve future water provisioning ecosystem services and to protect wildlife species that respondents may never see (Milon and Scrogin 2006). More generally, a meta-analysis of surface water resource values indicated that non-use values can be a substantial component of the total economic value of water (Johnston and others 2003).

Economic measures of human well-being (value) rely upon the theory of consumer demand. Water supplies from forest landscapes contribute to the satisfaction of demands for specific ecosystem services (such as water for drinking and recreation). Economic valuation of water resources from forest landscapes generally requires estimates of the numbers of beneficiaries associated with each water resource ecosystem service and the per capita value associated with each ecosystem service. Because the biophysical characteristics of watersheds are heterogeneous, as are the characteristics of beneficiaries, the ecosystem service values of water resources from forest landscapes can vary greatly across locations and need to be carefully addressed in large-scale ecosystem service assessments.

The sustainable production of water resource ecosystem services relies upon many primary and secondary processes. These processes are called *ecosystem service production functions* and the services of the natural environment that directly affect human well-being are called *final ecosystem services* (Bateman and others 2011, Boyd and Banzhaff 2007, Brown and others 2007, Fisher and others 2009, Johnston and Russell 2010).[5] That is, final ecosystem services are end products of nature to be distinguished from ecological processes and intermediate

[5] For example, final ecosystem services have been defined as "components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf 2007).

ecosystem services on which they depend. By recognizing that human values are derived only from final ecosystem services, we can avoid the problem of double-counting that would occur if primary and intermediate services were valued independently in an economic assessment.

Ecosystem services provided by water resources include both intermediate and final services, and it is important to understand how the economic values of these services can be accounted for. Consider the ecosystem services enjoyed by a specific group of beneficiaries: trout anglers. In this case, tree cover may enhance the value of water resource ecosystem services by cooling and purifying cold-water streams, thereby increasing trout abundance (Johnston and Wainger 2015). Although water quality also influences organisms providing food sources for trout, the food web is an intermediate ecosystem service that is not directly consumed or used by anglers and should not therefore be valued independently. Rather, it supports a final ecosystem service that contributes to angler well-being.

Human well-being derived from the consumption of an ecosystem service is generally estimated using revealed or stated preference methods (Champ and others 2003). Revealed preference methods rely upon observations of behavior to infer economic value. For example, observations regarding the frequency of fishing trips to streams with differing water quality, in combination with distances that people travel for fishing access, can be used to infer the value of water quality as it relates to angling (Huppert 1989, Whitehead 1993). A second example of revealed preferences is provided by observations on housing prices for homes situated next to lakes with varying water quality. When entered into statistical models, these observations can reveal the value of water quality to homeowners (Poor and others 2001). In contrast, stated preference models rely on survey questions to infer values. For example, questions may ask survey respondents how much they would be willing to pay (WTP) for specific water quality changes (Young and Loomis 2014).

Complex Landscape-Riverscape Systems

Land uses influence water supplies and, in general, managed and unmanaged forests supply the cleanest and most stable fresh water supplies relative to all other land uses (Jackson and others 2004, Jackson and others 2005, Brown and others 2008).[6] However, efforts to quantify the impacts of forests on water resource ecosystem services involves disentangling the components of landscape-riverscape systems, which becomes increasingly complex as the scale of analysis shifts from individual, small watersheds to multiple watersheds at broad spatial scales. Understanding the influence of land uses on hydrologic systems is challenged by a number of factors, including: (1) natural gradients (e.g., soil type and typography) are correlated with human gradients (e.g., land uses) across the landscape, (2) stream ecosystems respond to land use changes at multiple spatial and temporal scales, (3) watershed responses to land use change may be nonlinear, and (4) influences from current land uses are difficult to isolate from historical land uses (Allan 2004). Although advances in GIS and remote sensing technology are helping to address these challenges (Johnson and Host 2010), scientific knowledge of causal landscape-riverscape linkages over broad spatial scales remains at an early stage of development.

The key to linking ecosystem service assessments with economic values is to assure that final ecosystem services are measured using the same indicators that are included in economic valuation functions (Boyd and others 2016, Olander and others 2015).[7] For example, water quality affects the demand for various recreational activities (such as swimming or boating), and one indicator of water quality is clarity (or turbidity). In this case, turbidity levels might be considered as a final ecosystem service linking water supplies from forest landscapes with an ecosystem service valuation function. In general, science-based metrics that link ecosystem service assessments with economic valuation functions will help governments and businesses (such as water and electric utilities) understand how forest conservation would benefit their constituencies, whether they are taxpayers or ratepayers.[8]

In the following, we first present an overview of the literature describing relationships between forest cover and various metrics of water resource ecosystem services across different spatial scales. Next, we present an overview of economic principles, and illustrations from the literature, describing how the economic value of water resource ecosystem services can be measured. This is followed by a discussion of how forestry agencies might go about obtaining ecosystem service values of water resources from forest landscapes. This section includes information on available data sources and decision support models.

ECOSYSTEM SERVICE ASSESSMENTS OF WATER RESOURCES FROM FOREST LANDSCAPES

Forest Impacts on Streamflow, Soil Erosion, and Chemical Contamination

From the standpoint of evaluating alternative land covers, the impacts of forest cover on the hydrograph (i.e., the temporal pattern of streamflow after rainfall events) and baseflows are more important than total annual flow because forests typically require more water for evapotranspiration (and hence, have lower

[6] Point source pollution from small areas of non-forest land uses embedded in largely forested watersheds is an obvious exception to this generalization.

[7] The term *benefit relevant indicator* has been suggested to link outputs from ecosystem service assessments with economic values (Olander and others 2015).

[8] Within the Southern United States, organizations such as The Conservation Fund, The Nature Conservancy, Conservation Trust for North Carolina, and the Carolina Mountain Land Conservancy are working with water and electric utilities to conserve forests in the region to protect water quality and quantity.

total annual streamflow) than land uses that have less vegetation cover to intercept and transpire precipitation inputs (e.g., urban areas, annual crops, and so on) (Sun and Lockaby 2012). Forest soils act as a buffer against heavy storms, slowing the rise of streams and minimizing flood risk. During dry conditions, water that has percolated into the forest soil is released gradually for streamflow and groundwater discharge. Timber harvest and management can have negative impacts on quantity, quality, and timing; however, decades of research provide best management practices that allow forests to be managed in ways that minimize impacts on processes and conditions that protect water resources. Examples include riparian buffers, road building and surfacing standards, and stream crossing design.

Conversion of forest cover to urban or agricultural uses alters hydrology. It often results in enhanced peak flows and stormflows, while both enhanced and reduced baseflows have been reported (Amatya and others 2008, Boggs and Sun 2011, Sun and Lockaby 2012). Such conversions reduce evapotranspiration and soil infiltration capacity due to compaction and impervious cover such as buildings, roads, and parking lots, resulting in greater overland flow (O'Driscoll and others 2010). As a result, characteristic changes in hydrology following forest conversion include greater annual streamflow and higher peak flows (Bosch and Hewlett 1982, Crim 2007, de la Crétaz and Barten 2007, Hibbert 1967, McMahon and others 2003, Schoonover and others 2006). Decreased infiltration lowers groundwater recharge rates; thus, baseflows may be reduced (Calhoun and others 2003, Rose and Peters 2001, Wang and others 2001). However, reduced evapotranspiration from lower forest cover may offset some of the baseflow reduction. Changes in hydrologic response depends on the amount (i.e., percentage loss of forest cover) and the location (i.e., headwater vs. riparian) of conversion.

Forest watersheds typically have lower stream channel erosion due to lower stream velocity and peak discharge. In addition, forests stabilize soils, so soil erosion and sediment delivery to streams can increase following the removal of forest vegetation and loss of forest floor and roots (Jackson and others 2004). In some stream channels, it can be difficult to differentiate sediment contributions from current land use and historical agricultural use within the watershed, as the legacy effects of historical land use can be observed decades later (Jackson and others 2004). In addition to generating sediment export from terrestrial sources, hydrologic changes due to current land use conversions have the potential to re-suspend legacy sediment that accumulated in the stream beds decades ago. Hence, even streams within a heavily forested watershed may exhibit degradation due to historical land uses (Harding and others 1998). As noted in the introduction, these long-term "legacy" effects of land use can constrain the ability to correlate existing land use and some water quality parameters. However, urban and agricultural watersheds typically exhibit stream sediment concentrations that are much higher than forested watersheds (Clinton and Vose 2006, Lenat and Crawford 1994, Paul and Meyer 2001, Schoonover and others 2005).

A variety of factors determine the differences in erosion rates between land uses, but the most important factor is the location and severity of disturbance (e.g., amount and location of soil disturbance, road density and condition, and stream crossings).

Forested watersheds are also associated with low streamwater concentrations of most chemicals (Larsen and others 2013). Changes in stream nutrients can be observed at relatively small levels of forest loss. For example, increases in stream nutrients have been observed at levels of imperviousness as low as 5 percent (Schoonover and others 2005, Crim 2007). Since most forests are deficient in one or more elements, forested systems are generally effective in retaining inputs of nutrients in soils and biomass. Net export (output in streamflow minus precipitation inputs) of macronutrients from undisturbed forested catchments is often negative, a scenario that indicates accretion of forest biomass (Likens and Bormann 1995, Swank and Douglass 1977). Non-point source related health risks from urban and agricultural land uses include elevated nutrients (e.g., nitrogen and phosphorus), fecal coliform, e-coli, metals, pesticides, and personal care products (Klapproth and Johnson 2000, Larsen and others 2013, Paul and Meyer 2001). Forests also play a critical role in enhancing water quality in watersheds with mixed land use. Higher quality water draining forested portions of a watershed can dilute lower quality water and improve overall water quality (Clinton and Vose 2006).

Biophysical Measurements at Stream and Watershed Scales

Streamflow and groundwater recharge—As noted above, when precipitation exceeds evapotranspiration, excess water is available to recharge soil storage, recharge groundwater, and/or contribute to streamflow. Regulation of the quantity and timing of streamflow and the amount of groundwater recharge are important ecosystem services provided by forested watersheds. This amount and timing varies from forest to forest and across watersheds with different soils, bedrock, topography, land uses, and climatic regimes. For example, the amount of water consumed by evapotranspiration by forests in the Southern United States ranges from 47 to 90 percent of precipitation (Vose and others 2015). Hence, it is difficult to generalize the quantity of streamflow from the wide diversity of forest types and landscapes in the Southern United States; quantification requires measurement, modeling, or some combination of the two. In addition, small forest stands may be dispersed within a matrix of mixed land uses; separating the contribution of forests to overall streamflow at larger spatial scales can be a challenge. Groundwater recharge is also especially difficult to quantify. The best approximation uses an approach that proportions excess water based on bedrock geology and soils (Wolock 2003).

Streamflow data are available for only a small sample of forests and regions. The most comprehensive dataset for U.S. streams is that for the network of stream gauges maintained by the United States Geological Survey (USGS) (waterdata.usgs.gov/nwis/rt) that includes near real-time estimates of streamflow.

Often these gauges are placed on larger streams draining mixed land uses, so as noted above, determining the amount of streamflow contributed by forests is difficult to quantify. Other Federal agencies such as the Forest Service, U.S. Department of Agriculture, also monitor forest streamflow (e.g., the Coweeta Hydrologic Laboratory; the Santee Experimental Forests) on small watersheds, and these data may serve as a good approximation of streamflow for similar forest types in the same geographic regions and climate regimes. These data also serve an important role for testing and validating hydrologic models.

If direct measurements are needed, streamflow can be estimated using either permanent or temporary instruments that quantify volume (ft^3 s^{-1}). Permanent gauges—weirs and flumes—are expensive and labor intensive, but they can provide very accurate long-term and fine temporal resolution measurements of streamflow (water.usgs.gov/edu/measureflow.html). Streamflow can also be estimated using measurements of stage height derived either manually or with automated sensors. To determine flow volume, stage height (i.e., the height of the surface of the stream above a given fixed point) is combined with manual measures of stream cross-sectional dimensions and velocity to develop a rating curve, where volume (ft^3 s^{-1}) is estimated from stage height (water.usgs.gov/edu/measureflow.html). If measurements are long term, have a high temporal frequency, and are taken during storms, they can also be used to develop flow regimes and storm hydrographs. For example, derived parameters such as total annual flow (total water supply), minimum daily flow (water supply risk and ecological flows), maximum daily flow (erosion and flood risk), and peak flow (flood risk) are all relevant for economic valuation.

When direct measurement is not possible or appropriate, models can be used to estimate streamflow and groundwater recharge. These models range from highly detailed and calibrated process models, to simple empirical models. As expected, modeling skills, data, and computing requirements vary greatly across the full range of modeling approaches. Caldwell and others (2015b) provide an excellent summary and review of various hydrologic models.

Water quality metrics—Water quality parameters important to valuation of water resource ecosystem services include a combination of physical, chemical, and biological metrics. Because stream nutrients are generally low (especially relative to other land uses; Larsen and others 2013) in both managed and undisturbed forest watersheds, total suspended solids (TSS) and water temperature are among the most important physical water quality metrics to monitor in forested watersheds because they can be impacted by forestry practices and other disturbances (Jackson and others 2004). TSS is a combination of suspended sediment and organic matter and is highly correlated with turbidity. Where erosion is (or has been) high, TSS is mostly comprised of sediment (Reidel and Vose 2002). High TSS levels can have a negative impact on aquatic organisms and can impact water treatment costs and reservoir storage (Dearmont and others 1998, Forster and others 1987, Holmes 1988). TSS can be measured directly with grab samples or automated flow proportional samplers (e.g., ISCO samplers). The advantage of flow proportional samplers is that TSS is sampled across the hydrograph and provides a better quantification of TSS due the strong relationship between flow and TSS. Some of this variation can be captured with frequent grab samples, but this requires timing sampling to occur during all stages of the hydrograph. In either case, further analyses in the laboratory are required to quantify the amount of TSS. Sampling approaches that do not require laboratory analyses include in situ optical sensors (e.g., YSI data sondes) that can collect data automatically and be linked with the hydrograph.

Water temperature can be measured using spot measurements with a thermometer or measured and logged continuously with a thermometer and data logger. Stream temperature can also be predicted from air temperature (which is influenced by forest cover), although these relationships are weak in areas where groundwater springs contribute substantially to streamflow (Caldwell and others 2015). Thermal pollution can have negative impacts on aquatic organisms (especially in cold-water streams) and contribute to secondary effects such as algal blooms.

Chemical and biological metrics include concentrations of chemicals—such as nitrogen, phosphorus, pharmaceuticals, pesticides/herbicides, and heavy metals—and biological metrics such as fecal coliform and e-coli. High levels of these constituents can have direct negative impacts on human health (i.e., from direct contact from swimming) and increase costs of water treatment (Larsen and others 2013). Streamflow from most forested landscapes has very low concentrations of most chemical and biological parameters so quantification can be difficult and perhaps not necessary in most cases. However, where monitoring is required, streamwater can be sampled using grab samples or automated flow proportional samplers, by laboratory analysis. Because many constituents are highly reactive (i.e., they undergo biological transformations while being stored in sample containers), there are strict guidelines for sample processing and storage.

Water Resource Ecosystem Services From Managed Versus Unmanaged Forests

Considerable information is available on the impacts of forest management on streamflow throughout the United States (Brown and others 2005, Jones and Post 2004). In general, removal of the forest canopy increases streamflow for the first few years, but the magnitude, timing, and duration of the response varies considerably among ecosystems. Sometimes, streamflow returns to pre-harvest levels within 10 to 20 years; whereas other times, streamflow remains higher, or can even be lower than pre-harvest flow, for several decades after cutting (Jackson and others 2004). This wide variation in response patterns is attributable to the complex interactions between climate and vegetation; the former can vary considerably from dry to wet climatic regimes, and the latter can vary in structure and phenology (coniferous vs. deciduous forest).

It is further expected that due to demands from a shrinking land base and emerging bioenergy markets, management intensity will increase on new and established plantations to meet wood fiber demands. In the Southern United States, evapotranspiration varies considerably among managed and unmanaged forests, and among forest types (Vose and others 2015). In general, coniferous forests have higher evapotranspiration (and hence lower streamflow) than deciduous hardwood forests due to a combination of greater interception and transpiration (Ford and others 2011). This variation is important for evaluating the implications of increasing pine plantation forests or fast growing woody species such as Eucalyptus because the magnitude of the effects on streamflow depends on the species, forest types, or land use being replaced (King and others 2013, Vose and others 2015).

Water Resource Ecosystem Services From Headwater Forests

Water flows along topographic pathways to form a stream network—headwater forests are located at the beginning of the stream network (or highest elevation) and typically contain, ephemeral and 1st and 2nd order streams. In the Southern United States, some of the water that flows into the Piedmont originates from the mountainous and heavily forested landscapes of the southern Appalachian Mountains. High rainfall, deep soils, forest cover, and steep terrain provides a perennial flow of high quality water to streams and rivers in the Piedmont and Coastal Plain.

Some of the headwater forests in the South are located on National Forest System (NFS) lands, which contribute about 3.4 percent of the total surface water supply in the South that serves some portion of the water supplied to 19 million people (Caldwell and others 2014). In comparison, State and private forest lands in the South contribute about 32.4 percent of southern surface water supplies providing some proportion of the water consumed by nearly 50 million people.

Water Resource Ecosystem Services From Riparian Forests and Wetlands

Hydrological functions of forested wetlands may include flood mitigation or short-term surface water storage; and to a lesser extent than forested wetlands in other regions of the United States, they abate storm damages and recharge groundwater (National Research Council 1995, Walbridge 1993). Biogeochemical processes of wetlands include the transformation and cycling of elements and retention and removal of dissolved substances and thereby the improvement of surface, subsurface, and groundwater quality (Blevins 2004, National Research Council 1995).

Functions of riparian forests also include hydrological, biogeochemical, and habitat aspects. Many studies have shown that riparian forests help to stabilize stream banks and also trap pollutants such as sediment, nutrients, bacteria, fertilizers, and pesticides from runoff (Anderson and Masters 1992, Binkley and Brown 1993, de la Crétaz and Barten 2007, Klapproth and Johnson 2000, Naiman and others 2005, USDA National Agroforestry Center 2008, Vellidis 1999). The hydraulic connectivity of riparian zones with streams and uplands, coupled with enhanced internal biogeochemical processing and plant uptake, make riparian zones effective buffers against high levels of dissolved nutrients from uplands and streams, while geomorphology and plant structure make them effective at trapping sediments (Naiman and others 2005).

However, an intact riparian corridor does not ensure stream protection, as this relationship is dependent on several other factors including watershed characteristics such as topography, hydrology, soils, and vegetation, residence time of pollutants in the buffer, depth and variation of water table, upland land use practices, and climate (de la Crétaz & Barten 2007, Groffman and others 2003, Tomer and others 2005, Walsh and others 2005). Use of the riparian corridor also affects the relationship, although the impacts of periodic timber harvesting, livestock grazing, and recreation are limited if best management practices (BMPs) are implemented appropriately (Anderson and Masters 1992).

Quantification of Water Resource Ecosystem Services at Multiple-Watershed Scales

Scientific interest in understanding landscape scale relationships between land uses and the condition of water supplies has intensified during the past few decades. This trend can be attributed to the pace and significance of changes occurring in land uses and land cover around the world, advancements in the concepts and tools used by landscape ecologists, and the increasing availability of spatially referenced data on land use/land cover as well as indicators of stream condition (Allan 2004). A common approach is to estimate correlations between land uses and indicators of stream quality using statistical methods. Several studies of this type have been reported for the Eastern United States, all of which show a positive correlation between the amount of forest cover and various metrics of water quality.

An early example of this type of analysis conducted in the Mid-Atlantic region showed that the proportion of stream miles in riparian forest cover had a strong negative effect on total nitrogen and suspended sediment in streams (Jones and others 2001). A second example, using multiple regression analysis of data on land use and macroinvertebrate abundance (an indicator of high water quality) in North Carolina, highlighted the influence of the specific physiographic region on estimated land use/water quality relationships (Potter and others 2004). In particular, the authors reported that the amount of forest cover in riparian areas was a good predictor of macroinvertebrate abundance in the Coastal

Plain region, whereas total watershed forest cover was a better predictor of this metric in the Piedmont region. Notably, forest cover was not found to be a good predictor of macroinvertebrate abundance in the southern Appalachian Mountains. These findings indicate the potential danger of transferring research results from one physiographic region to another.

It has been noted that the analysis of correlations between categories of land use and metrics of water quality are subject to some important caveats (Allan 2004). First, the analyses of land use-water quality relationships implicitly substitute space for time. That is, statistical approaches have typically explained variations in water quality across hydrologic units using data on the varying proportions of land use within those hydrologic units. This approach may obscure important determinants of water quality within specific hydrologic units that occur over time as land uses within those units change. Therefore, forecasts of changes in water quality for specific land units based on these sorts of cross-sectional analyses run the risk of omitting important influences, such as historical land use practices specific to those land units, and may result in biased estimates. Second, because categories of land use implicitly sum to 100 percent, various measures of land use may provide equally good models. For example, an increase in forest cover, as measured across land units, implies that other land uses necessarily decrease. So, metrics of forest cover may simply be revealing the relative prevalence or absence of other land uses. Third, it has been shown that spatial correlations among land use variables (for example, forest and agricultural lands may tend to occur together) can bias the interpretation of estimated parameters (King and others 2005).

Fourth, the influences of land uses on water quality are often nonlinear and parameters estimated using linear statistical models may be biased. For example, water quality generally responds nonlinearly to the amount of agricultural land, with streams remaining in good condition until the proportion of agricultural land within a catchment exceeds 30-50 percent (Allan 2004). Within urban watersheds, a nonlinear response occurs when impervious cover exceeds about 10 percent of land area (Sun and Lockaby 2012).

The nonlinear response of water quality to land use demonstrates the importance of understanding spatial context in estimating how changes in forest cover affect hydrological systems. For example, throughout many regions in the Southern United States, forests are being converted to developed land uses and the amount of impervious cover is increasing. If the existing proportion of forest is less than the threshold value at which changes in impervious cover begin to substantially impact water quality, conversions to impervious cover may have minor impacts on water quality (fig. 3.2). In this case, linear estimates of the relationship between impervious cover and water quality degradation would overestimate the response. Above the threshold, linear estimates of the rate of change in water quality with respect to changes in impervious cover would underestimate the true relationship.

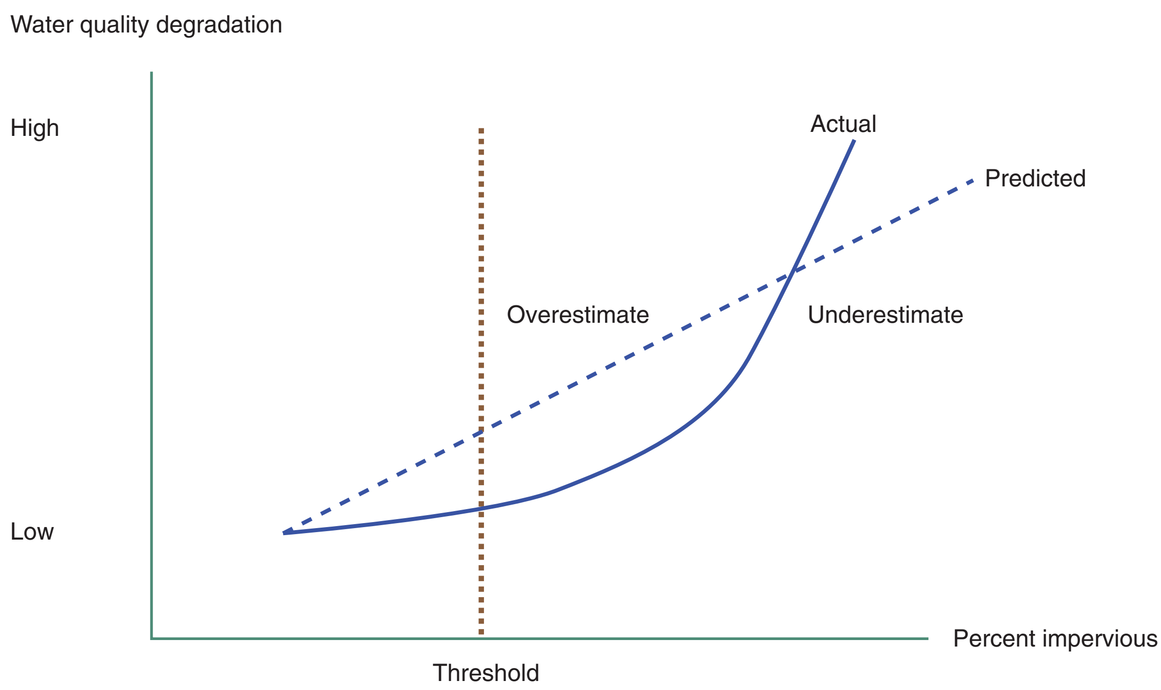

Figure 3.2—Landscape-riverscape production of water quality is nonlinear: assuming a linear relationship can result in potentially large errors.

Lastly, we note that scientific interest in the relationship between large-scale tree loss and alterations to the hydrological cycle is growing. Major structural changes in forest vegetation due to land conversion associated with afforestation, deforestation, forest degradation, desertification, and forest die-off are anticipated to alter complex, nonlinear feedbacks between land surfaces and the climate system in ways that are yet poorly understood (Bonan 2008). However, recent macrosystem ecological models linking forest losses with atmospheric fluxes indicate that changes in albedo and other components of energy balance will lead to significant increases/decreases in precipitation at both local and remote locations (Devaraju and others 2015). Better understanding of large-scale forest-atmospheric couplings is needed in an era of rapid and uncertain climate change (Stark and others 2016).

ECONOMIC VALUATION OF WATER RESOURCE ECOSYSTEM SERVICES

Water resources provide a variety of ecosystem services for both instream (e.g., fishing, boating) and diverted (e.g., residential, agricultural) uses. Some services conflict with each other, such as withdrawal for agricultural irrigation versus maintenance of natural flow regimes supporting endangered aquatic organisms. Other services are complementary, such as water stored in reservoirs for future residential consumption and water resource recreation such as swimming and boating. Decisions affecting the provision and allocation of water resource ecosystem services require measurements of the economic value of water in providing alternative benefits (Ward and Michaelson 2002, Young and Loomis 2014). Because forest cover influences water quality and quantity, land use decisions should consider how alternative land uses influence the suite of ecosystem services and values provided by water resources.

The concept of economic value often causes confusion in decisionmaking. This is because the economic value of a good or service often differs from its price. The market-clearing price for a given quantity of an ecosystem good or service can provide a good approximation of economic value if all of the inputs to its production are privately owned and the good is produced in a competitive market. This might be the case, for example, for bottled water produced from a privately owned spring. However, in many instances, hydrologic systems are public goods, not privately owned, and may be freely accessed (i.e., zero price). In this case, the economic value of water is what beneficiaries are willing to pay for it. WTP for public water resources is an expression of the demand for water resource ecosystem services by beneficiaries.

Valuation of Ecosystem Services: Production of Water Supplies

The beneficiaries of public water supplies include residential, agricultural, and industrial water users (Young and Loomis 2014). In the United States, water prices are often administratively determined and do not reflect actual supply and demand conditions. Statistical techniques have been frequently used to estimate water demand (WTP) functions that reveal the true economic value of public water supplies (Ward and Michelsen 2002).[9] In practice, WTP for water typically exceeds the amount paid for it, often by very large magnitudes (Olmstead 2010). Therefore, administered water prices do not provide reliable estimates of this ecosystem service value.

Estimates of the demand for public water supplies can be combined with information on the long-run marginal costs (LRMCs) of supplying increasing amounts of water to determine socially efficient prices for, and social value of, diverted water uses. LRMCs of water supply include the costs of collection, reservoir storage, treatment, distribution, anticipated future capital costs for new facilities, and the opportunity cost of water for other potential uses (Olmstead and Stavins 2009). Contributions of forest cover to reducing the LRMC of water supplies increase the social value of water by lowering the socially efficient price.

The ecosystem service value of water quantity can also be measured in hydrological systems where water is an input into a production process in which changes in water quantity ultimately influence productivity. For example, in many regions of the United States, water is used to irrigate crops. The economic benefits from alternative amounts of water being supplied for irrigation purposes can be measured using information on change in the value of agricultural crops produced and the change in the cost of production (Ward and Michelsen 2002).[10] A similar approach could be used to measure the economic benefits of water supplies to industrial users.

The impact of forest cover on flood risk is a topic of increasing concern as more extreme precipitation is expected to accompany a warming climate (Donat and others 2016). Economic damage assessments from flooding provide estimates of losses sustained by a variety of economic sectors including private households, industry, agriculture, and infrastructure (Merz and others 2010).[11] The greatest challenge in measuring the influence of forest cover on flood protection ecosystem services is to understand how

[9] As noted by Ward and Michelsen (2002), this method requires adequate variation in administered prices and quantities consumed in order to estimate water demand.

[10] Economists have shown how preservation of (tropical) forest cover boosts agricultural production by increasing baseflow (Pattanayak and Kramer 2001).

[11] These estimates typically include the costs of repair and recovery but do not consider how much people, or industries, would be willing to pay to avoid such damages.

forests influence water balances during extreme precipitation events. Understanding these relationships would allow the ecosystem service value of forests on flood risk to be measured using estimates of the damage cost avoided.

Valuation of Ecosystem Services: Production of Water Quality

Water quality can either be an intermediate input into a final ecosystem service or a final ecosystem service by itself. We first present an example of water quality as an intermediate input before going on to provide examples of water quality as a final ecosystem service.

Improvements in fish habitat—Consider an ecosystem service causal chain in which water quality influences fish habitat, which influences fish mortality and reproduction rates, which influence fish abundance, which influences the number of fish caught by anglers (Olander and others 2015). Because water quality is an intermediate input in the causal chain, the value of water quality to a specific class of beneficiaries (i.e., anglers) is estimated using information on its contribution to fish abundance and the economic value of fish abundance as an input to fishing.

Hundreds of economic studies have been conducted estimating anglers' WTP to catch fish (some studies are for marine and others are for freshwater resources). The results of these studies are summarized in a meta-analysis that allows WTP values for catching a variety of freshwater (bass, muskellunge, pike, trout) and anadromous (salmon) fish species to be estimated (Johnston and others 2006). An example of how these data can be used to estimate the ecosystem service value of riparian reforestation along rivers in a watershed in south coastal Maine is provided by Johnston and Wainger (2015). By calibrating the relationship between riparian tree cover and brook trout abundance, and using results from the metaanalysis, the authors concluded that each 47 acres of riparian canopy restoration per 1,000 ft^2 of river would increase the value per angler per fish caught by about 50 percent. The authors provide a very useful discussion of the many assumptions that were necessary to reach this conclusion. They also note that other ecosystem service benefits of riparian canopy restoration (such as aesthetics) were not estimated.

Water treatment costs—Water quality is an input that is combined with capital and labor inputs in the production of potable water. Drinking water that meets quality standards is valued by consumers and is also required by Federal standards. One approach to valuation is to calculate the additional cost of assuring that drinking water meets those standards with degradation of water quality at water treatment plant intakes. Several economic analyses have been conducted to evaluate the impact of water quality on the cost of water treatment (Dearmont and others 1998, Forster and others 1987, Holmes 1988, Murray and Forster 2001). Each study used turbidity (water clarity) as the metric of input water quality. In these studies, multiple regression analysis was used to isolate the impact of turbidity on water treatment cost by controlling for other relevant variables (e.g., volume of water treated, wage rates, electricity costs). Although the impact of turbidity on water treatment cost varied across studies, each study found that increases in turbidity in raw water resulted in higher water treatment costs.

Water treatment cost studies have been used to estimate the value of specific water quality changes in water basins other than those included in the original research studies (Elsin and others 2010). Using the Neuse River Basin in North Carolina as their study area, the authors estimated the net present value of future cost reductions to treatment facilities in this basin if specific turbidity level reductions were attained.[12] Cost reductions associated with turbidity reductions in this watershed of 5 to 30 percent were computed, resulting in cost savings ranging from approximately $1 million to $16 million. However, these authors did not describe how such reductions in turbidity levels might be attained.

Direct linkages between forest cover and water treatment costs have been recently explored in two studies. First, a study conducted in northeastern France using statistical analyses of spatially explicit data concluded that a 1-percent increase in regional forest cover (with equal reduction in agricultural land) reduced average water supply costs by about 1.3 percent (Abildtrup and others 2013). A similar approach was used in a study examining the impact of forest cover on water treatment costs in Malaysia (Vincent and others 2015). Using data on actual treatment costs and GIS data layers on virgin and logged forests, the authors found that avoiding conversion of 1-percent of virgin forest to non-forest use reduced treatment cost by 0.47 percent, and avoiding conversion of 1 percent of logged forest decreased treatment cost by 0.31 percent.

Within the United States, a recent study by the American Water Works Association (Warziniack and others 2017) surveyed 37 water utilities in forested ecoregions to assess the impact of land use on water treatment costs. The study paired survey data on chemical costs for treatment (typically alum or other coagulants, polymers, copper sulfate, corrosion control chemicals, and disinfection chemicals) with data on water quality at the intakes and land use in the watershed. Figure 3.3 shows scatterplots of their data. They found that costs increased with both total organic carbon (TOC) and turbidity, and that both TOC and turbidity decreased with forest cover (the relationship between TOC and forest cover was negative but not statistically significant). While

[12] This was accomplished by assuming that the total change in turbidity level would take 5 years to accomplish and would be sustained for 25 years.

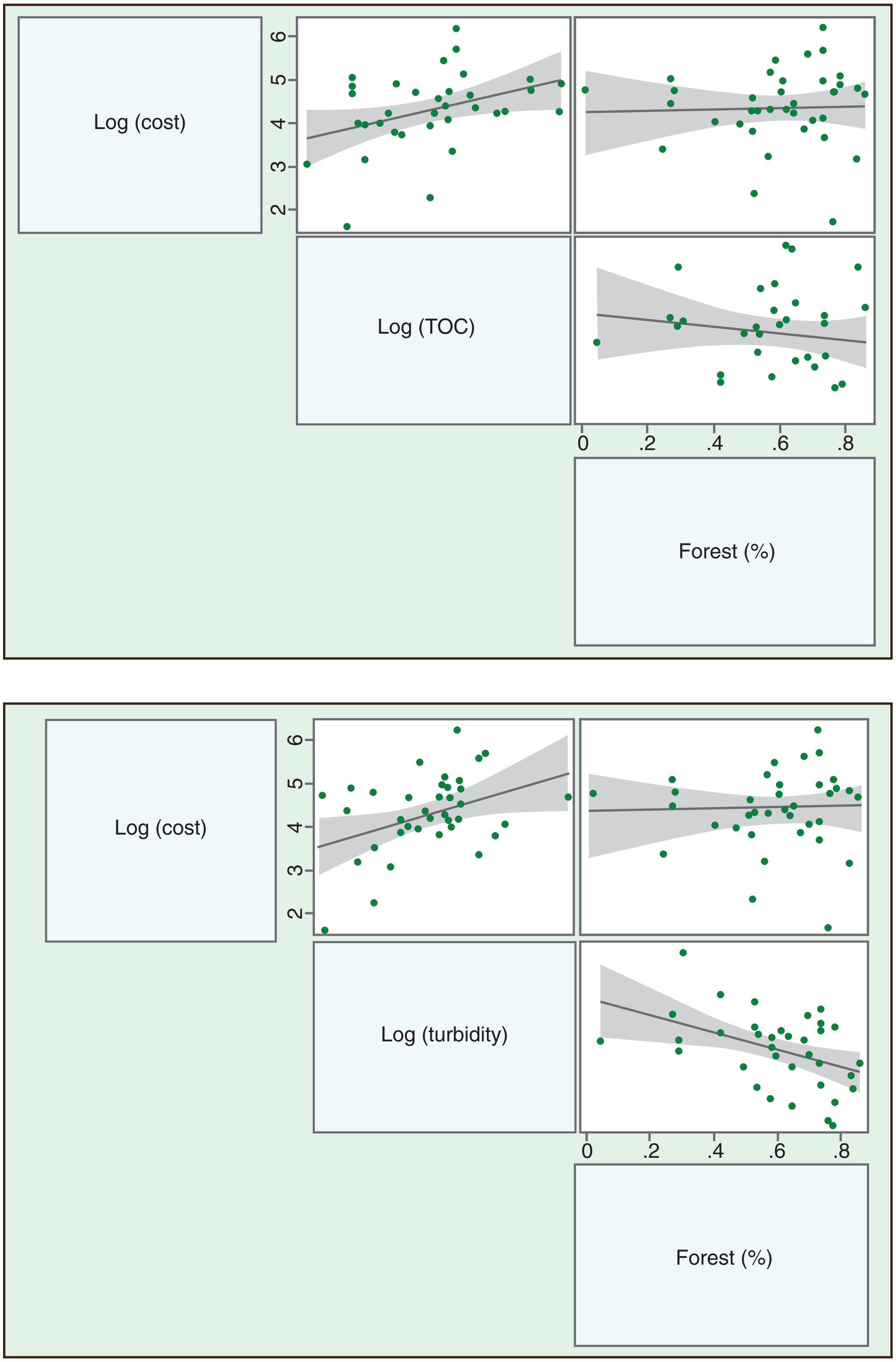

Figure 3.3— Empirical relationships between percent forest cover, total organic content (TOC), turbidity, and water treatment costs (cost) (Warziniak and others 2017).

the study was able to link forest cover to water quality, and water quality to treatment costs, there was too much noise in the data to directly link forest cover to treatment costs.

Water resource recreation and non-use values—Within the Southern United States, recreational uses of water resources are enjoyed by millions of people. About one-third of the population in the South, 16 years of age or older, engage either in boating or fishing activities and more than one-half engage in outdoor swimming (Cordell and others 1999). These usage rates are very similar to national averages and provide an indication of the importance of protecting water quality in the region and across the United States.

Hundreds of economic studies of the value of water quality have been conducted in the United States. Factors that influence the economic value of water quality have been identified using a statistical approach, known as meta-analysis, which summarizes the results of many previous studies (Johnston and others 2003, 2005, 2016; Van Houtven and others 2007). The fundamental conclusion of these studies is that the economic value of water quality depends on the characteristics of the water bodies being studied as well as the characteristics of the population of people who use or care about those hydrological systems. For example, WTP typically varies with the size of the water quality change and average household income.[13] Non-use values are also consistently found to be an important component of total economic value of water quality. However, the non-use values for water quality improvements are generally less than the use values.

GUIDELINES FOR ESTIMATING THE ECOSYSTEM SERVICE VALUES OF WATER RESOURCES FROM FOREST LANDSCAPES

The goal of ecosystem service valuation is to link, in a meaningful way, ecosystem service production functions (or assessments) with economic valuation functions (fig. 3.1). Here we describe the general steps to be followed in conducting large-scale (e.g., statewide) ecosystem valuation assessments of water resources from forest landscapes. The guidelines we present are similar to general ecosystem service valuation assessments described elsewhere (Johnston and Waigner 2015, Olander and others 2015).

Step 1: Scoping

The first step in assessing ecosystem service values of water resources from forest landscapes is to identify the objectives of the analysis (fig. 3.4) This includes a description of the policy or management problem facing decisionmakers, consideration of the general issues that need to be addressed by analysis, and articulation of alternative approaches to providing desired results. Examination of trends in forest cover and the condition and use of water resources could help identify current and emerging problems. Engaging stakeholders at this stage can clarify how they would be affected by any potential changes in the provision of water resource ecosystem services. The result of the scoping phase will include a detailed description of the specific hydrological systems to be evaluated, what ecosystem services are to be measured and included in an economic assessment, and a good understanding of the specific groups of beneficiaries who would be impacted by changes in the flows of water resource ecosystem services.

Statewide analyses of alterations in water resource ecosystem services resulting from changes in forest cover need to identify specific locations where forest changes are anticipated to occur, perhaps using models of land use change. This element is critical in that forest loss (or gain) can have differential impacts on water quantity or quality depending upon the existing levels of forest cover and the physiographic region (Allan 2004, Boggs and others 2015, Sun and Lockaby 2012). This geographic-specific information could then be used to help identify the beneficiaries of changes in the flow of ecosystem services.

Step 2: Data Collection and Analysis

Once the goals and objectives of an ecosystem service valuation assessment have been identified, and the specific ecosystem services to be included in the analysis have been selected, the next step is to identify information needs. This step will likely involve an extensive review of the literature relevant to the specific ecosystem services selected. Ultimately, it is necessary at this stage to decide whether primary data, secondary data, or some combination of both, will be used for analysis. This requires decisions to be made on specific modeling approaches that will be used.

Primary data collection and analysis—Sophisticated measurement tools and modeling approaches are available to measure or predict water resource condition for specific forested watersheds (as described previously). These approaches are often used to assess the success or failure of a land management activity in research settings, to measure the effectiveness of BMPs, or to identify critical watersheds for conservation purposes. There may be some circumstances where direct measurements would be useful and a worthwhile investment for valuing water supplies from forest landscapes (e.g., comparing the relative value of land use choices for a specific watershed).

[13] We note that Johnston and others (2016) found that economic value of water quality improvements in the Southeast United States exceeded values in other regions, and that a 1-percent increase in water quality increased water values by about 0.28 percent. This estimate of elasticity is similar, but smaller than, the elasticity estimate (about 0.42) reported for water quality in an earlier meta-analysis (Van Houtven and others 2007).

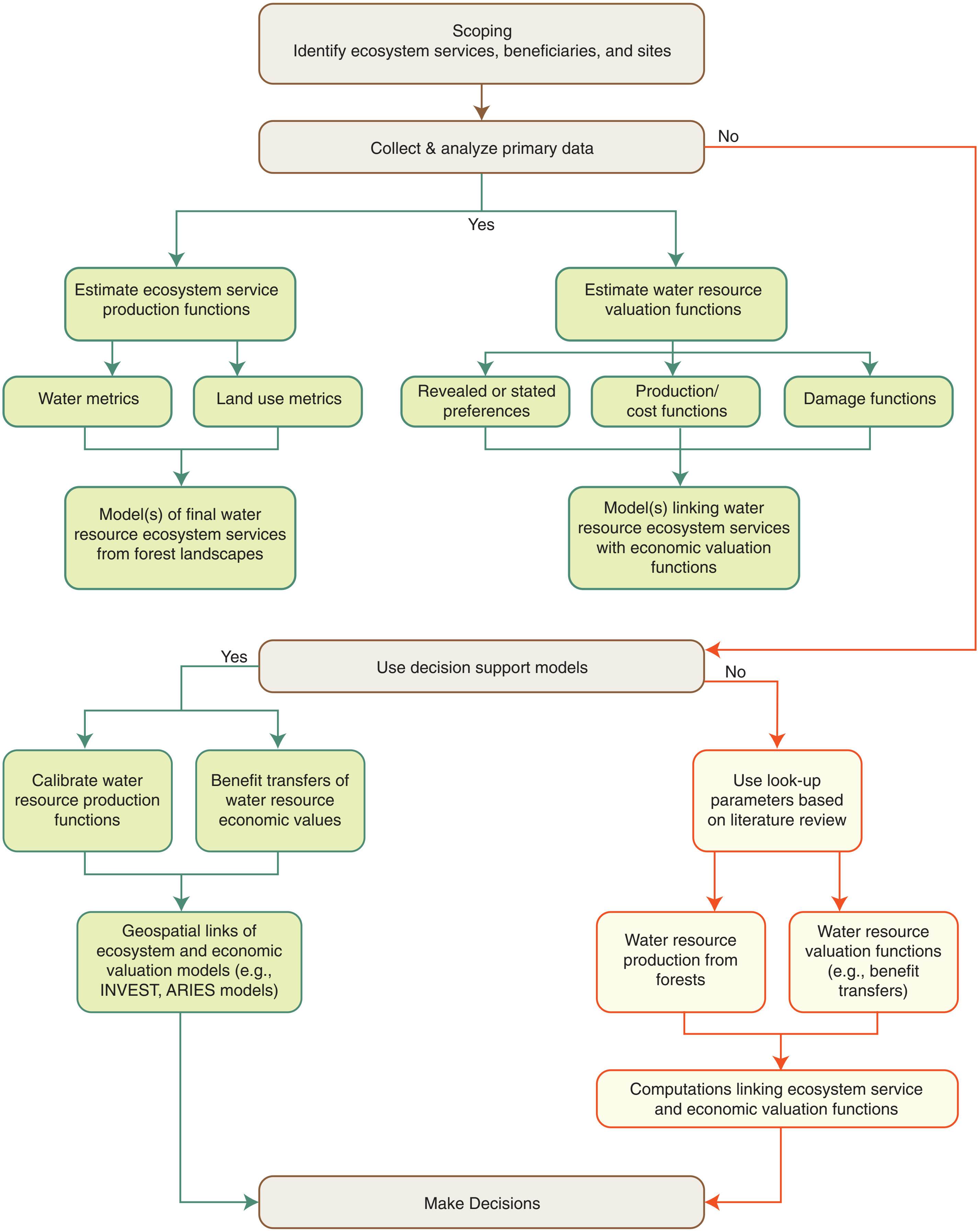

Figure 3.4—Decision tree for estimating water resource economic values from forest landscapes.

Similarly, situations may arise where measurement of economic values of specific watersheds using revealed or stated preference methods would be warranted.

If primary data are to be collected and analyzed to provide a large-scale ecosystem valuation assessment, a sampling plan must be developed so that experimental results can be generalized to sampled populations. Watershed sampling plans should recognize and capture the diversity of forest types occurring in and across watersheds (such as riparian, plantation, and wetland forests) and the protection status of forests (such as public forests, private forests, and conservation areas). Economic valuation studies of forested watersheds need to consider the population of beneficiaries who value water resources (such as recreational users, people utilizing public water supplies, and people who value the existence of healthy hydrologic systems). It is also essential to keep in mind that the ecological indicators of inputs to the economic valuation functions (such as water clarity or frequency of floods) must be the outputs of the ecosystem production function. Without well-specified indicators of ecosystem services, it is not possible to estimate the value of water supplies provided by forest landscapes.

Secondary data collection and analysis—The major constraint to collecting and analyzing primary data at the statewide level is obtaining an adequate research budget, as such studies could easily cost hundreds of thousands of dollars. Thus, in most situations, large-scale analyses of the ecosystem service values of water from forest landscapes will rely upon the collection and analysis of secondary data (fig. 3.4).

The simplest approach to the use of secondary data would be to conduct a literature review, beginning with the literature presented in previous sections of this chapter. The goal of the literature review is to identify published scientific studies conducted in regions similar in character to the region that is the focus of the valuation study (called the *policy area*). Relevant studies can then be used to implement a process known as a *benefit transfer* (Johnston and others 2015, Rosenberger and Loomis 2003). The simplest benefit transfer methods apply unit values (such as average economic value per unit consumed) from an original study site to the policy area under consideration. This method ignores differences in characteristics between the original study site(s) and the policy site(s), as well as differences in the characteristics of beneficiaries and can lead to large errors. Consequently, it is not recommended unless no alternatives are possible or if the characteristics of the policy site and original study site are very similar.

A better approach is to use a *function transfer* that is based upon functions or statistical models developed in original studies that define relationships between dependent variables (such as WTP for water quality improvements) and a set of explanatory variables (such as the characteristics of the hydrologic system that was valued and the characteristics of beneficiaries). Function transfers are preferable to unit value transfers as they help to match the conditions in the original study area to the policy area. They require information on the values of the explanatory variables in the policy area.

Even more informative than the function transfer approach is to use the results of meta-regression analysis, which is a statistical model that summarizes the results of numerous original studies. Applications of this approach are limited to cases where an adequate number of high quality original studies are available to estimate a meta-regression model. Fortunately, for the purpose of economic valuation of water quality, several meta-analyses have already been conducted which may provide usable information for policy areas of interest (Johnston and others 2003, 2005, 2016; Van Houtven and others 2007).

For the purposes of ecosystem service valuation of water supplies from forest landscapes, a benefit transfer process could be decomposed into the following steps:[14]

1. Confirm the feasibility of conducting a benefit transfer. This methodology depends upon the availability of high quality information from primary studies on both the ecological production function and the valuation function. If such information is not available, then primary data collection methods should be considered. Also, primary studies often report results for “iconic” sites, such as might be found in a National Park or Wilderness Area. If the policy site to which values are to be transferred to is more ordinary, then benefit transfer may not be appropriate.

2. Confirm that specific ecosystem services and beneficiaries are similar. If, for example, an original study provides economic values for improvements in water quality to recreational users of a reservoir, then using those values to describe the benefits of improvements in water quality to people who use the reservoir for drinking water would probably induce large errors.

3. Evaluate how the effects of changes in forest land use on water quality or quantity parameters might be quantified using secondary studies. Similar to concerns regarding the adequacy of economic value transfer methods, this step necessitates the availability of high quality information from primary studies linking forest cover with water quality or quantity.

4. Assure that the ecosystem service representing the output of the hydrological system(s) being evaluated is identical, or very similar, to the input(s) included in the economic valuation function.

[14] The following is largely based upon Johnston and Wainger (2015).

5. Choose the value transfer method and conduct the transfer. Decide whether unit values, transfer functions, or the results of meta-analyses are to be used both for the ecosystem service assessment and for economic valuation.

6. Use the selected methods to compute how historical or anticipated changes in forest cover in the policy area impact resulting water resource economic values. This step necessitates identification of the number of beneficiaries of the selected ecosystem services so that economic values can be aggregated over that population.

7. Conduct sensitivity analysis. Many assumptions may have been made in conducting both the ecosystem service assessment (i.e., impact of changing forest cover on water quantity and quality) and in economic valuation. Repeatedly recalculate ecosystem service values under alternative assumptions.

8. Report results. The results of analysis are reported to the relevant stakeholders. This could be policymakers, land managers, scientists, or the general public. Comparisons of management or policy alternatives may be facilitated by the use of *alternative matrices* or maps describing how ecosystem service values are impacted under alternative scenarios (Olander and others 2015).

Although benefit transfer methods can save costs, and are commonly used for ecosystem service valuation, researchers should be aware of two potential sources of error that can be introduced using this approach (Johnston and Wainger 2015, Rosenberger and Stanley 2006). *Measurement errors* occurring in primary studies used for transfer will carry over to transferred values, and these errors can be significant. Further, lack of similarity between site characteristics, valuation context, and human populations at the study sites and policy sites can cause *generalization errors*. Researchers using benefit transfer methods for valuing ecosystem services should attempt to minimize these errors to the extent possible.

Models and water resource data—In addition to using secondary studies to conduct value transfers of ecosystem service production and valuation functions, data and models are available to assist more complex ecosystem service valuations and decisionmaking. The emerging prevalence of spatially explicit data and GIS systems has supported the development of decision support tools to help agencies understand how management directed toward one ecosystem service affects other natural resource values (e.g., Bagstad and others 2012). The ecosystem service components of these models are typically complex and data intensive and require detailed information on selected ecosystem processes. In contrast, the economic component of these models is typically very simple and relies upon unit value transfers of ecosystem service values (Johnston and Wainger 2015).

Usually, preserving forests to maintain water quality improves other ecosystem services; that is, many ecosystem services are complementary to the provision of clean water. Sometimes, however, investments in watershed health come at the expense other ecosystem services. Grazing cattle, for example, is an important economic use on public lands, but it has traditionally had negative impacts on water quality.

Decision support tools have been used to understand spatially explicit ecosystem service flows and tradeoffs. The most widely known model for studying the landscape's ability to provide ecosystem services and for analyzing tradeoffs in management activities is InVEST, produced by the Natural Capital Project (http://www.naturalcapitalproject.org/invest/). InVEST can be run independently, but it is most often used as a plug-in to a GIS program (for example, the ArcGIS ArcToolbox environment). The model populates predetermined ecological production functions with user-provided data to determine economic values of ecosystem services. InVEST models have been developed that link land use and land cover with water quality and quantity (Kareiva and others 2011). Nelson and others (2009), for example, use InVEST to evaluate impacts of land use changes on water quality, peak storm runoff, soil conservation, carbon sequestration, biodiversity, and marketed goods (timber, housing) in the Willamette Basin, Oregon. InVEST water quality and quantity models may be relatively simple (e.g., using data representing annual averages for entire watersheds) or more sophisticated (e.g., using measures of daily hydrology and water resource infrastructure). In general, the economic valuation component of these models is very simple and depends upon benefit transfer of unit values.

Balances and tradeoffs between water supplies and carbon sequestration can be evaluated using a decision support tool (Water Supply Stress Index) developed by the USDA Forest Service (http://www.forestthreats.org/research/tools/WaSSI). This model can be used to predict how climate, land cover, and changes in human populations may impact water availability and carbon sequestration at the watershed level. Other models capable of showing spatial tradeoffs in ecosystem services include ARIES (http://www.ariesonline.org/) and the Forest Ecosystem Services Toolkit (FEST) (http://forestecoservices.net/).

The U.S. Environmental Protection Agency has developed a public-domain Watershed Management Optimization Support Tool to model the effect of management decisions on watersheds.

This decision support tool is designed to help local water resource managers and planners evaluate the economic costs, benefits, and tradeoffs involved with green infrastructure and land conservation decisions. The model addresses water flows but does not consider water quality. (https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=261780).

Agencies may also decide to conduct their own analyses of the value of water resource ecosystem services from forests using other data available for download. Some of the data sources that could be considered include the following:

Forests to Faucets — (http://www.fs.fed.us/ecosystemservices/FS_Efforts/forests2faucets.shtml). These data are provided by the Forest Service and are available for download from the Geospatial Data Gateway that includes 12-digit watershed boundary (HUC-12) data.[15] Within each HUC-12, data are provided on a number of variables including: population served by surface water intakes, mean annual water supply, percentage of forest, percentage of protected forest, percentage of National Forest System forest, percentage of private forest, percentage of forest highly threatened by insects and disease, percentage highly threatened by development, and percentage highly threatened by wildland fire.

Safe Drinking Water Information System — (https://www.epa.gov/waterdata/safe-drinking-water-information-system). These data include information on public water systems including: water quality violation information for water systems, enforcement actions by States, and sampling results for unregulated contaminants and for regulated contaminants when the monitoring results exceed allowed level.

EnviroAtlas — (https://www.epa.gov/enviroatlas). Data and interactive tools are available for understanding the benefits that people receive from ecosystem services. Geospatial water resource data provided at the HUC-12 level include variables such as: agricultural water use, domestic water use, industrial water use, number of aquatic animal species, number of aquatic plant species, percentage of forest, percentage of forest land in buffer, percentage of cropland, percentage developed, percentage impervious, stream length, and stream length impairment.

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[15] The United States is divided into nested hydrologic units. Each unit is identified by a unique Hydrologic Unit Code (HUC), ranging from regions (HUC 2-digit codes) to subwatersheds (HUC 12-digit codes). For hydrologic unit maps, see: http://water.usgs.gov/GIS/huc.html.

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