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Trees At Work: Economic Accounting for Forest Ecosystem Services in the U.S. South

Erin O. Sills, Susan E. Moore, Frederick W. Cubbage, Kelley D. McCarter, Thomas P. Holmes, and D. Evan Mercer



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Abstract

Southern forests provide a variety of critical ecosystem services, from purification of water and air to recreational opportunities for millions of people. Because many of these services are public goods with no observable market value, they are not fully accounted for in land use and policy decisions. There have been several efforts to remedy this by estimating the total value of forests in different States, with each effort including different bundles of ecosystem services and using different valuation methodologies. In this guide, we propose a more consistent and theoretically sound approach to 1) quantifying annual flows of ecosystem services, 2) developing a spatial catalog of the marginal values of changes in those flows, and 3) accounting for the total value of ecosystem services lost or gained as a result of changes in forest ecosystems. Four chapters—on cultural services, watershed services, air quality and carbon, and provisioning of non-timber forest products—provide guidance on best practices for quantifying and estimating the values of these services as provided by forests. Expert panels were convened to write each chapter. The guide as a whole was developed with input from the Southern Group of State Foresters and stakeholder meetings held in 2014 and 2015.

Keywords: ecosystem services, non-market valuation, green accounting, southern forests, non-timber forest products, watershed services, air quality, cultural services.

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Chapter 1

Introduction

Erin O. Sills, R. David Simpson, D. Evan Mercer

MOTIVATION

Given the economic importance of wood products in the U.S. South, the value of southern forests as timberlands is well known and acknowledged in State policy decisions. However, timber represents only part of the value of forests, and forestry leaders are increasingly interested in quantifying the full value of the South's forests, i.e., the value of all final ecosystem services produced by forests, including cultural (non-tangible) benefits, regulation of environmental quality, and provision of services (timber and non-timber goods). Recent efforts to quantify forest ecosystem services (FES) at the State level have considered different sub-sets of these services and employed different methodologies (Escobido and Timilsina 2012, Moore and others 2011, Paul 2011, H. Simpson and others 2013), which have limited the credibility of the results, the utility of cross-state comparisons, and the ability to report regional values. To address these issues, this guide describes best practices for quantifying four categories of FES (cultural services, regulation of water quantity and quality, regulation of air quality, and provisioning of non-timber goods) and a framework for estimating the total value of gains or losses due to changes in FES. While these four categories of services have been identified as the most important to stakeholders in the forestry sector in the Southern States, they are not comprehensive, e.g. pollination services are excluded here but discussed elsewhere (Iovanna 2017). In this introductory chapter, we explain the increasing interest in ecosystem services, discuss the motivations for valuation of FES in particular, and place this work in the context of economic accounting.

The primary intended audience for this report is State forestry officials charged with requesting, selecting, guiding, and evaluating the results of FES assessments in their states. We assume prior knowledge only of introductory micro-economics or forest economics. Thus, this report is not intended as a comprehensive guide to actually carrying out an assessment, but rather to obtaining an assessment that is methodologically sound and policy relevant. For readers who are interested in expanding their understanding of approaches used to assess and value forest

ecosystem services, we suggest consulting a recent state of the science review provided by the Northern Research Station, U.S. Forest Service (Binder and others 2017).

ANTECEDENTS

Interest in the economic value of changes to natural ecosystems has a long history, beginning in the 1960s (e.g., Clawson 1959, Krutilla 1967) and blossoming in the 1980s and '90s (Banzhaff 2010, Liu and others 2010). The 21st century has brought exponential growth in scientific publications on ecosystem or ecological services (Fisher and others 2009). One catalyst for this growth was a controversial article in the journal *Nature* by Costanza and others (1997), which has been cited over 15,000 times according to Google Scholar. Using benefit transfer techniques (often improperly [Plummer 2009]), Costanza and others (1997) estimated the world's ecosystem services to be worth \$33 trillion, \$15 trillion more than Gross World Product in 1997. A 1998 special issue of *Ecological Economics* contained eight stinging commentaries on Costanza and others (1997), while a 1998 column in *Nature* (vol. 395, page 430) called the article a "box office success ... panned by the critics" and "a political document ... of no use for policy analysis or implementation." Costanza and others (2014) conceded that the article was "clearly an awareness-raising exercise with no specific policy or decision in mind," and it certainly did raise awareness both of the value of ecosystem services and of the potential pitfalls in attempting to estimate total values.

In fact, the use of ecosystem service values in policy and management decisionmaking has been limited (Adamowicz 2004, Liu and others 2010, Laurans and others 2013, Rogers 2015). Laurans and others (2013: 241) concluded that the value of ecosystem services typically "receives no more than a cursory reference in the form of an expected, proposed or desired use," while Liu and others (2010: 73) state that "the contribution of ecosystem service valuation to ecosystem management has not been as large as hoped nor as clear as imagined." Recent Federal policy initiatives, however, have generated new interest

in using estimated values of ecosystem services as inputs to policy and management decisions (Richardson and others 2015). For example:

- In 2011, a report by the President’s Council of Advisors on Science and Technology (PCAST 2011) recommended:
 - establishing a Quadrennial Ecosystems Services Trends (QuEST) Assessment and
 - requiring Federal agencies to include valuation of impacts on ecosystem services in planning and management decisions.
- The U.S. Forest Service 2012 Planning Rule requires evaluation of ecosystem services associated with alternatives for managing the National Forests (USFS 2015).
- In 2013, the White House released “Principles and Requirements for Federal Investments in Water Resources” (CEQ 2013), requiring Federal agencies to use an ecosystem services approach to evaluate Federal investments in water resources.
- In 2015, the Council on Environmental Quality issued a memorandum directing all Executive Departments and Agencies to document how they would incorporate ecosystem services in their activities (CEQ 2015).

Most of these Federal guidelines call for agencies to consider how policy, management, and investment decisions affect flows of ecosystem services, quantified in terms of the value of those changes.

There is also increasing interest in mainstreaming ecosystem service valuation into accounts, or “green accounting.” Progress has been made both internationally, e.g., by the G8 (TEEB 2015), the European Union (Brower and others 2013), and multilateral organizations (WAVES 2015), as well as at the State level, e.g., by the Ecosystems Services Working Group (ESWG)

in Maryland (USDA 2015) and as reflected in adoption of the Genuine Progress Indicator by 20 States (Ceroni 2014). These efforts typically estimate total values of ecosystem services based on existing valuation literature.

PREVIOUS ASSESSMENTS IN THE U.S. SOUTH

Four Southern States have recently completed assessments of ecosystem services and FES in particular. The scope and findings varied from State to State, as summarized in table 1.1 and described in more detail below.

Following a scoping study by the Virginia Department of Forestry in 2006, the Piedmont Environmental Council funded a Yale University study to estimate the value of all ecosystems in Virginia. Annual ecosystem service benefits from Virginia’s forests were estimated to be \$15.3 billion (\$880/acre/year), as reported in table 2 of Paul (2011). Choosing a narrower scope, the Florida Forest Service commissioned the University of Florida to quantify the most important values generated by forest lands enrolled in the Florida Forest Stewardship Program. The present value of the ecosystem services from the 437,800 enrolled acres was estimated to be \$2.07 billion (\$5,030 per acre or \$151/acre/year at the assumed 3-percent discount rate) (Escobedo and Timilsina 2012). Funded by the Georgia Forestry Association, the University of Georgia assessed the value of all private forest lands in Georgia, resulting in an estimated annual value of \$37.7 billion (\$1,709/acre/year) (Moore and others 2011). The most recent effort, commissioned, funded, and implemented (with collaborators) by the Texas A&M Forest Service, found Texas’ forest ecosystem services to be worth \$92.9 billion/year (\$1,489/acre/year) (Simpson and others 2013). The wide range in the estimated annual value of ecosystem services generated by an average acre of forest land (from \$151/acre/year in Florida to \$1,709/acre/year in Georgia) reflects both methodological differences and differences in the value of forests across the four States.

Table 1.1—Previous assessments of forest ecosystem services in Florida, Georgia, Virginia, and Texas

	Florida (Escobedo and Timilsina 2012)	Georgia (Moore and others 2011)	Virginia (Paul 2011)	Texas (Simpson and others 2013)
Value estimate	\$2.07 billion (\$151/acre/year)	\$37.7 billion (\$1,709/acre/year)	\$15.3 billion (\$880/acre/year)	\$92.9 billion/year (\$1,489/acre/year)
Scope	FSP lands	Private forests	All forests	All forests
Services valued	Water, carbon, timber, habitat	Water, climate, habitat, pollination	Water, carbon, forest products, habitat, recreation	Watershed regulation, climate, biodiversity, cultural values

FSP = Forest Stewardship Program.

One reason for the wide range in values is the different scopes of the studies. For example, the Florida study focused on the value of “the components of forests that are directly enjoyed, consumed, or used to produce specific, measurable human benefits,” while the Georgia study used the more general concept of “ecosystem services as the things nature provides that are of direct benefit to humans.” The authors of each State study made different choices about which services to include; for example, only Florida and Virginia included timber, and only Georgia and Virginia included pollination services. Most critically for the total value estimates, only Texas and Georgia included estimates of the non-use value of forests (e.g., aesthetic and cultural benefits). However, all studies estimated the value of forests for protecting water quality, regulating water flow, regulating climate change via carbon sequestration, and providing wildlife habitat or biodiversity. Water (including both quality and quantity dimensions) is a key component of the total value in all States: 66 percent in Florida, 54 percent in Georgia, at least a third in Virginia (varying by forest type), and 14 percent in Texas. This is consistent with findings from the stakeholder workshop described below, in which participants identified water as the most important ecosystem service to include in valuation studies.

The four studies varied in their methods. Although all used benefit transfer (or “value transfer”) (Richardson and others 2015), the Virginia study only transferred values from published revealed preference studies, while the other studies drew on a much broader literature. All of the studies transferred “unit values” for some services, multiplying the value (or average value) per acre reported in the literature by the number of acres of a given forest type. They varied in how far they were willing to extrapolate values from the literature to different forest types and different locations. For the Florida study, the researchers estimated meta-regressions to obtain values of water and habitat adjusted to conditions in Florida. The Florida study also used the spatial modeling tool InVEST (Integrated Valuation of Environmental Services and Tradeoffs) for water, nutrients, and timber. In Georgia and Texas, original stated preference surveys were conducted to estimate non-use values.

The studies also reflect different approaches to disaggregating forests in order to calculate values. In Georgia, researchers divided private forests into categories based on forest type, riparian status, rare species abundance, scenic visibility, public land buffer, development class, and geographic region. They then transferred values per acre of each ecosystem service from the literature for each category of forest. The other State studies differentiated per acre values differently for each ecosystem service, e.g., using FIA (Forest Inventory and Analysis) forest types for carbon, watersheds or proximity to water for watershed services, and broad regions for other services. In some cases,

researchers assumed that the supply of ecosystem services varies across regions, whereas in other cases, the demand was also assumed to vary. For example, the Florida and Texas studies estimated watershed and cultural values by multiplying household willingness-to-pay (WTP) by the number of households living in each region.

MOTIVATIONS FOR QUANTIFYING THE VALUE OF FES IN THE U.S. SOUTH

Often referred to as the “wood basket” of the world, the U.S. South produces over 18 percent of the world’s pulpwood and 7 percent of industrial roundwood, generating more than \$230 billion in traditional economic output and \$48 billion in wages for almost 1.1 million people (SGSF 2013). Southern forests also provide regulatory and cultural ecosystem services, including air and water purification, climate regulation, wildlife habitat, and recreational opportunities. Rapid population growth throughout the South is driving land use changes that put forests at risk, while simultaneously increasing the number of people in the region who value FES. The Forest Service, U.S. Department of Agriculture, forecasts that 23 million acres of forest land will be converted to other uses by 2060 (Wear and Greis 2013). This conundrum of simultaneously increasing threats and values makes it essential to quantify changes in FES in order to better manage change in the forest sector.

To identify the specific motivations and goals for valuation of FES in the Southern States, we reviewed the reports summarized in the previous section and convened a meeting in February 2014 with 38 stakeholders from Federal (3 participants) and State (15) government, NGOs (12), private industry (4), and academia (4) in 10 Southern States. Most (90 percent) participants had more than 10 years of experience in natural resources fields.

The stakeholders all agreed that one key purpose for valuing FES is to increase awareness of the importance of southern forests and the consequences of forest loss. Some hoped that reporting a single (large) dollar value for all FES in a State would catch the attention of the public and policymakers, thus replicating the global influence of the Costanza and others (1997) study at the State level. As noted above, there are concerns about the theoretical validity of this approach and its utility to decisionmakers. In the following section, we therefore review the theoretical basis and mechanics of constructing economic accounts. This leads us to suggest that this objective can best be met by quantifying each FES in physical units, such as percent of drinking water that flows through forests or number of recreation user days in forests, while using valuation to understand changes in FES.

Other goals expressed by stakeholders require understanding how values vary across space and time in order to manage change, specifically by:

1. Improving policy, land use planning, and decisionmaking by quantifying the value of forests relative to other land uses and identifying which forests are most important for which ecosystem services;
2. Catalyzing and supporting the creation of ecosystem service markets by estimating the values of those services; and
3. Reducing the inherent bias of valuing forests (and setting policy) only for commercial forest products rather than by assessing the value of all benefits from all forests.

Of course, each stakeholder group has its own particular concerns. State forestry agencies need to evaluate forest conservation programs. For example, the USDA Forest Stewardship Program (FSP) provides funding to States to help landowners plan and sustainably manage millions of acres of forest land for timber, wildlife habitat, watershed protection, recreation, and other ecosystem services. One goal of a Florida State assessment of FES was to quantify the value of ecosystem services provided by properties enrolled in the FSP. State forestry agencies also must respond to the natural disasters that regularly affect southern forests, including hurricanes, floods, tornadoes, ice storms, drought, and wildfires. The resulting timber losses are well-documented, but methods are needed to quickly and accurately estimate losses of other FES, to help make the case for public assistance for recovery efforts in forests that are not actively managed for timber.

Although decisionmakers in State government have been the primary target audience for FES assessments, conservation organizations and private landowners also find them important. For example, conservation organizations can use the results to direct resources toward properties with high ecosystem service values, while private landowners can use the results to report how their land management affects FES and to guide their transactions in ecosystem service markets (cf., Kareiva and others 2015). Thus, most stakeholder goals require understanding the value of changes in FES resulting from marginal losses or gains of particular types of forest in particular regions.

ECONOMIC ACCOUNTS

Given that all stakeholders expressed interest in an accounting of the total value of FES in their States, we review here how standard economic accounts are constructed and interpreted, as well as the challenges of expanding these accounts to include

FES. An accounting total is generally considered to be a “bottom line” measure that summarizes overall performance in a single monetary figure such as gross national product or income, which represents what economists call “utility,” or, more broadly, “welfare.” This figure is obtained by: 1) assembling data on quantities of “goods and services”; 2) assembling data on the value society attaches to these goods and services, as measured by “prices”; and 3) multiplying the quantity by the price for every good and service and summing over all goods and services.

Assuming that the quantities and values of all final goods and services have been identified and summed, the next question is how to interpret the results. Economic accounts are typically more useful as measures of changes from one year to the next than of any absolute level of well-being. For example, if the national product increased by \$300 billion from one year to another, one might conclude that the country is collectively \$300 billion (about \$1,000 per person) richer than the year before. On the other hand, the national income of the United States is about \$17 trillion per year. There is no reasonable basis for saying that we are, collectively, \$17 trillion, or about \$53,000 per person, better off than we would be if we produced and consumed literally nothing. Would the average American accept \$53,000 to do without literally everything for a year? Likewise, we might identify increases or decreases in the acres of different forest types in a State and multiply these by per-hectare values of FES. However, even with good estimates of the per-hectare values, we could not use this procedure to calculate a figure that accurately reflects the consequences of the complete elimination of all forests in the State, i.e., the total value of all of those forests.

Thus, we focus on changes in accounting totals, considering what they reveal about changes in well-being. Economists assume that people make decisions in order to maximize their utility,¹ or the net benefits (welfare) they receive as a consequence of their consumption choices. As shown mathematically in Appendix 1.1, the change in an individual’s welfare from one period to the next is just the change in the quantity of each good and service consumed by the individual, multiplied by its price, and summed over all goods and services. Thus, the change in the accounting total is the dollar value of the change in total welfare.

Next, we consider how to aggregate changes in individual welfare over the entire population. The simple answer is to add together the changes in aggregate consumption of each good weighted by its price. While this procedure is straightforward and often used, it has some fundamental problems. When we add up the changes in expenditures over all individuals, and call it a change in aggregate welfare, we are implicitly assuming that everybody has identical reactions to changes in their income (i.e., everyone has the same marginal utility of income). While unverifiable

¹ Or more accurately, people make decisions *as if* they were maximizing utility. According to a standard economic proof, if people prefer some alternatives over others in a consistent way, they behave as if their preferences were determined by a “utility function.”

empirically (since utility cannot be observed or measured), it is reasonable to assume that a \$1,000 increase in annual income would be much more important and valuable to someone with an annual income of \$20,000 compared to someone with an annual income of \$200,000. The only theoretical justification for adding up changes in individuals' expenditures to get some notion of the welfare of the population as a whole is the idea of "potential compensation." The "potential compensation" (or "Kaldor-Hicks") criterion holds that if the sum of accounting gains is positive, there is—at least in theory—a way to assure that everyone is made better off by redistributing the monetary gains (Johansson 1993). In practice, changes that increase the overall value of FES may have negative effects on the welfare of some groups even while they have positive effects on the welfare of other groups.

INCORPORATING FES INTO ACCOUNTS

While accounting appears to be a straightforward exercise, incorporating FES presents several complications. First, many goods and services that contribute to our well-being do not have prices that can be observed from transactions in markets. There are two likely reasons why we cannot observe prices for these ecosystem goods and services. One is entirely mundane, but always worth considering. We do not observe prices for things that are not worth much. This is a proposition that may seem at once counterintuitive and commonsensical. It may seem counterintuitive when applied to the goods and services nature provides, because goods and services such as fresh water and flood protection are essential to human life. However, common sense suggests that when small changes in the amounts of goods and services make no difference, they are not valuable. Things command high prices when a little more of them contribute substantially to our welfare, and this, in turn, happens when the things in question are in short supply relative to their demand. Fresh water is essential to life, but it is not very valuable when and where there is plenty of water available relative to the number of people who benefit from it.

The other possible explanation for the lack of observable prices is that the good or service in question is "public" in some or all aspects. A public good is one that is non-rival and non-exclusive. Non-rival means that when one person benefits from the good, it does not diminish the benefits for others. A good is non-exclusive when there is no feasible way to prevent others from benefiting from it. In this case, the value of an additional unit of the good is the sum of its value to all individuals who benefit from it. These characteristics of public goods undermine the functioning of private markets because the prices of any transactions in markets for public goods understate their value. There may not even be any transactions, if no one pays to provide something that confers a relatively small benefit to each of a relatively large number of people.

Non-market valuation techniques have been developed to estimate "prices" for goods and services that are valuable but not traded in markets. The following chapters discuss the most commonly used and appropriate techniques for different categories of FES. However, it is important to recognize that the prices determined through these techniques are highly context dependent. Real estate agents sometimes say that three things determine the price of a property: location, location, and location. Because a home is fixed on the landscape, much of its value is determined by its proximity to places of work, shopping venues, and attractive natural features. Likewise, the value of attractive natural features depends crucially on the number of people who can access them. Moreover, the values of ostensibly similar ecological features may vary tremendously depending on how much is available, now and in the future or under alternative policy scenarios. Market prices reflect this variation for most goods that are bought and sold in markets; the challenge is obtaining "prices" that reflect the local supply and demand for ecosystem services that are not traded in markets.

A second complication with accounting in general, and especially in accounting for ecosystem services, is that not all ecosystem goods and services are things we enjoy directly (sometimes called final goods and services or benefit relevant indicators) (Olander and others 2015). Some, called intermediate goods and services, are inputs required to produce final goods and services. Further, many of these inputs could be considered investments made today in order to produce final goods and services in the future. And furthermore, some of these capital investments may be for the purpose of averting or restoring damages.

The final challenge that we consider here is valuing investments—giving up some consumption now in order to enhance consumption possibilities later. Suppose that we define units so that, by giving up one "apple" now, we can acquire one unit of "capital." Let's suppose that "capital" can be used to enhance our future consumption of "bananas." So, what is the worth of the extra unit of capital we would get by foregoing consumption of an apple? It consists of the products of three things, which are first multiplied together, and then summed over the life of the capital investment:

- The utility derived from the additional banana consumption, times
- The extra banana production, afforded by the additional capital investment, times
- A discount factor that expresses the value of foregoing current consumption to increase consumption in the future.

If a consumer were to decide to eat an extra apple this year, we would multiply that apple by its price and add it to all the other consumption changes, weighted by their prices, in order to

estimate the change in welfare. If the consumer forgoes eating an apple in order to grow more bananas, though, she is showing that she regards that investment (or input) to be at least as valuable for her overall welfare as immediate consumption of an apple. Hence, it is reasonable to add to the ledger of changes in welfare the net present value of investment.² Assuming that consumers and investors made this calculation in deciding how much to invest, we can simply record the dollar value of investment as a measure of its contribution to well-being.

The calculation of this value is more daunting when considering investments in the production of public goods. Investments and disinvestments in forests may not be made optimally when, for example, a landowner plants trees that store carbon or protect downstream water quality. In this case, we should still record the value of the investment, but we must now estimate values for the public goods of carbon storage and water quality. This requires understanding the “extent of the market,” or the size of the population that benefits from these goods. Further, we must calculate these values over the life of the investment. This means, in particular, we must project the marginal value of the public good, the investment’s contribution to the provision of that public good, and the discount factor to be employed, over the time horizon of the investment.

Finally, we note that many of the effects of (dis)investments in natural capital are eventually reflected in traditional economic accounts. For example, if clearing forests increases the probability of floods, the damage will eventually be reflected in lost assets or reduced agricultural yields. Thus, part of the motivation for green accounting is to attribute a portion of changes in gross domestic product (or other summary figures) to changes in natural capital. Another motivation is to project the effects of current (dis)investments in natural capital on future well-being. Finally, clearly some (dis)investments in natural capital affect well-being directly and thus are additional to standard economic accounts. To obtain a comprehensive measure of long-term economic performance, we would need to account for all of these future returns to investments.

STRUCTURE OF THE GTR

The following four chapters on cultural (chapter 2), regulating (chapters 3 and 4), and provisioning (chapter 5) services exemplify the different approaches and different levels of information found in the literature. Non-market valuation methods can be categorized as either stated preference, meaning that they are based on surveys that elicit people’s values, or revealed preference, meaning that they infer values from prices in related markets. The chapter on cultural ecosystem services

describes both stated and revealed preference methods that can be used to estimate the value of these services in particular contexts and for particular populations. The chapter on water services gives greater emphasis to revealed preference methods. The chapter on air quality emphasizes benefit transfer approaches embedded in accounting frameworks for urban forests and for climate change mitigation. The next chapter discusses provisioning of non-timber forest products (NTFPs). Once harvested, NTFPs are private (rather than public) goods, but there are still significant challenges in quantifying harvest and deriving the associated value of the forest provisioning service.

These chapters lay the groundwork for the concluding chapter to discuss aggregation of such highly variable information (across services, forests, and people) and potential pitfalls or red flags involved in doing this. While the four chapters on different categories of ecosystem services describe options for estimating the marginal values of those services, the final chapter considers how to aggregate those marginal values in order to track changes in value over time and project changes in value as a result of policy alternatives. This should be considered both when planning and when reviewing results of an assessment (e.g., when writing terms of reference or assessing deliverables). The concluding chapter argues that FES assessments should account for total flows of ecosystem services from forests (in physical units), but should quantify the marginal value (in dollar terms) of the most likely changes in forest cover. This approach will help steer FES assessments around the numerous possible pitfalls of green accounting.

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² The modifier “net” before value is important. It is certainly possible to make *disinvestments*. One way to do this would be by failing to keep up with depreciation. Another is by liquefying capital assets, including natural capital such as forests.

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Appendix 1.1

To show that changes in accounting totals are indicative of changes in well-being, we consider how well-being can be expressed in terms of dollars-and-cents figures. First, consider an individual who buys apples and bananas and carrots and a host of other goods numerous enough to go through the alphabet many times over. Her utility, as a function of the number of apples, bananas, carrots, and so on, that she enjoys could be expressed as

$$U(x_A, x_B, \dots, x_N), \quad (1.1)$$

where N is now not the 14th letter of the alphabet, but rather, the very, very large number of different goods and services we enjoy. This expression can be decomposed to write changes in utility from one year to the next as follows:

$$dU = MU_A dA + MU_B dB + \dots + MU_N dN. \quad (1.2)^1$$

In this expression, dU is the difference between the overall level of utility a consumer experienced this year compared to last year, i.e., her change in well-being. It can be decomposed into MU_A , the marginal utility she gets from consuming an extra apple, times dA , the number of apples she consumes this year over and above the number she consumed last year, plus the same calculation for bananas, and so on through all the things she consumes.

While we could measure changes in the quantities of things a person consumes between one year and the next, we cannot measure the extra utility she realizes from consuming a bit more of something (the MU 's). We can, though, do a simple trick and

use a basic result from economic theory. The trick is to divide all the terms in the expression above by one of the marginal utilities—say, for convenience, the marginal utility of apples:

$$dU/MU_A = dA + MU_B/MU_A dB + \dots + MU_N/MU_A dN \quad (1.3)$$

The fundamental result from theory is that the ratio of the marginal utilities of any two goods a consumer buys is equal to the ratio of the prices she pays for them; that is,

$$\frac{MU_B}{MU_A} = \frac{p_B}{p_A}, \quad (1.4)$$

and so forth, for all other goods the consumer buys (recalling that we only have prices for goods that are bought and sold). The argument that this condition holds is a staple of basic economics textbooks and involves a proof by contradiction. If this condition did not hold, the consumer could buy more of the good that provides her with more marginal utility per dollar spent and less of that which provides less, and achieve higher overall welfare for the same monetary expenditure, contradicting the assumption that her choices are optimal.

So,

$$dU/MU_A = dA + p_B/p_A dB + \dots + p_N/p_A dN \quad (1.5)$$

or

$$p_A dU/MU_A = p_A dA + p_B dB + \dots + p_N dN \quad (1.6)$$

¹ In mathematical jargon, expression 1 is a *first-order approximation* to a change in utility. A second (or still higher)-order approximation might be more accurate, especially when the period-on-period changes in quantities are larger, as would likely be the case when the “periods” under consideration are longer. A more accurate approximation would involve the definition of many more terms, however, and would require empirical estimates of own- and cross-price elasticities of demand, which would be a daunting exercise.

Appendix 1.1 cont.

The right-hand side of expression 1.5 consists entirely of things we can observe: the prices of goods and changes in their quantities. To interpret the left-hand side, we need to apply a corollary to the result from consumer theory we just considered. Recall expression, which characterizes the conditions under which a consumer is maximizing her utility given the constraint that she cannot spend more money than her income allows. We can express it equivalently as

$$\frac{MU_A}{P_A} = \frac{MU_B}{P_B} = \frac{MU_C}{P_C} = \dots = \frac{MU_N}{P_N} \quad (1.7)$$

The intuition for this condition is that if the consumer got more additional utility for the last dollar she spent on apples than she did for the last dollar she spent on bananas, she could reallocate her expenditures from bananas to apples and be better off overall. So, if her choices are optimal, she must get the same marginal utility per dollar spent on all goods. That means all the ratios in expression measure the same thing: the additional utility realized from an additional dollar of expenditure. The ratios are all equal to the marginal utility of income: the increase in welfare due to an additional dollar of income.

If we denote the marginal utility of income as MU_y , we now can write expression (1.6) as

$$dU/MU_y = p_A dA + p_B dB + \dots + p_N dN \quad (1.8)$$

The left-hand side of this equation is now the dollar value of the change in utility, since units of utility (the dU) are divided by utility per dollar spent (MU_y).

To recap: we asked how we might represent a change in an individual's welfare from one period to the next. We started by decomposing it into changes in how much of each thing she consumes, multiplied by the price of each thing. We then rewrote this as the dollar value of the change in welfare, or change in utility, showing that it is the sum of the prices of goods multiplied by the changes in their quantities, which is just the change in the accounting total.

Chapter 2

Forest Ecosystem Services: Cultural Values

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WHAT ARE CULTURAL SERVICES?

How we define “culture” and societal well-being related to culture depends heavily on who is looking at it, but culture can be generally described as “the customs and beliefs of a particular group of people that are used to express their collectively held values” (Soulbury Commission 2012). In the context of forests, culturally derived norms, beliefs, and values help drive preferences for forested landscapes and forest-based benefits such as diversity and identity, justice, education, freedom, and spirituality (Farber and others 2002, Fisher and others 2009, Kellert 1996). Environmental policies and responsible forest management can enhance how forests help give rise to and support cultural ecosystem service values. Likewise, human components (e.g., customs and beliefs) determine how forests are to be culturally valued (fig. 2.1). This is somewhat different when compared to other types of services (e.g., regulating services) because human culture plays a central role in determining how people interact with forests and perceive their associated benefits. In other words, human culture gives important meanings to forests that are recognized as valuable, but the forests themselves do not inherently possess these meanings. Table 2.1 offers a topology of cultural ecosystem services and

associated ecosystem and human components. However, our understanding of the many factors that give rise to cultural ecosystem services is still a matter of ongoing investigation.

There is good reason for investigating the cultural ecosystem service values associated with forests: they are critical to our understanding of the value of forest land and the benefits of forest conservation. The U.S. South is expected to lose 30-43 million forest acres to urbanization between 1997 and 2060 (Wear and Greis 2002), and structural changes in southeastern ecosystems are expected to impact the provision of a wide range of cultural ecosystem service benefits (Bowker and others 2013). Concurrently, social trends also suggest that youth are spending less time outdoors compared to previous generations (Louv 2008). This means that the customs and beliefs held by future generations and how future generations value forests will likely be different compared to previous generations. To help maintain the forest estate, we may need to actively cultivate cultural ecosystem service values through outreach programs that encourage different population groups (i.e., younger and non-white populations) to take up new recreational activities (Poudyal and others 2008). Providing tangible experiences can

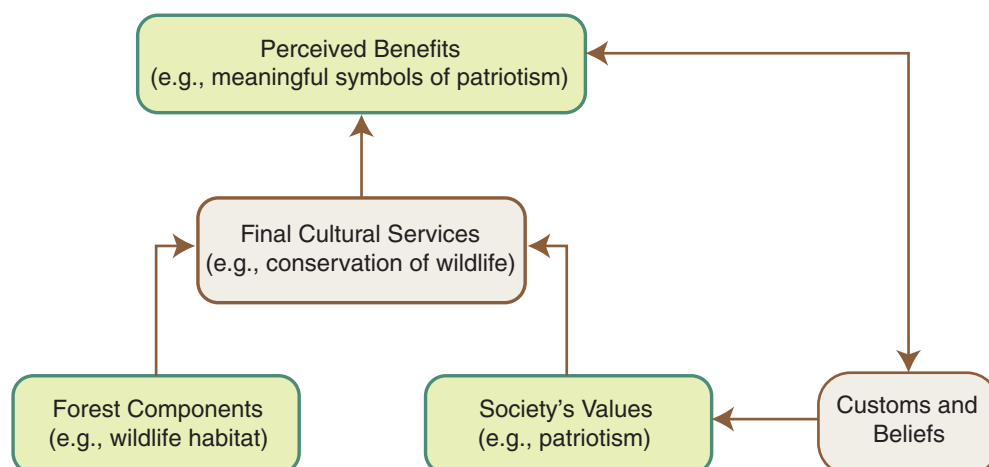


Figure 2.1—Relationship between forests and society that give rise to cultural ecosystem services and perceived benefits provided by forest lands.

Table 2.1—Typology of the components that give rise to cultural ecosystem services

Components intrinsic to visitors/users	Components intrinsic to the site	Potential benefits	Cultural ecosystem services
Social capital	Historic features	Health and well-being	Cultural identity
Skills	Biodiversity	Social contacts	Cultural heritage
Knowledge	Wildlife	Personal pride	Spiritual services
Values	Management	Education	Inspirational services
Beliefs	Structure	Inspiration	Aesthetic services
	Stories	Spiritual well-being	Recreation/Tourism
	Practices		
	Artworks		

Sources: de Groot and others (2005); Tabbush (2010).

help individuals gain a sense of place and an appreciation of the services provided by forests. Those who gain a sense of place are also more likely to perceive that they have attained the benefits they desire and will want to visit forests again in the future (Kil and others 2012).

While the importance of cultural ecosystem services has been recognized within land management and policy decisions, many services are not yet adequately defined, quantified, or integrated within the ecosystem services framework (Daniel and others 2012). This is largely because attempts to quantify these values require: 1) an understanding of how ecosystem components and associated artifacts (e.g., historic sites) interact with important human components (e.g., customs, beliefs), and 2) the suitability of the quantitative and valuation methods used to measure these interactions (de Groot and others 2005). Experts in most social science disciplines, such as anthropology, sociology, and psychology, are trained to address the first requirement, and economists are involved to address the second component. Notably, the valuation approaches used by economists tend to offer incomplete and lower bound estimates of ecosystem service values. Moreover, economists tend to use market data that is readily available but poorly suited to the task, particularly when public goods and externalities are involved (Costanza and others 1997). There is a need for decisionmakers to better understand the methods available to estimate non-market values for ecosystem services and the challenges associated with using these methods to understand tradeoffs in cultural ecosystem services.

The goal of this chapter is to provide an overview of how valuation methods can be used and applied within a cultural ecosystem services context. The first part will focus on different types of values (use and non-use values) and the ways cultural ecosystem services may be integrated within the ecosystem services framework. Second, we will examine the policy context and purpose for producing economic estimates of ecosystem services value. Third is a non-technical description of important

theoretical frameworks and challenges associated with economic estimation efforts. Finally, we will provide readers with a non-technical description of commonly used non-market valuation methods and benefit transfer and will discuss these approaches within a cultural ecosystem services context. Consistent with the theme of this book, we will focus on the Southeastern United States; however, the overall concept of cultural ecosystems services, and the methods used to identify and measure the economic value of these services, are applicable well beyond this region.

CONNECTING CULTURAL ECOSYSTEM SERVICES WITH THE ECOSYSTEM SERVICES VALUATION FRAMEWORK

Cultural Ecosystem Services and Use Values

The cultural aspects of a society are often expressed through the activities, practices, skills, and events that occur in the forest, or that involve artifacts that originated from the forest (Tabbush 2010). These activities may involve the consumptive use of forests (e.g., mushroom picking) as well as non-consumptive uses (e.g., hiking). To help readers understand use values in the context of cultural ecosystem services, this section will provide an in-depth discussion about recreational and aesthetic experiences, also known as amenity services (de Groot and others 2005). Since amenity services are often associated with public lands, the last part of this section will describe some of the cultural ecosystem services also associated with private forest lands.

The benefits of engaging in outdoor recreation activities on forest lands have long been important to people in the southeast region of the United States (Ownby 2014). Today, millions of people participate in non-consumptive recreational activities such as walking, hiking, camping, birdwatching, wildlife viewing,

and other such pursuits in forests every year (Cordell 2008). Southeastern forests are also habitat for many game animals and fish sought by consumptive recreationists, such as hunters and anglers, and serve as the source of rivers and streams used for recreational activities. It is unsurprising that the value of recreation in the southeast region has been the subject of numerous economic studies (see Appendix 2.1). The number of visitors to southeastern forests is expected to rise over time along with demand for different recreational activities (Bowker and others 2013).

Several authors have suggested that for accounting purposes, recreation (e.g., angling) should be thought of as a benefit that arises from the joint use of other ecosystem services (e.g., clean water) and conventional goods and services (e.g., fishing equipment) (Boyd and Banzhaf 2007, Fisher and others 2009). There is a long history of research on the components of and benefits from a recreation experience (Driver and others 1991, Manfredo and Driver 1996, Manfredo and others 1983, Manning 1999). In the recreation literature, the recreation experience is most often understood from the psychological perspective. Consequently, individuals who participate in outdoor recreation do so with the expectation that the recreation activity, done in a particular setting, will result in certain beneficial psychological and physiological outcomes (Manfredo and Driver 1996, Stein and others 2003). This is commonly referred to as the experiential or behavioral model of recreation where the outcomes of recreation rather than the activities themselves are deemed to be the benefits (Driver and others 1991). Interestingly, the majority of the benefits categorized as cultural ecosystem services by most authors (MEA 2003, DeGroot and others 2005) are tied to outdoor recreation (see table 2.2 in chapter 2 Moore and Driver 2005).

The key aspects of setting in the recreation experience model are defined as the physical setting (e.g., level of remoteness), the social setting (e.g., level of crowding), and managerial setting (e.g., level of management restrictions and facilities) (Driver and others 1987). Settings are expected to influence the activities that can be performed in any given location and the benefits that recreationists might expect to obtain (Pierskalla and others 2004). We have taken the outdoor recreation experience model and ungrouped the three setting attributes so that we can begin to elaborate on the physical setting (Morse 2010) (fig. 2.2). Presented this way, the biophysical setting is considered to be but one aspect of the recreation setting. Furthermore, the physical setting is more than simply a level of remoteness, but also a combination of geophysical, biological, and vegetative attributes that are spatially explicit (Morse and others 2009). It is this ecological portion of the physical recreation setting that is the final service (the forest, the water body, and the fish) recognized by Boyd and Banzhaf (2007), or the result of indirect and final structural and functional services of Fisher and others (2009).

By conceptualizing the recreation experience model in this way, we can identify clearly the contribution of the ecosystem as a critical aspect of the recreation setting. It also demonstrates that the ecosystem is but one of the attributes that combine to form a recreation opportunity setting, or the amenity service. Using recreation theory, we can also identify the many benefits (psychological, physiological, etc.) that are accrued from recreation experiences. Finally, the model clarifies the subjectivity of an experience (essential for cultural ecosystem services) that is dependent on how different individuals' motivations and expectations regarding their experience and desired outcomes can influence their value of the recreation experience.

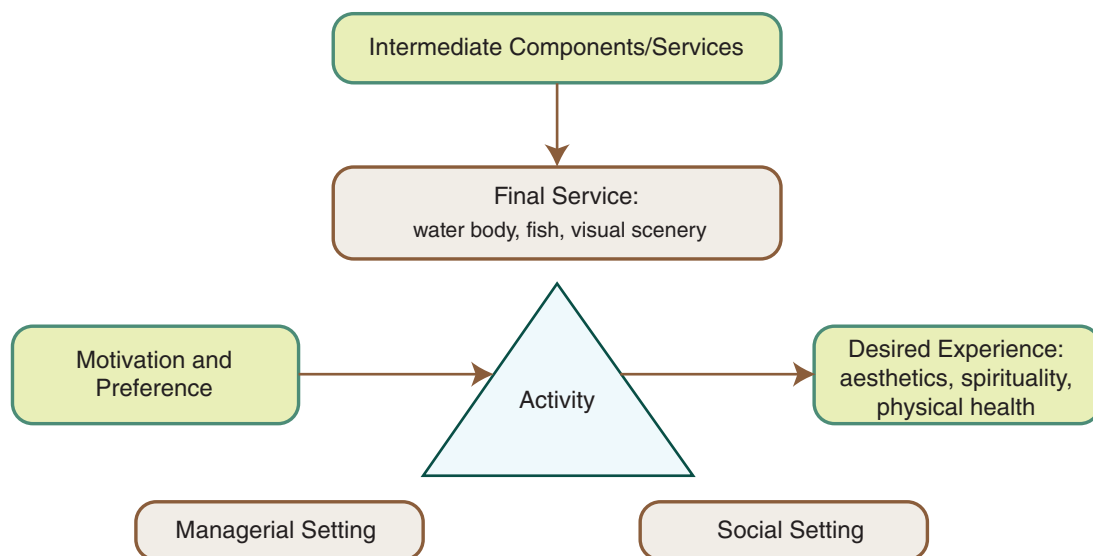


Figure 2.2—Integrating recreation theory with ecosystem services.

Public lands are often recognized as being central to the provision of cultural and amenity ecosystem services; however, private forest lands also play an important but less well understood role. In the southeastern region of the United States, the value of recreation on private lands is most often associated with lease and fee hunting (Mozumder and others 2007). Willingness to pay for hunting leases in Alabama was found to be steadily increasing and was last valued at \$1.29 per acre per hunter (Zhang and others 2004). Willingness to pay for fall and spring turkey hunting permits in Mississippi has been valued at \$11.00 and \$36.25, respectively (Brunke and others 2006). The value of the hunting leases can depend on a number of factors such as alternative hunting access options, hunter perception of crowding on public lands, availability of game species on leased lands, and duration of the lease agreement (Munn and others 2011).

Even when the public is not able to access private lands, certain benefits may still be provided in the form of spillover effects (Blitzer and others 2012). Private lands offer green landscapes and positive aesthetic experiences for neighbors and others in the community (Luttik 2000). Moore and others (2011) estimated total aesthetic value of Georgia’s private forests to the residents of Georgia at \$11.2 billion per year. Haefele and others (1991) found people were willing to pay \$100 per year and \$21 per year, respectively, for protecting high-elevation spruce-fir forests in the southern Appalachian Mountains from exotic insect infestations and air pollution damages. With continued forest fragmentation and a loss of forest cover, demand for green spaces can be expected to increase in the Southeastern United States (Griffith and others 2003).

Similar to aesthetic benefits, how land is managed can serve as salient symbols on the landscape, expressing important cultural priorities and values (Allison 1996, Sorice and others 2012). For example, planting even-aged pine stands can be seen as an homage to the South’s longstanding history in producing forest products (Zhang and Polyakov 2010), or more simply the virtue of productivity. Likewise, prescribed burning can help express the value of living in harmony with nature or the importance of maintaining certain management traditions (Putz 2003). The act of owning forest land also helps people express important ideals about identity, individualism, a sense of duty (toward nature or society) and skepticism about State control (Brook and others 2003, Burton 2004). Maintaining the ability for self-determination has important psychological benefits and can enhance perceptions

of personal well-being (Ryan and Deci 2000). Those who help shape the landscape may also have a greater sense of place and assign a higher value to the cultural benefits associated with private lands (Kil and others 2012).

Due to changing cultural values and perceptions, there is an increasing tendency to create landscapes with high amenity values (for aesthetic and recreational use, for example) at the expense of traditional landscapes with high cultural and spiritual ecosystem service values, and to disregard those with traditional knowledge about forest management (de Groot and others 2005). When changes are made to the landscape, either through market forces or government interventions, this can be perceived as a threat to certain cultural ecosystem service values and the psychological well-being of forest owners (Kreye and others 2016). Government interventions that seek to improve public welfare through the management of private forest lands should also recognize the cultural ecosystem service values and traditional knowledge associated with maintaining private lands. Policy efforts that recognize the unique benefits associated with maintaining private forest lands can help improve the quality of human life within rural communities while simultaneously achieving forest conservation and sustainable use goals (de Groot and others 2005).

Cultural Ecosystem Services and Non-use Values

Some of the cultural ecosystem services provided by forests do not have to be *used* for the benefits to be recognized or considered valuable. Non-use (or passive use) values are ideals that serve as a guiding principle in how people make decisions about the environment and are an expression of important cultural values (Dietz 2015). Understanding people’s attitudes (positive or negative) toward different types of non-use values can improve our understanding of public priorities for less tangible cultural ecosystem services. Table 2.2 provides a list of the non-use values most often recognized in the research literature (Dietz 2015).

Bequest value, or beliefs about the importance of passing ecosystems on to future generations in good condition, relates to the heritage services provided by conserved or restored forests (Broadbent and others 2015). What separates bequest value from a general concern about environmental integrity or altruistic

Table 2.2—A topology of non-use values

Option value	Bequest value	Existence value	Altruistic value
The value of maintaining an asset or resource to have the possibility that it may someday be used.	The value of maintaining or preserving an asset or resource so that it is available for future generations.	The benefit people receive from knowing that a particular environmental resource, organism or thing exists.	The value of maintaining an asset or resource that is not used by the individual, so that others may make use of it.

Source: Dietz (2015).

value is the additional desire that a specific group (i.e., future generations) be given the opportunity to benefit from forests (Walsh and others 1984). In the southeastern United States, longleaf pine (*Pinus palustris*) ecosystems, and associated wildlife such as the gopher tortoise (*Gopherus polyphemus*), are considered by many to be part of the cultural heritage of the region. However, less than 6 percent of the original extent of longleaf pine ecosystems remains in the South (Guldin and others 2016). Participants in advocacy organizations, such as the Longleaf Alliance, recognize the bequest value associated with these ecosystems and seek to promote longleaf pine restoration on public and private lands.

Society's beliefs about the treatment of non-humans and ecosystems are an important part of the spiritual and cultural identity ecosystem services provided by forests (Davidson 2013, de Groot and others 2005). Some people psychologically benefit from knowing that a certain organism exists, because the organism is an important symbol within their community or personal life (de Groot and others 2005). Some also believe we have an ethical responsibility as a society to protect the integrity of ecosystems (Pienaar and others 2013). To illustrate, households in South Carolina were found to have a willingness to pay of \$10.64 annually for the benefits associated with protecting the endangered red-cockaded woodpecker (*Leuconotopicus borealis*) (Loomis and White 1996). Research has also found that people in the United States, including the Southeast, tend to have biocentric attitudes toward forests (i.e., a concern about the well-being of all organisms) (Kreye and others 2014, Pienaar and others 2013).

The beliefs people hold about *how* ecosystem services should be distributed to society (e.g., government interventions, markets) can also contribute to the cultural identity and heritage ecosystem services associated with forests in the Southeastern United States (Johnson and Duke 2007, Kreye and others 2016, Poudyal and others 2015). Beliefs about government control over the distribution of services can impact the perceived legitimacy and public acceptance of environmental policy decisions (Petersmann 2004) as well as estimates of willingness to pay for ecosystem service provision used in decisionmaking (Kreye and others 2014, 2016b). Most government approaches to ecosystem service distribution involve the permanent modification of private property rights (e.g., land acquisition, conservation easements). Government entities may also use laws and regulations to oversee the fair distribution of cultural benefits, such as access to outdoor recreation and other resource amenities by minority groups (Floyd and Johnson 2002, Johnson and others 1998). However, for some goods and services, government involvement may be considered cumbersome or less effective compared to free market approaches. Government involvement in service provision may also involve certain tradeoffs, such as the psychological benefit of being assured that resources will be protected into the future (e.g., option value) but the loss of the psychological benefits associated with maintaining private lands (e.g., self-determination). Recent research has found public values for forest-based services in

Florida and Georgia are indeed influenced by the policy processes (e.g., conservation easements) and the institutions (e.g., State agency) used to bring about the services (Kreye and others 2016, Poudyal and others 2015). While a broader body of social science research has recognized the relationship between social trust and ecosystem management, this relationship is rarely incorporated into valuation efforts and is not well understood as a contributor to cultural ecosystem services (Kreye and others 2014).

Why Assess the Economic Value of Cultural Ecosystem Services?

Given the difficulties in defining and quantifying certain cultural ecosystem services, one may question the rationale behind attempts to estimate economic values for these services. Some advocate that multi-criterion and deliberative evaluation procedures be used instead of strict economic strategies to clarify tradeoffs and synergies involving cultural ecosystem services (Daniel and others 2012). Moreover, some argue that traditional market approaches are inappropriate for identifying and measuring cultural ecosystem services (Kumar and Kumar 2008, Peterson and others 2009). Still, we see evidence every day of people expressing important cultural values through consumer and market behaviors (e.g., visiting National Parks, buying organic food), but this can only occur when markets exist that allow people to express these values. The metric that can best express changes in human well-being for decisionmaking purposes often depends on the policy context and stakeholder needs.

In most policy contexts, the need to obtain economic values can arise when (1) there is a need to justify management costs for current levels of service provision, or (2) when a common metric (i.e., dollar) is needed to assess tradeoffs in welfare across dissimilar services (e.g., clean water benefits versus agricultural goods). In the first case, a single value estimate may be all that is needed for use in a cost-benefit analysis. In the second case, each policy alternative may need to account for other associated factors such as the number of people impacted by a change in service provision and the distribution of benefits over time.

Economic values are also central to helping advance market-based approaches (i.e., payments for ecosystem services) to ecosystem service provision. In the past, the valuation of cultural ecosystem services, primarily recreation, has been limited to public lands. Increased population growth and demand for forest-related benefits in the Southeastern United States has now compelled decisionmakers to also understand the full value of private lands. Traditionally, the value of private forest lands has been associated with agricultural productivity (i.e., timber) and landowners receive little or no external reward for securing services that help provide cultural benefits. As such, ecosystem services with either off-site or non-use values are likely to be undersupplied by private forest landowners because it is difficult for them to capture those values. Helping landowners diversify income by offering a variety of nature-based activities that

capitalize on cultural ecosystem services could play an important role in encouraging private landowners to maintain recreational benefits and culturally valuable ecosystem services. Key to this approach is understanding public willingness to pay (i.e., demand) for cultural ecosystem service provision and landowner willingness to accept compensation (i.e., price) for providing cultural ecosystem services.

Economic Concepts and Theoretical Assumptions

Economic theory attempts to describe the values people hold for different goods and services under different contexts (e.g., a change in quality, the availability of substitutes) and when choices are constrained by cost (e.g., income). Because of these constraints, monetary values do not represent the inherent value of an ecosystem, only its value in relationship to other levels of provision or other goods (Kallis and others 2013). It is assumed that the benefits derived from cultural ecosystem services provide the individual with some utility or measure of satisfaction. Random utility theory states that the level of utility associated with a good or service will vary randomly across a population (Train 1998). It is also assumed that measured estimates of willingness to pay (WTP) are an appropriate representation of individual utility. Therefore, the most commonly accepted and applied metrics for valuing cultural services are individuals' maximum WTP in dollars for ecosystem service provision. In some cases, economists may also choose to measure the minimum willingness to accept (WTA) to forego the provision of a good or service (Brown 2007).

The difference between what a consumer is willing to pay and what they actually paid for a good (i.e., its price) is known as consumer surplus. In cases where market prices do not exist, but goods or services have recognized value, individual consumer demand is represented by WTP. When consumer surplus is aggregated across a population, an estimate of the welfare gain to consumers of that good or service is obtained. The analogue for producers is known as producer surplus and is typically thought of as profit—the difference between the price and the cost of providing the service (i.e., supply curve). Producer surplus can help describe how producer welfare can be improved through the development of missing markets. Market-based solutions to welfare—or ecosystem service—distribution are considered efficient when supply equals demand. Change in consumer and producer surplus is the measurement that economists provide to policymakers to help describe how public welfare is impacted by changes in ecosystem service provision (Brown 2007).

To facilitate a successful market transaction, it is important that goods and services embody certain characteristics (Brown 2007). One important characteristic is the ability to be rivalrous, or when the good is consumed by one person it prevents simultaneous consumption of the good by others (e.g., timber). Another characteristic is the ability to exclude or prevent people who have not paid for a good or service from having access to it

(e.g., fee-based access to a State park). Goods and services that have a direct use value and are rival and/or excludable tend to be already traded in a market setting. However, use values are only part of the total economic value (TEV) associated with a good or service. Non-use values are also considered important because of how these values relate to important beliefs and ideologies that help drive consumer behaviors and preferences for less tangible cultural ecosystem services. Non-use values may also constitute an important part of the total value associated with certain forests, such as wilderness preserves (Loomis and Richardson 2000, Walsh and Loomis 1989).

When attempting to link land management decisions with certain cultural ecosystem service values, it is useful to employ Lancaster's theory of values, which postulates that the utility we expect from a good or service is dependent on the characteristics, or bundle of attributes, that make up the good or service (Lancaster 1966). This is to say, consumers decide which services (or goods) are preferable by examining the attributes or features that make up the service, along with the price of the service and the income they have available to spend on it. Advanced valuation approaches, such as an attribute-based choice experiment and hedonic methods, are often structured around this theory. The advantage of this approach is that both attributes on the landscape—created through management actions—and attributes describing cultural components can be linked with changes in benefits or utility. Attributes on the landscape may include forest plant diversity and structure, trails and historic sites, and spatial relationships among important features (e.g., distance between historic sites). Attributes that describe important cultural components may include the customs of service users (e.g., level of hiking activity) and/or beliefs that people hold about service outcomes (e.g., health benefits, importance of passing forests to future generations). Unfortunately, attributes that describe the human components of cultural services are less often examined or understood in a valuation context, more of which will be discussed later on in the chapter.

The theoretical frameworks and assumptions presented above also come with some important limitations. One limitation is that WTP estimates are a meaningful proxy of utility only for those who have a sufficient amount of disposable income. This means that valuation efforts may not be appropriate for addressing environmental justice concerns in poor communities. Another limitation is the occurrence of lexicographic preferences or when a person is unwilling to forego a good in exchange for any other amount of another good based on ethical reasons. There is also the assumption that people, as consumers, are primarily interested in maximizing their utility and make decisions outside of social contexts. In reality, some people are willing to be losers at times (e.g., pay higher taxes) if they believe that this will help society or their community as a whole succeed. Similarly, some landowners may be motivated by personal ethics (e.g., being a good land steward) to assume the costs of conservation and are unwilling to accept financial incentives for providing certain cultural ecosystem services.

The theoretical issues described above illustrate the difficulty of understanding human behaviors toward the environment using a strictly economic approach. Still, valuation methods are continually being improved and offer a useful way of comparing tradeoffs in utility associated with ecosystem service provision. For example, non-market valuation approaches have been the subject of investigation in 130 countries for over 50 years and have resulted in over 7,500 published research papers (Carson 2012). It is our recommendation that those who plan to employ valuation methods consider how the associated assumptions may impact their efforts to produce useable estimates of economic value within a given policy situation. In some policy contexts, non-monetary metrics or a mixed methods approach (i.e., using both quantitative and qualitative methods) can instead be used to understand policy impacts and inform decisionmaking.

VALUATION METHODOLOGIES

This section of the chapter provides a general description of the non-market valuation methods currently used to generate estimates of economic value. Revealed preference methods, such as travel cost and hedonic pricing, rely on data from actual spending choices or behaviors to infer underlying relative utilities for non-market goods and services. Stated preference methods use hypothetical market contexts to quantify non-market values (e.g., by asking respondents to state their maximum WTP or WTA) for a non-market good or service (e.g., using a survey approach). We also provide a comparison of the outcomes of an economic impact analysis with non-market valuation methods so that readers can better distinguish between the approaches.

Limitations of an Economic Impact Analysis

Economic impact analysis is a quantitative approach often used by government agencies to justify management or program costs. This approach focuses on policy driven changes on the landscape which result in changes in business revenue, business profits, personal wages, and/or jobs. For example, recreation on Federal lands contributes about \$13.6 billion to the Nation's gross domestic product each year and supports approximately 205,000 jobs, many of which are located in rural areas (Weldon 2014). The total economic impact of hunting activities, wildlife viewing, and recreation in the southern Appalachian region was estimated at \$594 million, \$407 million, and \$6 billion, respectively, in 1996 (Barnhill 1999). It is also reported that expenditures made on wildlife related recreations (e.g., fishing, hunting, and wildlife-watching) generated 0.7 percent of regional employment and gross output in the Southeastern United States (Munn and others 2010).

Clearly, public demand for cultural ecosystem service provision has an important economic impact on surrounding communities. However, these types of economic metrics (i.e., jobs created, revenue earned) do not offer a suitable description of consumer or producer surplus necessary for describing changes in utility for a

given service. Moreover, economic impact estimates are based on existing market values for a limited number of services and fail to describe the value associated with related non-market goods and services. To use economic estimates within an ecosystem services framework, at a minimum the estimates need to be comparable across services and describe consumer surplus for both market and non-market values.

Revealed Preference Methods

Travel cost methods—Travel cost methods are primarily used to estimate the value of recreation offered at a specific site (e.g., a State park). Economists assume that the economic cost necessary to reach a recreational site, such as travel expenses and associated user fees (e.g., entrance fees), is an estimate of user WTP for recreation. In other words, it is assumed that people perceive and respond to changes in travel costs and amenity qualities the same way that they would respond to changes in market price and qualities of purchased goods. The values generated using this method can allow decisionmakers to compare estimated benefits with the costs of maintaining the site for visitors (Parsons 2003).

A simple zonal travel cost method can be applied by collecting information from visitors about the number of visits and their travel distance to the site. Because travel costs tend to increase with distance, researchers are able to construct a demand curve by calculating the number of visits purchased at different prices or travel costs (fig. 2.3). Consumer surplus for users is then estimated by quantifying the area above the price line and below the demand curve.

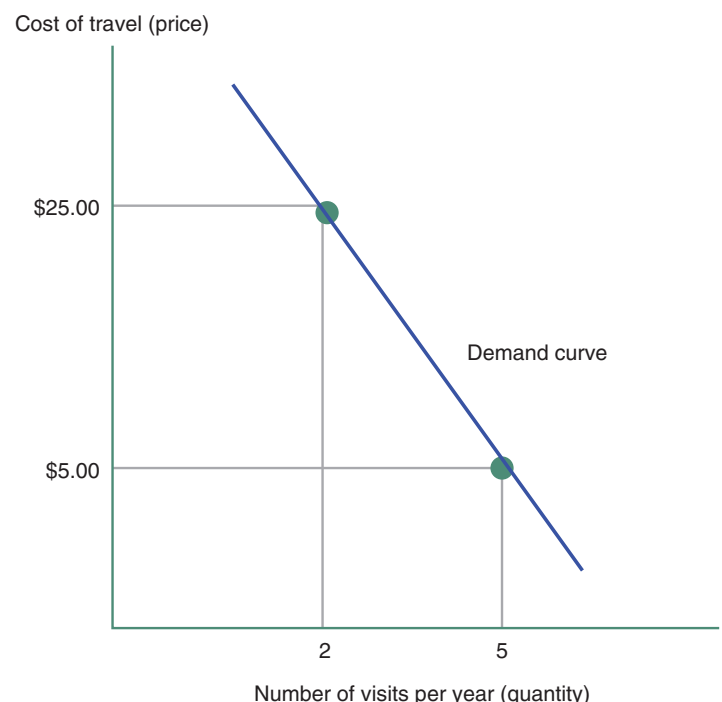


Figure 2.3—Example of a demand curve based on the cost of travel (purchase price) and number of visits to the site (quantity).

The merits of the travel cost method are that it is often relatively inexpensive to apply and uncontroversial because it is based on standard economic techniques for measuring value. The challenge with using the simple zonal approach is accurately estimating the travel costs of visitors. Transportation costs often include gas prices and—if traveling a long distance—possibly food and hotel accommodations. The value of the visitor's time is sometimes also considered by including estimates of wages lost while traveling and visiting the site. These average costs are included in the travel cost calculation as average distances, gas prices, and proportion of wage values (e.g., Amoako-Tuffour and Martinez-Españeira 2012). A potential source of error in the value estimates occurs when cost information is not collected directly from the visitor, but instead is approximated by the researcher. Error can also be associated with the assumption that the visitor's sole purpose for the trip was to recreate at that site. Similarly, the availability of substitutes and their effect on preferences are also not usually considered (Randall 1994).

By focusing only on costs, this simple valuation method tends to provide an incomplete understanding of the associated benefits. Many natural areas offer multiple activities (hiking and boating), which a given visitor, based on their customs, may or may not use. Linking travel costs with specific psychological or physiological benefits is also difficult unless additional data are collected to better understand how users perceive their experiences. It is also important to consider that the simple travel cost approach only provides values associated with current benefit levels, not changes in benefits levels, which would be more useful for planning. Finally, travel costs can only examine the use values associated with the park and not the non-use values. In short, simple travel cost estimates provide an incomplete understanding of consumer surplus and potential tradeoffs in social welfare associated with different management outcomes.

The simple zonal travel cost method could be improved by using an individual travel cost or hybrid travel cost/contingent valuation approach with a more detailed survey of visitors' characteristics and expected benefits (Armbrecht 2014, Cameron 1992, Randall 1994). Survey questions can pertain not only to individual travel costs but the purpose of the visit (e.g., hiking and/or fishing) and how changes in benefit levels or costs may impact visitor behaviors. For example, a recent study used the individual travel cost method to compare the value of non-trail backpacking activity with trail backpacking activity at Allegheny National Forest (Cho and others 2014). In another study, a series of questions describing different activities and benefit levels was used to cluster visitors into user groups, such as “backcountry enthusiasts,” “do it all adventurers,” “windshield tourists,” and a “creature comfort” group (Benson and others 2014). In these cases, the data collected not only provided park managers with estimates of economic value, but also a better understanding of how management funds should be distributed to maintain certain benefits.

Even with advanced sampling and analysis methods, the travel cost approach may not always be appropriate for estimating the value of cultural ecosystem services in all policy contexts. When the focus is on private or local forests, the travel cost approach to valuation is more likely to result in the underestimation of consumer surplus. To build a demand function, the resource needs to be far enough away to have an effect on the income of visitors, which may not be the case for local visitors. Also, those who live nearby may place a relatively higher cultural ecosystem service value on local resources due to place attachment (Kil and others 2012). This may be a particular concern in the rural Southeast where there are fewer acres of public lands per capita, compared to the West, and recreational activity on private lands is poorly or infrequently documented.

Hedonic pricing—The hedonic pricing method takes advantage of the fact that sometimes non-market values for cultural benefits such as aesthetics, recreation, and other environmental amenities (e.g., clean air and water) are embodied in the price of other goods and services and can therefore be derived (Dietz 2016). This method is most effective when there has been a distinct change in environmental quality that has impacted the quality or satisfaction associated with a related good. For example, it is assumed that home buyers select houses based on their preferences for different attributes of the house including location (e.g., distance to parks and schools), features (e.g., number of bedrooms/bathrooms), and surrounding environment (e.g., air and water quality). After controlling for variation in price, location, and house features, it is assumed that the remaining difference in home prices is related to a difference in environmental quality. The implicit price estimates are then used to estimate inverse demand functions or marginal WTP.

How hedonic values are used in a policy context depends on how the associated environmental benefits are perceived. Values associated with green space and aesthetic amenities can be used to justify changes to zoning laws and other regulations that help control urban sprawl and land use change (Gibbons and others 2014). Values associated with increases in pollution or a decline in health benefits can be used to justify costs associated with remediation or health care provision. Hedonic models have long been used by the Environmental Protection Agency to derive the value of living in neighborhoods with high or low levels of air quality (Smith and Huang 1995). A few studies have also estimated the economic value of trees or forests on property price in the Southeastern United States (table 2.3). Using the hedonic property price models, Anderson and Cordell (1988) analyzed the economic value of urban trees and found certain trees were associated with a 3.5–4.5 percent increase in homes sales price. Likewise, Mansfield and others (2005) found proximity to forest types and proportion of parcel that was forested increased home sales prices in Research Triangle, North Carolina.

Hedonic modeling uses regression analysis to determine how changes in the attributes of the good, the users, and the environment (i.e., independent variables) are correlated with

Table 2.3—Economic value of forests in the Southeast region

Study	Measurement used	Location	Method	Results
Anderson and Cordell (1988)	Number of large, small, pine, and hardwood trees in front yards of residential single family properties	Athens, GA	Hedonic property price	Trees were found to be associated with a 3.5%–4.5% increase in homes sales price
Mansfield and others (2005)	Percentage of residential single family parcel that was forested, acres of forest on a parcel, percentage of forested land within 400-m, 800-m, and 1,600-m buffers around parcel, distances to private and institutional forests	Research Triangle NC	Hedonic property price	Proximity to both forest types and proportion of parcel that was forested increased home sales prices; increasing forest cover on parcel by 10% adds less than \$800 to home sales prices while adjacency to private forests add more than \$8000

changes in price. Market data for housing is often secured from the property appraiser's or tax offices. Measured changes in environmental quality (e.g., water pH, turbidity) can be secured from the State or Federal agencies responsible for monitoring those changes. Census data is often used to describe relevant changes in social components such as demographic characteristics, annual household income, and employment. Sometimes, it may be necessary to use GIS spatial software to map the location of the house relative to the site where the service (or disservice) originates (e.g., a nature park or an industrial park). Because changes in environment quality and the housing market can be masked by other influences on the market, large amounts of data often needed for analysis and estimates depend heavily on model specification (Lipton and others 1995).

The main strength of the hedonic approach is that, as a revealed preference method, the non-market value estimates are derived from actual choices and reflect the cultural customs of homebuyers. Property markets have been found to be relatively efficient in responding to information and property records can be reliable in reporting market transactions. Still, outside influences like taxes, interest rates, or housing market bubbles can affect housing prices, masking the influence of the environmental amenities. There are also limitations to the hedonic method in regard to informing policy. Researchers can only examine the environmental qualities that are directly associated with the good (e.g., housing) and apply these values to policy contexts with a similar physical setting. For example, the value of wildlife habitat provided by trees on a parcel of land in a neighborhood may not be comparable to the value associated with habitat located in large, unbroken parcels of forest (Willis and Benson 1988).

Hedonic pricing also provides a limited understanding of the expected benefits, which is important for characterizing the cultural ecosystem service values associated with forests. It is assumed that potential homeowners are aware of the surrounding environmental conditions and will make their selection based on how it will enhance their quality of life or reduce discomfort

or effort (Dietz 2015). Moreover, hedonic pricing can only *suggest* the presence of certain self-enhancement benefits (e.g., self-seeking analysis of costs and benefits) and not the self-transcendent attitudes (e.g., altruistic and biocentric attitudes) often associated with non-use values and benefits (e.g., existence value) (Dietz 2015). Similar to travel cost methods, hedonic pricing can only account for certain use values, and therefore it can describe only part of the total economic value of a good or service. One way to improve the hedonic approach is to use it in combination with stated preference methods. Using a combined methods approach, Johnston and others (2001) found that the optimal scale and design of population growth management policies depended upon many factors, including the distribution of benefits and costs among heterogeneous groups.

Revealed preference methods are often preferred by economists because the observations included in the analysis are based on actual market data. However, to improve our understanding of cultural ecosystem service values, there is a need to better measure and quantify certain human components (i.e., beliefs and perceived benefits) that market behaviors alone cannot describe, and relate these measurements to changes in forest management. Given the flexibility, stated preference methods may instead provide the appropriate tools needed to meet this challenge.

Stated Preference Methods

Stated preference methods estimate non-market values by querying individuals on their WTP for improvements (or willingness to accept payment for losses) to an environmental good or service using choice trials and surveys. These WTP estimates can then be used to estimate consumer surplus and associated demand curves. These methods include several approaches (e.g., contingent valuation, contingent ranking and rating, contingent behavior, choice experiments, and paired comparisons) that construct hypothetical markets for ecosystem services and observe study participant behavior to infer their preferences and subsequent economic value.

Given their reliance on practitioner-defined hypothetical market contexts, stated preference methods can be applied in virtually unlimited ways to inform a diversity of policy decisions (Bennett 2011). However, an important concern associated with this method is that valuation estimates are vulnerable to several biases. For example, with hypothetical bias, respondents may consider their responses to be less consequential (and therefore respond differently) than if they had to actually pay. Response bias can occur if the people who answered the survey are not representative of the larger population. Strategies for dealing with bias concerns have been the subject of ongoing research since the early 1990s when it was recommended that stated preference methods be used to inform natural resource damage assessments in U.S. Federal courts (Arrow and others 1993, Carson and others 2003). Since that time, stated preference methods have been improved and are shown to provide valid, useful, and scientifically defensible value estimates (Haab and others 2013). Indeed, for services that lack markets altogether, stated preference methods are the only adequate link to the economic value and public preferences for these services. Stated preference approaches also offer a way to link valuation estimates to perceived (versus actual) benefits, which are critical for understanding how various groups in society may value cultural ecosystem services differently and the role of education and knowledge in enhancing perceived benefits.

Since surveys are the primary mode of data collection, survey design and associated data analysis are critical considerations for appropriately assessing the value of cultural ecosystem services. Researchers often use interviews and focus groups to inform a more thorough survey and as part of quantitative or mixed methods research approach. The data that is collected is then used to parameterize a hypothesized utility function and the design of the WTP questions. A typical utility function assumes that the utility associated with a good or service is impacted by the attributes that make up the service, the unique preferences of the respondent for that service, and the cost or price of the service. Ultimately, the data collection tool should contain opportunities

to observe respondents’ preferences to descriptions of the hypothetical changes to an ecosystem service and associated prices. Information on respondent characteristics are later integrated into analysis as additional explanatory variables.

When setting up a stated preference study to examine cultural ecosystem service values, it is important to determine which data types will best deliver the desired information for the given policy context (Carson and others 1994). Below we describe the two most commonly applied stated preference methods: contingent valuation and choice modeling. As described in the sections above, these procedures can also be combined with revealed preferences methods to meet certain data objectives.

Contingent valuation—In a contingent valuation question, the respondent is presented with two alternatives—a status quo, with respect to the current provision level of one or more ecosystem services—and the policy alternative with a specified payment vehicle (e.g., tax) and dollar amount. This method initially utilized simple question formats, such as open-ended questions (e.g., fill in the blank) or bidding procedures to elicit WTP values (table 2.4). In the more commonly applied referendum approach, the investigator presents a single offer and records the participant’s “yes” or “no” response. These responses are typically analyzed with dichotomous choice models (logit or probit) from which a welfare measure can be derived (table 2.4). The contingent valuation approach has some practical limitations in describing important tradeoffs; therefore, this approach may be suitable in contexts where decisionmakers need to understand the welfare associated with a single policy or a policy that has already been implemented. To understand tradeoffs in relative utility and WTP among different services or service levels, for planning purposes, a choice experiment approach may be more suitable.

Choice modeling—Choice modeling methods, or choice experiments, arose from conjoint-analysis procedures and have been employed in the marketing, transportation, and psychology

Table 2.4—Main forms of contingent valuation

Open-ended question	Iterative bidding	Payment card	Dichotomous choice/discrete choice	Double-bounded dichotomous choice
Respondents are directly asked their maximum willingness to pay.	Respondents are asked whether they are willing to pay a certain amount and then extra or reduced increments (bids) until their maximum WTP is reached.	Respondents are given a card that contains several payment bids and they choose one.	Known as the referenda model, respondents are asked if they support a change in environmental quality given a specific additional payment.	Similar to dichotomous choice, but then depending on whether they responded, respondents are asked if they are willing to pay a slightly higher or lower amount.

Source: Morrison and others (1997).
WTP = willingness to pay.

literature for some time. In this approach, respondents are asked to choose from a selection of alternative bundles of attributes, which is more comparable to an actual market setting where tradeoffs are assessed and decisions are made (Adamowicz and others 1998). Rather than being questioned about a single event in detail, subjects are questioned about a sample of events drawn from a universe of possible alternatives. It is assumed that the order of preferences (A over B; B over C) for different services (or levels) generated using this approach is related to the respondent's latent scale of utility (Boyle 1990). Compared to contingent valuation, the choice experiment approach offers a richer description of the attribute tradeoffs individuals are willing to make and the estimated WTP values generally have smaller variances relative to the mean (Adamowicz and others 1998).

The effect of changing the level of a single attribute is known as a main effect. To understand main effects on the WTP decision, a main effects orthogonal array can be used to design a series of choice sets (Street and others 2005). To illustrate this procedure, a list of key attributes describing a forest-water protection program was developed through a series of interviews with the members of the public and experts in the field (see table 2.5; Kreye and others 2016b). Attributes of the clean water service provided by forests, including use and non-use values, were arranged into 16 different choices sets, each containing one attribute level from every attribute category.

Latent utility for ecosystem service attribute levels can be measured using several different methods (Carson and others 1994). In a dichotomous choice question format, the respondent is able to accept or reject the ecosystem service bundle or outcome described in each choice set (but without a specific reference to the status quo, as with contingent valuation) (fig. 2.4). This approach is similar to how consumers make decisions, which are often binary (e.g., category or brand decision, buy now or wait, etc.), and binary decisions are often consistent with economic demand theory. To produce a direct measure of relative utility for each attribute level, a ranking procedure may be used. In best-worst scaling, the respondent is asked to indicate which attribute is the “best” and which is the “worst” within each choice set (Louviere and Islam 2008). A novel approach to survey design is to combine ranking and dichotomous choice methodologies in the same choice experiment question to generate both direct and indirect estimates of utility and allow for internal validation procedures (Kreye and others 2016b, Soto and

Table 2.5—Survey items describing attributes of a forest-water protection program

Attribute category	Attribute levels	Item code
Non-use benefits	Ensure clean water in the future (option value)	1
	Ensure all people benefit (altruistic value)	2
	Pass along to grandchildren (bequest value)	3
	Maintain an unpolluted ecosystem (existence value)	4
Use benefits	Recreation	5
	Groundwater recharge (drinking water)	6
	Commercial fisheries	7
	Storm protection	8
How the program will be implemented	Purchase private forest land	9
	Conservation easement	10
	Technical assistance for landowners	11
	Financial assistance for landowners	12
Who will implement the program	Federal agency	13
	State agency	14
	County government	15
	Private non-profit organization	16
Proposed monthly utility tax payment	\$0.50/month	17
	\$1.00/month	18
	\$3.00/month	19
	\$5.00/month	20

Source: Kreye and others (2016b).

Water-Forest Protection Program A

Purchase private forest land

Implemented by a State agency

Ensures clean water in the future

Provides recreational benefits (e.g., swimming)

Funded via a utility tax of \$0.50/month (\$6.00 annually)

Would you vote to accept Program A (all characteristics)?

☒ Yes

☐ No

Figure 2.4—Example of a dichotomous choice experiment question presented in a valuation survey.

others 2013, 2016). Other methods that measure latent utility using a choice modeling approach include: ranking alternatives in order of preference, series of paired comparisons, bundle selection (i.e., pick some number of the best alternatives), judgmental ratings, and fixed-sum allocation tasks (Carson and others 1994).

Data analysis procedures for choice modeling are regression-based and produce statistically defensible estimates of value based on calculated probabilities. The independent variables in the model reflect the specified utility function and include specific attributes of the service (or multiple services) and of the respondent (e.g., demographic characteristics). Depending on how the dependent variable is characterized, the data are most often fitted to either a multinomial logit or probit model, but linear, quadratic, or log response functions may also be chosen (Boyle 1990, Johnson 2013). Advanced functional forms of these models (i.e., the conditional, random effects or mixed models) are largely recognized as being more effective in accounting for unobserved random utility or realistic variation in WTP estimates (Wooldridge 2013). Table 2.6 describes the outcome of a random-effects logistic model containing the attributes listed in table 2.5.

The part-worth value, or marginal rate of substitution, for each attribute and the expected mean WTP value can be calculated using the estimated model coefficients (Hanemann 1984). Confidence intervals around mean WTP can be calculated using the variance-covariance matrix and a simulation approach (Krinsky and Robb 1986). Table 2.7 reports mean WTP estimates

for some of the non-use values and policy processes found in table 2.5 and illustrates how individual WTP for a given policy alternative can be extrapolated to a larger population when it is multiplied by the number of households at the policy site. The estimated total annual value describes potential market demand for the associated non-market service.

Stated preference and cultural ecosystem service values—

Stated preference methods are well suited to assess cultural ecosystem service values because expected benefits can be part of the valuation question, whereas with revealed preference methods, expected benefits are often inferred. For example, using a choice experiment approach, researchers found that Danish recreationists experienced the greatest aesthetic benefits, and WTP, in forest stands that contain a mix of conifers and broadleaf trees (Nielsen and others 2007). The study presented in table 2.5 described public values for clean water benefits by including descriptions of non-use values and beliefs about the effectiveness of certain conservation processes (e.g., conservation easements or technical assistance for landowners) as attributes in a choice experiment (Kreye and others 2016b).

The stated preference approach can also include metrics that describe important attitudes or knowledge as explanatory covariates in the predictive model (Manning and others 1999, Walsh and others 1990). Attitudes toward the environment, as described by the new ecological paradigm scale (i.e., biocentric, anthropocentric) have been found to have an important influence on WTP behaviors for environmental goods and services (Dunlap

Table 2.6—Dichotomous choice model using random effects logistic regression to estimate household willingness to pay (WTP) for a forest-water protection program in Florida

Variable	Coefficient	S.E.	p>Z	Δ in monthly WTP ^a	Mean annual WTP
Payment	-0.5070	0.3200	0.0000		\$29.64
Ensure all people benefit	-0.1640	0.0840	0.0490	-\$0.81	
Ensure clean water in the future	0.2300	0.0990	0.0210	\$1.13	
Groundwater recharge (drinking water)	-0.3880	0.0820	0.0000	-\$1.89	
Conservation easement	0.3670	0.1000	0.0000	\$1.78	
Technical assistance for landowners	-0.2370	0.0850	0.0050	-\$1.15	
Private non-profit organization	0.3380	0.0990	0.0010	\$1.78	
County government	-0.2190	0.0830	0.0080	-\$1.08	
Weight	1.1970	0.3150	0.0000		
Constant	-0.9610	1.7760	0.0000		

Adapted from Kreye and others (2016b).

Number of observations = 4,140.

^a Difference from mean monthly WTP estimate of \$2.47 (2014 dollars).

Table 2.7—Total annual benefits estimated for 50 percent of households for two policy alternatives to protect forests and water resources in the State of Florida

Policy alternatives	Household		Percent of households	Number of households	Annual total
	Monthly WTP	Annual WTP			
<i>Alternative 1:</i> Ensure clean water in the future using conservation easements	\$5.38	\$64.56	50%	3,573,508	\$230,705,676
<i>Alternative 2:</i> Ensure all people benefit from clean water by offering landowners technical assistance	\$0.51	\$6.12	50%	3,573,508	\$21,869,868

Based on Kreye and others (2016b).
WTP = willingness to pay.

and others 2000, Kreye and others 2016b, Spash and others 2009). Metrics, such as level of ecological and environmental knowledge and level of environmental concern, are also related to important attitudes and beliefs that make up cultural values and often used by researchers to explain consumer behaviors for environmental goods (Mostafa 2007, Nielsen and others 2007).

As a caution, standard descriptions of the population, such as demographic characteristics (e.g., age, race, household income, employment, place of residence) may offer an incomplete description of the human components that make up cultural ecosystem service value. Estimates of WTP and attitudes toward forest ecosystems have been broadly correlated with demographic characteristics (Tarrant and Cordell 2002, Torgler and others 2008, Vaske and others 2001). However, this pattern may not always hold across all populations or be useful in all policy contexts. For example, campers and hunters in Canada tend to have strong biocentric attitudes, and these attitudes are not strongly correlated with the socio-economic characteristics of the campers and hunters (McFarland and Boxall 2000).

Valuation studies nearly always include variables describing population characteristics but only intermittently include variables measuring associated attitudes or psychological benefits. This is because these aspects of human behavior are usually outside the training of a traditional economist who often works with market data. Economists with training in stated preference methods and who work in interdisciplinary fields such as psychology, marketing, behavioral and ecological economics are often better equipped to provide decisionmakers with an understanding of the human components that give rise to cultural ecosystem services and impact WTP estimates.

Benefit Transfer Method

A benefit transfer is a procedure that transfers existing estimates of non-market values to a new study or policy context that differs from the original study. Benefit transfer is often used in legal

proceedings and government policy analysis when there is not sufficient time or funds available to develop original benefit estimates using primary data specific to the issue at hand (Boyle and Bergstrom 1992). Rosenberger and others (2017) describe its application to estimating the economic benefits of outdoor recreation in the National Forest System. The following sections provide a description of common benefit transfer procedures and important considerations for using benefit transfer to estimate cultural ecosystem service values.

The simplest form of benefit transfer is when the unit value(s) is directly transferred from the study site to the policy site with only a few minor adjustments (e.g., inflation). An improvement on this method is a preference calibration transfer that calibrates the preference structure (i.e., utility model) described in the original study to match the characteristics of the policy site (Smith and others 1999). In cases where there are many studies available, a meta-analysis benefit transfer function may be used. In a regression-based meta-analysis, the dependent variable is the estimated WTP and the information provided in the study are the independent variables or factors found to influence WTP estimates (e.g., age, income). The entire equation (function) is then transferred to the site by adjusting the model to fit the policy context or the population affected by the policy. Figure 2.5 illustrates the general differences between these three benefit transfer methods.

Unit value transfer—A direct unit value transfer is one of the more commonly used methods to help inform agency decisionmaking (Brown and Shi 2014, Gilmore 2014, Kroeger 2005). This is done by applying a single statistic (usually an average from one or more study sites) to the policy site. For example, a mean WTP estimate for lake fishing from a single study can be transferred to assess the value of fishing on similar lakes. In the context of cultural ecosystem service values, transferring a simple mean value across relatively similar sites may only be appropriate when the benefiting population is relatively the same (Bateman and others 2011).

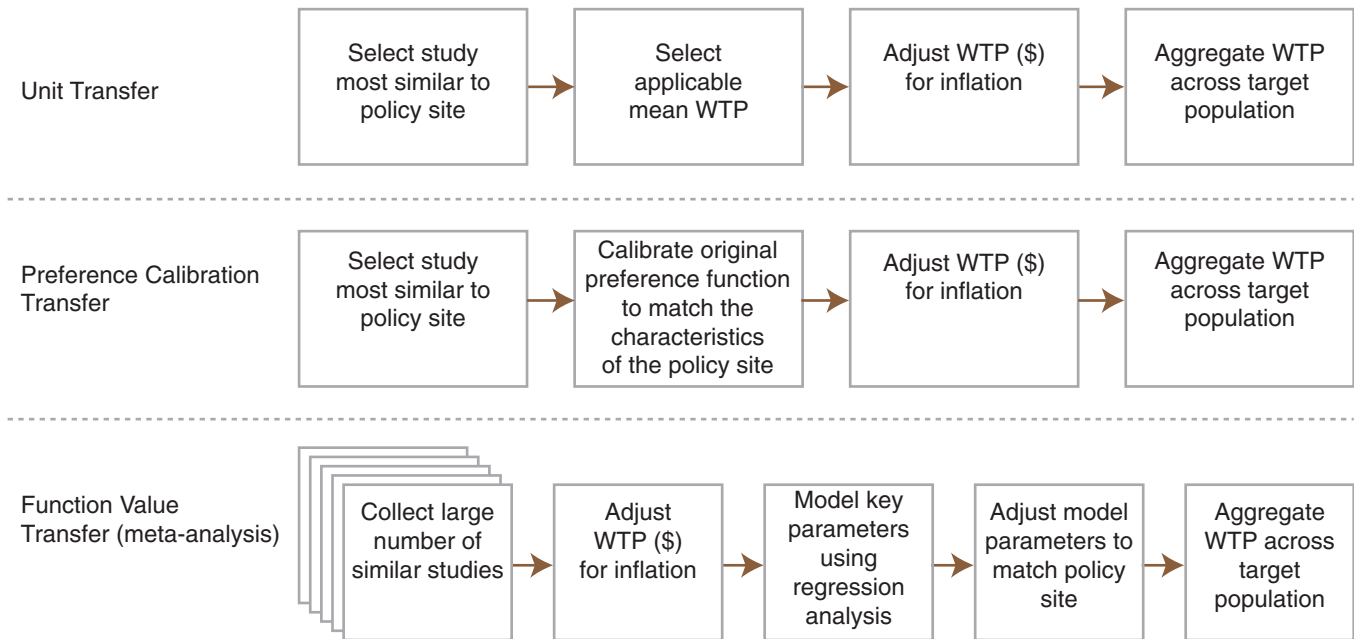


Figure 2.5—Comparison of general benefit transfer methods.

Preference calibrated benefit transfer—A benefit transfer approach known as preference calibration offers an improvement over the direct unit value transfer approach by linking the fundamental preference structure of individuals with WTP estimates. Through an algebraic exercise, the researcher can “back out” the parameters of the original function presented in a single study; the function is then calibrated and used in a new WTP function at the policy site (Smith and others 1999). Assuming that the ecosystem characteristics and benefits generated at the study and policy sites are similar, this approach allows researchers to address variation in preferences based on the features in the household budget constraint, such as household income, average travel cost, and rent. This approach does require researchers to be well founded in utility function theory and have advanced algebraic skills. Williamson and others (2007) provides an example of how to perform a preference calibration transfer for changes in water quality using hedonic property values, travel cost, and contingent valuation values.

Benefit transfer meta-analysis—A meta-analysis is a statistical method for analyzing results from existing studies with a set of related research hypotheses and has widespread use in several areas including health sciences, psychology, education, marketing, and social sciences (Nelson and Kennedy 2009). In economics, meta-analyses are often used to synthesize numerous studies that place economic values on environmental goods and services (Rosenberger and Loomis 2000). The results of a meta-analysis can be used to predict estimates of value constructs (i.e., influential factors, or drivers) while controlling for important differences among studies. Given that the sources of variation

in WTP can occur at both the site and population level, a meta-analysis approach may be preferred to other benefit transfer approaches (Rosenberger and Loomis 2000, Walsh and others 1990). As such, the meta-analysis approach to benefit transfer is provided in the next section.

Meta-analysis approach—A meta-analysis is a regression-based modeling approach and the studies that are selected and included in the analysis formulate WTP using the same methodological approach (e.g., contingent valuation) for the same services or benefits (e.g., hiking) in the same general type of ecosystem (e.g., a forested landscape). Researchers may also consider if the studies are conducted within a given region and within a designated time frame (e.g., Rosenberger and Loomis 2000). The value estimate (WTP) selected from a set of related studies (e.g., contingent valuation studies) serves as the dependent variable and variables describing study methods, target population, and site characteristics serve as the independent variables (Rosenberger and Loomis 2000) (fig. 2.6).

Standard error, standard deviation, and sample size are useful proxies for study quality and are often used as a weighting variable to address sample bias or to reduce the impact of lower quality studies on model performance (Nelson and Kennedy 2009). The data can be fitted to a wide variety of models including linear and weighted regression and ordinary least squares models. The resulting coefficients are then adjusted to meet the characteristics of the policy site and back calculated to obtain estimates of WTP (see Johnston and Besedin 2009). Confidence intervals can also be calculated using the equations

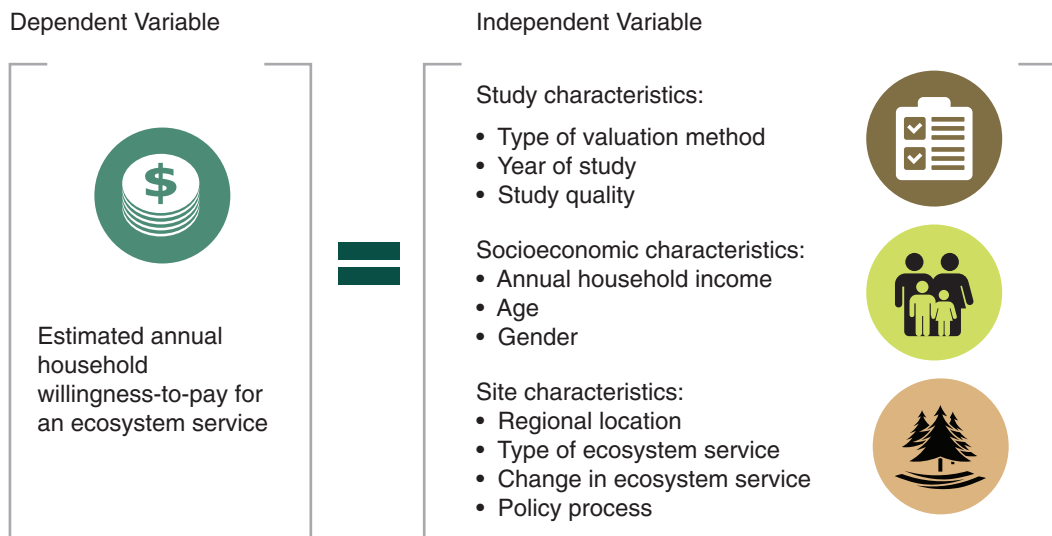


Figure 2.6—Conceptual model of a meta-analysis of valuation studies.

and the statistics commonly reported in regression analysis such as the margin of error and standard error. For more examples of a meta-analysis benefit transfer of forest values, see Chiabai and others (2009), Ding and others (2010), Escobedo and Timilsina (2014), Kreye and others (2014), and Zandersen and Tol (2009).

Benefit transfer and cultural ecosystem service values—The primary challenge of benefit transfer is the presence of important and unaccounted differences between the study site and the policy site. Selecting studies that help minimize these differences can be particularly challenging in the context of cultural ecosystem service values, as many valuation studies often fail to appropriately measure important human components (e.g., perceived benefits, beliefs) that give rise to cultural ecosystem service values. Without an understanding of the underlying cultural benefits, or the values associated with different benefits, we cannot adequately determine how changes in forest management or service provision impact public welfare. Even with calibration procedures, transfers across different sites are subject to less error than a transfer across populations for a given site (Morrison and others 2002). In other words, WTP estimates are more often impacted by the characteristics of the population valuing the forest, rather than the characteristics of the forest itself. As discussed earlier in the chapter, relying on demographic data alone to describe perceived benefits may provide an incomplete description of important cultural ecosystem service values (Bengston 1994). Another challenge is that many benefit transfer studies do not account for differences in perceived risk and the availability of substitutes across sites, which can also impact WTP behaviors and lead to the under or overestimation

of transfer of values (Batabyal and others 2003, Johnston 2007). Until the body of research examining cultural ecosystem services can provide a more contextualized description of the human components that give rise (e.g., beliefs, attitudes) or reinforce (e.g., perceived benefits) cultural ecosystem service values, the benefit transfer method will likely remain problematic in producing accurate and reliable estimates of value for many types of cultural ecosystem services.

CONCLUDING REMARKS

This chapter reviewed ways that human culture gives important meaning to forests that society recognizes as valuable. We also described commonly used approaches for assessing the economic value of cultural ecosystem services and highlighted important challenges associated with these valuation methods. Clearly, efforts to understand tradeoffs in the provision of cultural ecosystem services is a formidable challenge. Despite this, integrating cultural ecosystem service values into decisionmaking remains a pressing issue. Failing to account for diminished cultural services that result from forest management decisions or loss of forest lands will likely lead to less effective forest policy and reduced social welfare. Future research should work to advance valuation methods and approaches while continuing to investigate how cultural ecosystem service values are mediated by perceptions, customs, and beliefs. Providing decisionmakers with improved value estimates can lead to better policy and program design and help ensure that society continues to benefit from the full spectrum of ecosystem services provided by public and private forest lands.

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Appendix 2.1

Recreation and tourism values of forests in the Southeast region

Study	Geographic scope of values	Basis for valuation	Value estimates
Walsh and Loomis (1989)	Virginia	Use value of wilderness protection (contingent valuation)	\$12/visitor day
Zhang and others (2004)	Alabama	Hunters' WTP (contingent valuation)	\$1.29 per acre per hunter or \$23 per acre per hunter
Munn and others (2011)	Mississippi	Hunters' WTP (contingent valuation)	\$0.65 to \$6.40 per acre
Brunke and others (2006)	Mississippi	Hunters' WTP (contingent valuation)	\$11 and \$36.25 for fall and spring turkey hunting permits
Moore and others (2011)	Georgia	Non-use value of private forests	\$11.2 billion per year
Haefele and others (1991)	Southern Appalachian region*	Non-use value (WTP for protecting forests)	\$100 per year and \$21 per year for dichotomous choice and payment card estimates
Talberth and Moskowitz (1998)	Southern Appalachian region	Expenditures on hunting equipment and trip expenses	\$1.3 billion in 1988
Loomis and Rochardson (2000)	Southern Appalachian region	Consumer surplus of recreational fishing	\$237 million to \$637 million in 1997
Barnhill (1999)	National Forests of the Southern Appalachian region	Economic impact of recreational fishing	\$407 million
Barnhill (1999)	National Forests of the Southern Appalachian region	Total economic impact of hunting and wildlife viewing	\$594 million and \$407 million, respectively, in 1996
Barnhill (1999)	National Forests of the Southern Appalachian region	Economic impact of recreation on the National Forests	\$6 billion annually

* The Southern Appalachian region consists of Alabama, Georgia, North Carolina, South Carolina, Tennessee, and Virginia.

Chapter 3

Forest Ecosystem Services: Water Resources

Thomas P. Holmes, James Vose, Travis Warziniack, Bill Holman

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-Watauga 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.

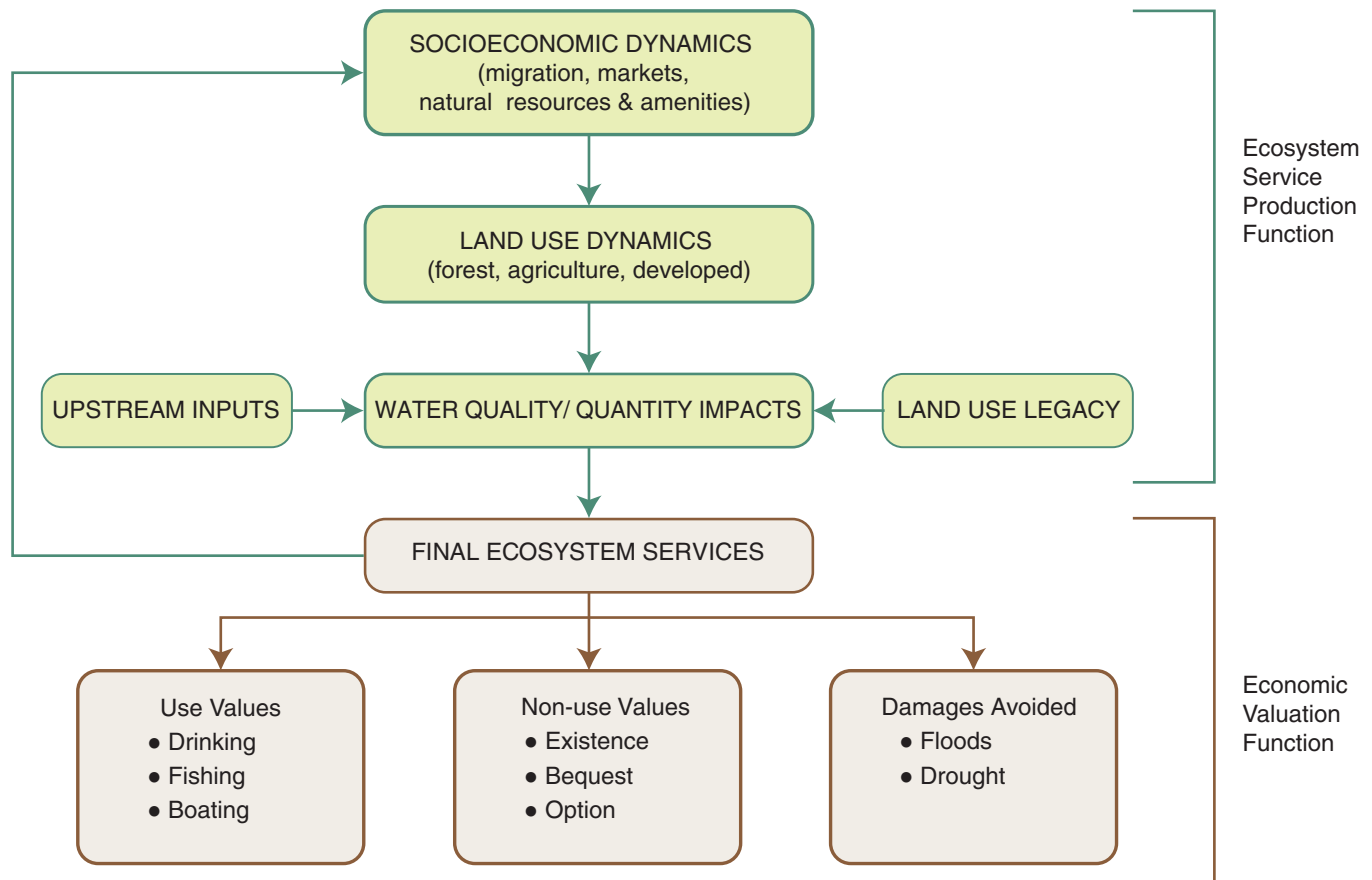


Figure 3.1— Ecosystem service assessments can be linked with economic valuation functions to measure how changes in forest landscapes affect the economic value of water resources.

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 Banzhaf 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 topography)

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 Cr  taz 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 ($\text{ft}^3 \text{ s}^{-1}$). 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 ($\text{ft}^3 \text{ s}^{-1}$) 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 (Dearthmont 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.

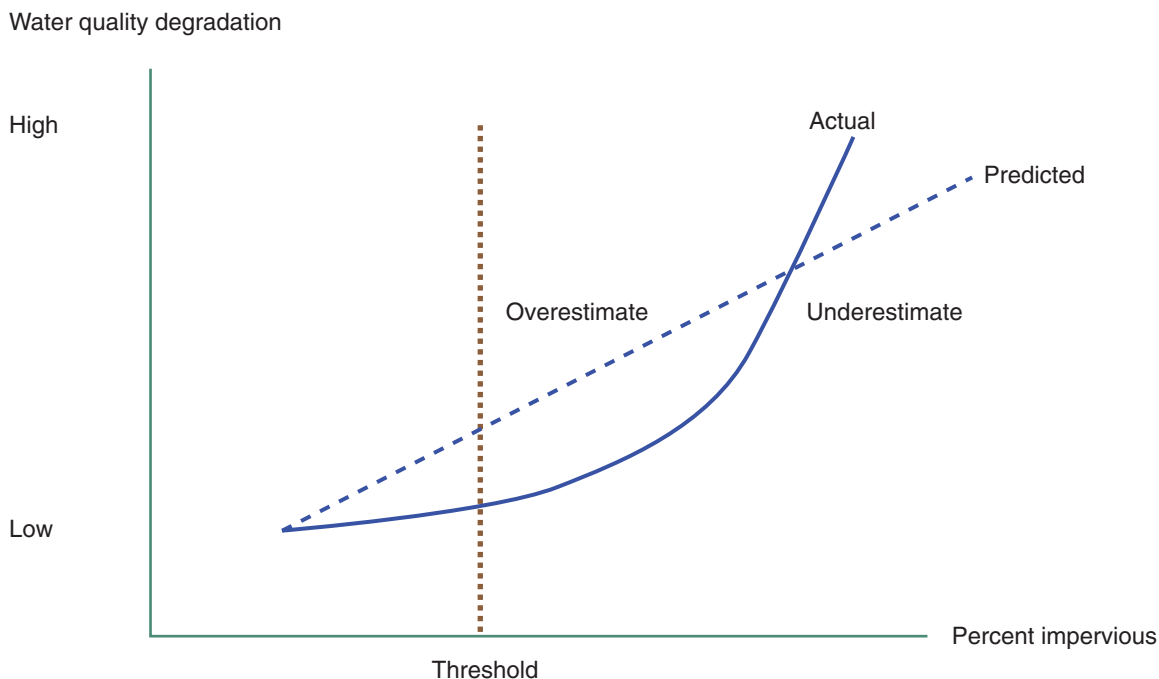


Figure 3.2—Landscape-riverscape production of water quality is nonlinear: assuming a linear relationship can result in potentially large errors.

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 ft² 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.

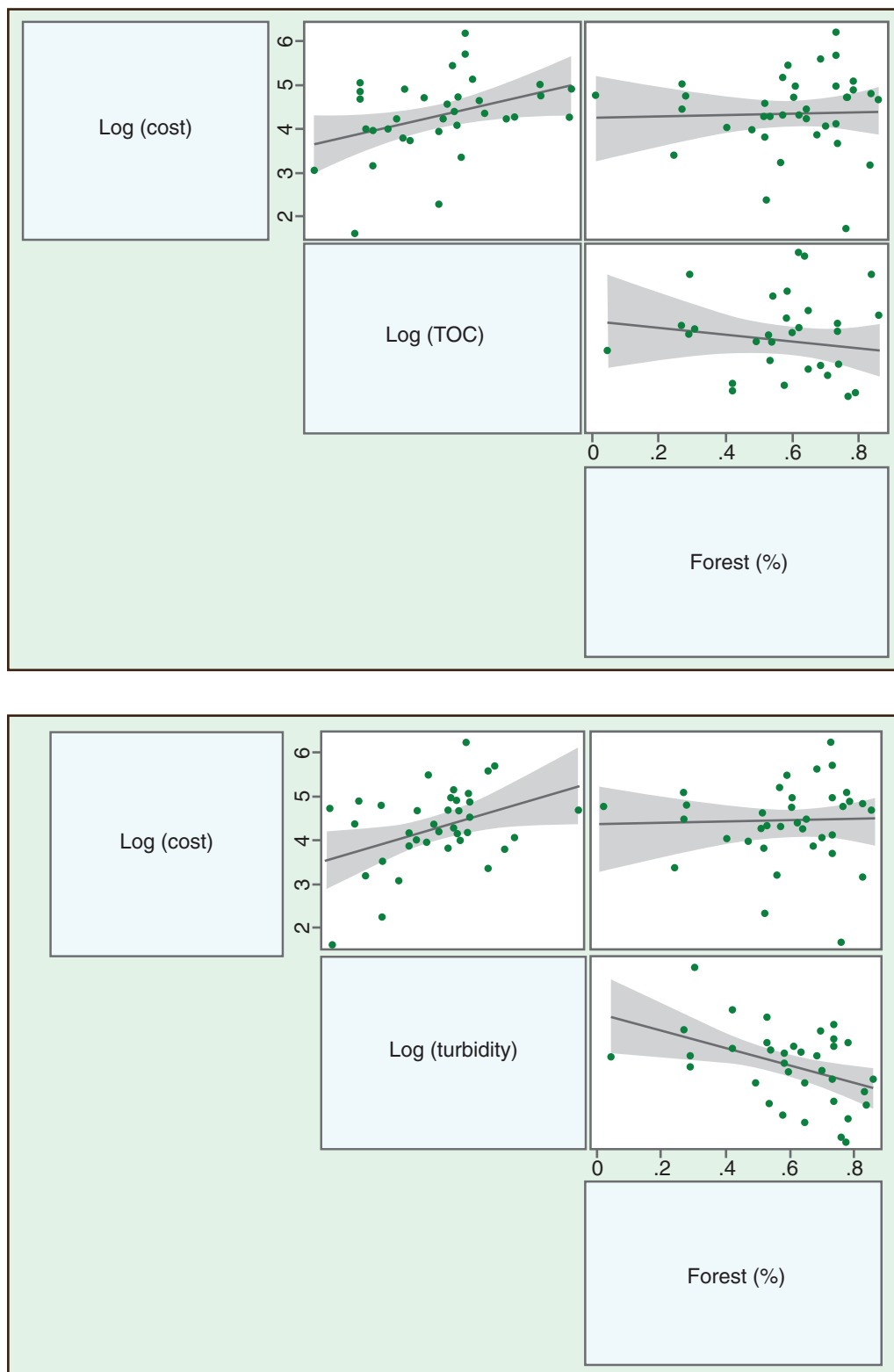


Figure 3.3— Empirical relationships between percent forest cover, total organic content (TOC), turbidity, and water treatment costs (cost) (Warziniak and others 2017).

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).

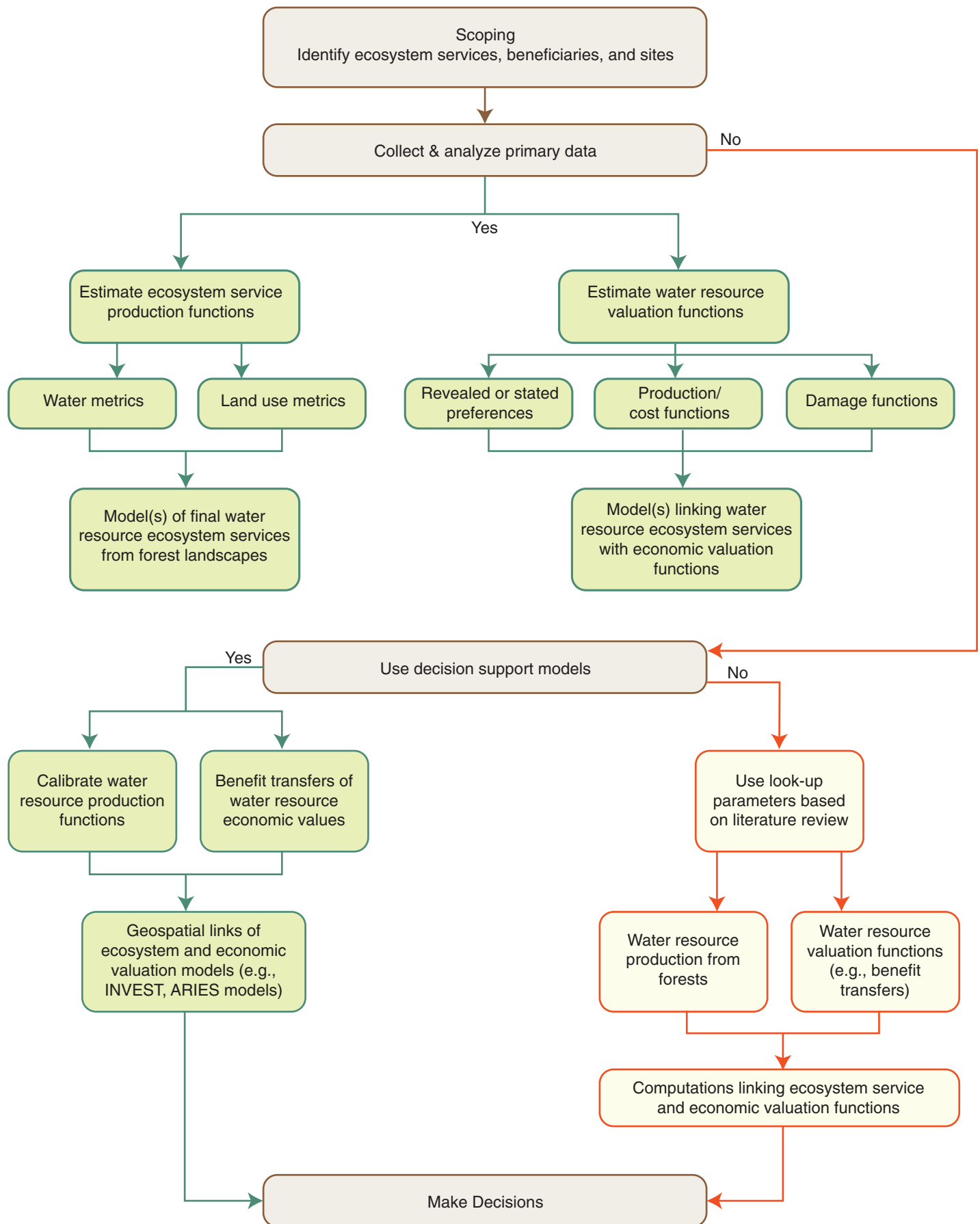


Figure 3.4—Decision tree for estimating water resource economic values from forest landscapes.

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 of 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/ecosystems/services/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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Chapter 4

Forest Ecosystem Services: Carbon and Air Quality

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INTRODUCTION

Forests provide various ecosystem services related to air quality that can provide substantial value to society. Through tree growth and alteration of their local environment, trees and forests both directly and indirectly affect air quality. Though forests affect air quality in numerous ways, this chapter will focus on five main ecosystem services or disservices related to air quality that have the potential to be estimated for forest stands:

- (1) Air pollution removal and its effect on air pollution concentrations,
- (2) Volatile organic compound emissions,
- (3) Pollen emissions,
- (4) Carbon sequestration, and
- (5) Air temperature reduction.

The objectives of this chapter are to:

- (1) Provide a background on how forests influence each of the above ecosystem services,
- (2) Recommend methods on how to quantify the magnitude of these ecosystem services, and
- (3) Review new approaches in assessing the value associated with these ecosystem services.

BACKGROUND

For each of the five air quality ecosystem services, this section will provide a brief description of: a) how forests impact the service, b) past forest ecosystem service assessments and approaches to value the ecosystem service, and c) challenges associated with estimating the service and values. However, before assessing ecosystem services and values derived from a forest, it is critical to assess the forest structure, as structure strongly influences the ecosystem services.

Assessing Forest Structure and Cover

There are four main steps needed to quantify ecosystem services and values from forests:

1. Quantify the forest structural attributes (e.g., number of trees, tree cover) that provide the service for the area of interest.
2. Quantify how the structure influences the ecosystem service (e.g., tree density, tree sizes, and forest species composition are significant drivers of carbon storage).
3. Quantify the impact of the ecosystem service, because it is typically the impact of the service on human health or other attributes of the environment that provide value to society.
4. Quantify the economic value of the impact of the ecosystem service.

In quantifying the forest structure (step 1), there are various sub-steps that could be followed:

- a) Delimit the boundaries of the forest area of interest (study area) and determine the area of forest land.
- b) Determine the percentage or amount of tree cover within the study area. This information can be derived from the National Land Cover Database (NLCD) (USGS 2015), but the 2001 NLCD data tended to underestimate tree cover (Nowak and Greenfield 2010). Cover data can also be photo-interpreted (e.g., Nowak and Greenfield 2012) using i-Tree Canopy (www.itreetools.org), which allows users to easily interpret Google images. However, depending on image resolution, all forest areas may not be interpretable. Tree cover maps have an inherent error that may or may not be known (often photo-interpretation is used to determine the map error). NLCD 2001 tree cover layers, on average, underestimate tree cover by 9.7 percent nationally, but the differences vary by region and land cover class (Nowak and Greenfield 2010). These data layers can be adjusted to meet photo-interpreted estimates, but there will be errors in the locations of adjusted tree cover. High resolution tree cover

layers or data often produce more accurate maps, particularly when LIDAR is used, but also have errors that are often hand corrected. These hand-corrected data sets can have error rates < 5 percent. With photo-interpretation, the cover attributes are assumed to be classified without error and standard error of the estimates are reduced with increased sample size.

- c) Determine the structural characteristics of the forest area (e.g., number of trees by species, diameter and condition class) by sampling the area of interest. This information can often be obtained from USDA Forest Service Forest Inventory and Analysis (FIA) data, particularly for rural forests. The most important forest structural attributes used in assessing forests effects on air quality include total tree biomass, tree condition, crown competition, and leaf area and leaf biomass by species.
- d) If field data are not available, structural characteristics can be estimated by extrapolating a regional average of characteristics per unit tree cover (e.g., number of trees per hectare of tree cover) to tree cover in the area of interest.

Estimates based on measurements in the field assume that plot/tree data are measured without error and sampled properly (e.g., random samples). Estimates from these data have an associated estimate of sampling error. When extrapolating regional standardized values per unit tree cover to the study area, additional uncertainty is added by assuming that the regional average applies to the condition of the study area, and there is also an additional sampling error in estimating tree cover in the study area (which is often quite small and can be calculated).

From these basic forest structural data, estimates of ecosystem service flows and values can be derived through process models and economic valuation procedures.

Air Pollution Removal and Its Effect on Air Pollution Concentrations

Biophysical service—Trees affect air quality through the direct removal of air pollutants, by altering local microclimates and building energy use, and through the emission of volatile organic compounds (VOCs) that can contribute to ozone (O₃), carbon monoxide (CO), and particulate matter formation (e.g., Chameides and others 1988). However, integrative studies have revealed that trees, particularly low VOC-emitting species, can be a viable strategy to help reduce urban O₃ levels (e.g., Taha 1996). While all plants can impact air quality, trees tend to have greater impacts due to their larger leaf surface area. In general, the best tree species for improving air quality are species with a large healthy leaf surface area, relatively low VOC emissions, low maintenance needs, and a long lifespan (are adapted to the site conditions). Species that transpire more water will have a greater capacity to reduce air temperatures and remove gaseous pollutants. Species with more textured or waxy surfaces and smaller leaves are generally better at capturing particulate matter. In addition, evergreen species offer the ability to capture particles year-round.

Trees remove gaseous air pollution primarily by uptake through leaf stomata, though some gases are removed by the plant surface area. For O₃, sulfur dioxide (SO₂), and nitrogen dioxide (NO₂), most of the pollution is removed via leaf stomata. Once inside the leaf, gases diffuse into intercellular spaces and react with inner-leaf surfaces or may be absorbed by water films to form acids (Smith 1989). Trees directly affect particulate matter in the atmosphere by emitting particles (e.g., pollen), intercepting particles, and resuspending particles captured on the plant surface. Some particles can be absorbed into the tree, though most intercepted particles are retained on the plant surface. The intercepted particles are often resuspended to the atmosphere, washed off by rain, or dropped to the ground with leaf and twig fall. During dry periods, particles are constantly intercepted and resuspended, in part, dependent upon wind speed. The accumulation of particles on the leaves can negatively affect photosynthesis (e.g., Darley 1971) and therefore potentially negatively affect gaseous pollution removal by trees. During precipitation, particles can be washed off and either dissolved or transferred to the soil. Consequently, vegetation is only a temporary retention site for many atmospheric particles, which are eventually moved back to the atmosphere or moved to the soil. Once in the soil, some chemical elements can be retained for substantial periods in slowly decomposable woody debris (Aber and Melillo 1982, Bieby and others 2011).

In addition to pollution removal via dry deposition, forests also affect local meteorology. Trees influence air temperature, radiation absorption and heat storage, wind speed, relative humidity, turbulence, surface albedo, surface roughness, and the atmospheric mixing-layer height. These effects consequently impact emission of pollutants from various sources and the concentration of pollutants in the atmosphere. For example, lower temperatures will reduce the emission of numerous biogenic and anthropogenic VOCs and other temperature-dependent pollutant emission sources (Cardelino and Chameides 1990). In addition, altering the local environment (e.g., air temperature reduction, shade, altered wind speeds) will affect building energy use and consequently emissions from power plants. Reductions in wind speed can reduce the dispersion of pollutants, which will tend to increase local pollutant concentrations as the pollutants are not dispersed as much with lower wind speeds. Subsequently, with slower winds the volume of the atmosphere where the pollutants mix can be reduced. This reduction in the “mixing height” will also tend to increase pollutant concentrations as the same amount of pollution is now mixed within a smaller volume of air (e.g., Nowak and others 2000).

Pollution removal by urban trees in the United States has been estimated at 711,000 tonnes (t) per year with average percentage air quality improvement in cities during the daytime of the season that vegetation is in-leaf typically < 1 percent (Nowak and others 2006). A more recent assessment of pollution removal by trees across the conterminous United States estimated pollution removal at 17.4 million t in 2010 (range: 9.0-23.2 million t) with

96 percent of the pollution removal occurring in rural areas. This pollution removal also equated to an average air quality improvement of < 1 percent (Nowak and others 2014).

There are many factors that determine the ultimate effect of trees on air pollution. Integrative studies of tree effects on ozone pollution have illustrated how these various factors affect air quality. One model simulation illustrated that a 20-percent loss in forest cover in the Atlanta area due to urbanization led to a 14-percent increase in ozone concentrations for a typical summer day (Cardelino and Chameides 1990). Although there were fewer trees to emit VOCs (chemicals that can contribute to ozone formation), an increase in Atlanta's air temperatures due to the increased urban heat island, which occurred concomitantly with tree loss, increased VOC emissions from the remaining trees and other sources (e.g., automobiles) and altered the chemistry of ozone formation (e.g., reaction rates) such that concentrations of ozone increased.

Another model simulation of California's South Coast Air Basin suggests that the air quality impacts of increased urban tree cover can be either positive or negative with respect to local ozone concentrations. However, the net basin-wide effect of increased urban vegetation is a decrease in ozone concentrations if the additional trees are low VOC emitters (Taha 1996).

Modeling the effects of increased urban tree cover on ozone concentrations in several cities from Washington, DC, to central Massachusetts revealed that urban trees generally reduce ozone concentrations in cities but tend to slightly increase average ozone concentrations regionally (Nowak and others 2000). As previously explained, the effects of trees on the physical and chemical environment demonstrate that trees can cause changes in pollution removal rates and meteorology, particularly air temperatures, wind fields, and mixing-layer heights, which, in turn, affect ozone concentrations. Changes in urban tree species composition had no detectable effect on ozone concentrations (Nowak and others 2000). Modeling of the New York City metropolitan area also reveals that increasing tree cover 10 percent within urban areas reduced maximum ozone levels by about 4 parts per billion (ppb), which was about 37 percent of the amount needed for air quality standards attainment (Luley and Bond 2002).

Though reduction in wind speeds can increase local pollution concentrations due to reduced dispersion of pollutants and mixing height of the atmosphere, altering wind patterns can also have a positive effect. Tree canopies can potentially prevent pollution in the upper atmosphere from reaching ground-level air space. For example, measured differences in ozone concentration between above- and below-forest canopies in California's San Bernardino Mountains have exceeded 50 ppb (equivalent to a 40-percent improvement below the canopy) (Bytnerowicz and others 1996). Under normal daytime conditions, atmospheric turbulence mixes the atmosphere such that pollutant concentrations are relatively

invariant with height. Forest canopies can limit the mixing of upper air with ground-level air, leading to significant below-canopy air quality improvements. Standing in the interior of forest stands can offer cleaner air if there are no local ground sources of emissions (e.g., from automobiles). Various studies have illustrated reduced pollutant concentrations in the interior of forest stands compared to outside of the forest stand (e.g., Cavanagh and others 2009, Dasch 1987). However, where there are numerous pollutant sources below the canopy (e.g., automobiles), the forest canopy could increase concentrations by minimizing the dispersion of the pollutants away from ground level (Gromke and Ruck 2009, Salmond and others 2013, Vos and others 2013, Wania and others 2012). This effect could be particularly important in areas with heavy tree canopy and vehicle traffic.

Economic valuation—The values associated with reduced air pollution concentrations are generally related to improved human health, improved visibility, and reduced damage to materials, plants, and ecosystems. Some studies have used “externality” values to estimate the value of pollution removal. For example, the value of the 711,000 t removed per year by U.S. urban forests was estimated at \$3.8 billion using externality values (Nowak and others 2006). In this context, “externality” values are the estimated cost of pollution to society that is not accounted for in the market price of the goods or services that produced the pollution. There are a few studies that have linked pollution removal and improved health, including one in London where a 10 × 10 km grid with 25-percent tree cover was estimated to remove 90.4 t of PM₁₀ annually, which equated to the avoidance of two deaths and two hospital admissions per year (Tiwary and others 2009). In addition, Nowak and others (2013) reported that the total amount of PM_{2.5} removed annually by trees in 10 U.S. cities in 2010 varied from 4.7 t in Syracuse to 64.5 t in Atlanta. Estimates of the annual monetary value of human health effects associated with PM_{2.5} removal in these same cities (e.g., changes in mortality, hospital admissions, respiratory symptoms) ranged from \$1.1 million in Syracuse to \$60.1 million in New York City. Mortality avoided was typically around one person per year per city, but was as high as 7.6 people per year in New York City. Most of the health values came from reduced mortality, which was estimated based on the value of a statistical life (e.g., Viscusi and Aldy 2003). The human health value of the 17.4 million t of air pollution removed by conterminous U.S. forests in 2010 was \$6.8 billion (Nowak and others 2014). Sixty-seven percent of the pollution removal value occurred in urban areas. Health impacts included the avoidance of more than 850 deaths and 670,000 incidences of acute respiratory symptoms. Health valuation is based on the U.S. Environmental Protection Agency (EPA) BenMAP model procedures that estimate the health impacts and monetary value when populations experience changes in air quality (Abt Associates 2010, Davidson and others 2007, U.S. EPA 2012). The health value varies spatially based on changes in pollution concentration and the number and age of people receiving that change in concentration.

Challenges in estimating air pollution impacts—Computer modeling has overcome some of the challenges related to quantifying the impacts of trees on air pollution concentrations. Though the models can always be improved, the greatest challenges are related to quantifying the secondary effects (i.e., tree effects on energy use, pollution emission and formation, and effects of tree VOC emissions on secondary pollutant formation [see below]). Tree effects on ozone concentrations are particularly challenging to quantify due to the numerous influences that trees have on this secondary pollutant. In addition, modeling could be refined to explore marginal returns of pollution removal to determine potential diminishing returns per unit tree cover with additional tree cover.

Volatile Organic Compound Emissions

Biophysical service—Trees can reduce air pollution by changing the local microclimate and directly removing pollution, but trees can also emit various chemicals that can contribute to air pollution, such as volatile organic compounds (e.g., isoprene, monoterpenes). These compounds are natural chemicals that make up essential oils, resins, and other plant products, and may be useful in attracting pollinators or repelling predators (Kramer and Kozlowski 1979). Complete oxidation of VOCs ultimately produces carbon dioxide, but carbon monoxide is an intermediate compound in this process. Oxidation of VOCs is an important component of the global carbon monoxide budget (Brasseur and Chatfield 1991).

Emissions of VOCs by trees and other sources can also contribute to the formation of ozone and secondary aerosols (e.g., Posch 2005). Because VOC emissions are temperature dependent and trees generally lower air temperatures and remove ozone, increased tree cover can lower overall VOC emissions and, consequently, ozone levels in urban areas (e.g., Cardelino and Chameides 1990, Nowak and others 2000, Taha 1996). VOC emissions from urban trees generally are < 10 percent of total VOC emissions in urban areas (Nowak 1992).

VOC emission rates vary by species (e.g., Guenther and others 1994). Seven tree genera that have the highest standardized isoprene emission rate, and therefore the greatest relative effect on increasing ozone, are: sweetgum (*Liquidambar* spp.), black gum (*Nyssa* spp.), sycamore (*Platanus* spp.), poplar (*Populus* spp.), oak (*Quercus* spp.), black locust (*Robinia* spp.), and willow (*Salix* spp.). However, due to the high degree of uncertainty in atmospheric modeling, results are currently inconclusive as to whether these genera will contribute to an overall net formation of ozone in cities (i.e., whether ozone formation from VOC emissions are greater than ozone removal or whether increasing tree cover reduces VOC emissions through temperature reduction).

Globally, average emission factors for isoprene are 12.6 mg per m² per hour for broadleaf trees and 2.0 mg per m² per hour for non-broadleaf evergreen trees (Guenther and others 2006). In the

United States, emission factors for monoterpenes are 449.2 µg per m² per hour for broadleaf trees and 872.6 µg per m² per hour for needle leaf trees (Sakulyanontvittaya and others 2008).

Economic valuation—The negative impacts of biogenic VOC emissions are often not directly associated with the emissions themselves, but rather the formation of secondary chemicals due to the VOCs emission (e.g., ozone, carbon monoxide, particulate matter). Thus, the valuation of VOC emissions is more dependent upon the impacts of the secondary chemicals that the VOCs help form (e.g., human health effects). The valuation of VOCs would likely best be done by valuing the impacts of the secondary pollutants as detailed in the air pollution removal section, but instead of the positive effect from pollution removal, the valuation would be a negative effect due to pollution formation.

Challenges in estimating the negative impacts of VOC—Due to the complexity of the atmosphere and chemical reactions, it is challenging to quantify the amount of secondary pollutants formed due to VOC emissions. To a lesser extent, quantifying total VOC emissions from a forest area is also challenging due to the number of VOC chemical species emitted. Methods for quantification of isoprene, monoterpenes, and some other VOC emissions by trees have been extensively addressed (e.g., U.S. EPA 2015, Washington State University 2015).

Pollen Emission

Biophysical service—Pollen emission is another air quality issue related to forests. While pollen plays an important ecological role, it does have the potential to negatively affect humans by causing allergic reactions (Puc 2003). While the proximity of trees to people is an important factor related to pollen allergies, various attributes of the forest influence allergic responses to pollen production. These attributes include: a) plant species composition, size, and abundance; b) allergenic potential of species (i.e., the relative potential of the pollen to cause an allergic reaction based on its shape and composition) (e.g., Ogren 2002); and c) length of pollination period (Carinanos and others 2014). Various studies have analyzed allergic responses to common tree species (e.g., Strandhede and others 1984).

Economic valuation—Valuation of the negative impacts of forest pollen production is difficult. Several factors need to be considered including pollen exposure to humans by species with varying levels of allergenicity, quantifying the impact of that exposure to human health, and then determining the economic cost of the health impacts.

Challenges in estimating the negative impacts of pollen—The first challenge in quantifying the forest's role in pollen formation is quantifying the pollen allergenicity of forest trees (e.g., Ogren 2002), then estimating the exposure of people to the pollen from the forest. Further challenges relate to quantifying the health and economic impact on the human population that is affected by the forest pollen.

Carbon Sequestration

Biophysical service—Trees, through their growth process, remove carbon dioxide from the atmosphere and sequester the carbon within their biomass. When a tree dies and the wood is allowed to decompose or is burned, most of the stored carbon goes back to the atmosphere, though some of the carbon can be retained in soils (or wood products). Thus, the net carbon storage in a given area with a given tree composition will cycle through time as the population of trees grows and declines (e.g., through aging and harvest). When forest growth (carbon accumulation) is greater than decomposition, net ecosystem carbon storage increases.

Human influences on forests (e.g., management) can further affect CO₂ source/sink dynamics of forests through such factors as fossil fuel emissions from machinery used for management and harvesting/utilization of biomass (Nowak and others 2002). Management choices such as fertilization and rotation length also affect carbon dynamics (Johnson 1992, Noormets and others 2015). For example, soils are often lacking in nitrogen or phosphorus, so the addition of these fertilizers can significantly increase tree growth and carbon sequestration (Oren and others 2001). However, fertilization during a drought period can worsen the drought impacts and significantly reduce carbon sequestration due to reduced transpiration and canopy conductance per unit leaf area, possibly due to structural and physiological changes in fine root area or hydrologic conductivity (Ward and others 2015). Prescribed burning immediately releases some carbon to the atmosphere but can also release nutrients tied up in understory vegetation that in turn make the nutrients more available to the trees and increase forest growth and carbon sequestration (Johnson and others 2014). Both air quality and climate change affect tree growth and consequently carbon sequestration by forests (Aber and others 1995, Sitch and others 2006). Longer growing seasons and increased precipitation are predicted to increase southern U.S. pine forest productivity (McNulty and others 1996). Through their influence on air temperature, trees also affect building energy use and consequently alter carbon emissions from sources such as power plants.

Above- and below-ground biomass in all forest land across the United States, which includes forest stands within urban areas, stored approximately 20.2 billion tonnes of carbon in 2008 (Heath and others 2011). Factors that influence carbon storage and sequestration include tree size, species, tree density, tree health, and tree growth rates.

Economic valuation—Current carbon valuation is typically based on the social cost of carbon as reported by the Interagency Working Group on Social Cost of Carbon or SCC (2013). Social cost associated with a pollutant (e.g., CO₂) refers to an estimate of total (global) economic damage attributable to incremental increase in the level of that particular pollutant in a given year. The current value (in 2015) is \$38 per metric ton of CO₂ based on a 3-percent discount rate. The cost of carbon emissions varies through time. The market price of carbon offset credits on

commercial and regional trading platforms has also been used to represent the value of avoided carbon emissions to landowners (Hein 2011). Using the SCC, the total value of avoided damage attributable to potential destruction of all forests could be obtained by multiplying \$38 by the CO₂ equivalent of all of the carbon stored in forests. On the other hand, if the objective is to estimate value of avoided damage attributable to a flow (of carbon to atmosphere) rather than stock, the cost could be multiplied by the annual rate of carbon sequestration.

Challenges in estimating carbon sequestration—Given the amount of forest data related to tree size, density, species composition, etc., collected by the Forest Service FIA program (USDA Forest Service 2016), the various forest carbon calculators (e.g., USDA Forest Service 2015) and carbon calculation procedures (Eve and others 2014), this service is relatively easy to calculate and value using either the SCC or the current price from an offset market.

Air Temperature Reduction

Biophysical service—Air temperature affects human health and well-being both directly and indirectly through its influences on the environment. These influences include effects on building energy use, human comfort and health, evaporative cooling, ozone production, and pollutant emissions. Urban areas tend to create heat islands that, on average, tend to be warmer than surrounding rural areas (e.g., Howard 1818, Oke 1973). Reducing air temperatures by a few degrees can have a significant economic impact through reduced energy use and improved human health.

Heat waves in cities can cause hundreds and sometimes thousands of human deaths. More than 700 deaths in Chicago were attributed to a heat wave in July 1995 (U.S. EPA 2006). Over 30,000 excess deaths were related to the heat waves in Western Europe during the summer of 2003 (Golden and others 2008). Forests reduce air temperatures mainly through transpirational cooling, shading of surfaces, and altering wind speeds. While forests can increase air temperature in winter relative to open spaces, they tend to reduce average and extreme high temperatures during the summer (Boggs and McNulty 2010, Karlsson 2000, Spurr and Barnes 1980).

Air temperature affects numerous attributes of the environment. It affects other ecosystem services such as evapotranspiration, and it also impacts biogenic emissions, anthropogenic emissions, and pollution formation. Air temperature also directly affects human comfort and human health (e.g., Harlan and others 2014, Martens 1997).

Economic valuation—Valuation of the effects of air temperature could be done by quantifying the impact of air temperatures on energy use and resulting emissions and human morbidity and mortality. Once these relationships are determined, the impacts could be valued based on energy costs, externality costs of

emissions, and the statistical value of human life. Recent studies have used multivariate regression techniques to directly link forest coverage with mortality, while controlling for other factors that impact human health (e.g., Walton and others 2016).

Challenges in estimating forest impacts on air temperature—

The relationship between forests and air temperature are well understood (Huang and others 1987, Kurn and others 1994). However, challenges still remain in modeling air temperatures and their impacts on other ecosystem services and human health and comfort.

BIOPHYSICAL QUANTIFICATION OF ECOSYSTEM SERVICES

Three of the five air quality ecosystem services can currently be assessed (air pollution removal, VOC emissions, and carbon sequestration), while others need more research and development (i.e., pollen, air temperature reduction). The following text details how these services can or could be assessed.

Air Pollution Removal and Impacts on Air Pollution Concentrations

Pollution removal by trees and forests, along with its health effects and values, has been estimated for the conterminous United States using models that incorporate U.S. hourly weather and pollution monitor data, national tree cover maps, and U.S. Census data (Nowak and others 2014). These data are incorporated within i-Tree Canopy, Design and Landscape to allow managers to roughly estimate the quantity and health value of air pollution removal by trees and forests.

In addition to pollution removal by trees, various model calculations in i-Tree are used to estimate the consequent effects on pollution concentrations and human health (e.g., Nowak and others 2013). The U.S. EPA BenMAP program was used to estimate the incidence of adverse health effects (i.e., mortality and morbidity) and associated monetary value that result from changes in NO₂, O₃, PM_{2.5}, and SO₂ concentrations due to pollution removal by trees. The model estimates assume that: a) the measured inputs (meteorology and pollution concentrations) represent the conditions in the region around the monitors; b) the boundary layer is well mixed; c) input variables are correctly measured; d) the dry deposition model in i-Tree accurately portrays pollution removal (Hirabayashi and others 2011, Morani and others 2014); and e) BenMAP accurately estimates and values health effects due to changes in pollution concentrations.

Under most situations, atmospheric acid deposition of NO_x and SO_x are below levels that cause reductions in forest health. However, a combination of high acidic deposition and/or low soil acid buffering capacity can result in forest soils exceeding the critical acid load level in places. At this point, the forest soil is termed to be “in exceedance” of the critical acid load. The trees

growing on soils that are in exceedance are much more likely to experience reductions in growth and increased risk of early mortality. Additionally, excess acids can leach into streams and thereby reduce water pH and aquatic biodiversity. Simple mass balance equations are typically used to estimate critical acid loading and to identify forest areas that are in exceedance of their critical acid load. For example, McNulty and others (2007) applied this approach to the conterminous United States at a 1-km resolution and found that the region with the largest proportion of forest area in exceedance of the soils critical acid load was the Northeast. Forest harvesting and base cation fertilization (e.g., the use of calcium carbonate lime) are two methods used to either remove acids from the ecosystem or neutralize acidic soil. Through a combination of forest growth and harvest, trees can be used to remove excess acid from forest soil, which will improve both terrestrial and aquatic ecosystem health.

Volatile Organic Compound Emissions

VOC emissions from forests can be quantified using either a) the Biogenic Emissions Inventory System (BEIS) model methodology (U.S. EPA 2015), which is also incorporated into the i-Tree model (www.itreetools.org), or b) the MEGAN Model - Model of Emissions of Gases and Aerosols from Nature (Washington State University 2015). Both models estimate VOC emissions from trees and forests, making various assumptions and using various structural and meteorological inputs. While assumptions can always be questioned, these models are likely the best available options for estimating plant VOC emissions. Neither model estimates the secondary impacts (e.g., ozone formation) or the value of changing VOC concentrations. More research is needed on these topics.

Pollen Emissions

There are databases on pollen allergenicity of numerous tree species and methods to estimate pollen allergenicity index for areas with tree cover. For example, the i-Tree model has a prototype allergenicity index that could be used to rate allergenicity of forests. This module has not been released yet and is similar to the index used in Carinanos and others (2014). However, these index values have not yet been linked to impacts on human health (e.g., allergies). More research is also needed on this topic.

The main assumptions of the pollen index approach are that the allergenicity ratings of tree species are correct and that pollen emissions are related to plant crown/leaf volume or biomass.

Carbon Sequestration

Carbon storage and sequestration by forests can be estimated using Forest Service forest carbon calculators (e.g., USDA Forest Service 2015), urban forest calculators (e.g., i-Tree), or carbon calculation procedures detailed in Eve and others (2014).

There are various carbon equations used to estimate carbon storage by trees, and it is assumed that these equations accurately estimate carbon based on the measured structural data (e.g., species, diameter, height). Sequestration rates are based on estimates of annual tree growth. Although there are limitations, the procedures used to estimate carbon stocks and flows are pretty well established and accepted.

Air Temperature Reduction

Satellites can be used to measure surface temperatures (e.g., NASA 2015, Sobrino and others 2004). For example, land surface temperature maps can be downloaded at neo.sci.gsfc.nasa.gov/view.php?datasetId=MOD11C1_M_LSTDA&date=2015-01-01. Various models can also be used to estimate air temperatures (e.g., Chen and others 1993, Yang and others 2013). However, there are limited procedures or models showing how forests and others surfaces affect air temperatures.

Use of land surface temperatures is based on a reasonable assumption that these temperatures are related to air temperatures. However, while surface temperatures influence local air temperatures, they can be quite different from air temperatures. Models used to estimate air temperatures make various assumptions about the atmosphere and earth surfaces to produce reasonable estimates of hourly air temperatures and temperature variations, but more research is needed on this topic.

ECONOMIC VALUATION OF ECOSYSTEM SERVICES

Once the flow or provision of the good or service (e.g., carbon, air pollution removal) is quantified, various methods of market as well as non-market valuation can be applied to characterize their value. Since none of these services (perhaps except for carbon sequestration) are currently traded in the market, non-market valuation methods are used to estimate their values. Methods of non-market valuation can be pecuniary or non-pecuniary. We focus on pecuniary methods that obtain the money equivalent value of a service. There are also non-pecuniary methods for characterizing value in other than monetary terms, such as number of human lives saved, number of clear days observed etc.

This section introduces market and non-market valuation methods that are potentially applicable in valuing one or more air quality services (table 4.1), provides guidance on choosing the most appropriate method, and discusses important considerations in benefit aggregation.

Market Price Method

This method relies on the prevailing price for the service as traded in the local market. Since many of the air quality services are not currently traded in the market, this method is not applicable except in the case of carbon sequestration. Carbon offset credits are currently traded in national and international cap-and-trade institutions and other “over-the-counter” voluntary

transactions. Market prices are typically less than the SCC, because the transactions are motivated by the expected benefits and costs to individual sellers and buyers, reflecting both their preferences and the regulatory framework, and hence do not necessarily take account of all of the social benefits associated with sequestration or reduced emissions. Price of emission permits for other kinds of air pollutants (e.g., SO₂) represent the value of abatement service that forests provide, provided we can attribute such service to forests and assuming that the current policy regime will continue. However, the market prices of such services could be distorted by lack of information and failure to capture external benefits of production. Moreover, the price of such a permit or offset credit is strongly influenced by the regulatory environment (i.e., whether the emitters are subject to mandatory regulation to offset their emission).

Contingent Valuation Survey Method

This method relies on asking people to state their willingness to pay (WTP) to consume or enjoy the benefit of a given air quality service by forests. Once the service provided by forests to improve the air quality is defined and quantified, a randomly selected sample of potential beneficiaries is asked to express their WTP to maintain a specified area of forest for the purpose of providing the service. The most common approach is to split the sample and ask different groups whether they would be willing to pay a specified amount that varies across groups. Typically, the WTP estimated at household level is extrapolated to an entire population to get the total value of the service attributable to a forest area. Dividing the total estimate by total hectares of the specified forest could yield value on a per-hectare basis, which provides an approximate measure to evaluate the marginal impact of forest area change. A similar approach can be taken by surveying how much forest landowners are willing to accept (WTA) in compensation for delaying or forgoing their harvest in favor of air pollution removal or carbon storage. Such an agreement would require landowners to forgo all other economic interests for a specific period of time, and their stated WTA to commit to this agreement is considered a proxy of the production cost of air quality. A few studies have applied the contingent valuation method to estimate landowners' WTA and buyers' WTP for carbon services (Poudyal and others 2012, Tsang and Burge 2011).

This method assumes that people will do what they say, and it is sensitive to how the WTP question is designed and how non-responses (or refusals) are factored into benefit estimation. The questionnaire needs to be very specific in attributing the benefit to the forest of interest. It is worth noting that Zhang and Li (2005) have argued that the value of a good or service can be more accurately approximated by deriving a shadow price from the opportunity costs of the resources involved, rather than eliciting producers' WTA. However, caution is required here because the full value, as captured by WTA, may be greater than the costs of acquisition. Despite the limitations, when there are no prior

Table 4.1—Summary of applicable valuation methods and example application by forest ecosystem service types

Method	Ecosystem service	Applicability ^a	Example application (where available)
Market price ^b	Air pollution removal		
	VOC removal		
	Pollen emission	✓	
	Carbon sequestration	✓	Jerath and others (2012), Hein (2011)
	Air temperature	✓	Pandit and Laband (2010)
Value of statistical life saved	Air pollution removal	✓	Levinson (2012)
	VOC removal	✓	
	Pollen emission	✓	
	Carbon sequestration	✓	
	Air temperature	✓	
Contingent valuation	Air pollution removal	✓	
	VOC removal	✓	
	Pollen emission		
	Carbon sequestration	✓	Carlsson and others (2010), Jerath and others (2012)
	Air temperature	✓	Carlsson and others (2010)
Replacement or substitute cost method	Air pollution removal	✓	
	VOC removal		
	Pollen emission		
	Carbon sequestration	✓	Platinga and Miller (1999), Richards and others (1993)
	Air temperature		
Hedonic method	Air pollution removal	✓	Luechinger (2009)
	VOC removal		
	Pollen emission		
	Carbon sequestration		
	Air temperature	✓	Pandit and Laband (2010)
Damage cost avoided ^b	Air pollution removal	✓	Hein (2011)
	VOC removal	✓	
	Pollen emission	✓	
	Carbon sequestration	✓	Interagency Working Group on Social Cost of Carbon (2013), Jerath and others (2012)
	Air temperature		
Life satisfaction or happiness method	Air pollution removal	✓	Levinson (2012), Welsch (2009), Luechinger (2009)
	VOC removal		
	Pollen emission	✓	
	Carbon sequestration	✓	
	Air temperature	✓	Rehdanz and Maddison (2005), Levinson (2012)
Benefit transfer	Air pollution removal	✓	
	VOC removal	✓	
	Pollen emission	✓	
	Carbon sequestration	✓	Moore and others (2013)
	Air temperature	✓	

^a Applicable to ecosystem service if checked.^b Interrelated, when the damage involves market goods/services.

studies of the value of a good and no related market behaviors that indirectly reveal the value, contingent valuation may be the only way to obtain an estimate of the value of the service.

Replacement or Substitute Cost Method

This cost-based method relies on the assumption that the value of a given ecosystem service provided by forests is equivalent to the cost of providing such a service by some alternative or artificial means. For example, the value of forest carbon sequestration, as typically measured in per metric ton of CO₂ sequestered, would be estimated as the cost of offsetting equivalent CO₂ from forest projects such as afforestation (Platinga and Miller 1999) or non-forest projects such as methane capture, agriculture soil carbon, or renewable energy. Similarly, various kinds of indoor ventilation and air purifying or conditioning systems have been designed to regulate VOCs in air, and the cost of acquisition and operation of such devices could offer a surrogate “price” for estimating the value of services like VOC removal by forests.

This method is reliable only if: (1) the service provided by forest is the same (in nature and quality) as that provided by an alternative project; (2) the alternative is the next least-cost means for providing the service; and (3) the benefits of the service provided by the alternative project exceed the cost of the alternative project (Brown 2017). While air quality and carbon-related services provided by forests and alternative means are likely to be similar in nature and quality, their level of provision and therefore their costs are likely to be determined by the regulatory framework, meaning that the benefits do not necessarily exceed the costs. While these are significant limitations on use of this method to estimate values, it can be used to quantify the cost-effectiveness of forests as a way to meet externally set standards such as national ambient air quality standards.

Hedonic Method

Even though no markets exist to observe financial transactions involving forest ecosystem services related to air quality, their benefit can be revealed by examining whether and how the change in supply of such a service is compounded in a related good for which market prices are available. The hedonic method is particularly applicable to valuing air pollution removal, due to the local nature of the benefits. In the case of air quality or volatile organic compounds, the price (or rent) of a residential house is a market value that relates directly to the air quality in the neighborhood. By regressing prices of properties against their structural features and some measure of air quality (e.g., air pollutant concentration), one could estimate the implicit value of a marginal unit reduction in a pollutant. The implicit marginal willingness to pay (WTP) for air quality improvement can be obtained from the regression. The value of this service can be obtained on a per-hectare basis by multiplying the resulting WTP by the rate of pollution removal (amount removed by a hectare of forest).

Damage Cost Avoided Method

When a damage can be avoided with a given ecosystem service in place, then the avoided cost of the damage is one value of that ecosystem service. One measure of the damage cost is the amount that society is willing to pay to avoid the damage, and thus insurance premiums are sometimes used to estimate the cost. In other words, if society incurs a cost to avoid a damage, then the worth of an ecosystem service that prevents that damage is at least as much as the cost incurred.

The SCC is a type of damage cost avoided, as it involves projecting the physical damage (including change in agricultural productivity, human health, property damage, etc.) from CO₂, and then estimating the monetary value of that damage (Pizer and others 2014). Avoided damage due to the sequestration (capture and long-term storage) of carbon by a forest can be interpreted as the value of carbon service for a given forest stand. The global SCC is estimated at \$38 per metric ton of CO₂ based on a 3-percent discount rate (Interagency Working Group on Social Cost of Carbon 2013).

This method could also be applied to estimate the value of the pollen emission disservice. If one knows how the amount of pollen emitted per forest area relates to the number of allergy-related illnesses, then the total public expenditure on allergy medications (i.e., cost of illness) could be used to obtain a lower bound on the value of the disservice (assuming that the total cost, or utility lost, due to allergies is greater than the expenditures).

Value of Statistical Life Saved Method

This method can be considered similar to the method of damage cost avoided with particular relevance in human health. It has been widely applied in risk and health economics studies (Viscusi and Aldy 2003). Forest ecosystem services like reduction in air pollution can be linked with positive health outcomes among the human population. The assumption here is that people are WTP for the marginal reduction in mortality attributable to a change in forest cover or condition. The benefit of a marginal reduction in mortality is the monetary equivalent of the reduction of mortality risk from one to none. For example, if the removal of x metric tons of air pollution as a result of expansion of y hectares of forest reduces the mortality risk by 1 in 1 million for 2 million people, the air pollution control service provided by y hectares of forest is 2 avoided deaths. Multiplying this value by the statistical value of a life saved yields the total value of the service attributable to y hectares of forest. The statistical value of life has been estimated by contingent valuation and by hedonic wage models, in which a regression analysis quantifies the tradeoffs people are willing to make between fatality risks and wage rates. Federal agencies like the U.S. EPA and U.S. Department of Transportation use \$7.4 million (mean estimate based on several wage-risk studies and contingent valuation survey) (U.S. EPA 2010) and \$9.1 million (based on wage-risk study), respectively, (Revez 2014) as the monetary value of avoiding the death of a single person.

Life Satisfaction or Happiness Method

The marginal effect of air quality on self-reported life satisfaction or happiness can be analyzed to reveal the marginal value of air quality, if the income equivalent of changes in life satisfaction or happiness can also be estimated. If one could quantify the change (better or worse) in carbon storage or air quality attributable to change in forest area or condition, this change could be linked with some measure of life satisfaction as reported by the beneficiaries. The basic assumption of this method is that measures of life satisfaction are reliable proxies of people's underlying utility, and the value of an ecosystem service such as air pollution removal can be estimated by evaluating how people tradeoff between the service and income. For example, if a 25-percent increase in carbon sequestration or air pollution removal (as a result of expansion of forest area or a similar increase in growing stock of an existing forest) increases the life satisfaction measure by 1 unit, and it takes a \$10,000 increase in annual household income to achieve a comparable increase in people's life satisfaction, then this method would conclude that the total value of the service provided by the increased forest area or improved forest condition to beneficiaries would be \$10,000 multiplied by the number of beneficiaries. By estimating a production function for the service, its value can easily be estimated on a per-hectare basis. Luechinger (2009) and Levinson (2012) have already applied this method to value reduction in air pollution and for changes in air temperature.

Benefit Transfer Method

When there is a budget or time constraint on collecting new data, benefit transfer is typically used to quantify the value of an ecosystem service. In a unit value benefit transfer, the value of a specific service (e.g., carbon sequestration per hectare of given forest type) in a given "policy site" is derived from an average of such values reported in one or more study areas of similar characteristics elsewhere (study sites) (Loomis 1992; also discussed in chapters 2 and 3 of this publication). Values reported in study sites would be estimated by one of the methods described above. These would be per-hectare value of a given ecosystem service provided by a given type of forest. The analyst would then 'transfer' them to the policy site by multiplying the average per-hectare value by the total area occupied by that type of forest. Many of the recent studies that characterized the value of forest ecosystem services in Southern States have used this approach (e.g., Moore and others 2013, Simpson and others 2013). A key consideration is that the ecological and socioeconomic characteristics of the "study site" and "policy site" should be as similar as possible because species composition (hardwood, softwood), location of forest relative to source of pollution, and beneficiaries can make a substantial difference in the level of supply and public value of the service. Several reports including Troy and others (2006), Liu and others (2010), and Costanza and others (2014) provide per-area values of air quality related to ecosystem services that may be applicable to regions of similar characteristics.

Aggregation

Aggregation involves extrapolating the value of each ecosystem service across the entire geographic region of interest (e.g., State) or the beneficiary population (e.g., State population). If the value is estimated per hectare, simply multiplying the per-hectare value by the total area of forest would yield a statewide total value of that particular service. Services such as air temperature reduction in urban areas are often estimated on a per-household basis rather than per hectare of forest. In this case, the per-household value of a service could be extrapolated to the total number of households that benefit from the service to get a statewide estimate. While this approach to aggregation is easy to implement, it is subject to a number of caveats.

First, the marginal value of a forest ecosystem service benefit can vary (impact of the loss of first hectare vs. the last hectare of forest). Unfortunately, there is no general guidance on how to incorporate nonlinearity of marginal benefits in valuation. Second, since neither the supply (nature of forest) nor the demand (preference of beneficiaries) of ecosystem services are homogenous across entire States, it is important to stratify the State into relatively homogenous regions and to use the most appropriate or adjusted value for each region (e.g., Moore and others 2013) before summing up regional estimates to get the statewide total value. Again, it is difficult to provide guidance on exactly how to stratify the State, i.e., how far it must be disaggregated.

Finally, the statewide total value of all services can be obtained by a sum of the statewide value of each type of service. Considering the interrelatedness of production process and consumption, it is important to avoid double counting and hence overestimating the total benefits. Only services that are mutually exclusive should be added. Likewise, it is important to account for tradeoffs between services, and at the extreme, to avoid summing services that are mutually exclusive (e.g., wilderness values and timber production).

IMPROVING FUTURE VALUATION

There is interdependence between forest structure and ecosystem services and values. Valuation is dependent upon accurate estimates of the magnitude of the service provided; service estimates are dependent upon accurate estimates of forest structure and how structure affects services. The key starting point to valuing services provided by forests is quality data on forest structure. Services and values cannot be adequately valued without accurate forest data. Combining accurate forest data with valid procedures for quantifying ecosystem services will lead to reliable estimates of the magnitude of ecosystem services provided by the forest. Finally, with accurate estimates of forest ecosystem services, values of the services can be estimated using valid economic estimates and procedures. Thus, three critical elements in sequence are needed to value forest ecosystem

services: structure – services – values. Errors with precursor elements will lead to errors in subsequent estimates (e.g., errors in characterization of forest structure will lead to errors in estimating services and valuation). All estimates and means of estimation can be improved to varying degrees.

Quantifying Forest Structure

Mapping of tree cover from NLCD 30-m resolution data can be improved by creating or utilizing high resolution tree cover data. These tree cover data are being created in some areas at the city or county scale (e.g., www.nrs.fs.fed.us/urban/utc/status/). Another option for calculating tree cover is to use photo-interpretation. With photo-interpretation, one can often get tree cover estimates with a standard error of around 1.4 percent with 1,000 interpreted points (about one day's worth of work). This accuracy is often better than that produced by cover maps, but photo-interpretation does not map specific locations of tree cover. There is a free photo-interpretation tool (i-Tree Canopy) that provides users an easy means to interpret and analyze tree and other cover types using Google maps (www.itreetools.org/canopy/index.php).

For estimates of local forest structure (e.g., number of trees, species composition), FIA collects plot data for forest areas at a density of one plot every 2,428 hectares. For small study areas (e.g., where sample size does not meet a minimum threshold), there are likely not enough plots to make a reasonable estimate of forest structure. In this case there are two options:

- 1) Collect more local data. This can be done using FIA procedures with plot intensification or by collecting your own data (e.g., using i-Tree Eco),
- 2) Use FIA plot data from the region to determine the forest structural attributes (e.g., number of trees) per unit of tree cover within the FIA sampled area, and then apply these standard values to the amount of tree cover in the study area. Plot data from FIA have been imputed to non-plot areas (e.g., Wilson and others 2013). This approach is less accurate, but it applies regional average values to local tree cover and could illustrate regional differences in forest structure and functions.

Air Pollution Removal and Its Effect on Air Pollution Concentrations

Forest effects on air pollution concentration have been estimated using i-Tree Landscape for urban and rural areas in counties of the conterminous United States, with valuation of those effects based on health impacts and values from BenMAP. However, health values only occur when people live in close enough proximity to forests to receive the improved air quality. As there are fewer people in rural areas, the health value of the air quality improvement decreases substantially from urban to rural areas.

As air quality affects more than human health (e.g., visibility, plant health, damage to materials), another approach to valuation has been to use “externality values” calculated for each air pollutant (e.g., Van Essen and others 2011). An externality value is a constant value per ton of pollution removed (\$ per t) derived from the literature. Externality values should be higher than health values as they should include health and other values. For example, pollution removal by trees in the conterminous United States in 2010 had a health value of \$6.8 billion but had an externality value estimate of \$86 billion (Nowak and others 2014).

While methods for modeling pollution removal by trees and forests are well developed, further research is needed on secondary pollutants that are formed through chemical reactions (e.g., ozone), including understanding the role of VOC emissions. For all pollutants, secondary effects of trees on pollutant emissions (e.g., via altering air temperatures and energy use) need to be quantified to provide a more comprehensive analysis of forest impacts on air quality.

Volatile Organic Compound Emissions

Though improvement can always be made, the BEIS and MEGAN procedures are likely the best available procedures to estimate plant VOC emissions. If VOC emissions can be converted to tons or concentrations of particulate matter, ozone, and carbon monoxide, then valuations can be done similar to pollution removal.

Pollen Emissions

Compared to other air quality (dis)services, there is relatively less certainty about pollen emissions. Current methods only produce a relative index value of pollen for forests but do not estimate actual pollen emission by species. Developing emission rates for each species by time of year would be helpful in improving pollen estimates. To model human health impacts, better understanding is needed of how species emissions affect pollen concentrations and their specific impact on allergic responses in humans. Given current knowledge, relating the relative index values to health impacts (e.g., increased allergies) and then health values associated with these effects could be used to value the impact of pollen emissions.

Carbon Sequestration

Improvement can always be made by developing more and/or better carbon estimation equations and developing better estimates of tree growth. However, the procedures used are pretty much the standard in estimating carbon storage and sequestration from trees and forests. The current (2015) value of forest carbon sequestration is \$38 per metric ton of CO₂ based on a 3-percent discount rate (Interagency Working Group on Social Cost of Carbon 2013).

Air Temperature Reduction

As air temperatures affect many attributes within forest ecosystems (e.g., transpiration, growth, photosynthesis, VOC emissions) and have secondary effects that impact air quality (e.g., photochemical rates, pollutant emissions), this service requires more research to improve model estimates of forest effects on air temperatures and associated secondary effects on other ecosystem services and air quality.

Changes in air temperature affect human health directly (e.g., heat stress) and also affect chemical emissions from numerous sources, which can influence human mortality. If these impacts can be quantified, then they can be valued either using the valuation procedures related to air pollution removal or using the statistical value of a life.

Economic Valuation

Most of the valuation methods presented in this chapter are established methods and have been widely used by resource economists. A bigger challenge in ecosystem service valuation lies in accurately quantifying the benefits. However, a number of issues still exist in modeling (users' preference, demand) and benefit aggregation that lead to potential over- or underestimation of benefits. Nonetheless, more rigorous methods are evolving with advancement in the field of econometrics. On the other hand, valuation studies so far have relied on cross-sectional data (market or household survey data of one particular point in time), but parameters estimated from such models may not be stable enough to be useful in forecasting and benefit transfer purposes. When the benefit transfer method becomes the only choice in valuation, researchers should attempt to get a mean estimate of benefits (\$/hectare) from as many studies as possible.

Improvements in the valuation approach could also be made by integrating economic methods presented here with other disciplines such as regional science and public health. For example, there currently is no single method accepted as standard practice in estimating the value of heat reduction services provided by forests. However, the per-person value of human life has been estimated by several studies worldwide (Viscusi and Aldy 2003), whereas the marginal benefit of increasing forest area in reducing heat-related deaths is estimated in regional science studies (e.g., Walton and others 2016). Taken together, these two estimates (value per life and number of lives saved) can be combined to characterize the total value attributable to heat reduction services. Considering the dynamics of ecosystem service production functions and the complexity of data collection and modeling, efficient approaches in ecosystem service valuation in the future may have to utilize existing information that are valid, reliable, and reasonably accurate for the study area.

SUMMARY AND CONCLUSIONS

Forest management and land use decisions in the past have rarely acknowledged the value of ecosystem services. This omission is largely because our society has taken those services for granted. These services could be undersupplied, especially in the Southeast, due to a lack of proper incentives to private landowners. This chapter reviews published and theoretically defensible methods and techniques to quantify and value several air quality services of forest ecosystems.

With the market for ecosystem services slowly emerging, it is becoming increasingly possible to estimate the value of some services (e.g., carbon) based on market price of offset credits. On the other hand, most other air quality services are currently not traded in the marketplace and require non-market valuation methods. Part of the difficulty in monetizing some services is quantifying the magnitude or impact of forests on the services. Hence, more research is needed to better understand the production function of those services, as well as the flow of benefits to natural and human systems. There are some user-friendly, computer-based tools (e.g., i-Tree) that are at the disposal of practitioners to explore and understand the production and value of some of the services discussed in this chapter.

Despite the available methods and tools, quantification and valuation of many specific services are still challenging because of the secondary effects (both in cost and benefit) of some of the services. In other words, the interrelation in production process of multiple ecosystem services complicates the process of characterizing value and aggregating benefits at the forest landscape level. For example, management practices that increase carbon sequestration will likely impact water flow and quality. Thus, when aggregating the values of these services, the analyst needs to avoid double counting and hence overestimating the total economic value.

While the valuation methods discussed here are standard, the procedure to be followed in the quantification of services and aggregation of benefits could be specific to the context or objective of valuation. For example, if the objective is to estimate the value of change in a provision of services as a result of proposed policy (e.g., increasing forest area or improving forest health), then appropriate caution should be exercised to estimate the level of service between two States (say, State A and B) and to multiply that by the value of benefit-per-unit evaluated for the respective State where possible. The difference of products can be interpreted as the public value of the policy.

The methods presented in this chapter do not constitute an exclusive list of prescribed methods for quantification and valuation of services but are intended to serve as guidelines to aid in quantifying forest and environmental resource values. Reliability and validity of many of these valuation methods are

still being debated. Moreover, as the methodological research in ecosystem service continues to grow, several other approaches and tools are likely to be tested and proposed in the future.

Each method has its own strengths and limitations and relies on a unique set of assumptions regarding the ecological processes that produce a service, as well as economic principles of utility and welfare. While “benefit transfer” from the existing literature is a lower cost option, it is only reliable across areas with identical characteristics. In other words, a benefit function developed in a study area should only be applied to new areas that fall within the range of characteristics of the study area. Hence, the need for a new valuation study depends on the uniqueness of the ecological and socio-cultural context of the area of interest. Valuation methods for these unique cases can be improved through further integrated and site-specific research that involves both ecologists and economists.

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Chapter 5

Forest Ecosystem Services: Provisioning of Non-Timber Forest Products

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INTRODUCTION AND BACKGROUND

The purpose of this chapter is to describe approaches to calculate a conservative and defensible estimate of the marginal value of forests for non-timber forest products (NTFPs). “Provisioning” is one of four categories of benefits, or services, that ecosystems provide to humans and was described by the Millennium Ecosystem Assessment (MEA 2005) as “products obtained from ecosystems,” which include food, fuel, wood, and fiber, as well as products such as remedies and crafts. Clearly, any valuation of ecosystem benefits must include the tangible items, or products, that they supply.

There are very few valuation studies of NTFP provisioning. The main reason is that NTFPs—which are also called “special forest products” in the United States—are typically secondary in importance to timber, so they lack the profile and attention wood products receive. In the United States, forest valuation is almost always based on timber production and a few other items such as grazing, minerals, or hunting leases. Timber price and quantity data have been tracked for many years and are easily available. In contrast, data are available for only a few NTFPs even though there is a wide variety of products and markets. Quality data related to geographic and temporal distribution of NTFP species and their harvests are quite limited.

Most studies of NTFPs report methodological roadblocks in getting reliable and accurate values. The basic information needed to estimate NTFP values begins with some sort of accounting of the flow or stock of the resource. Typically, values are estimated based on prices received by harvesters, although the contributions that NTFPs make to harvesters’ income, diet, and health also represent values to households and communities. Non-market valuation based on these types of contributions, while not easy, has been used in the absence of market prices.

Ecosystem service valuation studies should use past research as a starting point. While a comprehensive literature review is beyond the scope of this chapter, appendix 5.1 summarizes select valuation studies of NTFPs in the United States and around the world. Appendix 5.1 presents studies that may be relevant for an analyst attempting to value NTFP provisioning services.

Definitions

NTFPs come from plants (e.g., herbs, lichen, moss, shrubs, trees, vines) and fungi (e.g., mushrooms). They include food, medicine, decorations, and materials used in fine arts and crafts. Table 5.1 lists common NTFPs from the U.S. South. These products often are collected and consumed for personal and cultural uses (Cordell and others 2012, Robbins and others 2008) and provide significant economic contributions to users in the United States (Alexander and others 2001, Alexander and others 2011).

As the term NTFP specifies, these products are not timber and come from forests. This leads to questions about the definitions of “timber” and “forest.” This publication considers industrial wood chips and poles to be timber products, while wood extracted for traditional firewood and artisanal crafts are considered NTFPs. Definitions and quantification methods focus on goods produced from natural forests (e.g., Adepoju and Salau 2007, Famuyide and others 2013, Godoy and Lubowski 1992, Sullivan 2002), but they could be used for products grown in plantations solely for their non-timber value. Often, products grown in tree plantations or cultivated on an industrial scale, such as fruits, nuts, or Christmas trees, are not considered NTFPs, yet the valuation methods presented are applicable to these products.

Non-biotic products that may come from the forest, such as rocks and minerals, are not classified as NTFPs. In some parts of the world, animal products such as game meat are considered NTFPs, but in the United States, wildlife is generally treated as a separate category. We do not deal with animals or animal products here, restricting NTFPs to plants and fungi, although many of the same principles would apply. For more information on the valuation of wildlife for recreation such as hunting, see chapter 2.

Many past studies on NTFPs report “total market value” or “economic value.” Total market value is the total quantity harvested or traded in a locality, State, region, or country, multiplied by its market price. Economic value refers to the sum of the consumer and producer surplus for a good or service. While such estimates may provide information about the overall importance of NTFPs in a particular economy or policy changes that affect the entire source of a product, they are not described

Table 5.1—Major non-timber forest products of the Southern United States

Product	Part of plant	Market segment	Product	Part of plant	Market segment
Pine straw	Needles	Landscaping	Virginia snakeroot	Root	Medicinal
Pine cones	Cones	Decorative	Wild yam	Root	Medicinal
Conifer boughs	Boughs	Decorative	Black cherry	Bark	Medicinal
Sweet grass	Stem	Art	Trillium	Whole Plant	Medicinal
Sphagnum moss	Whole Plant	Art	Autumn olive	Fruit	Culinary
Mistletoe	Whole Plant	Art	Pawpaw	Fruit	Culinary
Vines	Stem	Decorative	Black walnut	Fruit	Culinary
Maple syrup	Sap	Culinary	Tulip poplar	Bark	Decorative ^a
Ramps	Whole Plant	Culinary	Persimmon	Fruit	Culinary
Fox grapes	Fruit	Culinary	Butternut	Bark	Medicinal
Galax	Leaves	Decorative	Eastern red cedar	Leaves	Medicinal
Moss	Whole Plant	Decorative	Witch hazel	Bark & Leaves	Medicinal
Grape vine	Vine	Art	White oak	Bark	Medicinal
Dutchman's pipe	Vine	Art	Sassafras	Leaves	Culinary
Mountain laurel	Whole Plant	Landscaping	Morels	Fruiting Body	Culinary
Winterberry holly	Leaves and Berries	Decorative	Other mushrooms	Fruiting Body	Culinary
Pitcher plants	Whole Plant	Decorative	Princess pine	Whole Plant	Decorative
Aletris	Root	Medicinal	Spanish moss	Whole Plant	Decorative
Bethroot	Root	Medicinal	Staggerbush/ Crooked-wood	Branch & Stems	Decorative
Black cohosh	Root	Medicinal	Blueberries/ huckleberries	Fruit	Culinary
Bloodroot	Root	Medicinal	Blackberries	Fruit	Culinary
Blue cohosh	Root	Medicinal	Fraser fir transplants	Whole Plant	Landscaping, Decorative ^b
False unicorn	Root	Medicinal	Rhododendron/ azalea transplants	Whole Plant	Landscaping
American ginseng	Root	Medicinal	Cypress knees	Stems	Art
Goldenseal	Leaves & Root	Medicinal	Wood for carvings	Wood	Art
Lady's slipper orchid	Root & Whole Plant	Medicinal	Wood for baskets	Wood	Art
Saw palmetto	Fruit	Medicinal			
Slippery elm	Bark	Medicinal			

^aTulip poplar bark is used for siding for houses.^bFraser fir is used extensively for Christmas trees.

in this chapter. Rather than total values over a geographic region, this chapter explains how to find the best estimate of *per-acre* values resulting from small changes in forest area, or marginal value. This type of information is the most relevant for policy decisions that affect a limited forested area, as well as for tracking changes in the value of provisioning services over time.

Market participants who move NTFPs from forest to the first point of sale are generally called “harvesters” (alternatively, “collectors,” “pickers,” “gatherers”). They are analogous to “loggers” in the timber industry. Harvesters can be an important source of information about NTFP activities. They represent the primary knowledge base to locate the resource, provide extraction rates, and identify what the market wants and what prices can be obtained for different qualities of products. They

observe changes in the forests that they know are critical to sustaining their livelihoods as well as the continued existence of the NTFP resource. Harvesters are the foundational players that initially define the market structure. They gather the products and consume, share, or sell the products as raw materials.

“Dealers” can play a key role in setting the prices and value received by harvesters of NTFPs. Dealers can be sub-classified into “primary buyers” who purchase directly from harvesters, consolidate transactions, and sell to regional or national buyers. “Secondary buyers” may consolidate transactions and sell to national or international buyers, or they may produce a final product for retail consumers. “Third buyers” in the market would then produce a final product for the retail consumer. The “final buyer” in the market chain typically is the retail consumer.

Greenfield and Davis (2003) found dealers of medicinal forest products (e.g., American ginseng), floral products (e.g., galax) and edible forest products (e.g., ramps) to operate full time, seasonally, and part time depending on the product. Some prices are reported daily by the dealers (ginseng) while others are set seasonally by buyers (e.g., galax and bloodroot). Blatner and Alexander (1998) and Schlosser and Blatner (1995) found that data on industries engaged in the marketing, processing, and distribution of NTFPs in the Pacific Northwest were limited due to the small, informal characteristics of NTFP markets and high price variability.

NTFPs can be produced by a continuum of forest management intensities. Those for which little or no management occurs are “wild-harvested” from natural populations. “Managing wild populations” does not mean establishing a species in a forest, but it does mean undertaking activities that favor a species or individuals over others to increase yield or quality of product. Intentionally establishing and maintaining NTFP species within a forest is called “forest farming.” Forest farming can take numerous approaches, including “wild-simulated,” a low-input method of establishing NTFPs with only minimal alterations and management, and “woods-cultivated” or “woods-grown,” which refers to more intensively established and managed systems.

Conceptual Model

The value of a provisioning ecosystem service is not the same as the value of a product. To see this, consider that all NTFPs are the result of a combination of production inputs that include at least ecosystem functions and human effort. The value of the provisioning ecosystem service is most clearly understood to be that portion of the products value that is due to ecosystem function, that is, the “residual value” after factoring out human and other production inputs. We remind the reader that the measure of value we demonstrate here is the *marginal* value. Where competitive markets exist, market price is the best starting point for determining marginal values.

Smith and others (2010) describe an appraisal system for estimating the residual value of NTFPs, based on wholesale prices and total harvest cost, which is used by some National Forests. An NTFP cost analysis program was initiated by Region 6 (Pacific Northwest) of the National Forest System in 2000 in response to national legislation that required the National Forests to charge “fair market value” for harvest permits (Smith and others 2010). The tool provides an Excel spreadsheet model that uses a residual-value appraisal of permitted products (Smith and others 2010, USDA Forest Service 2016). The residual value appraisal uses data from interviews with harvesters on delivered wholesale market prices for products and all associated harvest costs to achieve fair and unbiased base rates or a minimum fee for permits. While this tool only includes information related to products common in the Pacific Northwest, a similar approach, as described in this and following sections, could be used to find residual values in other parts of the country.

Timber stumpage—the price per acre of standing trees before harvest—roughly corresponds to a provisioning ecosystem service marginal value (Alexander and others 2002a). Sometimes, harvesters may pay landowners per acre to harvest an NTFP on their property, in the same way loggers pay stumpage fees, or hunters pay per-acre lease fees, to landowners. If the landowner has not invested any time or resources into the production of that NTFP, then this per-acre price is the same as the ecosystem service marginal value: it is stated on a per-acre basis and already has other production inputs factored out. Cases where rights to harvest NTFPs are paid on a per-acre basis are relatively rare, but this system is used occasionally for a few products such as pine straw and saw palmetto. Forest managers of public and private lands have many options in deciding how to allocate harvest rights for non-timber products (Alexander and Fight 2003).

Cases where no such per-acre transaction occurs and the observed price is per unit of harvested product are more common. In such cases, the analyst must factor out of the price all human inputs including harvest, establishment, and management costs (Godoy and others 1993) to obtain an estimate of value for the ecosystem function portion of the production inputs. This approach is similar to what economists define as net present value (NPV). Equation (1) presents the basic NPV formula (see Alexander and others 2002a):

$$NPV = \sum_{i=1}^t \frac{P_i \cdot MQ_i - (EC_i + MC_i + HC_i)}{(1 + \rho)^i} \quad (1)$$

where:

NPV = the per-acre net present value over the “rotation” period t , on forest land of a given forest type and ownership class

t = the time period of the NTFP production “rotation” (time from establishment to final harvest)

ρ = the discount rate

MQ_i = the quantity of NTFP produced in year i on a marginal acre of forest land of a given forest type and ownership class (see “Valuation of Marginal Changes in NTFP Provisioning Services” later in this chapter)

P_i = the market price or other estimated shadow price per unit of NTFP at the first point of sale in year i (see “Estimation of Production Costs” later in this chapter)

EC_i , MC_i , and HC_i are the establishment, management, and harvest costs per unit in year i (see Scaling Up/Aggregating” later in this chapter).

Typically having an annualized value (\$ per acre per year) would be preferable. Indeed, much of the present guidance is based on the idea of an annual flow of goods and services. In the (relatively unusual) case that an NTFP is established, managed, and finally harvested at one time (e.g., forest-farmed American ginseng), the NPV from equation 1 is first converted to an infinite-horizon soil expectation value (SEV) as shown in equation 2, then annualized into equivalent annual income (EAI) by multiplying by the discount rate (c.f. Alexander and others 2002a) (eq. 3):

$$SEV = \frac{NPV}{(1 + \rho)^t - 1} \quad (2)$$

$$EAI = SEV \cdot \rho \quad (3)$$

In other cases, NTFPs are harvested from the same plot annually. Such might be the case for fruits, or if only a small quantity of product (approximately equal to the annual growth of the product) is harvested. Perhaps the most common case is wild-harvesting, where there are no establishment or management costs, so equation 1 simplifies to (c.f. Godoy and others 1993):

$$EAI = (P - HC) \cdot MQ_{An} \quad (4)$$

where MQ_{An} is the marginal annual quantity harvested per acre on a sustainable basis.

If current harvest were so high as to be unsustainable, current estimates of value would overstate annual contributions to NPV, so estimated NPV should be adjusted down if current harvest levels are considered unsustainable. Godoy and others (1993) suggest adding a depletion premium to the harvest costs in the case of unsustainable harvest. Alternatively, one could adjust the marginal annual quantity harvested by the amount of the overharvest (Oh):

$$EAI = (P - HC) \cdot (MQ_{An} - Oh) \quad (5)$$

NTFPs are most commonly priced as delivered (to primary buyer) or as roadside prices. Since delivered or roadside prices are marginal values for a product that has had value added through harvest and transport, they are an overestimation of the forest's provisioning service value. In many cases, harvest

costs are primarily comprised of the opportunity cost of labor. If the analyst can reasonably assume that other harvest costs are minimal, and if it would be costly to collect the data required to adequately estimate those harvest costs, the analyst could defensibly argue that harvest costs are virtually zero, simplifying equation 1 further to:

$$EAI = P \cdot MQ_{An} \quad (6)$$

Unfortunately, since this last equation unambiguously overestimates the ecosystem service value (harvest cost must be non-zero), it does not meet the principle of finding a conservative estimate when uncertainty exists. For this reason, it is a best practice to find a reasonable estimate of harvest costs. These can be assessed through methods such as interviewing harvesters or utilization studies, as described in the "Estimation of Production Costs" section.

Challenges

NTFPs are found in most forests but are often neglected and not recognized in forest policies or management. One reason for this is that very little is known about the production, markets, industry, trade, and contributions to society of NTFPs. Realizing the actual contribution and value of NTFPs would likely enhance their status and generate greater attention in forest policy and management decisions. This section discusses the shortcomings of available data on the quantities and values of NTFPs in the U.S. South.

Valuing the ecosystem service of provisioning NTFPs presents numerous challenges compared to valuing the service of provisioning timber. First, NTFPs may be traded in formal or informal markets or used for personal benefits with no markets (McLain and others 2008). For products that are traded in formal markets, a convenient starting point for valuation is the market price, based on available transaction records and data on price and quantity. For NTFPs traded through informal markets, prices and quantities may not be tracked or recorded. Informal markets are less transparent, and the trade of many NTFPs occurs in the transitional region between formal and informal markets. Many NTFPs are consumed by harvesters or their family and friends with high cultural or recreational values placed on the act of collection or consumption. Finding the correct way to estimate and add together these values is not easy.

Second, NTFPs may be poached from natural forests or stolen from forest farms. Valuable products, such as ginseng, are reportedly poached frequently and extensively. Other products may be harvested without permits or permission, or out of season.

This illegal harvest production is not documented, but it certainly has value that affects States' economies. Law enforcement agencies may keep records that could help document the amount of NTFPs illegally harvested, yet this represents an unknown portion of the total illegal harvest. We do not provide guidance here about how to estimate value of poached products, but it is clear that when poached NTFPs are excluded from valuation, the resulting total values are certainly underestimates.

Third, scalability of research is a challenge. Results of many local level studies are not scalable to State, regional, or national levels. Findings and recommendations from studies that focus on local ecological or sociological systems are difficult, if not impossible, to apply to larger geographic areas. This may be because the locations chosen for NTFP valuation are not representative of the broader region. Survey respondents may be drawn from sampling frames that are particular to case study locations and markets, making survey results difficult to extrapolate to a State or region. Inadequate sampling of plant populations may not provide full representation from all forest types. Attaining an accurate sample of an entire forest or region may be difficult due to funding or accessibility issues.

Geographic variability in the presence and frequency of specific NTFPs, even within the same forest type, makes it very difficult to impute NTFP values on specific acres. Any particular acre might have more or less NTFP quantity than the average. Further, even if product is available, its value may be diminished by factors that make it less accessible or more costly to harvest. These factors are difficult to estimate with existing data.

Fourth, most studies have been one-time efforts and therefore do not provide the information required for trend analysis. Most NTFP valuations could be considered one-time spot reports that become outdated with changing economic conditions. Only a few NTFPs, such as American ginseng, are regulated and monitored, resulting in time series data. The importance of temporal data is that NTFP markets and trade fluctuate with changes in demand for "natural" products, as does the biological production cycle given external vagaries such as weather. Many NTFPs have only had one, if any, formal valuation study and more often than not the valuation is many years old (Blatner and Alexander 1998). Survey data and value estimates from one point in time do not account for variation in supply and demand.

There are numerous challenges with measuring quantities and prices received for NTFPs in the forest. Problems with quantification relate to both biological and social conditions. Many NTFPs are seasonal, so there may be a very short amount of time to adequately quantify, interview harvesters, and assess product value. Ramps (*Allium tricoccum*; wild onions) are a good

example, as they are available for harvest for only a few weeks during the year. When interviewing people about how much they harvest or what prices they receive, the timing and location of interviews are important. During the season, when recollection is best, harvesters may not want to be bothered or may be too busy to respond to requests for information. If outside the season, recollection may be more challenging. NTFP harvesting is a part-time seasonal occupation for many people who combine it with other more formal employment. The diversity of NTFP harvesters makes it challenging to sort out who is doing what.

The NTFP industry depends a lot on trust, and market participants are wary of outside surveys and interviews (Greenfield and Davis 2003). Harvesters, buyers, and other market players are reluctant to provide information especially when they are unsure of how it will be used. Harvesters may be reluctant to share information if they do not trust the interviewer to protect their knowledge. People involved with NTFP markets may be reluctant to provide reliable information if they are concerned about the product becoming regulated or losing competitive advantage. In addition to these factors, survey response rates may be low due to the nature of the business.

A partial accounting of harvest volumes can be obtained from the permitted harvests on public lands such as National or State forests. However, these data often have limitations. The product categories of the USDA Forest Service (National Forests) "cut-and-sold" reports often are too general to identify species. For example, the product code "foliage" is used across all National Forests. This code is applied to products sold to the floral industry, but there is no way to identify the species harvested.

IDENTIFICATION AND SELECTION OF NTFPS FOR VALUATION

Almost any plant or fungi gathered by the occasional forest visitor could be classified as a non-timber forest product for personal use. NTFPs gathered for personal use may be associated with specific market segments (food, medicine, decorations) or not (spiritual gain, luxury, subsistence, or recreation) (Emery and Pierce 2005). Southern forests also are the source of a plethora of medicinal herbs for a multi-billion dollar international market that has contributed to the region's economy since trade began with Europe, Asia, and other parts of the United States. Sassafras roots, harvested from what is now Martha's Vineyard, were formally traded in the global herbal market as early as the 1600s. Other NTFPs also have a long history of trade; for example, the longleaf pine forests of the Southern United States were a global source of pine resin and associated products from colonial times until the early 20th century.

Today, non-timber products are harvested from the forests of every Southern State. Table 5.1 provides examples of common non-timber forest products found in the region.

Some forest types, because of their biological diversity, have more NTFPs than other forest types. For example, the Appalachian hardwood forests are high in diversity and have many different medicinal forest products. However, we recommend focusing valuation efforts on the NTFPs that are most significant to each State. As knowledge and expertise about NTFPs increase, and the infrastructure and resources to value NTFPs improve, then additional products can be evaluated.

In this section, we discuss key information required to identify and select NTFPs for valuation, including commercially and non-commercially harvested NTFPs. Major market segments are identified to allow for categorizing products by their purpose. States also may find it useful to classify and prioritize NTFPs for valuation by forest type. Forest ownership is discussed to allow for further segmentation of analysis. Finally, we present criteria for selection of NTFPs for valuation.

The decision tree in figure 5.1 provides a framework for selecting priority NTFPs for valuation.

Without actually knowing the values of NTFPs in the State a priori, decisionmakers may need to rely on expert knowledge to select priority products for valuation. This can be accomplished by convening a diverse group of practitioners, researchers, and

managers who can provide insights and recommendations. These experts may also provide valuable advice toward identifying data sources, and, if necessary, sampling and survey methodologies. Their knowledge of harvest routines and market structure will facilitate the selection of an efficient sampling strategy. Discussions should cover non-commercial values, as these may tend to be overlooked in comparison with commercial values.

Market Segments of NTFPs

There are many ways to classify non-timber forest products. In general, NTFPs can be classified into five broad market segments (based on Chamberlain and others 1998): (1) culinary; (2) medicinal; (3) decorative; (4) nursery stock and landscaping; and 5) fine arts and crafts. Additional categories may evolve as knowledge about this industry develops. A brief summary of each segments is provided below.

Culinary forest products—Edible forest products include berries, nuts, saps, ferns, tubers, and bulbs. In the South, ramps are one of the most popular edible forest products found in the mountains. Edible fungi, particularly mushrooms, also are well-documented edible forest products. The geographic distribution of edible forest products is dependent on ecological conditions. Maple syrup is primarily produced in the Northeastern United States, from Ohio to Maine, but production extends south into Virginia. Many edible forest products are collected for personal consumption, although there are markets also for most products, such as the nuts of black walnut (*Juglans nigra*), which are harvested throughout the tree’s natural range (Chamberlain and others 1998).

Medicinal forest products—The use and trade of herbal medicines derived from forest plants has a long history and may constitute the highest valued segment of the NTFP industry. The American Herbal Products Association (Dentali and Zimmermann 2012) tracks 21 plant species that are used in commerce. Table 5.2 summarizes information on 13 that are native to eastern hardwood forests. As previously noted, the markets for these products vary with changes in demand. Some products have increased in volume harvested (black cohosh [*Actaea racemosa*], slippery elm [*Ulmus rubra*], trillium [*Trillium erectum*]), while others have decreased (saw palmetto [*Serenoa repens*], bloodroot [*Sanguinaria canadensis*]). The roots of American ginseng (*Panax quinquefolius*) have been harvested from eastern hardwood forests for over 250 years. Nineteen States are certified by the U.S. Fish and Wildlife Service to export wild-harvested ginseng. Of those, seven are located in the South (table 5.3). Saw palmetto, which is used to treat benign prostatic hyperplasia, is sourced primarily from the pine forests of Florida (table 5.4).

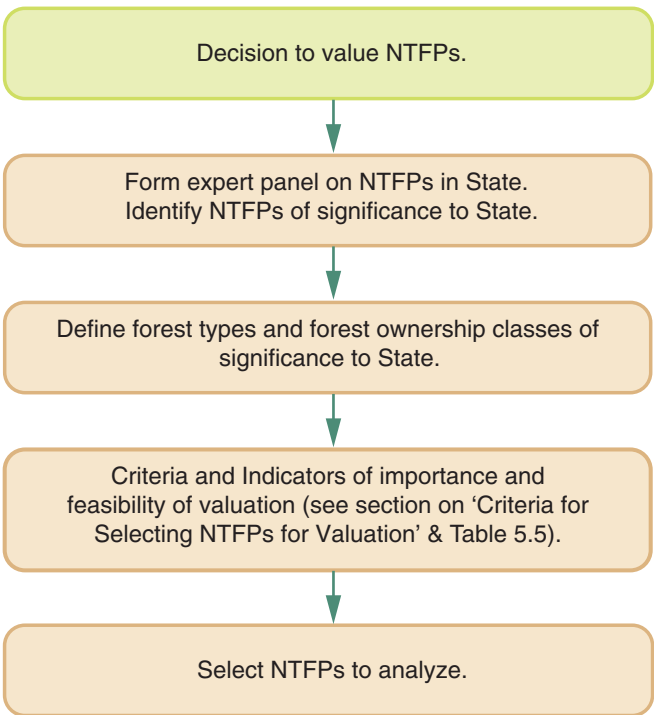


Figure 5.1—Decision tree that provides a framework for selecting priority NTFPs for valuation.

Table 5.2—Average annual harvest of medicinal forests products tracked by American Herbal Products Association and found in southern forests

Latin name	Common name	Plant part	Average annual harvest ^a 2001–2005	Average annual harvest ^a 2006–2010	Percent change
<i>Actaea racemosa</i>	Black cohosh	Root	224,072	284,162	26.8%
<i>Aletris farinosa</i>	White colicroot	Root	1,012	690	-31.9%
<i>Aristolochia serpentaria</i>	Virginia snakeroot	Root	121	43	-64.2%
<i>Caulophyllum thalictroides</i>	Blue cohosh	Root	6,651	5,169	-22.3%
<i>Chamaelirium luteum</i>	Fairywand	Root	4,688	4,541	-3.1%
<i>Cypripedium</i> spp.	Lady's slipper	Whole plant	51	48	-4.3%
<i>Dioscorea villosa</i>	Wild yam	Tuber	33,422	37,692	12.8%
<i>Hydrastis canadensis</i>	Goldenseal	Root & Leaf	73,619	74,708	1.5%
<i>Panax quinquefolius</i>	American ginseng	Root	62,294	63,461	2.0%
<i>Sanguinaria canadensis</i>	Bloodroot	Root	24,823	5,056	-79.6%
<i>Serenoa repens</i>	Saw palmetto	Fruit	3,293,377	2,432,841	-26.1%
<i>Trillium erectum</i>	Red trillium	Whole plant	1,099	1,445	31.5%
<i>Ulmus rubra</i>	Slippery elm	Bark	182,435	304,207	66.7%

^aAverage annual wild harvest (pounds dry weight) for 5-year periods.

Sources: AHPA (2012) and Chamberlain and others (2013b).

Table 5.3—Volume by weight in pounds of dry wild American ginseng harvested by States

State	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Total
Alabama	457	1,025	749	221	761	318	717	1,345	474	454	476	626	7,623
Arkansas	2,073	2,632	1,770	504	927	989	1,190	1,796	1,195	487	238	1407	15,208
Georgia	266	426	263	402	167	280	406	293	212	158	361	346	3,580
Kentucky	15,085	22,583	16,717	9,392	13,713	11,345	11,839	19,246	15,041	13,176	15,276	20,025	183,439
North Carolina	8,790	6,548	4,271	5,602	7,060	12,378	11,402	10,531	8,041	9,716	8,765	7,849	100,953
Tennessee	5,815	10,826	8,690	5,280	8,153	8,695	8,435	14,642	11,464	9,322	10,145	13,867	115,334
Virginia	3,801	4,675	3,435	1,571	2,878	3,050	2,918	4,081	3,610	3,856	4751	4370	42,996
Total	36,287	48,715	35,895	22,973	33,659	37,055	36,908	51,934	40,037	37,169	40,012	48,490	469,133

Table 5.4—Volume by weight in pounds of dry saw palmetto berries

	Wild harvested dried (pounds)	Conversion factors	
		Times 2 for under-reporting ^a	Times 3.3 for fresh fruit (pounds) ^b
1997	1,253,280	2,506,560	8,271,648
1998	1,966,685	3,933,370	12,980,121
1999	1,082,594	2,165,188	7,145,120
2000	4,663,613	9,327,226	30,779,846
2001	2,206,157	4,412,314	14,560,636
2002	2,877,519	5,755,038	18,991,625
2003	3,397,465	6,794,930	22,423,269
2004	2,918,940	5,837,880	19,265,004
2005	5,786,806	11,573,612	38,192,920
2006	2,277,504	4,555,008	15,031,526
2007	4,199,685	8,399,370	27,717,921
2008	2,644,813	5,289,626	17,455,766
2009	1,581,106	3,162,212	10,435,300
2010	1,461,125	2,922,250	9,643,425
Total	38,317,292	76,634,584	252,894,127
Average	2,736,949	5,473,899	18,063,866

^a Industry representative estimates that reported harvest volumes are one-half what is actually harvested.

^b A factor of 3.3 is the standard for converting from dried to fresh product.

Source: Dentali and Zimmermann (2012).

Decorative forest products—Many forest plants and their parts are used in decorative arrangements, to complement and furnish the backdrop for flowers, and as the main component of fresh and dried ornaments. The end uses for forest-harvested floral decoratives include fresh/dried flowers, greenery, basket filler, wreaths, and roping. Galax (*Galax urceolata*) from western North Carolina is an iconic floral product from the hardwood forests. The city of Galax, Virginia, is named after the plant, reflecting its historical importance to the local economy. Spanish moss from southern forests also are in this market segment. Conifer boughs may be the most widely sold decorative forest product in the United States (Chamberlain 2000). The products in this category contribute significantly to regional economies. For example, in 1995, the United States exported more than \$14 million in forest-harvested moss and lichens, most of which originated from Appalachia and the Pacific Northwest (Goldberg 1996).

Nursery and landscaping forest products—Live forest plants are collected for the nursery and landscaping industry. These may be marketed as bare root stock or balled live plants for direct planting. Common examples include rhododendron (*Rhododendron* spp.), azaleas (*Azalea* spp.), mountain laurel (*Kalmia latifolia*), and the endemic Fraser fir (*Abies fraseri*).

The Christmas tree industry has realized tremendous growth over the last decades, due in part to the germplasm from Fraser fir, growing in a very limited distribution in western North Carolina and southern Virginia. Its value to these States could be significant. Pine straw, harvested throughout the South, is used extensively in landscaping. The National forests report the permitted harvests of plant materials used for nurseries and landscaping. In 2013, more than 43 percent of the total recorded harvest in all National forests came from North Carolina. Collection of forest understory wildflowers for horticultural sales is a cottage industry in the southern Appalachian region, and many wildflowers are readily available through internet sales and brick-and-mortar stores (Botanical Wonders Nursery 2015, Cullina 2000, Mainely Crafts 2015).

Fine art and craft forest products—Artisans using non-timber products to craft luxury items find the ingredients for their creations in forest plants. The use of NTFPs for fine arts and crafts is limited only by the artisans' imagination. Wood collected from forests may be formed into carvings, turnings, walking sticks, utensils, and containers. Moss, lichens, and seeds may be formed into jewelry. Vines are crafted into wreaths, sculptures, and statues. Fine baskets are crafted from splints of wood or grass stems. The outlets for these fine arts are varied as well. For some artisans, the preferred venue is local and regional craft fairs. Others may market their products through specialty retail stores or internet based shops. NTFPs used for fine arts and crafts contribute to a multi-million dollar handicraft industry. Determining the proportional value of the provisioning ecosystem service to this industry is problematic because of the challenge of differentiating the value of the forest input from the value of human artistry.

Forest Type Classifications

Each forest type supplies distinct NTFPs. Appendix B of the Forest Inventory and Analysis (FIA) manual for southern forest inventories identifies nine forest types of the South, along with their associated species (USDA Forest Service 2010). These forest types and associated tree species are provided in appendix 5.2 of this chapter. The FIA manual provides information on the tree species found in each forest type, but information on the associated plants and fungi is lacking.

Forest Ownership Classifications

The ownership of southern forests reflects historical and contemporary trends in economic sectors, stakeholder preferences, and technological developments. While the ownership class (public, corporate, family) of forests may affect the quantity of NTFP harvest, harvesting opportunities do exist across these ownership classes (Alexander and others 2011, Butler 2008, Salwasser 2006). Attempts to value NTFPs must consider the resource base and the interactions and impacts of its changing ownership and management. As competition for land continues in the South, changing forest land uses and management practices could impact forest ecosystems in ways

that affect the availability and values of NTFPs. If products become more/less available on private lands, their value from public lands will decrease/increase, as well.

Criteria for Selecting NTFPs for Valuation

Deciding which NTFP or suite of NTFPs to value in each State is the first step in estimating the value of NTFPs in the U.S. South. Harvest activity, demand, and a State's position as a producer of a particular NTFP will help determine whether to value a particular NTFP in that State. The perceived importance to a State's cultural heritage, economy, and ecological diversity should also be criteria for targeting valuation efforts. The criteria and indicators presented in table 5.5 and discussed below provide guidance on deciding which product, or suite of products, to prioritize.

Criterion 1—Amount of harvest by volume. There is little incentive to devote limited resources for valuation of products that have insignificant demand. There needs to be an indication that large quantities, relative to the product's abundance, are being harvested. One good source of the amount being harvested are the National Forest "cut-and-sold" reports (USDA Forest Service 2015a). For example, the National Forests of North Carolina reported that more than 10,000 pounds of forest herbs were harvested in 2014. In the same year, the National Forests in Florida reported 48,000 pounds of "foliage" as being harvested. These harvest quantities suggest that it would be worthwhile estimating the value of these products, starting with determining which species are categorized under these product codes. Other sources of data that might help estimate harvest volumes

and identify priority NTFPs are export databases such as the Interactive Tariff and Trade DataWeb (U.S. International Trade Commission 2016).

Criterion 2—Availability of data. The basic data requirements to value a non-timber forest product are the harvested or standing stock volumes, measured in an appropriate unit (e.g., pounds), the prices paid to a clearly identified market player (e.g., harvester, primary buyer, secondary buyer), and the associated costs (labor, capital, transport, processing). Lack of volume, price, or costs data could thwart valuation efforts. However, just because data are not available does not mean that a particular NTFP should be disregarded. If there is evidence of large harvest volumes or values, lack of data may suggest that funding should be allocated to data collection. Potential data sources are listed in appendix 5.3. Further development of sources of data is necessary to value many NTFPs. Perhaps the most valuable type of data would be the amounts of per-acre payments to landowners for the rights to harvest a product, where relevant.

Criterion 3—Amount of potential product stock in State's forests. The forest types, and the amount, found in a State are indicators of the resource base and should be a factor in selecting which NTFP to value. A State with large amounts of pine forests that could be the source of pine straw may want to direct resources to estimating the value of provisioning that product. Upland hardwood forests, mixed mesophytic forests, and Appalachian hardwood forest types (classified as *oak/hickory* group and *maple/beech/birch* group by FIA) have tremendous

Table 5.5—Criteria and indicators for selection of NTFPs to estimate value

Criteria	Indicator(s)
C1. Amount of harvest by volume	I.1 Permit volume reported by National Forests I.2 Permit volume reported by State I.3 Harvest volume reported by industry I.4 Harvest volume reported by regulatory agencies
C2. Availability of data	I.1 Harvest volume reported by regulatory agencies I.2 Number of dealers I.3 Permit volumes reported by National Forests
C3. Amount of potential product stock in State's forests	I.1 Acres of potential habitat, as determined by forest type and other parameters
C.4. State's relative standing as producer or potential producer	I.1 Acres of potential habitat compared to other States I.2 Relative harvest volumes
C.5. Economic importance to specific communities	I.1 Expert social knowledge
C.6. Ecological vulnerability to over-harvest	I.1 Expert ecological knowledge I.2 State's threatened and endangered listings I.3 State's natural heritage listings I.4 Plant's conservation status

plant diversity that provision medicinal and edible plants. In States with large areas of forest types that produce American ginseng (as shown in table 5.6), valuation efforts should include that product. Information about the extent of different forest types in different States is readily available through FIA databases.

Criterion 4—State’s relative standing as producer or potential producer. An important criterion for selecting NTFPs to value is the market share that a State realizes from a particular product. A State that is the sole producer, or a major producer, should consider directing more resources to estimating the value or provisioning that product. For example, Florida is the sole, or major, producer of saw palmetto. North Carolina is the sole producer of galax leaves for the floral industry. Efforts by these States to value the respective products would contribute much to a comprehensive valuation of NTFPs in the South. Estimates of the marginal value of forest for production of NTFPs common in other States may be available in reports from those States, assuming that each State directs valuation efforts based on local harvest activities and the local resource base.

Criterion 5—Economic importance to specific communities. The perceived importance of NTFPs to a State’s rural economy should be considered in selecting NTFPs to value. How important are these products to rural people of each State? For example, sweet grass baskets from South Carolina are important to the cultural and economic well-being of African-American communities, who can trace this activity to their ancestors. Valuing this product would be important to South Carolina. Communities in western North Carolina and eastern Tennessee depend on annual ramp festivals to generate revenues for fire departments and other civic groups. There are at least 10 ramp festivals in western North Carolina and eastern Tennessee, and people travel for many hours to attend them. Large volumes of ramps are harvested during the spring and sold through the festivals and through farmers’ markets, roadside stands, and restaurants. These edible forest products are critical to the rural economy of these States.

Criterion 6—Ecological vulnerability to over-harvest. Priority consideration may be given to forest species that are vulnerable or at risk of over-harvesting or some other stressor that can

endanger the product. Common species that are widespread may be less of an immediate concern in valuation. For example, it may be more important to estimate the value of orchids rather than Joe-Pye weed (*Eupatorium purpureum*), both of which are used for their medicinal properties. Orchids are much less common than Joe-Pye weed, and the marginal value of forest for provisioning a relatively scarce product is likely to be higher. Species of ecological concern may be identified from State threatened or endangered lists.

QUANTIFICATION OF NTFP PROVISIONING SERVICES

While the challenges of accurately valuing ecosystem services are numerous, the results depend heavily on quantifying the amounts of NTFPs harvested per acre of productive habitat. Quantification can take the form of an estimate of the total physical standing inventory, or “stock,” of each NTFP present; or the rate of harvest, or “flow,” from those areas. Most researchers prefer flow measures (Godoy and others 1993, Tewari 2000). The flow approach relies on measuring the amount of NTFPs harvested from a region (e.g., forest, watershed, State) often through reporting of volumes sold to market entities. This provides insight of the volumes of products that enter the market. To estimate values, data are needed on both annual harvest volumes and prices paid to a clearly defined market player (preferably the harvester). In this section, we discuss quantification of harvest volumes and the challenges of estimating the marginal quantity harvested per hectare, following the decision tree in figure 5.2.

There are a few sources (appendix 5.3) of data on the volumes of NTFP harvest in a State, and each has limitations. One major challenge with existing data sources is that none measure or report marginal *per-acre* NTFP flows. They report at the forest, county, State, or national level. Estimating even average per-acre harvest volumes would require knowing how many acres of product are in the forest. This requires inventorying the respective NTFPs. We know of no inventories of standing stocks of NTFPs. Trees within the forest are regularly inventoried, and growth and yield models are developed for many tree species. Trees that provide NTFPs, such as tulip poplar (bark) and slippery elm (bark), can be assessed for their per-acre stock volumes, using FIA database. In general, this information does not exist for forest plants and fungi that are harvested as non-timber products. To

Table 5.6—Area of potential oak/hickory and maple/beech/birch habitat for ginseng by ownership group and State

Ownership group	State							Total
	AL	AR	GA	KY	NC	TN	VA	
National Forest	215,248	1,280,534	541,963	789,820	889,384	572,202	1,486,968	5,776,121
Other Federal	83,871	191,850	104,814	290,252	271,419	486,375	314,118	1,742,698
State and local	200,468	131,427	264,390	208,062	284,177	671,329	381,969	2,141,823
Private	6,604,943	6,281,779	5,543,069	9,300,586	5,926,558	8,659,048	7,886,110	50,202,092
Total	7,104,530	7,885,590	6,454,236	10,588,720	7,371,538	10,388,954	10,069,165	59,862,734

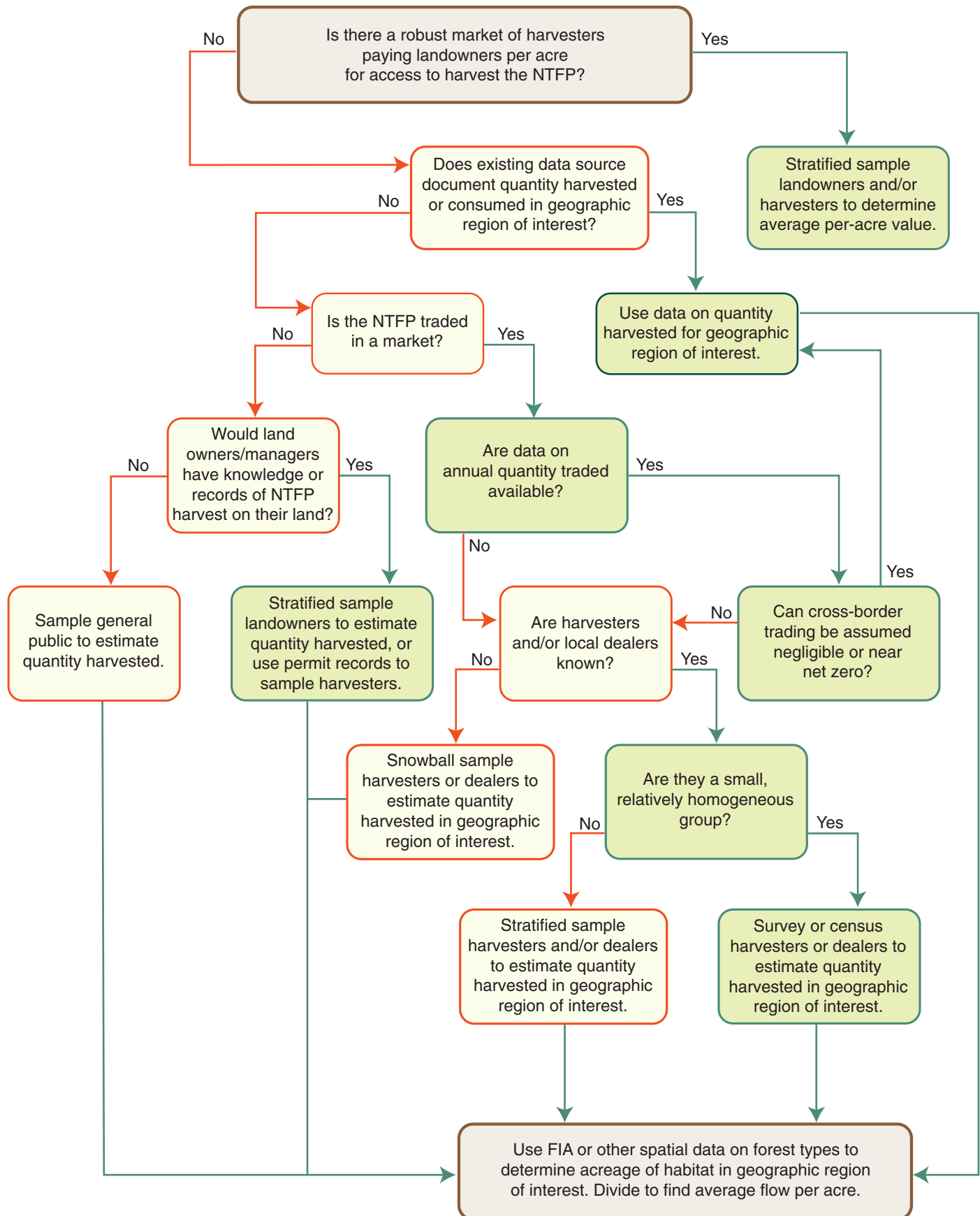


Figure 5.2—Decision tree showing the quantification of NTFP provisioning services.

obtain this information, forest agencies would need to develop methods and protocols to inventory these products. Efforts are underway to develop inventory protocols for some medicinal plants harvested for their roots (Chamberlain and others 2013a, Small and others 2011).

Given this lack of data, and the probable uneven distribution of individual NTFP populations and NTFP harvest within particular forest types, estimating the quantity of NTFP supplied by a “marginal” acre may not be possible at this time. An impact or land-use change on one “marginal” acre may not have any impact on NTFP flows at all, whereas a different acre might greatly impact flows. Thus, the best available estimate of the impact of a marginal change in acreage of forest of a given type on NTFP flows is the per-acre *average* flow for that forest type in a particular geographic region. However, it is important to keep in mind the fact that average and marginal are different measures and to identify conditions under which marginal values are likely to be substantially different than average ones. For example, there might be great geographic diversity where NTFPs are found and also where they are harvested. The areas that provide the highest net value of NTFP flows per acre may be those that are near access points (e.g., roads and trails), relatively closer to towns, and with productive sites, since costs would be lower. Also, the presence (or lack thereof) of sufficient substitute NTFP harvest sites may decrease (or increase) the marginal value of a specific site.

Another issue of particular concern for an ecosystem service valuation is variation in harvest (flow) over time. As table 5.2 demonstrates, aggregate NTFP harvests are quite variable. Variables explaining change in aggregate NTFP harvest levels over time have not been widely researched, but they likely include macroeconomic conditions, weather, and policy (Bailey 1999¹⁶). The challenge is that most ecosystem service studies estimate values only at a single point in time. That single point in time could be substantially higher or lower than the average, leading to erroneous estimates of the ecosystem service value. At its worst, the ecosystem service value might be based on a level of harvest that is unsustainable, and thus impossible over the long term. If conducting a single point in time estimate, the analyst should consult with individuals within the NTFP sector of interest to determine if the studied year is likely to be an over- or underestimate of the average, and if so, how to adjust it. If current harvest levels are unsustainable, they should be adjusted with a depletion premium (Godoy and others 1993) or by subtracting the amount of the overharvest. The overharvest adjustment would be equal to the difference between the annual amount harvested and the amount of new product that is generated every year.

The main sources of harvest volumes are the National Forests and the U.S. Fish and Wildlife Service. “Cut-and-sold” reports are available from the National Forests since fiscal year 1977 and are based on harvest permits issued by National Forests (USDA Forest Service 2015a). Data on NTFP harvest in the “cut-and-sold” reports for 13 National Forests in 12 Southern States (USFS Region 8) include Christmas trees, limbs and boughs, foliage, mosses, needles, cones (dry or green), transplants, roots, herbs, grass, vines, mushrooms, and other plants among non-convertible removals from National Forests for which a permit is required. Data from “cut-and-sold” reports reflect the permitted harvests amounts on National Forests only. Thus, they may not accurately represent the actual harvest amounts, and they may underestimate harvests for an entire State.

“Cut-and-sold” reports provide information on the volume and value of permitted harvests but not the number of permits. Also, the value of permitted harvests is based on the amount charged for the permit and does not reflect prices paid to harvesters for the product. With few exceptions, the “product” description and species codes do not provide enough information to identify specific species. American ginseng is the exception, though the use of the “species” code that identifies this NTFP is questionable. For example, the National Forests in Florida use the species code for ginseng in reporting “other plant” material harvested in 2014, even though the species is not found in the State. Other National Forests use the species code for ginseng in reporting permitted harvests of “transplants,” which is an unlikely use of the product. The National Forest System is aware of these challenges and is making efforts to improve the reporting system. However, even an improved system will provide data on harvests only from National Forests and no other public or private land ownership categories.

State forests may also be a potential source of information about NTFP harvests. Of the 13 Southern States, 7 (Alabama, Florida, Georgia, Kentucky, North Carolina, South Carolina, Tennessee) indicated that NTFP harvest on State forests is allowed, at least in some cases, while 5 (Arkansas, Louisiana, Mississippi, Texas, Virginia) indicated that no NTFP harvest is allowed on any State forest, and one State (Oklahoma) has no lands officially designated as State forests (Frey and Chamberlain 2016). State agency representatives indicated that NTFPs commonly harvested from State lands include pine straw, saw palmetto, pine cones, berries, medicinal plants, and transplants. In the States that allow harvest, activity is controlled by methods such as permits (both free and for a fee), leases, and other contracts (Frey and Chamberlain 2016). Unfortunately, in most cases the fees charged for harvest access are not set by the market, so most likely they bear little relation to marginal values. The quality and accessibility of harvest data from these State forests varies from State to State.

¹⁶ Also see: Frey, G.E.; Chamberlain, J.L.; Prestemon, J.P. 2017. Supply, demand, and regulation of wild American ginseng. Unpublished manuscript. On file with: Gregory Frey, Research Forester, USDA Forest Service, Southern Research Station, P.O. Box 12254, 3041 E. Cornwallis Rd., Research Triangle Park, NC 27709-2254.

Data are available for American ginseng in States that are certified, by the U.S. Fish and Wildlife Service, to export the roots. Ginseng buyers are required to register with the appropriate State agency, keep records of each transaction (volume and harvest location by county), and have ginseng roots inspected and certified by the State agency before the roots can leave the State. The records are sent to State agencies who report annually to the U.S. Fish and Wildlife Service. Web addresses are provided in table 5.7 for appropriate State agencies. Buyers of American ginseng regularly buy other medicinal forest products, but they are not required to report volumes of these other products.

Market Harvest Volumes

Many NTFPs are sold in markets and used, gifted, or bartered by harvesters (McLain and others 2008). When a particular NTFP is primarily harvested for sale, the analyst must decide if the market harvest volume in a State is a reasonable approximation of the total volume. Using only market harvest volumes may leave out some quantities that are never traded and used for personal use by the harvester, but for some products, this might be a negligible amount.

NTFP ecosystem values are produced at the location of harvest. If data on volumes traded are based on the location of sale rather than the location of harvest, the analyst must determine if cross-border (e.g., interstate) trading of NTFPs can be considered negligible or, alternatively, if the trade is roughly equivalent in both directions (net zero). If one of these two assumptions can be made reasonably, then data at location of sale are appropriate for estimating NTFPs harvested in the State. On the other hand, if trade goes mostly one direction (either mostly into or mostly out of a State), then a further survey will be needed to determine the location of harvest.

Using existing data sources—The Southern United States has a dynamic and significant commercial NTFP industry, with active and substantial trade within and across State borders. The States of the South are in many cases the global source of particular

non-timber forest products. Yet complete data are not available to fully assess the values of these products. Data are primarily available for medicinal forest products, gathered regularly by the American Herbal Products Association (AHPA). The AHPA surveys its members every 2 years and publishes the findings. To illustrate the significance of the medicinal plant industry in the South, we provide examples based on these findings. In addition to ginseng (described below), the average annual harvest (2006–2010) of another 21 medicinal forest products are tracked by the AHPA (Dentali and Zimmermann 2012). Quantities range from 43 dry pounds of Virginia snakeroot (*Aristolochia serpentaria*) to 2.4 million pounds berries of saw palmetto (*Serenoa repens*). Over 14 years (1997–2010), more than 19,730 dried tons of dried saw palmetto berries were harvested, primarily from the forests of Florida (table 5.4). Average annual harvest declined for about half of the products from the 5-year period 2001–2005 to the period 2006–2010.

We provide two examples of the flow (harvest) approach to quantifying NTFPs. First, American ginseng is harvested from forests in seven Southern States, and harvests are well documented. In this example, we start by estimating the total volume of ginseng harvest in each State. Once total volume is estimated, we calculate average volume per acre based on the number of acres of habitat (table 5.6). The wide range across States illustrates the importance of examining market share of a product to determine the urgency of valuing any given NTFP in a particular State. The second example, saw palmetto, is harvested almost exclusively from forests in Florida. Reporting of harvest volumes of saw palmetto is voluntary and facilitated through a biennial (every 2 years) industry survey.

Example: American ginseng—Data on ginseng harvest is available from the U.S. Fish and Wildlife Service, and to a lesser extent the National Forest “cut-and-sold” reports. States first need to determine if this product is worth valuing. An examination of the States that are certified to export ginseng reveals seven Southern States (i.e., Alabama, Arkansas, Georgia, Kentucky, North Carolina, Tennessee, Virginia) that produce significant

Table 5.7—Southern States certified to export American ginseng

State	Ginseng program
Alabama	http://www.agi.alabama.gov/divisions/plant-health/
Arkansas ^a	http://www.aad.arkansas.gov/ginseng-program
Georgia	http://www.georgiawildlife.com/GinsengProgram
Kentucky	http://www.kyagr.com/marketing/ginseng.html
North Carolina	http://www.ncagr.gov/plantindustry/plant/plantconserve/ginseng.htm
Tennessee	http://www.tennessee.gov/environment/article/na-ginseng-program
Virginia	http://www.vdacs.virginia.gov/plant-industry-services-ginseng.shtml

Note: this table provides links to State program Web sites showing the regulations enacted to manage the harvest of American ginseng [Date accessed: October 4, 2017].

^a On last date of access, the URL listed for Arkansas did not contain information, but it is the official website for the Arkansas ginseng program.

quantities of American ginseng (table 5.2). From 2002 through 2013, more than 469,000 pounds of American ginseng were harvested from the forests of these States (table 5.3). Of these Southern States, western North Carolina, eastern Tennessee, and eastern Kentucky accounted for approximately 85 percent of the southern harvest from 2002 through 2013 (fig. 5.3). Virginia accounted for about 9 percent of the South's total, while Arkansas accounted for about 3 percent.

The importance of ginseng to a State can be further assessed by examining permitted harvests on National Forests. Only four National Forests reported the permitted harvest of American ginseng (table 5.8) and then only for 2009 through 2013. During this period, the total amount of American ginseng harvested from National Forests in the southern United States was approximately 4,450 pounds. The National Forests of North Carolina generated more than 60 percent of this volume. The Cherokee National Forest generated 13 percent of the total volume of American ginseng from National Forests.

The next step in quantifying ginseng flow is to determine the volumes harvested in each State. These data are available through the U.S. Fish and Wildlife Service and can be complemented by National Forest data from the "cut-and-sold" reports. Table 5.3

summarizes volume of ginseng harvest by State and year (2002–2013). State level data can be disaggregated to the county level, while national forest data are for the specific forest.

The challenge now is to determine the per-acre marginal flow of American ginseng. To approximate this with the average production per acre, we need to know how many acres of appropriate habitat are producing ginseng in each State. American ginseng prefers mixed mesophytic forests (e.g., north-facing cove forests) but may be found in similar forest types. FIA classifies this forest type as *oak/hickory group* and *maple/beech/birch group*. Trees found in these groups under which ginseng grows include yellow-poplar, northern red oak, black walnut, black cherry, ash, yellow birch, and red maple. However, because it has been harvested extensively, American ginseng may not appear in forests that should support the plant.

Using FIA databases, we can estimate the area of appropriate forest habitat in each State (table 5.6). To estimate the volume per acre, the best available approximation may be to assume that harvest occurs evenly over the entire area of potential habitat. This assumption will likely lead to an underestimate of the volume per marginal acre where ginseng is actually being harvested.

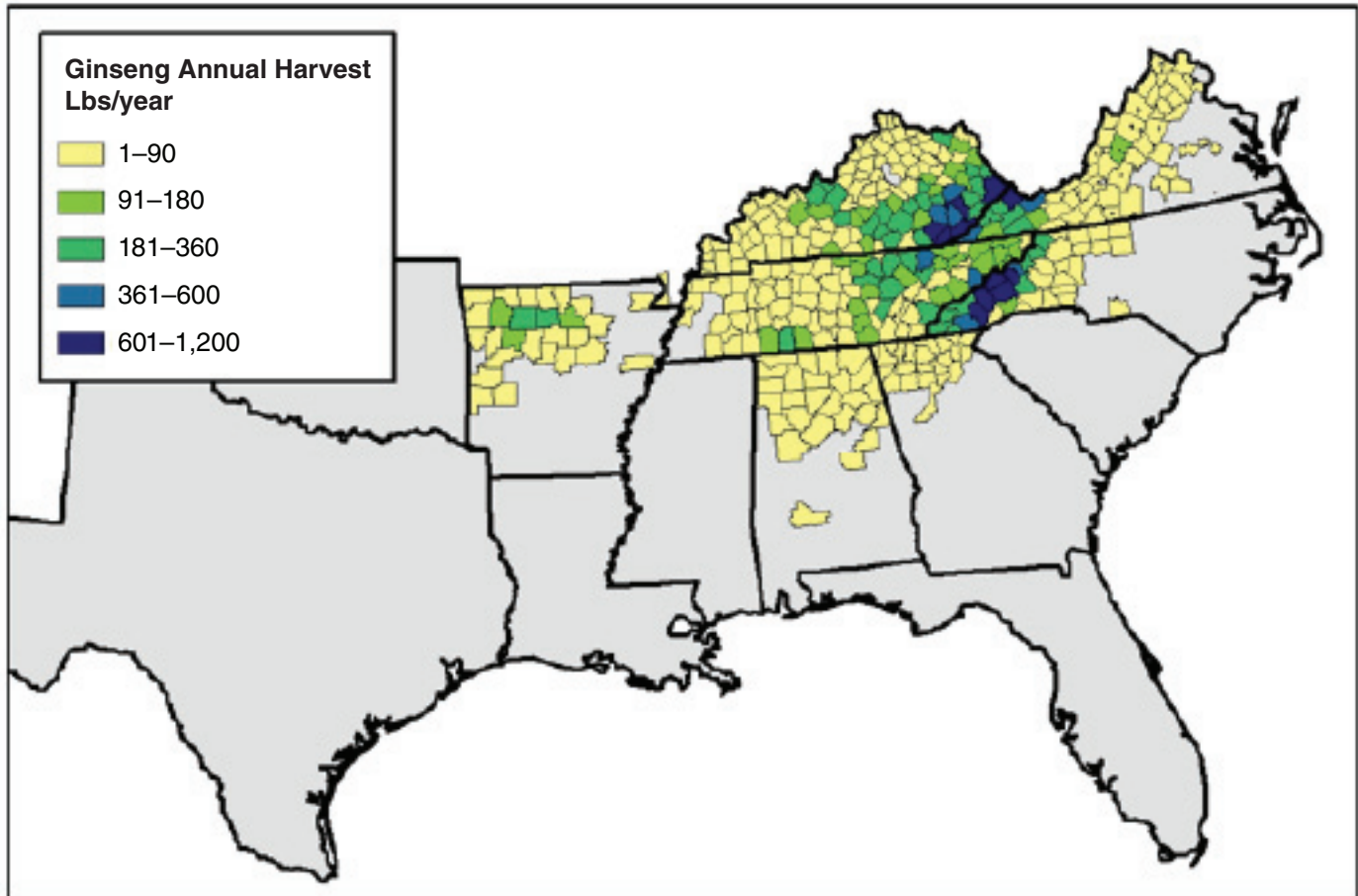


Figure 5.3—Concentration of American ginseng harvest for 2001–2007. Source: Chamberlain and others (2013).

Table 5.8—Pounds of permitted wild-harvested ginseng from National Forests, 2009–2013

State	National Forest	Pounds of permitted wild-harvesting ginseng					
		2009	2010	2011	2012	2013	Total
Georgia	Chattahoochee/Oconee	38	70	29	50	123	310
Kentucky	Daniel Boone	95	74	197	178	132	676
North Carolina	Pisgah/Nantahala	637	417	458	626	728	2,866
Tennessee	Cherokee	55	36	46	58	406	601
Total		825	597	730	912	1,389	4,453

Source: National Forest cut-and-sold reports.

Example: saw palmetto—Saw palmetto is endemic to coastal plains from South Carolina to southeastern Louisiana, including the Florida panhandle. It is found in every county in Florida (Smithsonian Marine Station at Fort Pierce 2015) and is considered a keystone species (Carrington and others 2000). Figure 5.4 shows saw palmetto growing in east Texas and north into Arkansas (Seiler and others 2015). Saw palmetto is a characteristic understory shrub in pine flatwoods, prairies, scrub, and live oak-sea oats communities (Duever 2011, Smithsonian Marine Station at Fort Pierce 2015). It is an indicator of poorly drained soils in pine flatwoods (Tanner and others 1996). The major center of harvesting is southwest Florida (Collier County), although harvesting also occurs in central Florida (Polk and Indian River Counties) and in southeast Georgia (Carrington and others 2000).

The berries of saw palmetto are harvested for their medicinal properties. The American Herbal Products Association includes saw palmetto in its biennial survey of the herbal industry and is the primary source of data regarding harvest volumes. Table 5.4 presents estimated quantity of harvest volumes (HVs) of saw palmetto for 1997 through 2010 as reported by Dentali and Zimmermann (2012). They argue that “under-reporting of saw palmetto berry harvests is a certainty” (Dentali and Zimmermann 2012) and a “reasonable estimate” is that harvest data capture only half of total dry weight. Estimates of harvest volume data are transformed to account for under-reporting (table 5.4). As harvesters are paid for fresh fruit, estimates of dried weight are multiplied by 3.3 (a standard industry conversion rate) to reflect fresh fruit HVs.

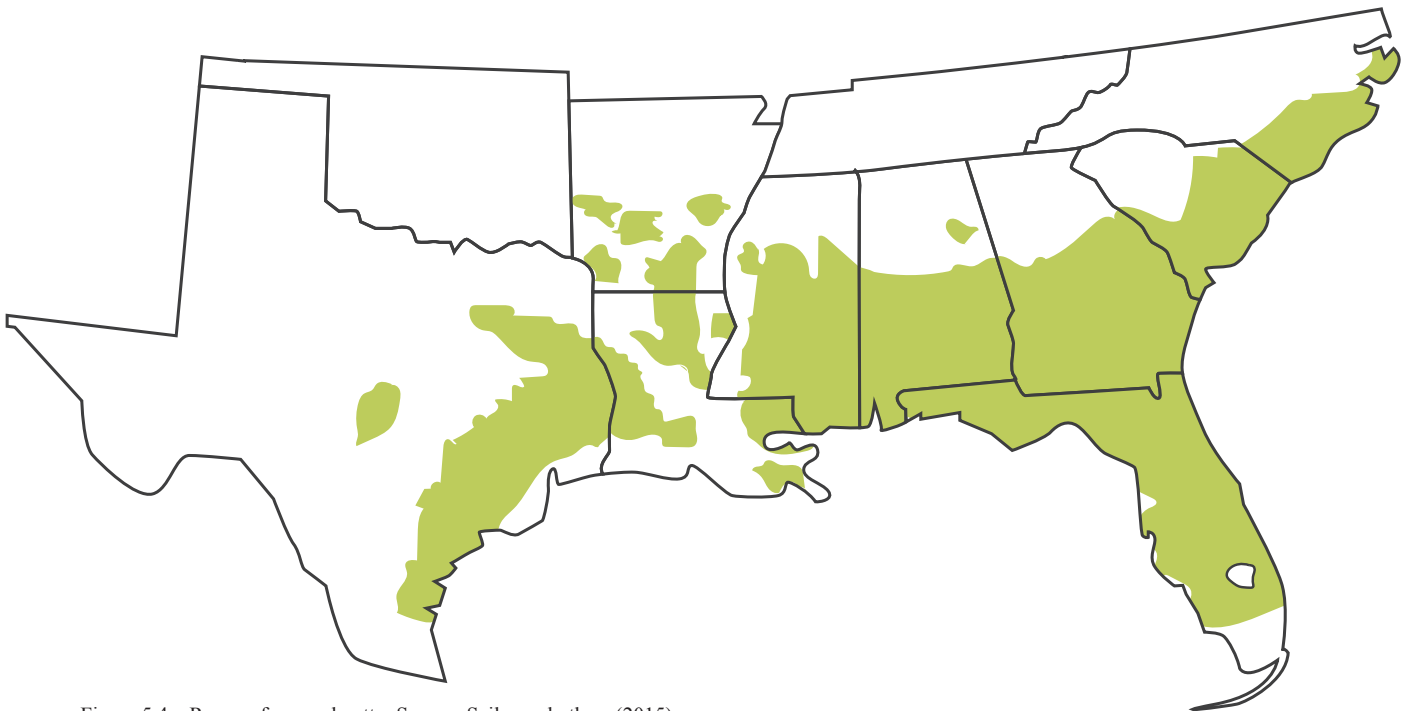


Figure 5.4—Range of saw palmetto. Source: Seiler and others (2015).

Determining per-acre marginal flow of saw palmetto depends on accurate estimates of acreage of the plant and yield per acre. Mitchell (2014) estimated the suitable habitat for saw palmetto in Florida at approximately 9,266,265 acres. We could use this estimate to calculate average harvest per acre if we assume that all available habitats are harvested. An inventory of actual harvest area is needed.

Estimating harvest volumes with no data sources—

Determining commercial HVs when there are no data available will require States to survey the industry. Surveying harvesters and primary buyers is probably the best way to get volume estimate at or near “farm gate.” Neither harvesters nor primary buyers are easy to identify as there are few if any directories of these market players. Unlike timber processors, there are no sampling frames available for non-timber product enterprises.

Significant thought and care should be put into determining the (human) population of interest and designing a sample frame and sampling technique based on the product(s) of interest. The sampling frame might then be stratified to account for variability in scale of operation, market participant type, sub-regions, and forest type. We present some guidance here on survey design and implementation, but an analyst considering a survey method should review the broader literature on surveys, including Dillman and others (2009).

Step 1: Identify sources of business information—Segmenting the NTFP industry by markets can help to focus development of sample frames. The segments discussed in “Quantification of NTFP Provisioning Services” provide a convenient and consistent framework. For medicinal forest products, the list of ginseng buyers provides a good starting place to craft a sample frame. Ongoing research with the USDA Forest Service and Virginia Tech substantiate that most ginseng buyers also buy other medicinal herbs. Dealers from around the State could be recruited to seek more information and could ask one or two simple questions to harvesters. Other sources of business information include trade associations (e.g., Christmas tree growers, florists and floral decorative, nurseries), U.S. Census Bureau (companies over a certain value are listed), State Chamber of Commerce, and the Internet. Also, some universities may have lists of businesses that have been included in previous studies.

Step 2: Determine what information is desired—Keep the requested information at a minimum and in an easy format to encourage maximum response rates. Surveys that are long and tedious typically do not get good response rates. Simple and straightforward surveys are much more productive.

Step 3: Create a sample frame—This can be a simple list of the market players (i.e., firms, enterprises, companies), with contact information, that you want to survey. This group of buyers can

help identify harvesters who bring products from the forest to them. Once the sample frame is completed, maintaining it is less costly.

Step 4: Determine the level of sampling needed to achieve representation of the population—In some cases, where the human population of interest is limited, it may be possible to undertake a census of all market players. In other cases, sampling the population will be necessary. For example, in 2002 there were 11 galax dealers in North Carolina and taking a census of these would be possible. Conversely, there are hundreds of ginseng primary buyers, and sampling the population may be more appropriate. Deciding on which approach to use will be easier once a sample frame has been created.

In some cases, the human population of interest is uncommon among the general population, and population members are unknown to the researchers. In these cases, one of a few methods for sampling rare populations may be used (see: Kalton and Anderson 1986, Wagner and Lee 2014). An example of these sampling methodologies includes “snowball sampling,” in which a few members of the population are identified by the researchers and asked to identify associates that are also within the population, a process which is continued until a relatively complete list of individuals is developed (Kalton and Anderson 1986, Wagner and Lee 2014). As an example, ginseng buyers may know other dealers who trade in non-ginseng botanicals, forming the basis for a snowball sample of botanical dealers. An indication that the process has “snowballed” sufficiently to cover the population of interest is when the same people are identified over and over again.

Step 5: Contact and evaluate non-respondents—Once the survey is implemented and responses have been received, efforts should be made to get data from non-respondents. Non-respondents could impact survey results if they represent a large portion of NTFP production. Examining this portion of the sample is critical to ensuring credible reporting. It allows the analyst to determine if the survey represents the population.

Example: pine straw—Pine straw markets operate at various levels of formality and scale. There may be a few large scale producers that collect needles from many properties in a year, as well as numerous smaller scale operations that collect pine straw on weekends or seasonally. Furthermore, there may be many private landowners who collect their own straw and sell to wholesalers or even retail the product directly. With pine straw, stratifying the sample of pine straw collectors into large scale, small scale, and landowners may make sense. Other stratification schemes may be more appropriate for another State. In the subset of large scale collectors, it likely makes sense to interview all the producers or as many as possible, since there are relatively few and each has a disproportionate impact on the market. Small scale producers and landowners might be identified by an expert panel

and using snowball sampling. Subsets of the population may be further stratified by other characteristics and a sample of each subset surveyed.

Some subsets of market participants may be extremely difficult to identify; for instance, landowners who collect straw themselves and retail directly. That portion of the market that is informal is difficult to quantify and track. State analysts may have to rely on an expert estimate of the informal market size or undertake a few select interviews with sellers that can be identified. Another option is to acknowledge that this segment of the market is insignificant and missed in the analysis.

Non-Market Harvest Volumes

Non-commercial NTFP uses include subsistence, recreation, cultural and spiritual uses, and education. While these uses are the most difficult to identify, quantify, and value, they may provide significant economic value to a State. Unfortunately, we know of very few data collected on non-commercial NTFP harvesting, and what is collected does not ask about quantities harvested and is limited in its reach to a single land ownership class. While this information alone is insufficient to estimate the quantity of NTFPs harvested, it could form the basis for a secondary survey.

One database that includes information on NTFP harvest is the National Visitor Use Monitoring (NVUM) Program, which collects data on recreational visitors to National Forests (English and others 2002). In summarizing the NVUM methodology, which is described in detail in English and others (2002), the USDA Forest Service (2013) states, “In essence, visitation is estimated through a combination of traffic counts and surveys of exiting visitors. Both are obtained from a random sample of locations and days distributed over an entire forest for a year.” These surveys ask whether part of the trip involved gathering of NTFPs, or “special forest products,” and whether this was the primary reason for the visit (Bowker and others 2009, USDA Forest Service 2013). It does not specify which NTFP was harvested.

Another source of NTFP harvest information is the National Woodland Owners’ Survey (NWOS), which collects data on the attitudes, perceptions, and management of family forests in the United States (Butler and others 2005, USDA Forest Service 2015b). The most recent iterations of NWOS include questions about whether NTFPs have been collected on these forests during the time they have been owned by the current landowners. It asks about market segment (edible, medicinal, etc.) and if the product was for sale or personal use. However, it does not ask about the quantity of product collected, so it would be impossible to use these data alone to estimate quantity of NTFP harvested on private lands.

Given the lack of publicly available data, if quantification of NTFP harvest for non-commercial purposes is desired, it is necessary to create and implement a survey. One approach would be to survey the general public in each State to determine how many people collect NTFPs, how much, and for what purpose it is used. For example, Maher and others (2013) used a survey of the general population in a region of Alaska to determine NTFP harvest quantities for 20 different NTFPs. Robbins and others (2008) undertook a phone-based random sample survey of residents of New England to examine personal use of NTFPs. An alternative approach would be to survey landowners, stratified by land ownership type and scale, forest type, and other characteristics.

If personal use, or “recreational” NTFP harvest is allowed via permits on public lands (National forest, State forest, etc.) in the State, it may be possible to access those permit records to contact harvesters. This approach was taken by Starbuck and others (2004) in Washington State. This would give a narrow sample and avoid the expense of surveying the general public, the majority of whom may collect no NTFPs at all.

VALUATION OF MARGINAL CHANGES IN NTFP PROVISIONING SERVICE

The value of anything is first identified as it fulfills the demands of humans. The majority of NTFPs fall within the three important categories of needs: physiological (food), safety (medicinal), and social (crafts, greenery, flowers) (Blatner and Alexander 1998, Greenfield and Davis 2003). When available, market prices are the most accurate measure of value at the margin—the value society places on one additional unit of a product or service. Unfortunately, NTFP markets may be informal with no data recorded, or NTFPs may not be traded at all. In this section, we discuss estimating the marginal value of products that are harvested for commercial and non-commercial uses.

Market Harvest Values

Assigning a price may be the most challenging part of valuation and it is best to use market price, if possible. It is usually necessary to have a basic understanding of the way the market for that product works, as the product changes hands from the harvester to the first point of sale, to various intermediaries and value-added processors, to the final consumer. To understand the intricacies of small, informal markets, studies frequently utilize interviews, written surveys, diaries, and face-to-face interactions with harvesters and other participants in NTFP activities (e.g., Alexander and others 2002b, Blatner and Alexander 1998, Carroll and others 2003, Davis and Persons 2014, Emery and others 2003, Greene and others 2000, Greenfield and Davis 2003, Jones and others 2004, Schlosser and Blatner 1995, Wolfe and others 2005). It is important to keep in mind that the results from these studies may be relevant only to the specific locations, products,

and time period, such that decisionmakers are cautioned not to apply valuations from one study to other areas or use them to aggregate values to a larger scale.

Prices (and harvest quantities) can vary greatly over time (Blatner and Alexander 1998, Schlosser and Blatner 1995) due to factors such as weather patterns, land use policies, supplies of competing food sources, and overall economic conditions in the region, creating uncertainty in what price to use. With an understanding of the market, sometimes one can track reasons for price and quantity variations. For example, in the Pacific Northwest, early and heavy snow decreases supply and increases price in floral greens. If prices are available there are a number of issues that need to be addressed. Prices will vary depending on a number of factors such as season, harvest volume, product quality, distance to processor or consumer, and locality. Many NTFPs are sold in small quantities, so it is hard to quantify and collect data across perhaps thousands of individual sellers. Therefore, prices for the same product in the same region can vary dramatically based on market conditions.

When possible, the ecosystem service value should use the price at the first point of sale after harvest; however, without a basic understanding of the market, this first point may not be clear. Many NTFP harvesters who sell their products have a point of sale near the forest and in the local area (Godoy and Lubowski 1992, Gram 2001). That first point of sale for the harvester is the primary buyer who may consolidate transactions and sell to secondary buyers, or may be a retailer (from a small roadside stand to a local brick-and-mortar store).

NTFPs such as ginseng and other medicinals, and florals for landscaping and nurseries, are part of formal markets, with dealers and distributors that move the products, with or without further value-added processing, along the supply chain. Gatherers of pine straw, bark, and other landscaping and nursery materials also work in formal markets. Local and regional landscaping and gardening nurseries and depots drive demand for much of the floral and greenery NTFPs. In addition to local and community gardening retail shops and self-employed landscaping businesses, harvesters sell generally to small dealers who, in turn, may transport wholesale quantities to regional businesses. Harvesters of these NTFPs likely realize a larger share of the overall value of the delivered product given that much of the intrinsic values of these products are reflected in the near natural state and form of the raw material itself.

The characteristics and behavior of market agents reflect the degree of structure in the market for the particular category of NTFP and the spatial scale of the market. In more formal markets, established agencies or organizations may track prices of particular NTFPs. A few data sources for NTFP prices are reported in appendix 5.2. For example, medicinal and herbal NTFPs such as black cohosh, bethroot, goldenseal, and bloodroot

are tracked by the American Herbal Products Association (AHPA 2007, Dentali and Zimmermann 2012). Past literature may also be useful; however, reported prices may not be up-to-date, or may not reflect local conditions.

If price data are not available, or are not adequate, from an existing data source, prices can be obtained through direct surveys of harvesters or buyers. Obtaining price data from harvesters or buyers could be costly in time and money across a broad swath of producers. Producers may be sensitive about revealing actual price information for competition and confidentiality reasons. Depending on the type of NTFP, having local dealers as a focal point to collect data may work to address the decentralized nature of the business. The section on “Estimating Harvest Volumes with No Data Sources” discusses some approaches for determining basic sampling and stratification approaches in the context of NTFPs. The same basic decisions apply here, as outlined in the decision tree in figure 5.5. These include snowball sampling; stratified sampling of harvesters, dealers, and/or landowners; or a survey or census of harvesters or dealers without stratification.

Example: American ginseng—In the American ginseng market, harvesters sell roots to primary buyers who sort, grade, consolidate, and market larger volumes to national and international buyers. Davis and Persons (2014) reported historical price ranges for American ginseng for 31 years (1982–2012). During the years 2000 through 2007, primary buyers paid, on average, wild-harvesters \$430 for a pound of dried American ginseng root (Chamberlain and others 2013b). Table 5.9 summarizes prices paid to harvesters for a pound of dried ginseng. There are other sources, including the State agencies with responsibilities for administering the ginseng program. These may provide only historical prices; current prices are best obtained directly by surveying dealers.

Example: saw palmetto—Estimating the value of saw palmetto requires having estimates of prices paid to harvesters. Carrington and others (2000) reported prices paid to “freelance” harvesters of \$0.10–\$0.11 (1997–98) per pound of fresh fruit. Mitchell (2014) recorded prices paid to harvesters ranging from \$0.10 to \$1.00, and notes that a price of \$0.50 per pound is average. Landowners, who hired crews to harvest, received \$0.20–\$0.21 per pound and paid harvesters about \$0.07 per pound of fresh fruit (Carrington and others 2000). To get up-to-date prices, State analysts need to visit saw palmetto buying stations during the harvest seasons and ask harvesters what they are being paid. Several visits are needed throughout the season to record variations.

Carrington and others (2000) also reported one value of particular note: the price *per acre* received by landowners from contractors who harvest the saw palmetto. In industries structured this way (pine straw may be another example where this type of per-acre contracting occurs), this per-acre value in a competitive market

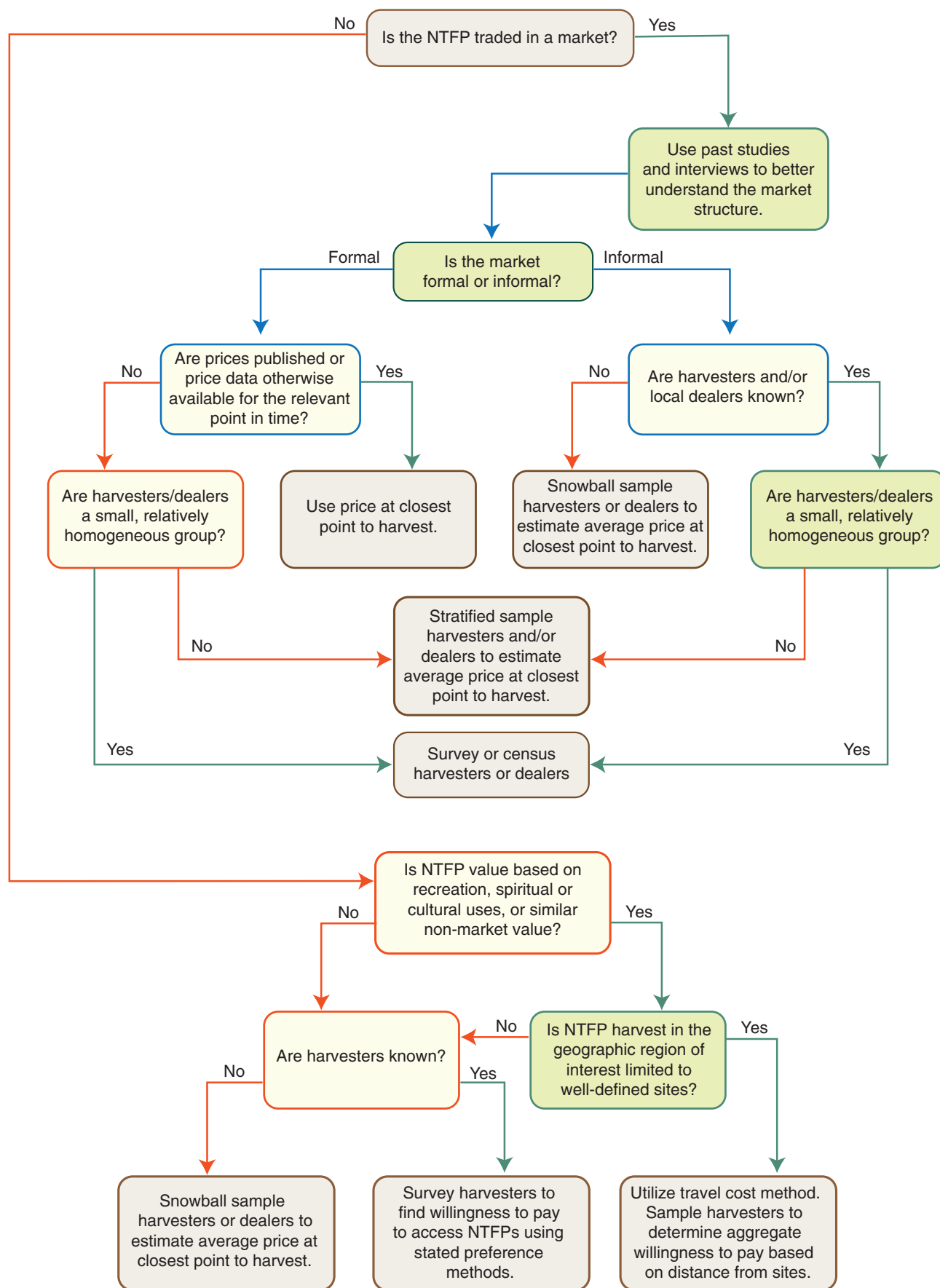


Figure 5.5—Decision tree for valuation of marginal changes in NTFP provisioning service.

Table 5.9—Wild American ginseng high and low yearly prices paid to harvesters

Price range	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
	<i>\$/dry pound</i>											
Low	250	300	250	250	300	400	250	350	350	320	400	800
High	500	400	500	550	600	1,150	1,000	600	1,100	750	1,250	850

Source: Davis and Persons (2014).

is essentially the ecosystem value as we have defined it, and may even obviate the need for collecting information on harvest quantity (see section on Conceptual Model). However, such per-acre contracting is not typical among NTFPs. Landowners who contracted for the harvesting of saw palmetto received about \$15.42-\$31.25/acre at 151-303 pounds per acre.

Non-Market Harvest Values

When traditional markets do not exist, valuing the provisioning services of non-commercial NTFPs poses difficulties. Non-market valuation techniques, as described in chapter 2, can be applied to NTFP harvesting opportunities. In many cases, NTFP harvest may be considered recreational in nature, and recreation valuation techniques can be applied (chapter 2). Even if the NTFP harvest is not considered “recreational” to the harvester (e.g., for subsistence, cultural, spiritual, educational, or other purpose), from a methodological standpoint, similar valuation techniques can be used. Contingent valuation and contingent behavior methods are the most popular examples of stated preference methods. Revealed preference methods for non-market valuation are based on actual behavior rather than stated intentions. Hedonic pricing uses data on real attributes to estimate how those attributes affect market prices. The travel cost method is the most commonly used revealed preference technique when valuing access to public lands for recreation activities.

Imputed values method—When the NTFP of interest has no market value, an alternative approach would be to use imputed values based on the market value of a good that is commonly bartered for the NTFP, or the market value of a close substitute for the NTFP (Godoy and others 1993). These would be good approaches if the analyst was confident that the estimated imputed or substitute prices were consistent enough across a broad human population that they could be relied upon. However, in most parts of the United States, bartering for lack of a market is not widespread enough to be consistently applied. Likewise, finding a true market substitute for a non-market product is unlikely. For example, one might propose that the value of store-bought (field cultivated) berries could be used as a substitute for non-commercial wild-harvested berries. However, if one were to ask the consumer, he or she likely places high value either on the harvesting itself (for cultural or recreational reasons) or on the fact that the berries are natural.

Hedonic pricing method—Similarly, hedonic pricing may have applicability for NTFP valuation in some rare cases, but the method does not seem promising. We are unaware of any studies that have used hedonic pricing for NTFP valuation. We do not want to dismiss the concept entirely, however. Hedonic pricing with NTFPs would rely on estimating the difference that people are willing to pay to access or purchase land that has NTFPs, compared to similar land that has equivalent characteristics but no NTFPs. Unfortunately, data on real estate sales do not include availability of NTFPs on the property, and the use of any proxy (e.g., forest type) for NTFP availability is just as likely to also be a proxy for many other characteristics other than NTFPs, all of which might impact price just as much or more than presence of NTFPs. One might consider a “market” for campsites or some similar access points, which have NTFPs or not, but it is hard to see how this could form the basis for a study or how the data could even be collected. Most campgrounds charge a flat rate regardless of the site; if sites with NTFPs are filled before other sites, this might be an indicator of some value, but trying to estimate that value in dollar terms is challenging in the least.

Stated preference methods—Stated preference methods could be used to value non-market NTFPs, although we are not aware of any studies in the United States that have done so. The analyst would use one of various approaches, described in more detail in chapter 2, to elicit people’s willingness to pay for a particular product, through a survey. In this case, a best metric would be to elicit a willingness to pay to access the product, assuming that the survey respondent would hypothetically have to harvest and process the product themselves. In this way, the respondent implicitly factors out harvest costs. Stated preference methods may be the most applicable for NTFPs when the population of harvesters can be identified. This would serve as the population for the survey.

Travel cost method (TCM)—If the harvest or collection of NTFPs is not traded commercially, and is mostly valued for recreational, spiritual, or cultural purposes, then the total quantity of product harvested may be less important to value than access to the resource. TCM can be used to estimate a value per trip or per visitor-day. Because the TCM is used to derive a demand curve for recreation, and if quantity of NTFP was calculated on a per trip basis, a dollar value per unit of NTFP or per acre can be calculated (Starbuck and others 2006).

To estimate demand for NTFP harvesting on southern forests using the TCM, a crucial assumption is made that the cost of travel to the site (including associated fees) is a proxy, or shadow price, for the harvesting activity. In the United States, TCM has been rarely used to value NTFP provisioning services, with two examples being Markstrom and Donnelly (1988) and Starbuck and others (2004). However, the methodology is not substantially different from how TCM would be used to value any recreational resource, and more guidance can be found in chapter 2. A comprehensive review of the development of recreation demand models and TCM can be found in Phaneuf and Smith (2005).

TCM involves surveying or observing recreational participants at one or more sites, with the goal of determining the cost of traveling to the site and the time spent at the site for each participant. These data are used to generate a demand curve, usually expressed in terms of numbers of visits at various levels of cost. For TCM to be effective, the analyst must be able to determine a distance from origin to destination to calculate travel costs. One approach to valuing NTFPs would be to use the ZIP code of origin to the forest centroid as the distance calculation upon which the travel cost is based. This allows TCM to be used to define a value per unit of harvest for a regional area (such as a forest area with a high density of NTFP harvesting). The inaccuracy introduced by using a forest centroid instead of a specific coordinate is well within the error of the estimate (Bowker and others 2009, English and others 2002).

While imperfect, TCM can yield important economic and demographic information about NTFP harvesting and provide a proxy for the value of the NTFPs associated with recreation. If a State has a number of forests that provide the NTFP in question, the analyst would want to survey visitors about which forest they visited and then aggregate across all the forests of similar type to estimate a total State value. For any non-market valuation, one must survey visitors and obtain information on behavior to estimate demand and hence monetary value. TCM can be applied to any spatial scale so long as the origin-destination information is obtained and there is sufficient information to calculate a cost per trip or per day value.

The sampling frame and visitor characteristics need to be carefully considered. When a survey is designed, map information that shows trail and parking access points must be used to identify where to “trap” the most representative sample. Questions regarding demographics and activities can then be compared to other sampling or population information to generate an estimate of the sample characteristics relative to the population. Numerous estimation methods exist to account for endogenous stratification and other sampling issues known to be exist in recreation data. With a well-designed sampling frame and survey protocol issues related to the reliability of the estimate resulting from sampling issues can be minimized and does not significantly increase the cost of generating a non-market value

for an NTFP. For TCM to be effective the survey must ask information about the purpose of the trip and what portion of trip time and expenditures are related to each listed activity. This is easily done by a well-designed survey instrument and does not pose a significant issue in the application of TCM.

The analyst must consider the number and distribution of harvest sites of the particular NTFP throughout the State, and whether only one site or multiple sites are of interest. In general, single-site analysis would be relevant for a narrow policy question involving only that site (for example, a change in a National Forest management plan) or in the rare case where only one harvest site exists. Multiple-site analysis would be appropriate for a statewide valuation if the NTFP is restricted to a relatively few, relatively well-known sites, such as National or State forests. Bowker and others (2009) constructed a multiple-site TCM using NVUM data from 120 National Forests, which included NTFP gathering, although it did not estimate a value for NTFPs independently. The complexity of harvesting location and site definition for a multi-site TCM are not insurmountable, and again relate to the development of the sampling frame and survey implementation. Bowker and others (2009) developed a rigorous sampling framing and survey method, and the NVUM results could have easily been extended to NTFP if a question on NTFP harvesting had appeared as a list of recreational activities.

In summary, for TCM to be effective, one must collect origin-destination information and demographic information. In a well-executed TCM, it is important to ask about the primary purpose of the trips and the types of activities and/or percentage of trip for each purpose in order to handle issues of substitution that can affect valuation (Freeman 2003, p. 424-425). These considerations provide initial insight and guidance about using the travel cost method to value NTFP provisioning. While it is has been infrequently used for NTFPs, the TCM may offer a way to value NTFPs that are harvested recreationally.

ESTIMATION OF PRODUCTION COSTS

Provisioning ecosystem service can be interpreted to mean the portion of value of a product that is generated by the ecosystem. In this case, the portion of value that is generated by humans, that is the human-incurred production costs, should be factored out (equation 1), leaving a “residual value” that can be attributed to ecosystem function. Human-incurred production costs include costs of establishment, management, and harvest. In the case of wild-harvested NTFPs, there are no costs of establishing or managing these populations. Ginseng harvesters are supposed to plant seeds that are found on plants that they harvest. In this case, the “establishment cost” would be the time it takes for the harvester to plant seed, which likely would be negligible in this case, and could be simply considered part of the harvest cost. Harvesters of other products have fewer restrictions. This section focuses on measuring costs incurred in production of NTFPs.

To estimate the cost of harvesting an NTFP, an analyst would need to interview or survey the harvesters. The costs of harvesting include fuel, any equipment needed to harvest, any food or lodging required (most harvesters in Southern United States probably do not overnight anywhere during harvest trips), and the harvesters' labor.

In many cases, labor is the principal cost of NTFP production and is not paid in terms of a wage. The value of this labor can be estimated as the "opportunity cost" of some alternative economic activity. In general, if the harvest activity is relatively low-skill, the average hourly wage for low-skill labor may be an appropriate value to put on this opportunity cost. This is usually somewhat higher than minimum wage (but could be lower than minimum wage if there is a larger informal labor market in the area). Certain specific types of NTFP harvest activities might correspond more closely with higher-skilled employment. The U.S. Bureau of Labor Statistics provides data on average wages by occupation and area, which can be used to find an appropriate opportunity cost of time for the typical NTFP harvesters in your area (U.S. Bureau of Labor Statistics 2016).

For certain products, there might be numerous harvest methods with varying levels of mechanization. For example, pine straw can be harvested with a rake and a wooden manual baler, or with a tractor and mechanical baler. For the former, equipment costs are minimal and labor might be the most important input, so using an appropriate opportunity cost of time is important. For the latter, harvest would also include such expenses as equipment depreciation, fuel, etc.

Determining production costs of NTFPs harvested from "managed" populations of NTFPs will require "utilization" studies, similar to what the USDA Forest Service FIA program does for timber. In these types of studies, the analysts actually spend time monitoring all aspects of establishment and management of the product. Care must be taken not to double count costs when estimating production costs for NTFPs that might be produced with other products (e.g., pine straw with timber and cattle). Also, the amount of labor required should be recorded and stratified by different tasks (e.g., site preparation, maintenance, harvesting, transport) along with equipment and materials costs. To make these studies relevant, they need to be replicated over time and locations so that generalizations can be made.

Some NTFPs are being grown in a "forest farming" system that have associated establishment and management costs. These costs include labor to undertake site preparation, planting, and maintenance. The costs of seed, equipment, pesticides, and other

inputs can be estimated directly by monitoring forest farming activities. A few studies have created "forest farming budgets" for selected NTFPs (e.g., Burkhart and Jacobson 2009, Davis and Persons 2014); however, unlike typical crop budgets published by State Cooperative Extension services, these sources are not updated periodically or replicated in different geographic areas.

Initially, efforts to estimate the value of natural forest ecosystems to provide NTFPs can disregard production costs. There is little evidence that most NTFP harvesting from natural populations is being managed, nor are there associated establishment costs. As forest management agencies begin to actually manage for these products, then associated costs will occur and can be estimated.

SCALING UP/AGGREGATING

The guidance here has been written with a State-level valuation in mind. We assume that the desire is in valuing NTFPs across the entire State by forest type and landowners. Many of the recommendations may be valuable for other spatial scales. To be sure, local (e.g., National Forest or district) level valuations provide valuable information for managers of those areas, but they often cannot be scaled up because of differences within the population of harvesters. For example, if a small number of individuals have an extra-large impact on the market, a simple random sample at low sampling intensity may not capture the value accurately. A small number of large harvesters can skew results of a valuation exercise. It is, therefore, critical that surveys of harvesters get a representative sample of all harvesters.

A State-level valuation should strive to account for variations in harvester populations by appropriately sampling from different operational scales, forest types, ownership classes, and/or management/regulation regimes. This is called stratified sampling, which strives to sample across the variety of players. After an initial survey of the populations, efforts should be made to make sure that each strata is represented. Also, efforts should be made to get responses from "non-respondents."

Results of small studies can be "scaled up" or aggregated to represent State-level spatial dynamics with effort, thought, and consideration. Small scale "pilot" studies allow the researcher or analyst to better understand factors that may inform larger studies of challenges that need to be addressed. Undertaking small studies across landscapes can provide valuable information on per-acre production, growth, and yields and operational costs. To scale these to State-level analyses requires that they capture variations across forest types and landowner classification. To aggregate small studies to the State level, they must represent all spatial and temporal variations.

CONCLUSIONS

Non-timber forest products contribute to the economies of every State in the South. They are important provisioning services of all forest ecosystems of the region. Yet, they have not been included in efforts to value forests. People collect these products for personal consumption and to generate income to support families. Different approaches are used to estimate market and non-market values of these products. Estimating market values is simply the product of quantity harvested (or standing stock) and market price (farm gate) as a proportion of habitat. Non-market estimation is more complicated and requires interviewing and surveying harvesters and translating results into comparable units of measure.

The major challenge with estimating the provision service of ecosystems for NTFPs is the lack of data. In general, there are insufficient data to estimate the provisioning of NTFPs from forest ecosystems. There are volume data for a few NTFPs, such as American ginseng, although this is the only NTFP for which county-level data is available. Price data exists for ginseng as well, but not for most other products. Determining standing stock of NTFPs will require inventorying plants within the forests, similar to what is done with trees. Determining the flow, or annual harvest rate, will require utilization surveys of market players.

Designing and implementing surveys requires thought and consideration for the designed end-use. The first task will be to develop a sample frame that represents various market segments (for market valued NTFPs) and personal use harvesters. These will take significant time and resources, but once these are completed, maintaining them is less costly. There are standard and well-accepted methods for designing and undertaking surveys (Dillman and others 2009).

The initial investment to estimate the provisioning of NTFPs by forest ecosystems may be significant for any one State. By collaborating on valuing specific priority (high valued) products, States can reduce their investment requirements. Further, after the initial investment has been made, replicating and refining the estimating efforts will be significantly reduced. All evidence indicates that NTFPs are economically and ecologically significant and worthy of valuing. The total ecosystem provisioning value of forests will remain elusive until NTFPs are included.

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Appendix 5.1

Summary of select valuation studies of NTFPs in the U.S. South and around the world

Citation	Geographic location	Stock or flow quantification	Market or non-market valuation	Marginal value estimates	Other methodological notes
Adger and others (1995)	Mexico			Multiple NTFPs \$330 per hectare per year Coffee and other NTFPs \$1,540 per hectare per year	
Alcorn (1989)	Mexico				
Alexander and others (2002a)	Pacific Northwest	flow	market	Matsutake, chanterelle, and morel mushrooms \$0.4-99 per hectare per year	Describes soil expectation value approach similar to that in section 1.2 here. Describes assumption and methodological choices. Variation in values due to differences in species, location, and methodological assumptions.
Chamberlain, Prisley, and others (2013)	U.S. South	flow	market	Not applicable	Calculates average annual harvest and revenue (2000-2007) for ginseng in the United States.
De Groot, Wilson, and Boumans (2002)	Worldwide	flow	market and non-market	Not applicable	Synthesis of meta-analysis which summarizes the relationships between ecosystem functions and monetary valuation techniques
Godoy and Feaw (1989)	Indonesia			Rattan \$495 per hectare per year	
Gram (2001)	Peru	flow	market and non-market	Fishing, hunting, gathering \$9 - \$17 per hectare per year	Analysis of strengths and weaknesses of valuation methods for economic importance of local extraction activities
Grimes and others (1994)	Ecuador	flow	market	Fruit, medicinal and handicrafts NPV \$1,257 - \$2,939 per hectare per year	Estimates sustainable harvest levels and net present value of fruit, medicinal and handicraft products on three plots
Greenfield and Davis (2003)	North Carolina	flow	market	Not applicable	Looks at 46 forest botanical products (medicinal, edible plants, floral plants, ornamentals) in 25 western counties, using an exploratory, inductive approach estimate volumes, sales, economic and socio-economic information and prices of NTFPs.

(continued to next page)

Appendix 5.1 continued

Summary of select valuation studies of NTFPs in the U.S. South and around the world, cont.

Citation	Geographic location	Stock or flow quantification	Market or non-market valuation	Marginal value estimates	Other methodological notes
Godoy and Lubowski (1992)	Sri Lanka Brazil Peru	flow	market and non-market	Florals \$50-\$420 per hectare per year Babassu palm products \$59 per hectare per year Wild camu \$167 per hectare per year	Summarizes net economic valuation studies in multiple countries
Paoli and others (2001)	Indonesia	stock and flow	market	Gaharu wood \$3.80 - \$18.56 per hectare Average of \$10.83 per hectare	Estimates net present value per hectare for five major forest types
Peters, Gentry, and Mendelsohn (1989)	Ecuador			Fruit and latex \$6,820 per hectare per year	
Robles-Diaz-de-Leon and Kangas (1998)	Chesapeake Bay Watershed, Maryland	stock and flow	non-market	Ornamentals, edible fruits, edible nuts \$60,694 per hectare per year	Calculates total gross income obtained from a riparian forest buffer
Starbuck and others (2004)	Gifford Pinchot National Forest, Washington	flow	non-market	Not applicable	Estimates recreation demand (consumer surplus) for NTFP gathering
Tewari (2000)		stock and flow	market and non-market		Explains the variation dynamic and static, income and wealth valuation models of NTFPs

Appendix 5.2

FIA Forest Groups and Associated Tree Species

Forest Type Group/Species	Forest Type Group/Species	Forest Type Group/Species
White / red / jack pine group Jack pine Red pine Eastern white pine Eastern white pine / eastern hemlock Eastern hemlock	Oak / pine group Eastern white pine / northern red oak / white ash Eastern redcedar / hardwood Longleaf pine / oak Shortleaf pine / oak Virginia pine / southern red oak Loblolly pine / hardwood Slash pine / hardwood Other pine / hardwood	Elm / ash / cottonwood group Black ash / American elm / red maple River birch / sycamore Cottonwood Willow Sycamore / pecan / American elm Sugarberry / hackberry / elm / green ash Silver maple / American elm Red maple / lowland Cottonwood / willow Oregon ash
Spruce / fir group Balsam fir White spruce Red spruce Red spruce / balsam fir Black spruce Tamarack Northern white-cedar Fraser fir Red spruce / fraser fir	Oak / hickory group Post oak / blackjack oak Chestnut oak White oak / red oak / hickory White oak Northern red oak Yellow-poplar / white oak / northern red oak Sassafras / persimmon Sweetgum / yellow-poplar Bur oak Scarlet oak Yellow-poplar Black walnut Black locust Southern scrub oak Chestnut oak / black oak / scarlet oak Cherry / white ash / yellow-poplar Elm / ash / black locust Red maple / oak Mixed upland hardwoods	Maple / beech / birch group Sugar maple / beech / yellow birch Black cherry Hard maple / basswood Red maple / upland
Longleaf / slash pine group Longleaf pine Slash pine	Aspen / birch group Aspen Paper birch Gray birch Balsam poplar Pin cherry	Tropical hardwoods group Sable palm Mangrove Other tropical
Tropical softwoods group Tropical pines	Oak / gum / cypress group Swamp chestnut oak / cherrybark oak Sweetgum / Nuttall oak / willow oak Overcup oak / water hickory Atlantic white-cedar Baldcypress / water tupelo Sweetbay / swamp tupelo / red maple Baldcypress / pondcypress	Exotic hardwoods group Paulownia Melaleuca Eucalyptus Other exotic
Loblolly / shortleaf pine group Loblolly pine Shortleaf pine Virginia pine Sand pine Table Mountain pine Pond pine Pitch pine Spruce pine		
Other eastern softwoods group Eastern redcedar Florida softwoods		
Exotic softwoods group Scotch pine Other exotic softwoods Norway spruce Introduced larch		

Appendix 5.3

SUMMARY OF DATA SOURCES

Data sources for non-timber forest products in the U.S. South are varied and disparate

Source	Data type	Notes
Interviews/surveys of harvesters and dealers	price, volume, acreage	
USDA Forest Service cut and sold reports	volume	includes value of permit fees
USDI Geological Survey models	acreage, geographic distribution	species habitat
Interactive Tariff and Trade DataWeb (U.S. International Trade Commission 2016)	volume	export data
Greenfield and Davis (2003)	price	North Carolina ginseng, goldenseal, galax, ramps
Davis and Persons (2014)	price	historic ginseng prices
U.S. Fish and Wildlife Service	volume, geographic distribution	ginseng, county and State
American Herbal Products Association	volume	17 Appalachian medicinal products
National Woodland Owners survey	acreage	By number of owners, type of sale, type of product
USDA Forest Inventory and Analysis	acreage, geographic distribution	forest type, bark
State Natural Heritage databases	geographic distribution	—
Wholesalers	price, volume	various NTFPs
Ginseng dealers	price, volume	by State
National Christmas Tree Association	price, volume, acreage	by total U.S., State
State agriculture departments	price, volume	pine straw
Farmers markets	price, volume	edibles
USFS and State forestry agencies	volume	also permit fees
Maple Syrup Association	price, volume, acreage	and other saps
National Agriculture Statistics Service	price, volume	maple syrup, blueberries
Green nursery/landscaping industries/associations	price, volume	forest florals, cones
Georgia Farm Gate Value Reports	price, volume	includes pine straw
Mitchell (2014)	price, volume, acreage	saw palmetto
Chamberlain and others (2013a)	price, acreage	black cohosh
Chamberlain and others (2013b)	price, volume	ginseng

For the purposes of this report, and for future analysis by State foresters, the above list provides a starting point of some specific research studies and general resources with regular collection and compilation of data on NTFPs.

Chapter 6

Conclusion

Erin O. Sills, R. David Simpson, D. Evan Mercer

This concluding chapter recommends a standardized approach to accounting for forest ecosystem services in the Southern States. First, we synthesize 10 principles from the preceding chapters. Next, we present a template for State forest ecosystem service assessments, recommending a staged approach with five outputs.

PRINCIPLES OF ECONOMIC ACCOUNTING FOR ECOSYSTEM SERVICES

1. **Accounting involves multiplying *total* quantities by *marginal* values.** Accounting requires understanding total annual flows of ecosystem services from forests; for example, the total volume of water that flows through a forest ecosystem, the sediment load in water exiting those ecosystems, or the net amount of carbon sequestered or released. Other examples include the total number of recreation user days or total volume of medicinal plants harvested. Weighting each of these physical flows by its marginal value provides an economically meaningful measure of the value of forest ecosystem services.
2. **Not all units of an ecosystem service have the same marginal value.** Just like a market price, the marginal value of an ecosystem service reflects its supply and demand in a particular time and place. Thus, the most useful measures of the value of ecosystem services are based on spatially disaggregated estimates of changes in the annual flows of those services multiplied by marginal values that reflect the level of supply (e.g., extent of forest ecosystem) and demand (e.g., number of people who can “consume” the ecosystem service) in different locations.
3. **Marginal values depend crucially on context.** Economic value reflects scarcity. When the supply of a good or service is limited, it is economically valuable. Goods and services with abundant supply are less valuable. Scarcity is relative, as it is determined by the relationship between the supply and the demand for a good or service at a particular time and place. Thus, values calculated for one time and place may be very different for another. A spatial catalog of values can serve for both tracking changes over time and for cost-benefit analysis of alternative policies.
4. **When ecosystem services are public goods, their value depends on the size of the “public.”** Demand—or willingness to pay—is, in general, greater the larger and the wealthier the affected population. Thus, in addition to quantifying the flows of ecosystem services in particular times and places, the analyst must consider the size of the population that benefits from those services. The relevant “public” can range from immediate downstream water users to the global population affected by climate change. State assessments should identify both the relevant “public” who reside in the State, and the total “public” of all people who benefit from ecosystem services generated by forests in the State no matter where they reside.
5. **Diminishing returns are critical and inescapable.** The other side of the valuation coin concerns the abundance of supply. Economic value is determined on the margin: the economic value afforded by an acre of forest is determined not by the services the forest as a whole provides, but rather, by what *additional* services that acre provides over and above those that would have been available in its absence (i.e., the forest’s marginal product). With rare exceptions (e.g., where additional acres provide habitat contiguity), the value of the ecosystem services provided by each additional unit of forest diminishes as the total area of forest increases, and vice-versa: i.e., an additional unit of forest is more valuable where there is less total forest area. Analysts should only use valuation methods that acknowledge diminishing returns, and they should caution against inappropriate use of their estimated marginal values. For example, they should discourage multiplying the total acres of forest in a State by the marginal value of ecosystem services from one additional acre of forest.
6. **Only *final* services should be included to avoid double-counting.** It is inappropriate to sum the value of an ecosystem service that contributes to some final outcome and the value of that outcome itself. For example, we do not need to estimate the value of the soil structure created by forests if we have an estimate of the value of the resulting stabilization of water flows, because the value of the soil structure as an input into the hydrological system is already reflected in the value of that watershed service. Likewise, accounts should not include both the value of a forest in filtering water so that expensive purification is not required and the cost savings from not having to purify water that has been cleansed by a forest. A corollary is that in order to fully account for forest values, we need estimates of the values of all final ecosystem services now and into the future.

7. **Some—but not all—final ecosystem services are reflected in traditional economic accounts.** The value of the inputs used to produce marketed goods and services is reflected in the value of those goods and services as recorded in GDP (Gross Domestic Product). Because inputs include forest ecosystem services, we can say that some part of GDP is attributable to those services. For example, the total value of agricultural production reflects the values of hydrological and pollination services provided by forests to the agricultural sector. In this case, accounting for forest ecosystem services means giving them credit for their contribution to GDP. Other ecosystem services affect well-being directly in ways that are not counted in GDP, e.g., by providing a more pleasant environment and creating a sense of place. Just as the value of home-cooked meals (and therefore the contribution of home cooks to our well-being) is excluded from GDP, so are the psychological benefits of the forest landscape (and therefore the contribution of that cultural ecosystem service from forests).
8. **Economic accounts sum up changes experienced by everyone in the accounting framework, e.g., everyone in the State.** A positive sum indicates the possibility that everyone could gain. For example, an increase in the total value of forest ecosystem services in a State means that there is a positive gain in value that *could* be distributed among everyone in the State. However, gains (and losses) are almost never distributed evenly, and concentrated gains or losses are often the most visible and most discussed. Both accounting totals and the distribution of gains and losses are relevant to forest policy decisions.
9. **Accounting for ecosystem services necessitates thinking broadly ...** Ecosystems give rise to economic values that span a broad spectrum. At one end are those that accrue largely to private individuals in consequence of private choices, such as recreational use of forests or collection of non-timber forest products. At the other end are values that may benefit people around the world, such as carbon sequestration. Some global values, for example the value of preserving endangered species, may not be associated with any market transactions. In between are services provided to local communities, such as flood protection and groundwater re-charge. Information from multiple sources is required to estimate quantities and values for the various types of services. Even in State studies focusing on values that accrue to State residents, values to populations outside the State should be noted, as they are likely to be relevant to the design of ecosystem service markets and national policies that affect the State.
10. **... but also prioritizing.** The ultimate goal of ecosystem accounting exercises is to express all ecosystem service values in dollar terms. This is a long-term aspiration that is not fully achievable in the near future. Meanwhile, State assessments can:
 - a) identify all relevant forest ecosystem services (and disservices),
 - b) note those likely to be most affected by changes in forest area,
 - c) compile estimates of the marginal value of those services from comparable prior studies, and
 - d) call for additional research on services that lack marginal values but that are likely to be both valuable and vulnerable to changes in forest ecosystems.

In this way, State assessments can focus future valuation efforts on ecosystem services with values that are likely to be substantial but that have not been estimated using theoretically and methodologically sound approaches. They should steer researchers away from “deriving more precise estimates of zero,” i.e., employing the best methods and data to estimate values that are too small to affect decisions about forests. While this future research is pending, State assessments should focus on quantifying annual flows of these ecosystem services in physical terms, rather than heroically extrapolating values from other studies in ways that ultimately cannot be defended. An adage worth remembering here is that “absence of evidence is not evidence of absence.” The forest ecosystem services that have been valued in monetary terms are just a subset of all ecosystem services relevant to decisions about forest policy and management.

RECOMMENDED STRUCTURE FOR ASSESSMENT OF FOREST ECOSYSTEM SERVICES

These 10 principles imply that economic accounting of ecosystem services is not a single well-defined task, but rather requires tapping numerous data sources, making numerous judgment calls, and tailoring the approach to numerous different ecosystem services. As a result, we recommend that States undertake ecosystem services assessments in stages (fig. 6.1), allowing for adaptive management of the process and generating several kinds of outputs.

State assessments of forest ecosystem services should, at a minimum, generate the following five outputs:

1. **A statement** listing the ecosystem services generated by forests in the State,
 - a) highlighting priorities for future valuation studies (ecosystem services that are expected to be important but that have not been valued),
 - b) identifying services that can be included in accounts by transferring values from prior literature, and
 - c) acknowledging other forest ecosystem services with currently unknown values.

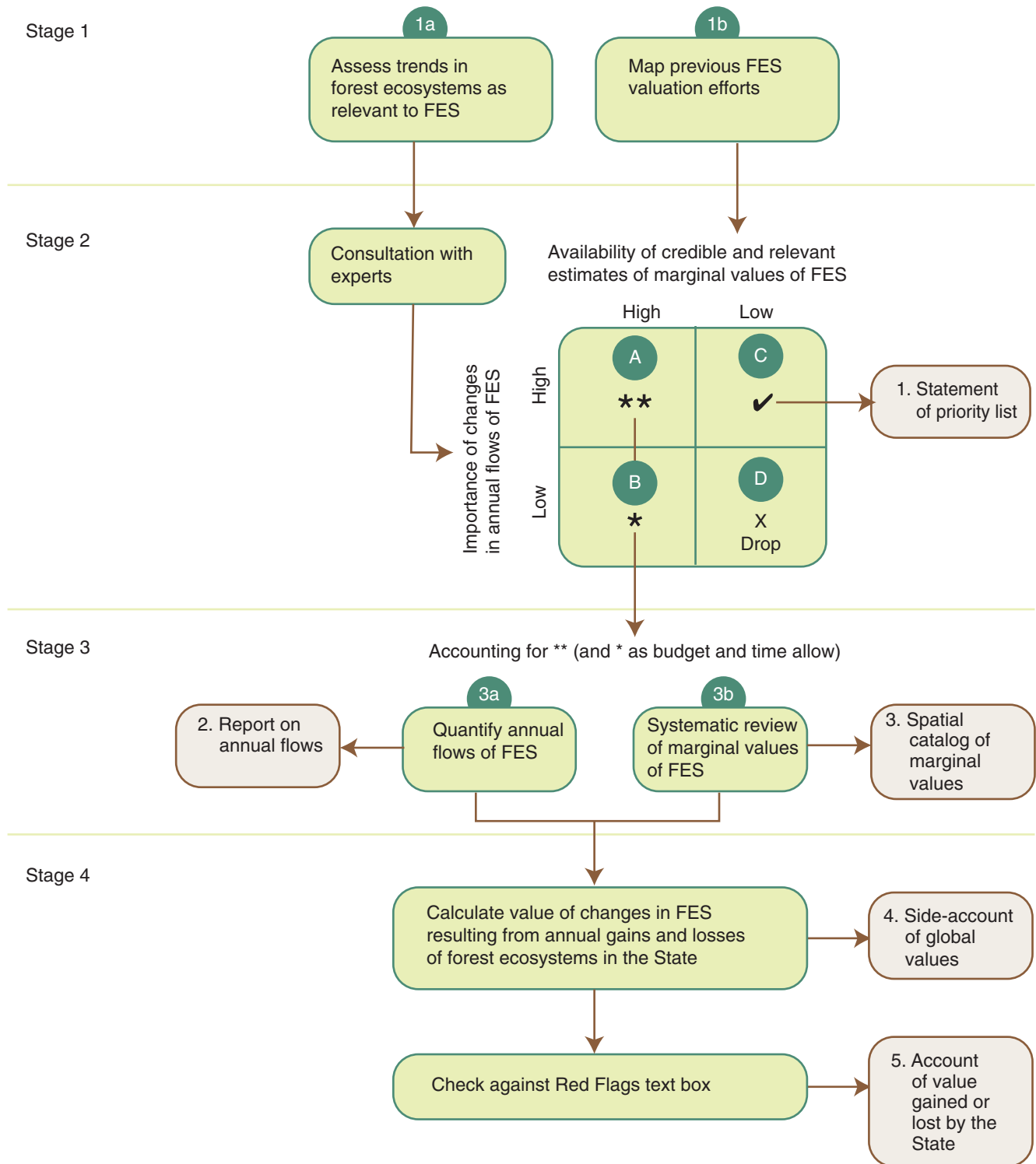


Figure 6.1—Template for Forest Ecosystem Service Assessment.

2. **A report** on the annual flows of tangible forest ecosystem services per acre and in total from each forest type in each spatial zone in the State.
3. **A spatial catalog** of the marginal values of forest ecosystem services, per unit of final ecosystem service or per unit of forest, for each forest type in each spatial zone.
4. **A side account** of national and global values (to people residing outside the State) for ecosystem services gained or lost due to changes in forest ecosystems in the State from year to year.
5. **An account** of the value to State residents of ecosystem services gained or lost due to changes in forest ecosystems in the State from year to year.

The first output, produced from scoping studies and consultations in the early stages of the assessment, should provide a complete list of ecosystem services generated by forests in the State classified on 1) the likely importance of changes in the service due to loss or gain of forest ecosystems and 2) the availability of credible and relevant estimates of the marginal value of the service.

Services of high importance but lacking prior estimates should be included in a priority list for future research to inform State funding decisions and encourage other funders and researchers to focus valuation efforts on priority forest ecosystem services. Ecosystem services that have high importance and high availability of prior estimates should be evaluated in the remaining stages of the assessment, using benefit transfer to develop accounts and analyze policy alternatives. Depending on budget, time, and State priorities, ecosystem services with relatively low importance but high availability of prior estimates could also be included in the remaining stages of the assessment.

The second and third outputs should be developed in tandem, using consistent definitions and measurements of final ecosystem services to quantify their marginal production by forests and their marginal value to State residents. For ecosystem services that are not tangible and not associated with any observable behavior (such as many cultural services), the most appropriate measure may be area of the forest ecosystem. However, in most cases, the area of forest ecosystem should be considered as an input to production of a final ecosystem service.

Specifically, the second output should quantify annual flows of final forest ecosystem services in physical units appropriate to each service, generating both spatially differentiated estimates of the marginal product of forests, and “headline” numbers such as the proportion of a State’s drinking water that flows through forest ecosystems. The third output compiles estimates of the marginal value of those same final forest ecosystem services from previous studies. The resulting spatial catalog of marginal values is the key tool for developing accounts of the economic

gains and losses due to changes in forest area over time and due to policy alternatives.

Finally, the fourth and fifth outputs are the typical outputs of green accounting exercises, depending on the accounting framework adopted. These accounts serve the purpose of monitoring, managing, and reporting the change in value of ecosystem services due to annual gains or losses in forest ecosystems in the State. The focus of State assessments is typically on the value of forest ecosystems to residents of the State. However, some ecosystem services from State forests also benefit people outside the State, at scales ranging from the population residing in watersheds fed by a State’s forest to the global population that benefits from mitigation of climate change through carbon sequestration in the State’s forests. We recommend tracking these values in a side account, recognizing that they can be only partially captured by State residents through ecosystem service markets or national policy.

To generate these outputs, assessments should proceed through the four stages illustrated in figure 6.1: 1) scoping studies, 2) consultation to set parameters, 3) in-depth studies, and finally 4) economic accounting of the value associated with changes in forest ecosystems in the State.

Stage 1

The first stage comprises two scoping studies to inform decisions about which ecosystem services to consider in the five outputs. Based on meetings with stakeholders from the Southern States, we recommend that at minimum, all assessments should include the following three categories of services:

- a) watershed services (regulation of flow and maintenance of quality),
- b) recreation (including tourism and hunting), and
- c) supply of forest products with the greatest market values.

Because the market value of timber is already well understood and incorporated into policy decisions in the Southern States, the added value of an ecosystem service assessment is to account for non-timber values generated by forests, including non-timber products and services. The three categories listed above have been identified as the most important regulating, cultural, and provisioning services of southern forests and should be included in any assessment.

Of course, these are just a subset of all forest ecosystem services, and the scoping studies are intended to identify other services that should be included in particular State assessments, such as other cultural services and regulation of air quality and climate. Specifically, scoping study (1a) identifies important changes in forest ecosystem services by quantifying changes in the area

of forest ecosystems over the past decade (or other recent time period), and scoping study (1b) surveys the literature on the economic value of ecosystem services, generating a “systematic map” of forest ecosystem service valuation.

For scoping study (1a), we recommend first identifying forest zones that 1) have the biggest effect on watershed functioning (see Chapter 3) and 2) are likely to support the most recreation, including hunting (see Chapter 2). Second, we recommend quantifying change in forest area over the past decade in the State as a whole and in those high value zones (for water and for recreation). Note that these zones are likely to overlap (see fig. 6.2). Change in forest area could be quantified as conversion into or out of FIA forest types (planted pine, natural pine, oak-pine, upland hardwood, lowland hardwood). Based on these changes in forest area in the State as a whole and in the high value zones, the analyst can identify forest ecosystem services likely to have undergone important change.

For scoping study (1b), we recommend a “systematic mapping” approach (Atmadja and Sills 2015, Haddaway and others 2016, James and others 2016, McKinnon and others 2015), supplemented by careful review of on-line catalogs, databases, and models designed to support benefits transfer (ESVD 2017, EVRI 2017, InVEST 2017, Loomis and others 2008, RUVD 2017, USGS 2017). The objective is to assess the availability of credible and relevant estimates of the marginal value of forest ecosystem services. Credibility should be assessed based on the criteria described in the relevant chapters of this guide, as well as

the general principles above and red flags (text box, next page) (e.g., the estimates should reflect benefits rather than costs, and the estimation methods should account for diminishing returns). Relevance should be based on the qualitative similarity of the study site to the State in terms of the type of forest, the relative scarcity or abundance of forest, and the size and income level of the population that benefits from the ecosystem service. The analyst can then determine whether there is relatively high or relatively low availability of credible and relevant estimates of the marginal value of each forest ecosystem service. This scoping study should also take note of the definition and measurement of the final ecosystem services that are valued, as an input to stage 3.

Stage 2

Both the managing agency and the research team should be involved in this stage, drawing on the results of the scoping studies and consulting with experts, in order to define the parameters for the rest of the assessment. In particular, the managing agency and other State forest experts should help verify and supplement the list of forest ecosystem services undergoing important changes in value. Specifically, they can offer insights about how future trends may diverge from past trends in forest ecosystems (e.g., due to socio-demographic and economic factors, policies and programs, or natural disasters and climate change) and other factors that shape the importance of ecosystem services (such as the cultural traditions and economic activities in the State).

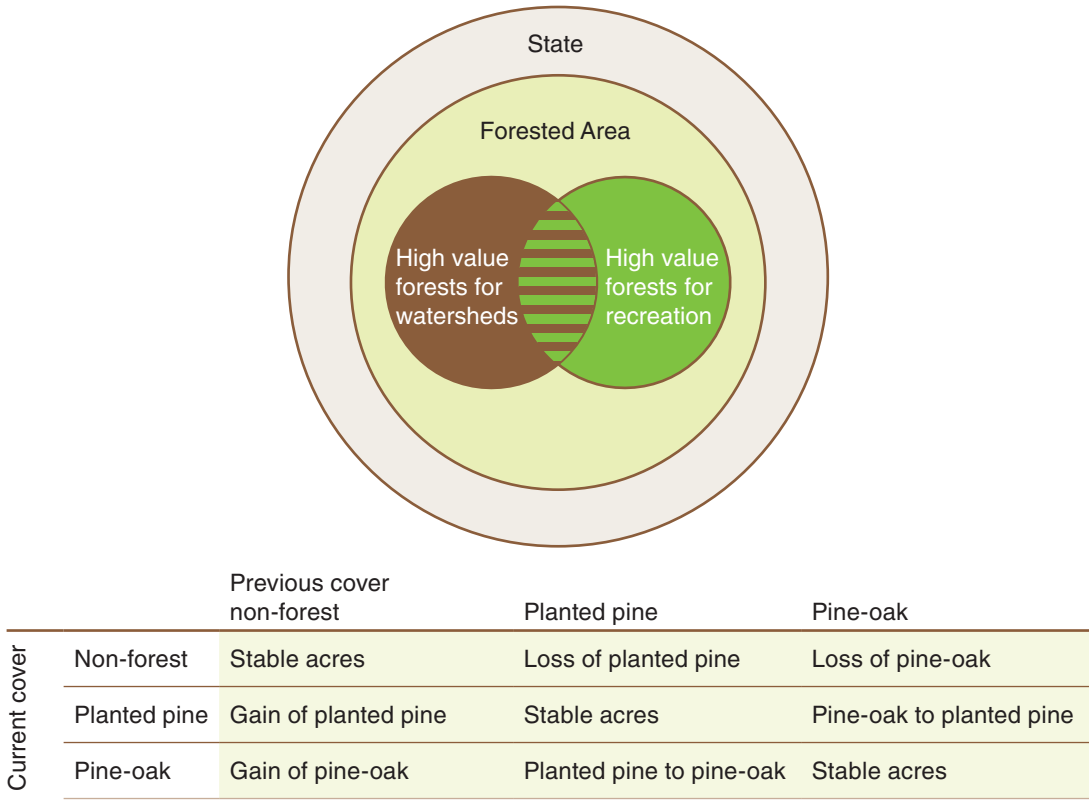


Figure 6.2—Scoping for important changes in forest ecosystem services: forest transitions in priority areas.

RED FLAGS

**Assuming forest is managed to optimize each ecosystem service.**

To estimate the annual flows of all ecosystem services that can be produced simultaneously by an acre of forest, it is important to make consistent assumptions about how that forest is managed. Management to optimize production of one ecosystem service will not necessarily optimize production of other services.

**Assuming a single fixed value for the marginal product or for the marginal value of an ecosystem service across all forest types and all locations.**

This is unlikely to be valid, because the relationship between the area of a forest ecosystem and the flow of forest ecosystem services is often non-linear, and marginal values are typically higher where ecosystem services are scarcer and where more people benefit. The principle of diminishing returns (i.e., diminishing marginal productivity and diminishing marginal utility) suggests that marginal values should generally be less than average values.

**Transferring marginal values from study sites with different forest types, different levels of abundance or scarcity of forest, and/or different income levels of the affected population.**

The study and target site should be similar for unit value transfers and should have an overlapping range of characteristics for predicting values based on benefit functions.

**Summing the values of intermediate and final services.**

This would almost certainly mean that some values have been double-counted because the value of final services includes the value of inputs (e.g., the values of regulating and provisioning), while cultural services include the value of supporting services (e.g., biodiversity and nutrient cycling).

**Summing the values of overlapping sets of services estimated by different methods.**

This results in double-counting services that appear in more than one set. In particular, stated preference surveys often encompass several services, some of which have also been the subjects of revealed preference studies.

**Combining estimates of values to different populations.**

State accounts should include values to State residents, with values to other people reported separately. For example, the social cost of carbon is an estimate of the global cost of climate change due to carbon emissions, and therefore should not be reported in the same account as air quality benefits to State residents. In this case, the market value of carbon offsets may be a better estimate of the value to State residents.

**Estimating ecosystem service values larger than relevant total values.**

For example, the value of avoided damage to a sector should not be larger than the total value added of that sector, and willingness to pay should be bounded by income.

**Combining measures of economic impacts or multiplier effects from regional economic analysis with estimates of economic values.**

These are based on fundamentally different assumptions and should not be added together, although they may provide useful complementary perspectives.

**Employing estimates of cost instead of estimates of value.**

In general, replacement or restoration costs are not valid estimates of value. Likewise, expenditures on recreation—including travel costs—measure costs, rather than value.

**Reporting single values with no indication of uncertainty.**

Accounts should not rely solely on point estimates, but rather should draw on confidence intervals where available and conduct bounding exercises (using minimum and maximum possible values) when confidence intervals are not available. Likewise, analysts should assess sensitivity to key assumptions, such as the size of the beneficiary population and the discount rate for calculating present value.

The next step is to plot the ecosystem services based on 1) their expected importance and 2) the availability of credible and reliable estimates of marginal value. They can then be sorted into the four categories shown in figure 6.1:

- A. Ecosystem services that are undergoing significant changes (e.g., due to changes in forest ecosystems) and that have credible and relevant estimates of marginal value: these should be the primary focus of accounting efforts, since they are both important and possible to value.
- B. Ecosystem services with fairly stable flows and with credible and relevant estimates of marginal value: these ecosystem services should be included in accounting efforts only if there is sufficient time and budget allocated to those efforts, because they are lower priority than the services in category A.
- C. Ecosystem services that have been significantly affected by changes in forest ecosystems but with no credible/relevant estimates of marginal value: these ecosystem services should be excluded from accounting exercises—thus avoiding the temptation to guesstimate or extrapolate to obtain marginal values. Instead, they should be added to a priority list of ecosystem services for future valuation studies. Depending on the budget, these studies may be part of the ecosystem service assessment. However, even if sufficient funding is not available, a formal list of State priorities, supported by a clear statement of the policy needs and information gaps (output 1), could influence and help researchers obtain funding for future research on these priority services.
- D. Ecosystem services with fairly stable flows and no credible and relevant estimates of marginal value: given likely low importance and lack of information, these ecosystem services should be acknowledged but not quantified in ecosystem service assessments.

To ensure that accounts provide a conservative or lower-bound estimate of the total value of forests, any significant disservices of forests such as water use by fast growing plantations, wildlife damage/hazards, or production of volatile organic compounds (VOCs) should also be acknowledged (cf., Escobedo and others 2011). If values of the (dis)services have been quantified in prior studies, they should be included in the accounts. Otherwise, they are candidates for the priority list for future valuation studies (category C).

In addition to defining which ecosystem services to include, this stage should also define how to disaggregate the State into spatial units for purposes of calculating changes in the annual flows of ecosystem services and their marginal values. At one extreme, each acre of forest might offer different marginal products and values of ecosystem services. At the other extreme, some previous State assessments have used a single average value

for all forests in their State. We recommend an intermediate solution, using FIA forest types (planted pine, natural pine, oak-pine, upland hardwood, lowland hardwood) as the highest level of disaggregation. Forest types should then be disaggregated as appropriate for each ecosystem service. These may overlap, as shown in figure 6.2 for watershed and recreation services. However, as long as they all nest in FIA forest types, they can be summed and aggregated.

Stage 3

This stage includes two studies that build on the scoping studies in stage 1 but focus on the smaller set of ecosystem services defined in stage 2.

Study (3a) quantifies total annual flows of ecosystem services from forests in each spatial unit defined in stage 2. Study (3b) generates a database of marginal values (and functions to compute marginal values) of each ecosystem service in each spatial unit defined in stage 2. These studies must be coordinated, so that the definitions and units of the final ecosystem service outputs quantified in study (3a) are the same as the final ecosystem services that are valued as inputs to production and well-being in study (3b). For some ecosystem services, marginal values may be reported for the service generated by an acre of forest, without specifying the quantity of service generated by that acre. This is particularly likely for non-tangible ecosystem services, such as a sense of place and natural heritage.

For most ecosystem services, study (3a) should quantify:

- a) the marginal product of an additional acre of forest (i.e., how many more units of each final ecosystem service are generated by an additional acre of forest),
- b) the extent of the market or the population that benefits from each service, and
- c) the total annual flow of each ecosystem service, in each of the spatial units defined in stage 2.

The marginal products and population of beneficiaries are used for accounting in stage 4. The total annual flows form the basis for a report on the magnitude of forest ecosystem services (output 2), including “headline numbers” such as the percent of drinking water that flows through forests, total air pollutants removed by forests, and number of recreational use-days in the State’s forests.

Study (3b) is a systematic review of marginal values of final forest ecosystem services reported in the scientific literature. Systematic reviews follow replicable procedures for searching the scientific literature and extracting consistent information from each study (Atmadja and Sills 2015, Moher and others 2009). In this case, the search protocol should screen for studies that

estimated marginal values of ecosystem services in similar forest biomes and socio-economic settings (i.e., temperate forest in developed countries). The information extracted should include:

- a) the marginal value or the function that describes that value, specifying the units and year in which estimated,
- b) which ecosystem services are included in the value,
- c) whether the estimated value is for an annual flow of services or the capitalized value of all future services that a forest will provide,
- d) the size of the population that benefits from services that are public goods,
- e) characteristics of the beneficiary population in the study site (spatial extent, education, income level), and
- f) characteristics of the forest in the study site, including forest type, extent, and how managed.

In addition, the reviewer should assess the credibility of the estimated values. This includes verifying that the study in fact estimated benefits (willingness to pay, change in welfare or utility) rather than expenditures or economic multipliers, and that it used a generally accepted method of valuation (as described in the previous chapters). It is also worth noting whether the estimated values have been assessed for plausibility (e.g., by comparing estimated total benefits with total income or total output).

Stage 4

The outputs of stage 3 can be used in a variety of ways to help monitor and manage change in forest ecosystems. This includes cost-benefit analysis of forest policy alternatives, selecting critical forest zones to target policy interventions, and identifying the best response to externally set standards (e.g., Federal standards for water quality). The first step is to quantify the extent of forest gain or loss in each spatial unit under each alternative considered (including the status quo). Both the type of forest and its management should be specified. Second, use the estimated marginal product of each final ecosystem service generated by a particular forest type under a particular management regime to quantify the changes in ecosystem

services. Finally, apply the marginal values appropriate to each final ecosystem service in each spatial zone to generate the economic value.

The primary objective of this accounting is to quantify the value to State residents of changes in the State's forest ecosystems. However, the literature also contains estimates of the marginal value of forests to the national or global population; for example, the social cost of carbon is an estimate of the total global cost of additional climate change due to one additional tonne of carbon dioxide emitted. These global values may be relevant to State residents if they are altruistic and care about the benefits that their State forests provide to the world, or if they expect to benefit from sales of those services in ecosystem service market. Thus, we recommend recording these national or global values in a side account (output 4), keeping them separate from the main account but ensuring that they are available for future negotiations.

There are many potential pitfalls when combining data and parameters from different sources in order to estimate values (see text box, p. 100). First, the marginal product and the marginal value must refer to the same final ecosystem service, defined in the same way, using consistent units. Second, some marginal values will have been estimated for bundles of ecosystem services, and in other cases for individual services. To obtain a defensible, lower-bound estimate, it is critical to avoid any double-counting. Last, we emphasize that the marginal values extracted from the literature are appropriate for valuing changes (not totals) in similar forest types and similar socio-economic settings as the study site.

As long as these red flags are heeded and pitfalls avoided, the report on changes in quantities of forest ecosystem services and the spatial catalog of marginal values of those services can be used to create an ecosystem services account that reports the value lost or gained by State citizens due to annual losses or gains of different forest types in different spatial zones (output 5). This account should be constructed from estimates of change in each spatial zone. We do not recommend reporting a "total value" of ecosystem services by summing the products of total forest area, marginal product, marginal values for each zone and for each service, because that would surely underestimate the total value of all forests, given the principle of diminishing returns.

Finally, we note that both outputs 4 and 5 report ecosystem service values both that contribute to GDP as conventionally measured (e.g., increased productivity of fisheries) and that represent additions to GDP (e.g., contribution of trout fishing to quality of life). These are worth distinguishing to facilitate interpretation and use of the accounts to understand the contribution of forests to the economy and to well-being.

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Southern forests provide a variety of critical ecosystem services, from purification of water and air to recreational opportunities for millions of people. Because many of these services are public goods with no observable market value, they are not fully accounted for in land use and policy decisions. There have been several efforts to remedy this by estimating the total value of forests in different States, with each effort including different bundles of ecosystem services and using different valuation methodologies. In this guide, we propose a more consistent and theoretically sound approach to 1) quantifying annual flows of ecosystem services, 2) developing a spatial catalog of the marginal values of changes in those flows, and 3) accounting for the total value of ecosystem services lost or gained as a result of changes in forest ecosystems. Four chapters—on cultural services, watershed services, air quality and carbon, and provisioning of non-timber forest products—provide guidance on best practices for quantifying and estimating the values of these services as provided by forests. Expert panels were convened to write each chapter. The guide as a whole was developed with input from the Southern Group of State Foresters and stakeholder meetings held in 2014 and 2015.

Keywords: ecosystem services, non-market valuation, green accounting, southern forests, non-timber forest products, watershed services, air quality, cultural services.



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