DISCUSSION PAPER

May 2011 ■ RFF DP 11-22

Ecosystem Good and Service Co-Effects of Terrestrial Carbon Sequestration

Implications for the U.S. Geological Survey's LandCarbon Methodology

James Boyd and David S. Brookshire

1616 P St. NW
Washington, DC 20036
202-328-5000 www.rff.org

RESOURCES FOR THE FUTURE

Ecosystem Good and Service Co-Effects of Terrestrial Carbon Sequestration: Implications for the U.S. Geological Survey's LandCarbon Methodology

James Boyd and David S. Brookshire

Abstract

This paper describes specific ways in which the analysis of ecosystem goods and services can be included in terrestrial carbon sequestration assessments and planning. It specifically reviews the U.S. Geological Survey's LandCarbon assessment methodology for ecosystem services. The report assumes that the biophysical analysis of co-effects should be designed to facilitate social evaluation. Accordingly, emphasis is placed on natural science strategies and outputs that complement subsequent economic and distributional analysis.

Key Words: ecosystem services, carbon sequestration, land use planning

Contents

Ecosystem Good and Service Co-Effects of Terrestrial Carbon Sequestration: Implications for the U.S. Geological Survey's LandCarbon Methodology

James Boyd and David S. Brookshire*

Executive Summary

This report reviews analysis of ecosystem good and service co-effects in the U.S. Geological Survey's LandCarbon (LC) assessment methodology (USGS; Zhu et al. 2010). The LC methodology is primarily focused on strategies to sequester carbon via land use change such as reforestation. But land use change has a range of ecological implications beyond carbon sequestration. The report describes a research strategy designed to capture the effects of land-based carbon sequestration interventions on the production and delivery of ecosystem goods and services (EGSs). It describes and advocates specific ways in which EGS analysis could in the future be included in terrestrial carbon resource assessments.

Ecosystem co-effects are important to many of USGS' audiences. Executive and legislative branch audiences are asked about co-effects by a range of constituents with diverse environmental interests; federal environmental trustees and planners are interested in how land cover change affects their regulatory and statutory mandates; and local stakeholders are concerned about how land cover change affects private property owners and the aesthetics of their communities. When sequestration interventions generate additional ecological benefits beyond carbon storage—positive co-effects—those benefits can be used to justify and motivate sequestration programs and identify the most beneficial locations for land cover and land management changes. When land cover change creates ecological losses—negative co-effects—those too should be taken into account. Can the losses be minimized via selection of different LC interventions, or

* Funding for this effort was provided by USGS and a USGS Cooperative Agreement entitled "Carbon Sequestration Valuation Methodology Development," contract # G09AC0042, with the Science Impact Laboratory for Policy and Economics (SILPE), University of New Mexico. James Boyd is a senior fellow and director of the Center for the Management of Ecological Wealth, Resources for the Future, Washington, DC. David Brookshire is professor of economics and director of SILPE at the University of New Mexico. The authors wish to thank Dr. Richard Bernknopf for his review and helpful comments.

interventions in different locations? Evaluators of the LC methodology, and interventions informed by it, are likely to have similar interests and concerns.

The report assumes that biophysical analysis of co-effects should be designed to facilitate social evaluation. Accordingly, emphasis is placed on natural science strategies and outputs that complement subsequent economic and distributional analysis.

The report proceeds as follows. First, EGS co-effects are defined in greater detail, with an eye toward definitions useful to analysis and a research strategy. Second, the analytical framework is applied to the existing LC methodology to identify consistencies, conflicts, and analytical gaps. Third, implications for future USGS analysis of EGS co-effects are discussed.

Several conclusions can be drawn based on this assessment of the LC methodology.

- First, co-effects analysis will be significantly constrained because of the current portfolio of outcome measures. This is understandable given the huge challenge posed by the LC effort generally, and given the lack of "off-the-shelf" data products and models that could be easily and directly applied to co-effects analysis. Most of the currently proposed outcome measures require further biophysical translation to facilitate social evaluation.
- Second, we identify a set of modeling and measurement gaps that, if filled, could leverage LC data products into a more robust assessment of co-effects.
- Third, the way in which ecosystem data products are presented and motivated in the LC plan suggest that USGS would benefit from a strategic reorganization of its co-effects efforts based around the analytical architecture described in our report.
- Fourth, we strongly encourage the proposed development of case studies to explore a wider range of ecosystem service co-effects, develop additional biophysical production and process models, and generate outcome measures at finer spatial resolutions. Such analysis will more effectively address the needs and expectations of LC's stakeholders and policy audiences.

1. Introduction and Motivations

This report defines and describes a research strategy designed to capture the effects of land-based carbon sequestration interventions on the production and delivery of *ecosystem goods and services* (EGS). The relationship of this research strategy to the LandCarbon (LC) methodology of the U.S. Geological Survey (USGS; Zhu et al. 2010) is made explicit. Although not currently a part of the LC methodology, the strategy describes ways in which EGS *co-effects*—the ecological effects of land use and land cover sequestration strategies—could in the future be included in a terrestrial carbon resource assessment based on the LC assessment methodology (Zhu et al. 2010).

There are several public policy motivations for this report. The LC methodology describes an approach to estimate the amount of carbon that could be sequestered terrestrially. The methodology is focused on the practical evaluation of a range of land cover and land management interventions and their ability to sequester carbon. This report concentrates on the land cover aspect of the methodology, but land management interventions will have similar effects. As the methodology notes, however, these interventions affect more than carbon sequestration. For example, the conversion of land from row crop agriculture to forest may have a range of consequences that reach beyond greater carbon storage. Land cover changes can affect things like water quality, aquifer recharge, the timing of surface water flows and flood pulses, fire risks, species location and abundance, and soil erosion.

These land cover co-effects are important economically, institutionally, legally, and politically. When sequestration interventions generate additional ecological benefits beyond carbon storage—positive co-effects—those benefits can be used to justify and motivate sequestration programs and identify the most beneficial locations for land cover change. When land cover change creates ecological losses—negative co-effects—those too should be taken into account. Can the losses be minimized via selection of different LC interventions, or interventions in different locations?

Co-effects are important to many of USGS' audiences. Executive and legislative branch audiences are asked about co-effects by a range of constituents with diverse interests in things like wildlife (environmental groups) and water (agriculture, municipalities, and the corporate sector). Other federal environmental trustees and planners are interested in how land cover change affects their regulatory and statutory mandates (e.g., the Endangered Species Act and wetland and surface water rules). Local stakeholders will be concerned about how land cover change affects private property owners and the aesthetics of their communities.

Evaluators of the LC methodology, and interventions informed by it, are likely to have similar interests and concerns. Regulatory analysts both inside and outside the federal government will scrutinize sequestration-driven land cover interventions on the basis of a full accounting of costs and benefits. Co-effects are difficult to measure—a point emphasized by this report—but failing to address them risks the longer-term acceptance and impact of the LC methodology.

Philosophically, this report assumes that a biophysical (natural science) analysis of co-effects should be designed to facilitate social evaluation. Accordingly, emphasis is placed on natural science strategies and outputs that complement subsequent economic and distributional analysis.

The report proceeds as follows. First, EGS co-effects are defined in greater detail, with an eye toward definitions useful to analysis and a research strategy. Second, this semantic and analytical framework is then applied to the existing LC methodology to identify consistencies, conflicts, and analytical gaps. Third, implications for future USGS analysis of EGS co-effects are discussed.

2. Ecosystem Good and Service Co-Effects: Definitions and Analytical Framework

Decisionmakers need to know how their choices *change* the delivery of EGS. It is not sufficient to describe EGS as they currently are. Rather, we need to be able to evaluate how our choices *increase* or *decrease* nature's ability to deliver them.[1] This requires us to measure directly or predict the effects of restoration, protection, land conversion, and management decisions on natural systems. This task falls to ecologists, biologists, hydrologists, and other natural scientists. However, natural science per se is not enough. Rather, it is necessary to explicitly design *policy-relevant natural science*, which describes the consequences of policy choices for biophysical outcomes that are meaningful to households, businesses, and communities.

[1] Actions that protect existing natural resources do not yield improvements relative to the current baseline; rather, they yield biophysical improvements relative to a future, degraded baseline.

2.1 Ecological Endpoints

Although the term *ecosystem services* is interpreted in a variety of ways, it conveys an important idea: natural systems are a tangible source of economic wealth and human wellbeing. Because EGS are usually public goods not traded in markets, we lack information on the prices paid for those goods and services—for example, we don't pay an explicit price for a beautiful view. Of course, just because something doesn't have a price doesn't mean it is not valuable. The challenge, then, is to encourage decisionmakers and stakeholders to reveal the values they place on goods and services that are unpriced.

A threshold question for ecosystem service analysis is therefore, what biophysical quantity units should we measure to facilitate economic valuation and other forms of social evaluation?

The centerpiece of policy-relevant natural science is the definition, measurement, and evaluation of *ecological endpoints*. Within the larger universe of biophysical outcome measures, ecological endpoints constitute a distinct set of outcomes: those that are meaningful and understandable to communities, businesses, households, planners, and other stakeholders. In general, natural systems can be thought of as collections of features, things, and qualities that interact via physical processes with other physical features, things, and qualities. Accordingly, almost anything we can measure in nature is an outcome of some underlying process.

> **Ecological endpoints** are biophysical outcome measures that require little further biophysical translation to clarify their relevance to human welfare. These endpoints are the essential bridge between biophysical and economic assessment.[2]

Three distinct economic issues are central to the definition of ecological endpoints and the way in which ecological changes can be integrated with economic assessment (for more detail, see Boyd and Krupnick 2009; Boyd and Banzhaf 2007; Boyd 2007). First, where nonmarket goods are concerned the units of quantity for what we consume or value are not consistent or even obvious. One of the nice things about markets is that they not only tell us the prices people pay for things, they also tell us about the quantity units on which people place a value. A grocery store is full of cans, boxes, loaves, and bunches; the number of these units bought yields a set of quantity measures to which prices can be attached. But public, nonmarket

[2] The term endpoint is used in many ways and refers generically to any modeled or measured outcome of a process, function, or relationship. We use the term more narrowly, to draw attention to the need for biophysical outcome measures that facilitate social evaluation.

EGS do not come in convenient quantity units. Ecological endpoints are akin to the quantity units we are accustomed to valuing in the market economy, but in a nonmarket setting must be derived and defended by the analyst,

Second, it is necessary to measure quantities whose value or importance can be meaningfully debated by stakeholders or detected by social scientists. In practice, this means choosing outcomes that are comprehensible and meaningful to nonscientists. Outcomes like biotic integrity indices, chemical water quality concentrations, hydrogeomorphic classifications, and rotifer productivity are of scientific interest but thwart social interpretation and evaluation. Ecological endpoints can be thought of as measures that your next-door neighbor would understand. Examples include the local abundance of certain species; the physical characteristics of viewable or accessible open space; expected risks of flood and fire; perceptible air and water quality; and the availability of water for drinking, irrigation, and recreation.

Third, the distinction between endpoints that are *directly valuable* and other outcomes that are *indirectly valuable* is important to any economic accounting system. Intermediate goods, for example, are those used to produce final goods. Final goods are what we count in the gross domestic product; their value embodies the value of the intermediate goods used to produce them. If we do not distinguish between endpoints and indirectly valuable inputs we run the risk of double-counting benefits associated with a particular ecological feature or quality (Boyd and Krupnick 2009). If we add the value of a car to the value of the steel we used to build the car, the value of the steel is counted twice – because the steel's value is part of the car's value. This in no way implies that intermediate goods are less valuable than final goods. But it does mean that we needn't count everything in nature—only those final EGS that embody the value of the whole system.

2.2 Interventions and Biophysical Production Functions

Two additional elements are necessary to policy-relevant natural science: first, the *interventions* or actions that trigger ecological changes and, second, the biophysical *production functions* that relate interventions to changes in ecological endpoints.

> **Actions and interventions** describe policy and management choices—land cover conversion, restoration, protection, and resource management—that affect natural resources and that trigger subsequent biophysical changes.

The LC methodology has already identified a clear set of interventions relating to land cover change and land management practices.

> **Biophysical production functions** are the biophysical relationships that link concrete policy choices to changes in socially meaningful biophysical outcomes (U.S. Environmental Protection Agency [EPA] 2009; Daily and Matson 2008; Boyd 2007).

The measurement and prediction of biophysical production functions is the most important aspect of an EGS research strategy. Economic analysis of co-effects cannot be done without it. The economic analysis of ecosystems depends entirely on our ability to measure these biophysical production functions.[3]

2.3 Depiction of the Analytical System

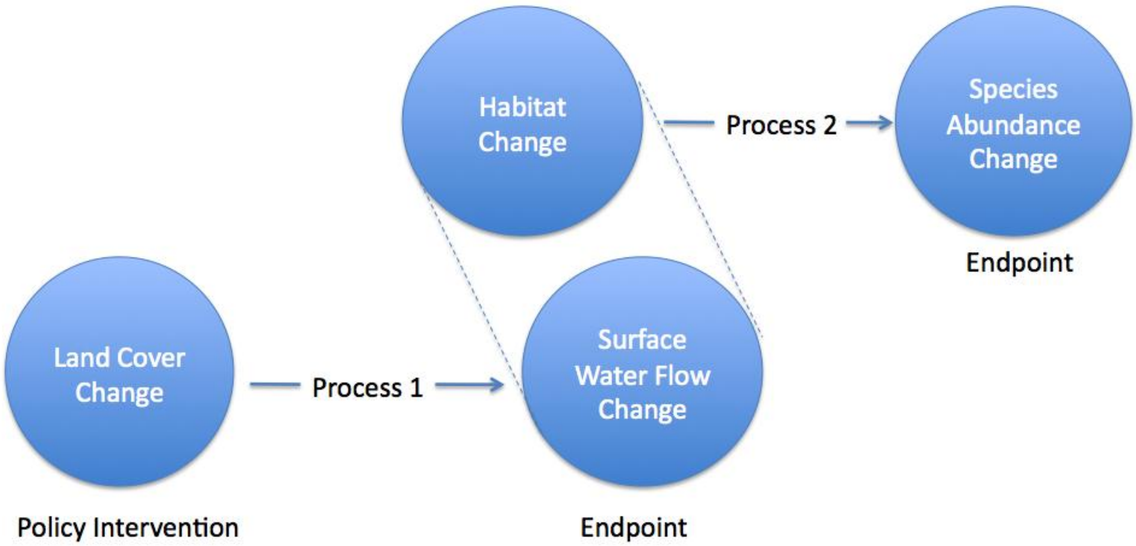
The biophysical underpinning of co-effects is a conceptual model of ecological and physical production. Starting with a policy action—such as land cover conversion—the production system describes the consequences of the action for subsequent biophysical changes. These production systems typically begin as theoretical hypotheses that are then validated or refuted by empirical observation and experimentation.

Figure 1 depicts a highly simplified version of such a production system. Land cover change, such as a conversion of agricultural production to forest, is a policy intervention that triggers subsequent changes in surface water flows (timing, speed, and volume) via hydrologic processes. Changes in the hydrograph are a biophysical endpoint because they describe flood risks and water availability for recreation, irrigation, navigation, and industrial production. In turn, changes in the hydrograph affect habitat conditions for aquatic, avian, and other species via a range of chemical and biological processes. Resulting changes in species abundance represent a second set of endpoint changes resulting from the policy intervention.

---

[3] *Biophysical production functions* as defined here are related to, but not synonymous with, the concepts of *ecological function* or *process* as understood within ecology. *Ecological function* or *process* is the broader concept, applicable to any relationship between biota or between biota and physical features. Biophysical production functions are a subset of those relationships—those that relate policy actions or choices to endpoint changes.

Figure 1. A Simplified Biophysical Production System



The key elements of this analytical system are:

- policy interventions translated into subsequent changes in ecological endpoints and
- a linked system of production where endpoints can play a dual role as both output and input.

2.4 Economic and Social Analysis of Endpoint Changes

Economic and social evaluation is built around the analysis of biophysical production. With relationships between policy intervention and endpoint change described, as in Figure 1, it is possible to evaluate how those endpoint changes lead to changes in social welfare. By design, endpoints are meaningful to decisionmakers and society generally. This means that changes in those endpoints can more easily lead to economic evaluation. There are several ways to approach economic analysis of endpoint changes.

First, economic studies derive monetary benefit estimates using hedonic, travel cost, and other econometrically sophisticated "revealed preference" methods.[4] Revealed preference studies

[4] For an overview of these methods see Freeman, (1993).

consider the prices people are willing to pay for marketed goods that have an environmental component. From those prices, inferences about the environmental benefits associated with the goods can be made. For example, when people purchase homes near an aesthetically pleasing ecosystem, home prices reflect that environmental amenity.[5] Alternatively, when people spend time and money traveling to recreation, they reveal a willingness to pay for the time and travel costs to access the recreational services. Travel cost studies are used to make a benefit estimate based on those expenditures.[6] The travel cost method requires data and analysis linking the number of trips to a site with the quality, size, or location of a site. Changes in these attributes can be valued if there is a perceptible change in the number, length, or cost of trips taken to the site.

Second, economic studies derive benefits via stated preference studies. These are particularly useful when—as is often the case—environmental benefits are not captured in market prices or in observable individual choices. Stated preference studies ask people, in a highly structured way, what they would be willing to pay for a set of environmental improvements. Contingent valuation studies are an example. Stated preference surveys are expensive, controversial, and are most reliable when the questions concern specific ecological services provided in specific contexts. The more complex and holistic the improvement, or change, the more difficult the methodological challenge. A principal drawback to this approach is the risk that people may misunderstand the precise service being valued when undisciplined by the need to spend their own real money. For the same reason, they may also overstate their willingness to pay.[7] Nevertheless, these methods are a distinct improvement relative to evaluation techniques that ignore social preferences.[8]

Third, analysts can use *benefit transfer* studies to harness the benefits of econometric estimation while minimizing the need for costly new site-specific analyses.[9] The benefit transfer method takes the result of a preexisting monetary study and translates it into a new environmental context. For example, if a study of trout fishing in Colorado yields a per-person

---

[5] Hedonic analysis is used in this type of study. See, for example, Mahan et al. (2000).

[6] There is a large methodological literature on this subject. See, for example, McConnell (1992).

[7] See generally Kopp et al. (1995), who present a collection of articles relating to the contingent valuation method.

[8] See Carson et al. (2001) for a review and defense of contingent valuation's role in the evaluation of EGS.

[9] For an overview of benefit transfer methodologies, see the 1992 special issue of *Water Resources Research* devoted to it. Also see Kirchoff et al. (1997) and Kopp and Smith (1993).

benefit of $100 a day, this result can be transferred, with some adjustments, to say something about the value of a fishing day in California. The challenge for—and hazard of—benefit transfer methods is that the value of EGS is highly dependent on the physical and social context in which they arise. It requires methodological and conceptual sophistication to credibly transfer values across the landscape.[10]

Fourth, analysts can evaluate social benefits using indicators of benefits that stop short of monetary valuation. Monetary valuation requires the use of data and methods that add substantially to the assessment burden. Typically, each benefit or cost stream arising from the natural landscape must be analyzed with different data and econometric methods. It is common in studies to see only a single environmental benefit monetized because of the costs of such studies. Also, econometric tools are opaque to most decisionmakers. And some audiences reflexively reject the monetization of benefits related to nature. It is thus useful to ask: is it absolutely necessary to conduct econometrically sophisticated studies to estimate the value and importance of EGS?[11] An underexplored alternative (or complement) to econometric analysis is the use of quantitative ecosystem benefit indicators (EBIs)—quantitative, countable features of the physical and social landscape.

> **Ecosystem benefit indicators (EBIs)** are countable features of the physical and social landscape that relate to and describe the value of endpoint changes. They can usually be derived easily from existing geospatial datasets.

EBIs are environmental and social features that influence—positively or negatively—the contributions of ecosystem services to human wellbeing. They convey information about the production of benefits involving ecological inputs. However, they tell us nothing about the underlying preferences for goods and services. Thus, EBIs provide *some* information relating to welfare but, by themselves, do not allow for monetary valuation.

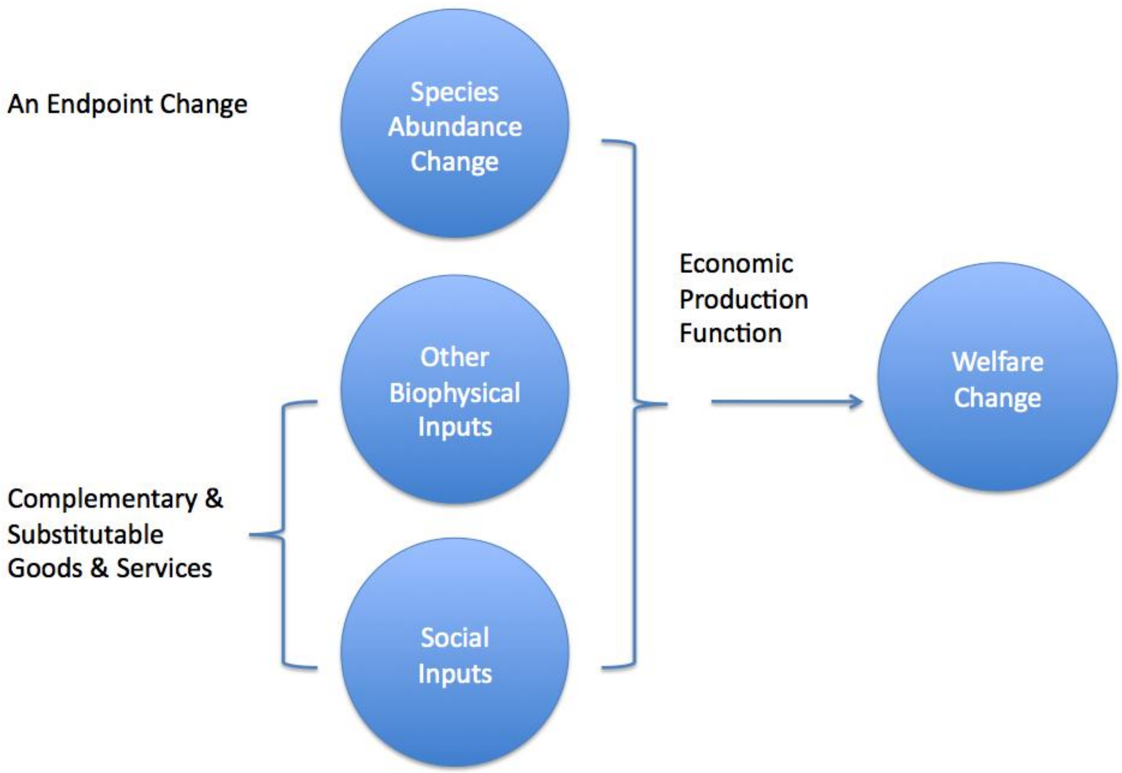
---

[10] For a description of the challenges associated with benefit transfer studies see Chapman and Hanemann (2001, 355): "It is sometimes claimed that the benefit transfer approach provides a convenient solution when the requisite data are lacking. But in this case there was considerable disagreement over basic issues, such as whether or not beaches in Florida are 'substantially dissimilar' from beaches in Southern California. If this benefits transfer is problematical, how much more so others!"

[11] The virtue of monetary valuation is that dollar benefit can easily be compared to other monetary costs and benefits. Thus, dollar values allow the ecological outcomes to be compared on the basis of a single metric.

EBIs relate to the ways in which ecological endpoint changes produce changes in human welfare (Figure 2). Like the analysis of *biophysical production*, the analysis of *economic production* describes how inputs combine to produce an output, in this case human welfare.

Figure 2. A Simplified Economic Production System



All else being equal, we can always say the following.

- The scarcer an ecological feature, the greater its value.
- The scarcer the substitutes for an ecological feature, the greater its value (substitutes are goods or services that at least partly satisfy similar wants or needs).
- The more abundant the complements to an ecological feature, the greater its value (complements are goods that “go together” or enhance each other).[12]

[12] Though note that not all ecological inputs require complements to yield a benefit.

The scarcity of, substitutes for, and complements to many EGS are relatively easy to assess. In many cases, metrics can be derived from existing social and biophysical geographical information system (GIS) data (Boyd and Wainger 2003).

Depending on the ecological feature, we can often go further than this. For example, the social value of some environmental features is often a direct and increasing function of the number of people with access to them. Similarly, the social value of some environmental features is often a direct and increasing function of the economic value they protect or enhance. Accordingly, we can often—but not always—say the following.

- The larger the population benefiting from an ecological feature, the greater its value.
- The larger the economic value protected or enhanced by the feature, the greater its value.

Relative to econometric benefit estimation, EBIs may be easier to develop because they can be derived from existing GIS data layers. They provide useful economic information in a cost-effective way. Linked to specific ecological endpoints, they can quickly inform decisionmakers and allow for more comprehensive evaluation of multiple goods and services given limited budgets for analysis.

2.5 Empirical Issues

Empirical measurement of these relationships is difficult. For example, it may not even be clear before the fact which endpoints will change as a result of a policy intervention. Empirical challenges include the following.

- Geographic separation between intervention and outcome. Natural resource interventions often generate effects at a significant distance from the intervention. Interventions that affect water quality, for example, can deliver water quality changes hundreds of miles away. Consider also that the speed and depth of flood pulses can be affected by interventions well up-watershed. Also, changes in habitat, particularly for migratory species, can affect species abundance over great distances.
- Temporal lags between intervention and outcome. Similarly, interventions may trigger endpoint changes over a period of time, rather than instantaneously. Lags can occur as a result of the natural life cycles of species, where population effects may take several generations to play out. Certain hydrologic processes, such as those affecting aquifer recharge and quality can also take years or more to become apparent.

- Production of EGSs usually requires a range of inputs in addition to those associated with an intervention. A given ecosystem endpoint is the product of factors beyond the policy intervention (e.g., a restoration project) itself. Species abundance, for example, is a function of broader habitat and forage resources. Similarly, flood pulses are affected by both natural and built hydrological relationships across the broader landscape. These complementary inputs typically vary across regions or natural systems.
- Lack of control groups. Related to the previous point, it can be difficult to find comparable situations for comparison of with-intervention and without-intervention outcomes. Particularly at larger geographic scales, biophysical systems may not be similar enough to construct control groups.
- The effects of small, marginal interventions can be difficult to detect. Policy interventions rarely occur all at once, and tend to be small in scale, relative to the natural systems they affect. Given time lags and spatial phenomena, this can make empirical detection of cause and effect very difficult.

Given these challenges, it is unrealistic to think that we can detect the magnitude of co-effects cheaply, quickly, and precisely. However, co-effects analysis presumes that empirical study is useful to policy evaluation.

Ideally, USGS and its partners will support and conduct studies to directly monitor intervention–endpoint relationships. This will require monitoring protocols and systems designed around the interventions contemplated by the LC national resource assessment and their most likely endpoint effects. As noted above, monitoring of endpoint changes cannot be limited to on-site monitoring because biophysical changes may occur over much broader spatial scales. Also, monitoring systems should be designed to provide information over longer periods of time—years and decades—to detect lagged effects.

Clearly, this kind of monitoring represents a significant investment. However, USGS already has in place many of the building blocks for just such an empirical system.

In the absence of direct monitoring of intervention–endpoint relationships, analysts may be able to extrapolate from known qualitative or empirical ecological relationships to create evaluations of co-effects. For example, empirical demonstration of an intervention–endpoint relationship in one region or watershed (say, the relationship between forest cover and surface water flows) can be cautiously applied to other regions and watersheds.

Figure 3. Biophysical Production Function Transfer

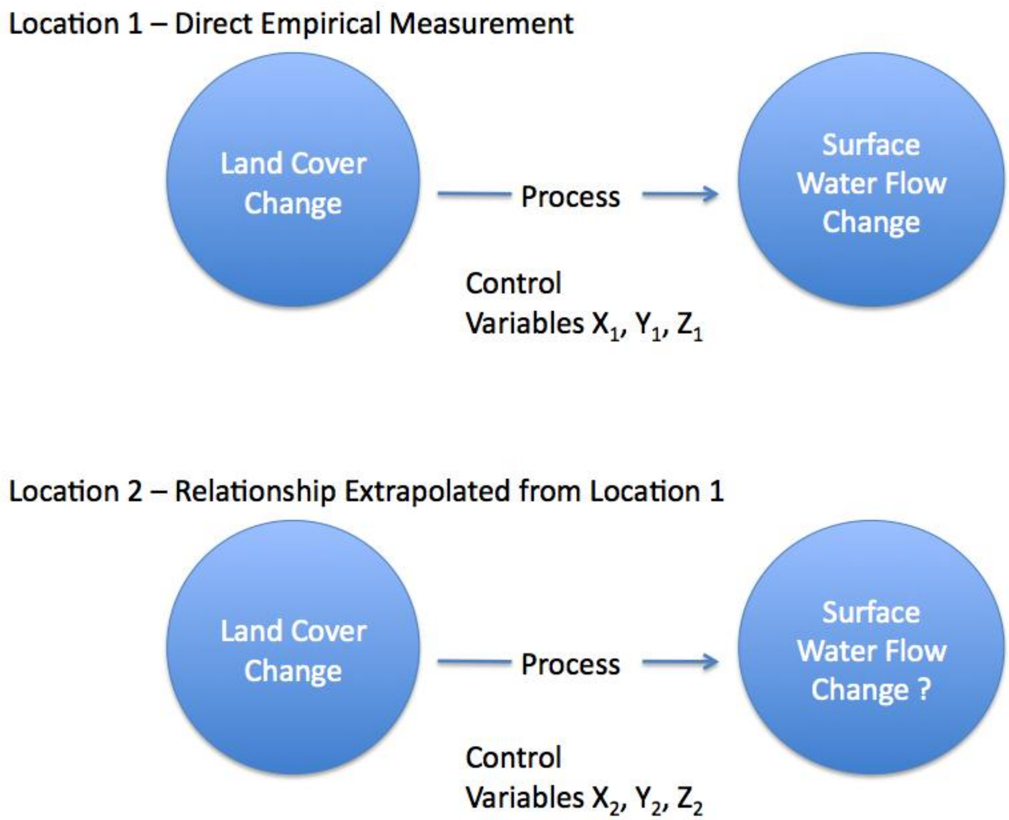


Figure 3 represents the transfer of a biophysical production function to other biophysical contexts (other locations). It emphasizes the need to statistically control for differences in biophysical setting that are most likely to affect the transferability of the production function. For example, the relationship between land cover change and surface water flows may vary across systems depending on the percentages of natural versus built land uses, climatic conditions, or other factors in the regions of interest.

The simplest, but least accurate, way to transfer production relationships is to assume that the measured relationship's magnitude applies in all biophysical settings.[13] A more defensible

[13] Note that the benefits of carbon sequestration are—unlike those of co-effects—independent of location. A ton of carbon sequestered in North America is biophysically and economically equivalent to a ton sequestered in Africa or Asia.

approach is to measure features (control variables) that differ across the systems and adjust the transferred relationship accordingly. To do this, analysts must measure production function relationships across a portfolio of systems to understand how different control variables affect the production function (this portfolio of studies is not depicted in Figure 3).

The transfer of production relationships could lower the costs of co-effects analysis. However, it still requires significant investment in monitoring and analysis.

Another approach to empirical assessment takes the existing science around interventions and nonendpoint outcome measures and translates known outcome effects into their subsequent implications for endpoint changes. Consider a known relationship between land cover and a chemical water quality measure such as nitrogen concentration. Nitrogen delivery is relatively well studied. The question is: how does surface water nitrogen translate into ecological endpoints relevant to social evaluation, such as species abundance, risk of waterborne disease, water quality, or water aesthetics?

Figure 4. Translation of Existing Production Function to Endpoint Change

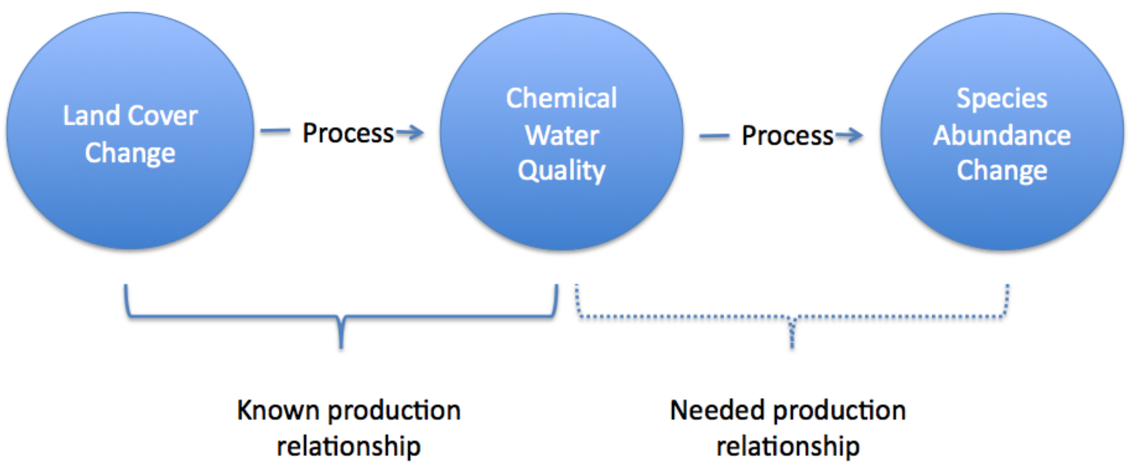


Figure 4 depicts this kind of empirical strategy, which builds on known relationships between interventions and nonendpoint outcomes. Often, existing models and monitoring can tell us about the relationship between interventions and proxies for, or precursors to, endpoint changes. The challenge, in this case, is to empirically relate the proxy or precursor to endpoints of interest to social evaluation. This can be done with a combination of new monitoring and modeled relationships.

All of these strategies are needed to empirically assess co-effects. None of these strategies is particularly simple, or easy, but some combination of them is required to do quantitative assessment.

A final note: monitoring, modeling, and experimentation are facilitated by the adoption of consistent units to describe interventions (restoration, conservation, management, and conversion), control variables, and outcomes (endpoints like species abundance, flood and fire risk, and water availability).

2.6 Which Ecological Endpoints Should Be the Focus of Analysis?

Estimating the co-effects of carbon sequestration interventions requires that we count goods and services produced (or lost) as a result of the interventions and then weight them according to their social value. Both of these tasks are complicated by the fact that most EGS are not market goods. The *missing prices* problem is a commonly acknowledged barrier to economic assessments of nature. Less well appreciated, but equally important, is the *missing quantities* problem.

Social and economic analyses of co-effects must somehow define the *environmental commodities* to which values are attached. These commodities are what we defined earlier as ecological endpoints. They are biophysical features, conditions, and qualities that people, communities, and businesses clearly understand are related to their welfare. But what are they *specifically*?

To answer this question, note that a combination of two factors makes a particular endpoint important to a given co-effects analysis:

- the endpoint is important or valuable to society (clearly we want to measure the things people care most about) and
- the endpoint's production is significantly affected by policy interventions.

The latter factor is a bit of a "chicken and egg" problem for analysts. How can we know there will be a strong production relationship if we haven't measured the relationship yet? In some cases, we can apply ecological function and process theory to generate hypotheses regarding likely production relationships. In other cases, however, we rely on serendipity and trial and error to discover production relationships that deserve greater analysis.[14]

The first factor—what is important to society—is an issue for social science and public policy. There are several ways to detect what directly matters to people. First, we can observe real-world choices. Consider just a few examples. For certain users (groundwater irrigators) water table depth is an endpoint because the cost of pumped irrigation is a direct and known function of that measure. We know that water availability in general is an endpoint because available water volume directly affects a range of users. We know that a species' abundance is an endpoint because declines in abundance trigger social concern. We know that flood probabilities are an endpoint because they affect water infrastructure, insurance, and residential and business location choices.

Second, we can observe legal and political conflicts involving natural resources. Water and land use conflicts reveal a range of ecological commodities that feature prominently in public deliberations. Many of these commodities (endpoints) are relatively clear. Abundant evidence, and common sense, suggests that water flows—water that is drinkable and useful for irrigation, soil quality, species abundance, open space, flood risks, and aesthetically pleasing flora—are biophysical commodities to which people attach value. Accordingly, they are all desirable endpoints for biophysical analysis.

Third, we can directly ask stakeholders what is important to them and what affects their economic or broader wellbeing. This information would provide input to a valuation exercise as well as an adaptive management framework.

In other cases, however, ecological endpoints may be more difficult to define, even though they are economically relevant and fit our definition of an ecological endpoint. This is particularly true when it comes to aesthetic, cultural, spiritual, and ethical values associated with nature. It is also a question for cognitive and social psychology as much as for nonmarket economics.[15]

---

14 This is similar to medical science, where the identification of an "underlying mechanism" is sometimes, but not always, the way to identify the cause of disease.

15 See the 2005 special issue of the *International Journal of Psychology*.

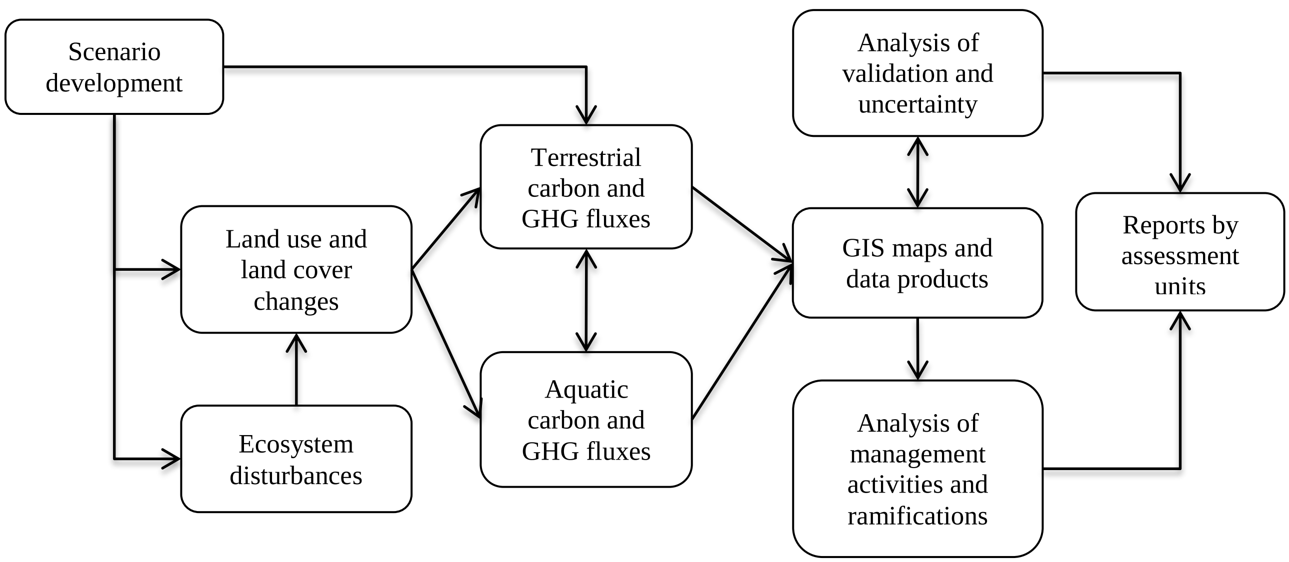
3. Application of the Co-Effects Analysis Framework to the LC Methodology

This section relates the co-effects analytical framework (denoted CEAF) to the LC methodology described in (Zhu et al. 2010).

3.1 Comparison of Frameworks at a General Level

Consider first LC's "scenario and assessment deliverables" depicted in Figure 5 below (Figure 3.5 in the LC methodology).

Figure 5. LC's Depiction of Relationships among Major Methods Designed To Achieve Scenario Runs and Produce Assessment Deliverables



Source: Zhu et al. 2010, Figure 3.5.

Is the co-effects framework described in this study consistent with LC's approach? At this level of generality, the two frameworks are consistent with each other. Note that both the LC and CEAF frameworks begin with a delineation of land use and land cover (LULC) changes, as shown in Figure 5. LC derives alternative policy scenarios based on these LULC projections. LULC change is driven both by policy interventions, demographic change, and ecosystem disturbances. These scenarios are then used to depict carbon and greenhouse gas (GHG) fluxes (changes). The analogous CEAF activity is to translate LULC scenarios into a different set of outcomes: those relating to ecosystem endpoint changes.

Figure 6. Relationships among Methods To Produce Assessment Deliverables

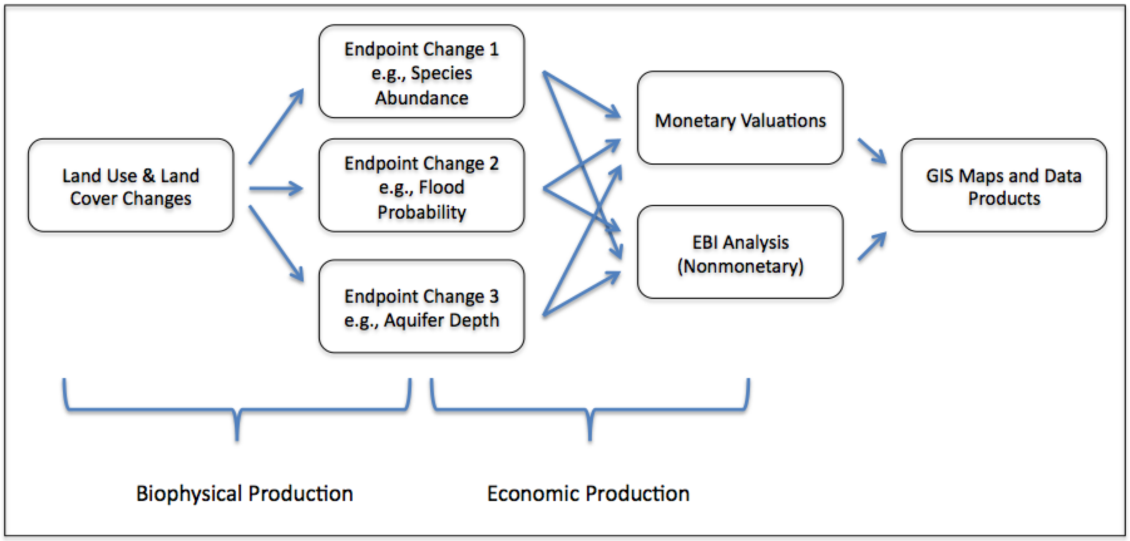


Figure 6 recreates Figure 3.5 in Zhu et al. (2010), but with ecological co-effects as the goal of analysis. Analysis of the biophysical production of specific ecological endpoints would proceed along the lines described in Section 2. Social evaluation of endpoint changes, whether via monetary estimation or nonmonetary quantification, follows. GIS analysis of the location of *delivered* ecological endpoints is desirable given the importance of location to the social value of endpoint changes.

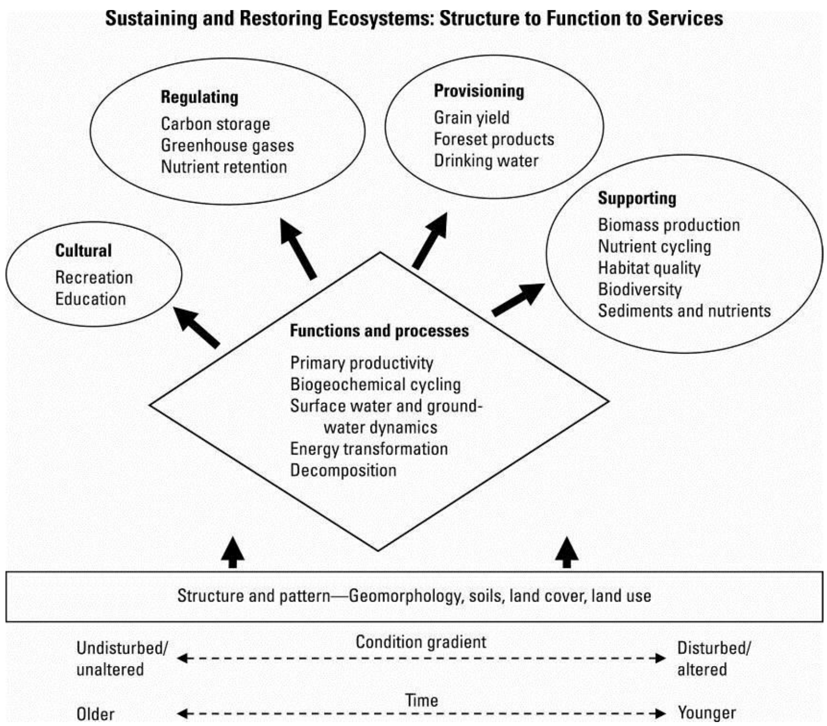
Again, the LC and CEAF approaches to assessment are similar and consistent at this level of generality. The major differences lie in the details. For example, unlike for sequestered carbon, there is no single, global/national value, or price, for a given ecosystem endpoint. This creates the need for an analysis of economic production and its associated data requirements.

As described below (Section 3.4), the spatial resolution of both biophysical production analysis and economic valuation is extremely important. As a general rule, biophysical outcome measures that are expressed only at the Omernick level II ecoregional scale (the LC methodology's basic reporting unit) will significantly inhibit ecosystem service analysis.

3.2 Comparison of Ecosystem Service Output Measures

The main differences between the LC and CEAF approaches are associated with the choice and interpretation of assessment data products and other ecological outcome measures. CEAF relies heavily on definitions proposed by the Millennium Ecosystem Assessment (MEA) to organize its assessment of co-effects. As will be argued, however, these definitions can confuse and thwart practical measurement of co-effects and suffer from certain theoretical weaknesses. The MEA's bundling of ecosystem service measures into categories such as "cultural," "regulating," "provisioning," and "supporting" services serves a pedagogical function. However, the definitions and organization of these services is inconsistent with the CEAF architecture and its emphasis on linked biophysical and economic production. Consider first Figure 7 (Figure 2.3 from the LC methodology, reproduced below).

Figure 7. LC's Conceptual Diagram of the Relations among Ecosystem Structure, Function, and Services



Source: Zhu et al. 2010, Figure 2.3.

Note first one similarity between the terminology of LC and CEAF: in particular, the importance of functions and processes to assessment. In both frameworks, function and process are important to descriptions of how policy interventions, demographic change and market forces, and ecological disturbances lead to subsequent changes in ecological conditions.

However, the figure features several internal inconsistencies in definitions and terminology that confuse practical measurement and application. This is true of the MEA definitions from which Figure 7 is drawn.

- **Outcomes versus the processes and functions that produce them**. The most obvious way to interpret Figure 7 is to think of the "functions and process" set as leading to the production of biophysical outcomes. In some cases, this is what is shown. For example, the listed biophysical functions and processes produce biodiversity, drinking water, and forest products. In other cases, however, processes and functions appear to be leading to other processes and functions, such as nutrient cycling and nutrient retention. It is unclear why these processes and functions are not in the "processes and functions" set.

  Recommendation: Distinguish between processes and functions and the biophysical outcomes (endpoints) they produce or mediate. Also, depict the relationship between linked functions and processes as a biophysical production system, where linkages are made explicit.

- **Biophysical measures versus vague definitions of social benefits**. Figure 7 refers to a set of categorically unlike things as *ecosystem services*. Consider that both recreation and sediments are described as ecosystem services. Sediments (or more specifically, sediment volumes) are a biophysical outcome measure. Recreation is not a biophysical outcome measure, or a biophysical process for that matter. Recreation is a label attached to certain human activities that depend on biophysical conditions. Co-effects analysis is accordingly interested in how ecological endpoints affect *the benefits of recreation*. But the benefits of recreation as something we measure are categorically different from ecological endpoints and processes.

  Recommendation: Define *ecosystem services* more carefully so that categorically inconsistent things—outcomes, processes, and benefits—are not confused. Terminological inconsistency thwarts analysis by confusing the organization and aims of different analytical activities.

- **Biophysical outcomes versus technological outcomes**. This issue is associated with the provisioning services—grain yield, forest products, and drinking water. All of these are the product of a combined biophysical and economic production process. They are not purely biophysical outcomes. Consider grain yield. Grain yield is a function of biophysical endpoints (precipitation, soil availability and quality, and the presence of pollinator species) *and* technological inputs, such as constructed irrigation, pumped groundwater, planting and harvesting machinery, hybridized and genetically modified seeds, and pesticides and fertilizers. Are grain yields increasing because ecological endpoints are improving, or because of changes in technology?

Recommendation: Again, define *ecosystem services* more carefully so that analysts can distinguish between changes in ecological endpoints and outcomes (e.g., harvests) that may have less to do with ecological conditions and more to do with technological inputs.

- **Intermediate versus final outcomes**. As described in Section 2.1 above, economic assessment requires an accounting framework that distinguishes between intermediate and final goods and services. Both intermediate and final goods and services are valuable. However, the value of intermediate goods and services is derived from the value of the final goods and services they produce. Consider the supporting services, habitat quality and biodiversity. Habitat quality is an *input to* biodiversity. When this linkage is not made clear, it confuses analysis. First, it obscures the underlying biophysical production relationship. Second, if the value of habitat quality and biodiversity are assessed independently, the value of habitat as an input to biodiversity will be double-counted.

  Recommendation: Using biophysical production models, distinguish between and make clear the relationship between ecological outcomes. In many cases, a given ecological outcome will be both an end in itself (a final good) and an input to subsequent biophysical production.

In a similar vein, the LC and CEAF approaches differ in the choice of assessment data products. The LC description of candidate ecosystem services and data products (Table 3.14 in Zhu et al. 2010) is adapted in Table 1 below.

Table 1. LC's Depiction of Candidate Ecosystem Services to Be Analyzed Using Results of the Assessment

Types of ecosystem services	Ecosystem service	Assessment data products
Supporting	Soil formation	Soil organic carbon
	Primary production	Net ecosystem productivity
Regulating	GHG mitigation	Soil organic carbon Carbon sequestration N_2O, CH_4 emissions
	Water quality	Soil erosion Nitrate retention
Provisioning	Food	Grain production
	Wildlife habitat	Species richness Occupancy and connectivity models Species climate vulnerability Metapopulation dynamics
	Fiber	Timber production
Cultural	Recreation	Species richness Occupancy models

Notes: CH_4, methane; N_2O, nitrous oxide.
Source: Zhu et al. 2010, Table 3.14.

For concreteness, note that the Tensas Parish study, described in Zhu et al. (2010) and shown in Table 2 below, derived specific outcome measures that correspond to some of the data products described in Table 1.

Table 2. LC's Preliminary Ecosystem Service Estimates for a Test in Tensas Parish, LA, and Claiborne County, MS, Using the A1B Storyline

Assessment data products	Unit of measurement	Baseline value (2001–2010)	R (2041–2050)		L (2041–2050)	
			Output value	ESCI	Output value	ESCI
Net ecosystem productivity	Grams of carbon per square meter per year	651	571	–0.123	575	–0.117
Soil organic carbon	Grams of carbon per square meter	5,433	6,153	0.133	6,155	0.133
Carbon sequestration	Grams of carbon per square meter	6,193	9,872	0.594	10,207	0.648
Timber production	Grams of carbon per square meter per year	4.89	9.70	0.985	3.61	–0.260
Grain production	Grams of carbon per square meter per year	70	57	–0.185	52	–0.252
Carbon storage	Grams of carbon per square meter	12,377	16,810	0.358	17,146	0.385
Carbon sequestration	Grams of carbon per square meter	148	91	–0.384	105	–0.292
N_2O emission	Gigagrams of nitrogen	24.3	21.6	0.112	21.7	0.110
CH_4 emission	Teragrams of carbon	0.163	0.133	0.183	0.143	0.125
Erosion	Tons per hectare per year	–0.062	–0.059	0.049	–0.061	0.008

Notes: CH_4, methane; ESCI, ecosystem service change indicator; L, "enhanced land use and land cover with reference land management scenario;" N_2O, nitrous oxide; R, "reference land use, land cover, and land management" scenario.
Source: Zhu et al. 2010, Table 3.15.

Not all of the data products listed in Table 1 were developed in the Tensas Parish example. Specifically, species-related analyses are not reported, presumably because of their relative difficulty.

The importance of Table 1 lies in its identification of specific outcome measures, or *data products* that are to be the focus of future assessment activity. How do these assessment products relate to the CEAF described in this paper?

Column 2 of Table 1—the "services"—presents the same semantic and conceptual issues as Figure 7 (inconsistent definitions of ecosystem service, inattention to joint production relationships, and so on). Column 3, the data products themselves, triggers the following observations.

Data products that are not co-effects measures

- Carbon sequestration and nitrous oxide (N_2O) and methane (CH_4) emissions

  Carbon sequestration and CH_4 and N_2O emissions are not co-effects, they are the principle focus of LC assessment methodology.

Data products that are input or precursor measures, not endpoints

- Net ecosystem productivity (NEP)

  NEP, the amount of energy trapped in organic matter, is an intermediate biophysical measure whose meaning is not clear or relevant to beneficiaries unless translated into other biophysical outcomes.

- Nitrate retention

  Nitrate retention is not an endpoint. Rather, it is a precursor to a range of endpoint outcomes related to surface water, groundwater, and marine conditions, such as species abundance, waterborne illness, and drinking water quality.

- Species richness

  Species richness is used to determine the sensitivity of ecosystems and species to natural and social disturbance. It is not an endpoint to which social value can be attached, absent subsequent translation into species-specific outcomes.

Data products that describe processes and functions

- Occupancy and connectivity models, metapopulation dynamics, species vulnerability assessment

  These products help explain and predict species abundance changes. Presumably, they will be used to generate abundance measures that are endpoints, but they are not themselves endpoint measures.

Data products that are endpoints

- Soil organic carbon

  If this measure is conceived of as a co-effect (rather than a sequestration measure) it is an endpoint because it affects pasture and harvest productivity, and that relationship is known to agriculturalists.

- Soil erosion

  Soil erosion is of direct relevance to agriculture, and is therefore an endpoint amenable to social evaluation. However, it is also an example of the need for geographic specificity. The social value of avoided erosion is highly place-specific.

Data products that do not distinguish between biophysical and technological production

- Grain and timber production

  As noted earlier, grain and timber production depend on a range of nonbiophysical factors, and therefore are problematic measures of ecological change.

3.3 Application of the CEAF Approach to LC Data Products—Examples

The previous section makes several recommendations for the choice of assessment products and the design of ecosystem service co-effects analysis. Specifically, we emphasize the need to:

- distinguish between processes and functions and the biophysical outcomes (endpoints) they produce or mediate and
- make explicit the relationship between linked functions and processes.

Given these recommendations, how do LC data assessment products fit into comprehensive biophysical production models designed to assess ecosystem services?

Consider first the three data products that are *input measures*: NEP, nitrate retention, and species richness. These three measures are input measures because (a) they are not outcome measures amenable to social or economic evaluation and (b) they are *precursors to* subsequent biophysical outcomes that are amenable to social and economic evaluation.

The CEAF approach asks us to translate these input measures, via ecological process models and additional data collection, into their associated ecological endpoints.

NEP and Species Richness

NEP, as a measure of energy trapped in organic matter, is a likely precursor measure for a range of species-related outcomes related to plant and animal existence, abundance, and location. Species-related endpoints include the abundance of commercially valuable species, the abundance of recreationally valuable species, and the avoidance of extinction events. These kinds of outcomes are socially interpretable and thus economically interpretable. However, the production relationship between NEP and those outcomes requires additional scientific validation. Without that validation, NEP changes cannot be interpreted economically. Consider Figure 8.

Figure 8. Two Data Products and Their Roles in Endpoint Production

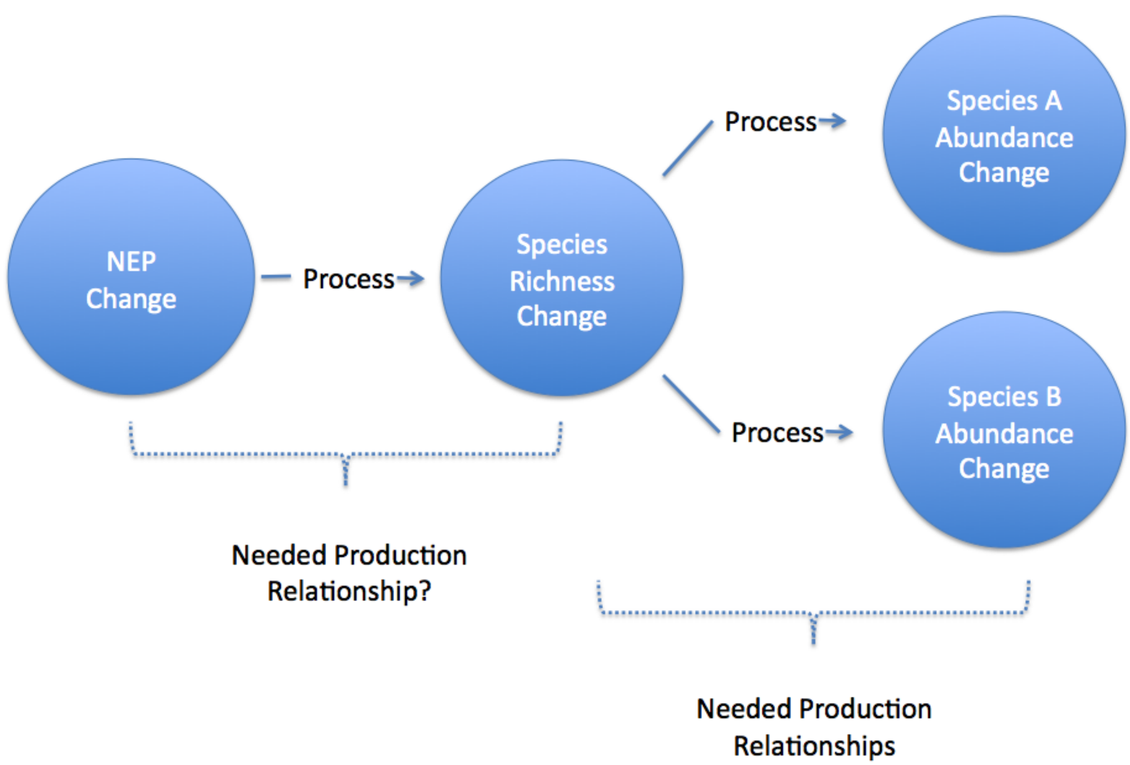
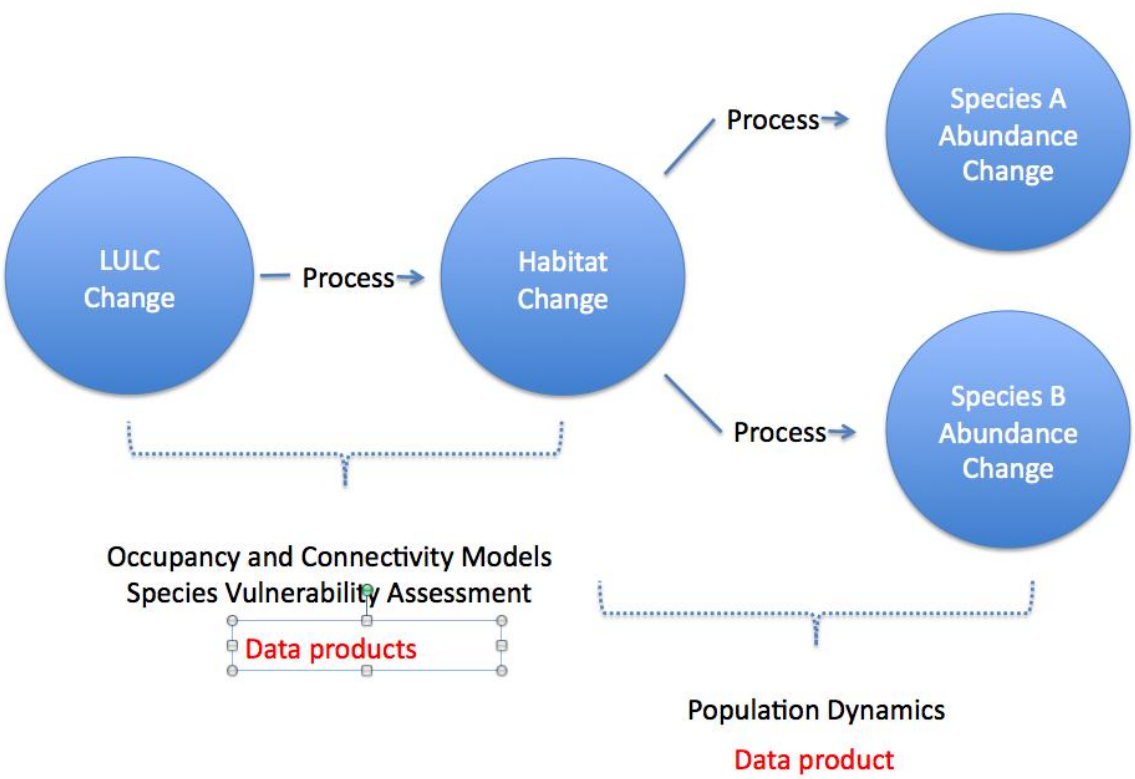


Figure 8 incorporates two LC data products—NEP and species richness—into an ecological production framework. The main point of this figure is that the right-hand side species production outcomes are ecosystem service outcome measures that are amenable to social evaluation. They are the outcomes that stakeholders understand, care about, and to which economic value can be attached. The goal of CEAF analysis is to organize empirical studies that relate LULC scenarios to changes in these outcomes. Because NEP and species richness are already LC data products, it is possible to use them as proxies for species abundance outcomes. However, without calibration of the linking production relationships, these proxies only allow for qualitative conclusions (e.g., more NEP means more species abundance). Quantitative analysis of ecosystem service benefits requires some kind of quantitative correspondence between NEP and a numerical change in species abundance.

A second observation is that NEP and species richness measures are not independent outcome measures. First, NEP may be a precursor measure for species richness. If species richness is ultimately determined to be the key dependent variable that affects species abundance, NEP may be a superfluous outcome measure. In other words, if species richness can be measured directly (rather than inferred from NEP), why measure NEP as an ecosystem service measure at all? Second, NEP and species richness are likely to have interactive effects on species abundance. If so, both should be measured, but they should not be treated as distinctly relevant ecosystem service outcome measures. Rather, they should be described (and interpreted) as part of a vector of factors that predicts species abundance changes.

Figure 9. Process-Related Data Products



Occupancy and Connectivity Models, Species Vulnerability Assessment, and Metapopulation Dynamics

Now consider the role of three other data assessment products from Table 1: occupancy and connectivity models, species vulnerability assessment, and metapopulation dynamics. Figure 9 describes their role in a CEAF assessment. As noted earlier, these data products are not outcome measures, but rather modeling approaches that help describe the production of species-related outcomes. For example, occupancy and connectivity models are used to describe the relationship between LULC changes and the characteristics of species habitats. Do LULC changes lead to spatial patterns of habitat that support species' reproduction, forage, predation, and migratory needs?

Species vulnerability assessments identify species threatened with extinction or significant loss as a result of habitat losses associated with climate change. Vulnerability assessments are important because they illuminate likely habitat losses and help target land use change priorities. However, they do not, by themselves, describe abundance changes. Metapopulation dynamics describes cross-species interdependencies and relates habitat features to the local existence and abundance of species. These types of analyses represent or inform the analysis of biophysical production. They are important and necessary, but they are not ecosystem service endpoints that can be socially or economically interpreted as outcomes.

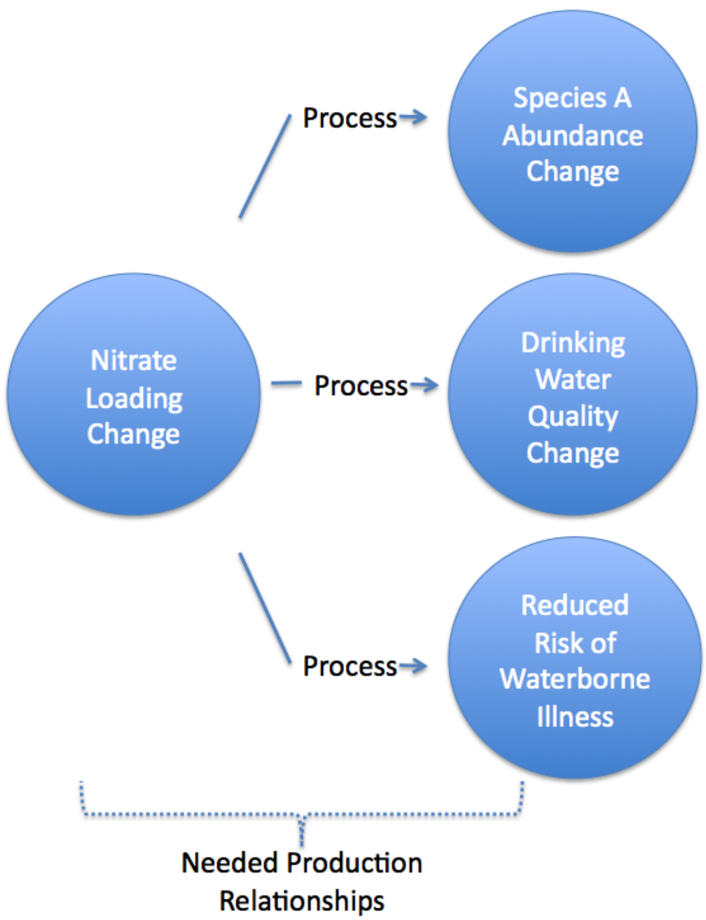
Finally, it is worth noting that the relationship between these process and model products and NEP and species richness measures is not clear. Presumably, there is a relationship between all three of these data products and the species richness outcome measures. The relationship is not made clear, however. Again, a biophysical production framework (CEAF) can be used to clarify these relationships. Input measures (e.g., NEP and species richness) and the process and model products should be described and interpreted as elements of a linked production system designed to assess species-related endpoints.

Nitrate Retention

Nitrate loadings are directly interpretable by one particular set of economic actors: water treatment plant operators with mandates to reduce nitrate loadings. This community can directly translate loadings into control requirements with associated treatment costs. This was the approach taken by Jenkins et al. (2010), who valued reduced nitrate loadings from land conversions using the nitrogen removal costs inferred from hypothetical trades between wastewater treatment plant operators and agricultural nonpoint sources. However, and as the authors note, this is not the preferred method for valuing nitrate reductions. Avoided treatment costs are certainly relevant, but they only capture a fraction of the benefits associated with nitrate removal.

Nitrate retention is an important input measure to co-effects analysis because excessive nitrate loadings affect species abundance, risks of waterborne illness, and drinking water quality. Alone, nitrate retention measures inhibit social and economic evaluation of these beneficial outcomes. Qualitatively, we can say that lower nitrate levels lead to more fish, better drinking water, better aesthetics, and fewer illnesses. If co-effects are to be quantified, however, these relationships require further attention and estimation, as in Figure 10. Figure 10 does not depict all of the endpoints affected by nitrate loadings; instead, it depicts a set of example production functions needed to translate the nitrate loading data product to socially and economically interpretable outcome measures.

Figure 10. Nitrate-Related Production Functions



The remaining LC data products and their relationship to a CEAF are described in the previous subsection. Briefly, however, carbon sequestration and N_2O and CH_4 emissions do not relate to ecological co-effects in any direct way; rather, they are GHG-related outcome measures. Grain and timber production are also not co-effects outcomes, but rather commercial outcomes arising from the LULC scenarios. Soil organic carbon and soil erosion are endpoints of direct relevance to agriculture and silviculture. We note, though, that if soil erosion metrics could be translated into subsequent surface water sediment concentrations and delivery, those outcome measures would facilitate the analysis of co-effects. Sediment delivery outcomes would be of direct relevance to loadings that affect dam and reservoir operations, for example. They could also be used to describe the habitat (and thus species abundance) consequences of sediment delivery.

3.4 Additional Data Products Related to the LULC Scenarios

The previous section describes the role of existing LC data products in CEAF assessment and identifies a set of production relationships and outcome measures needed to evaluate the data products via social and economic analysis. The LULC scenarios produced by the LC assessment will have additional implications for ecosystem service co-effects that are not captured in the existing data products.

Consider the land use changes associated with scenario L in the LC report (Zhu et al. 2010, 43): restore forested wetlands where previously they have been used for agriculture; increase afforestation by converting marginal agricultural land; eliminate deforestation; eliminate the loss of wetlands; increase the time between forest harvests; and reduce rates of clear-cutting. These land use changes will trigger a broader set of ecological endpoint changes than is captured in LC's existing data products.

Table 3. Additional, Desirable LULC-Related Endpoints

Implication of land use change	Example ecological endpoints (outcome measures)	Social and economic relevance
Forest cover affects local air quality via shading and non-GHG pollutant sequestration	Temperature, respirable suspended particulates, ozone, carbon monoxide	Human health, energy usage (temperature)
Forest and wetland cover affects subsurface hydrology	Water table volume, depth	Availability and extraction costs for residential, agricultural, and commercial users of well-drawn water users
Forest and wetland cover affects surface water hydrology	Seasonal flow rates, channel depth and width, probability, depth, and speed of flood events	Affects energy production, recreational opportunities, property damages from flooding, and species abundance

Table 3 describes a range of other biophysical consequences of LULC change, associated endpoints, and their relevance to social and economic evaluation of co-effects. Note that the translation of LULC scenarios into changes in these endpoints will require additional process and production modeling as well as monitoring of the endpoints themselves.

An additional category of social benefits that could be assessed is the impact of LULC scenarios on aesthetics. Forest and wetland cover can both positively and negatively affect recreational benefits, property values, and cultural or community experiences. Aesthetic outcome measures can be derived fairly directly from the LULC scenarios themselves. For example, a community, park, or highway's *viewshed*—based on topographic data analysis—can be intersected with LULC scenarios to generate changes in the "viewable" landscape for different types of users.

3.5 Limitations Imposed by Aggregation at the Omernick Level II Scale

LC data products are produced and reported at a high level of spatial aggregation. With the nation divided into 52 level II ecoregions, outcomes will be depicted at a roughly state-sized resolution. For two basic reasons, this degree of outcome aggregation will significantly inhibit the analysis of ecosystem service co-effects and will limit the policy relevance of the LC assessment. First, the homogenization of biophysical processes and outcomes implied by such a high degree of aggregation undermines the biophysical realism of the assessment. Second, the social benefits of ecosystem service co-effects are highly dependent on the spatial context in which co-effects are delivered to beneficiaries. A third cause of concern is the limited relevance of the ecoregional boundaries themselves for the analysis of ecosystem service production.

We acknowledge that aggregation is important to the practicality of the LC assessment. It is also important to note that coarse aggregation of sequestration outcomes is relatively unimportant. In other words, ecoregion level II aggregation is less of a concern when it comes to sequestration-related outcomes. But this is because sequestration processes (and their benefits) are not as dependent as co-effects on spatial phenomena. Sequestration potential is primarily a function of vegetative species (and their rates of growth) associated with different types of land cover. This makes ecoregional aggregation appropriate because, by design, level II Omernick ecoregions delineate areas with similar vegetation types, qualities, and quantities.

Biophysical Concerns with Level II Aggregation

Co-effects, and the spatial delivery of ecosystem service outcomes, are much more dependent on biophysical processes that are not uniform within a given level II Ecoregion (U.S. EPA 2009). As noted earlier, co-effects analysis requires a translation of LULC features into subsequent ecosystem endpoints via *spatial* biophysical production and process models that describe:

- the dependence of species on the configuration of lands and waters needed for their reproduction, forage, and migration;
- aquifer water availability and quality as a function of subsurface hydrological processes;
- surface water volumes, peak events, and quality as a function of LULC configurations;
- the dependence of aesthetic qualities on LULC spatial configuration; and
- the dependence of soil availability on topology and on hydrologic and land cover features.

Numerous landscape factors influence these production functions. For example, nutrient retention capability is strongly influenced by a range of factors that are highly variable across the landscape (Baker et al. 2006; Litke 1999; Lowrence et al. 1997). In general, the science of ecosystem-based management predicts that the ecological consequences of management actions are *not* limited to targeted species and areas. Rather, “interconnected ecosystems can propagate, amplify, or attenuate site-level actions” (Guichard and Peterson 2009, 74). It is taken as a given that fine-scale spatial analysis is necessary for ecosystem service assessment because their biophysical production functions depend on the landscape context in which those functions and services arise (Bockstael 1996). Recent empirical studies confirm this. For example, a study of Oregon’s Willamette Basin found that the spatial pattern of development and conservation dramatically altered the economic and ecological outputs provided by alternate landscapes (Polasky et al. 2008).

Conservation biology, for example, emphasizes the importance of habitat connectivity and contiguity to the productivity and quality of that habitat, measured through species diversity, richness, or other measures (Noss 1990; Gardner et al. 1993; Gustafson 1998; Richards et al. 1996). Terms like *connectivity* and *contiguity* are inherently spatial. They refer to the overall pattern of land uses, surface waters, and topographic characteristics in a given location or region. Often, a minimum size and connections or pathways to other resources are needed to support migration, reproduction, and foraging (Flather and Sauer 1996; Roberts et al. 2001; Green et al. 2007). Wetlands filtering nutrients in riparian zones have been shown to have a greater ability to prevent nutrient deposition than wetlands further inland (Lowrence et al. 1997; Correll et al. 1992). Moreover, threats to biodiversity tend to be a function of the spatial configuration of nonnatural land uses. For example, the proportion of a watershed covered by impervious surfaces is a known risk factor for aquatic habitats, as impervious surfaces create greater runoff volumes and shorter runoff times, leading to more pollutant deposition and warmer surface waters (Soil Conservation Service 1975).

These kinds of factors that so strongly influence the production of co-effects cannot be effectively evaluated based on uniform outcomes expressed at the scale of a level II ecoregion. Moreover, the spatial production of ecosystem services will routinely cross Omernick regional boundaries (the delineation of these regions is not based on or motivated by ecosystem service production). In particular, water- and species-related outcomes will be “exported” across regional boundaries. This will tend to confuse the interpretation of LC data outcomes because it does not illuminate the dependence of co-effects outcomes in one region on LULC factors in another. Finally, ecosystem service production in one region will also be affected by “imports” (e.g., water deliveries or species movement) from other regions. In effect, the ecoregional reporting construct arbitrarily divides systems of ecosystem service production in a way that confuses, rather than facilitates, analysis of biophysical processes and functions.

Economic and Policy Concerns with Level II Aggregation

The spatial location of GHG sequestration does not affect its value. A ton of carbon sequestered in Indiana has the same beneficial effect on climate processes as a ton sequestered in Montana. For this reason, level II aggregation of sequestration outcomes is not a concern for economic or policy analysis. However, the value of ecosystem services is highly dependent on the features (social and biophysical) of the landscape in which they are delivered. As economic commodities, EGS resemble real estate, rather than cars or rolls of steel. The value of real estate is highly dependent on its location—specifically, the features of the surrounding neighborhood. This is because (a) a given house or building cannot be easily transported to another neighborhood and (b) the house's value is dependent on location-specific variables (parks, schools, and shops) that are also immobile. In contrast, cars and rolls of steel can be easily transported, so their value tends to be independent of their location.

Section 4 (below) describes in more detail the range of location-specific social and biophysical factors that affect ecosystem service benefits. These factors include the number of beneficiaries with access to the good or service and the relative scarcity of the service in a given location. Unlike tons of GHG sequestered, the value of ecosystem services is closely tied to co-location with the populations and economic activities they support. And like any other good or service, an ecosystem service is a function of its scarcity, available substitutes, and complementary inputs. Co-effects benefits are often location-dependent because substitutes and complements are themselves not transportable. For example, if a lake is to have recreational value, people must have access to it. In other words, the lake must be spatially bundled with infrastructure—roads, trails, and parks—that are themselves not transportable. Substitutes for a given recreational experience depend on a recreator's ability to reach them in a similar amount of time. Thus, the location of nonfungible substitutes is important.16 The value of surface water irrigation is a function of the location and timing of alternative, subsurface water sources. If wetlands are plentiful in an area, then a given wetland may be less valuable as a source of flood pulse attenuation than it might be in a region in which it is the only such resource.

The scale at which these spatial factors are relevant depends on the specific service being valued, but are typically quite local, such as the scale of a particular farm, neighborhood, park, or

[16] An important issue in travel cost studies, for example, is the definition of relevant substitutes for the sites in question. See Arrow et al. (1993, 4608): "omitting the prices and qualities of relevant substitutes will bias the resource valuations."

business. In some cases, the relevant scale at which landscape factors matter will be broader. An example is certain recreation-related services where household members may drive an hour or more to enjoy a park or beach. The landscape scale over which to evaluate substitutes for this service is thus similar parks and beaches within an hour's drive. Even so, the scale of these ecosystem service areas is of much finer resolution than a level II assessment will permit.

Accordingly, level II outcome aggregation will not permit the evaluation of spatial factors that affect the value of ecosystem services. Importantly, it will also not permit the identification of specific beneficiaries. From a public policy standpoint, this is of great concern. If the LC assessment is to be of use to natural resource managers, communities, businesses, and other stakeholders, finer-resolution outcome measures are essential. If stakeholders cannot identify the co-location of co-effects with specific groups of beneficiaries, social and policy evaluation of co-effects cannot occur.

Put differently, LC's co-effects outcomes are not reported at a resolution consistent with the resolution at which real-world planning and policy evaluation occur. Most governmental, private sector, conservation, and household planning occur at a parcel-level scale. Why is this? Because the parcel-level scale is the scale at which actual policy, business, and economic decisions are made. Until LC outcomes can be delivered at that resolution, their relevance to policy and economic planning will be limited.

3.6 Planned Case Studies and Their Relationship to the CEAF

Zhu et al. (2010) identify a set of possible future case studies through which to explore ecosystem service co-effects in more detail. These case study regions will probably include the Mississippi Alluvial Valley, Prairie Pothole Region, southern Florida, and the Chesapeake Bay watershed. Because these case studies are still under development, and because details of their execution are not presented in the LC methodology, it is difficult to comment substantively and with specificity on their relationship to the CEAF approach. However, there is reason to believe that these case studies and their eventual data products will much more closely correspond to the principles and objectives advocated in this paper.

Consistent with the conclusions of this study, the case studies are motivated by "the need to have regionally specific information and our limited understanding of the complex relationships among ecosystem processes, land management actions, climate change and ecosystem services" (Zhu et al. 2010, 60). The descriptions of the case studies also refer to the construction of "biophysical production functions" (60) and analytical platforms to "better understand biophysical response and tradeoff analyses" (138). The "distributed geospatial-model-sharing platform" described in Appendix F (though not described in enough detail to evaluate with specificity) strikes us as a highly promising strategy given that it is motivated by the need to "share and integrate geospatial disciplinary models" (173). The integration and sharing of such models is necessary for the quantitative depiction of the biophysical production models central to the CEAF approach.

Also, it is notable that specific mention is made of LC linkages to models such as the Soil and Water Assessment Tool and the use of this model to estimate outcomes including soil erosion, groundwater recharge, water flows, and sediment and nutrient delivery across aquatic systems. These kinds of outcomes more closely correspond to the endpoints concept we describe. This is particularly true because the case studies description appears to emphasize the spatially explicit analysis of service delivery.

Some of the specific examples given do remain a concern for us. For example, the case studies description advocates the use of "duck energy days" (the amount of energy required by one mallard-sized duck for one day) as an ecosystem service outcome measure. Duck energy days is a classic example of an ecosystem service outcome measure that is inconsistent with social and economic evaluation. We understand that it is a computable measure with qualitative relevance to duck abundance (i.e., more duck energy days implies more ducks). But it cannot be given quantitative economic relevance unless it can be subsequently—and quantitatively—translated into an outcome measure that is economically interpretable and comprehensible to stakeholders (i.e., increased duck abundance).

3.7 Summary of Data Assessment Products and Co-effects Assessment

This review of LC's data products suggests that co-effects analysis will be significantly constrained given the current portfolio of outcome measures. This is understandable considering (a) the huge challenge posed by the LC effort generally and (b) the lack of "off-the-shelf" data products and models that could be easily and directly applied to co-effects analysis. This review has identified a set of modeling and measurement gaps that, if filled, could leverage LC data products into a more robust assessment of co-effects. Until those gaps are filled, however, expectations regarding the ability of analysts to translate LC data products into ecosystem service analyses should be minimized. Most of the outcomes claimed as ecosystem service outcomes do not in fact allow for social or economic evaluation of co-effects. Also, the way in which ecosystem data products are presented and motivated in the LC plan suggest that USGS would benefit from a strategic reorganization of its co-effects efforts based around an ecosystem service production architecture (à la CEAF) and its modeling and measurement implications.

We feel much more positively about the proposed case studies and their apparent aspirations to (a) describe a wider range of ecological consequences associated with the LULC, disturbance, and climate change scenarios; (b) develop additional biophysical production relationships via integrated modeling and measurement; and (c) describe the delivery of ecological changes with greater spatial specificity. These aspirations more closely correspond to the needs and expectations of stakeholders and policymakers and appear to be more consistent with the CEAF described in this paper.

4. Social and Economic Assessment of Endpoint Changes

This section describes how changes in ecosystem endpoints can be socially and economically evaluated. Social and economic evaluation serves several purposes. First, it helps decisionmakers understand the benefits of desirable endpoint changes (and the costs of undesirable changes) by focusing on how specific ecological outcomes contribute to economic production and household and community wellbeing. Economic valuation can help us see potentially undervalued ecological services, which otherwise may be underappreciated, in policy or management deliberations. Second, social and economic evaluation describes the *relative* benefits and costs of alternative policy choices and management scenarios.

Ecosystem-based management inevitably requires trade-off assessment and priority setting. Resource management decisions never result in a single ecosystem consequence (i.e., a change in a single ecosystem endpoint). Invariably, they create a diverse array of incommensurate ecological outcomes whose relative importance must be evaluated (Barbier 2009). For example, changes in LULC will trigger a range of changes in ecosystem endpoints, as described in the previous section. Some of these changes will be more important than others. How are we to assess relative importance and set priorities? A primary goal of public policy is to “do the greatest good for the greatest number.” Economic assessment helps us rank these various ecological outcomes, communicate and assess trade-offs among them, and determine distributional impacts. Without economic assessment (or some kind of social evaluation) there is no way to rank or prioritize among management options.

We describe below two strategies for conducting economic assessment of endpoint changes: (a) monetary valuation of endpoint changes and (b) application of EBIs as a substitute for, or complement to, monetary valuations.

4.1 Application of Economic Valuation Studies to Biophysical Production Analysis

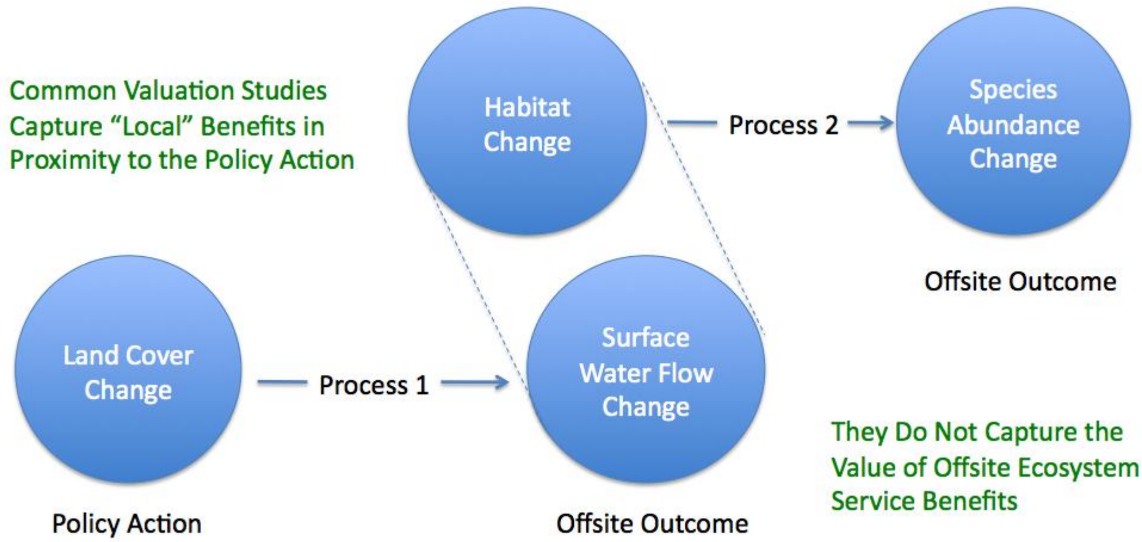
The first, traditional approach to economic assessment is to apply existing, or conduct new, monetary valuation studies to biophysical scenarios. We emphasize throughout this report that economic evaluation is conducted in reference to *changes* in ecological outcomes, rather than being applied to the value of an entire system. Why is this? First, the goal of economic assessment is to prioritize among policy-relevant choices. Policy-relevant choices trigger discrete changes in ecological systems; they do not involve the addition or removal of entire systems of ecological production. Second, economic valuations infer value by looking at revealed human behavior and responses to discrete choices or alternatives or by surveying responses to *choice scenarios*. Again, the focus of economic assessment is on the evaluation of plausible choices or scenarios that bear at least a resemblance to plausible changes in ecological conditions.

A fairly large set of existing economic valuation studies place dollar values on wetlands, open space, forest, and other types of land use change. These studies use the techniques described in Section 2, in some cases looking at property values in proximity to forests, wetlands, or open space, or evaluating travel and expenditure behavior related to recreation in a particular type of resource area. These types of valuation studies are an important piece of the evaluation puzzle because they (a) reveal that forests, wetlands, and other land cover types are economically valuable and (b) can help identify areas where they are *most* valuable.

However, care must be exercised when existing valuations are applied to a given ecological resource or change in that resource. For example, the amenity value of open space to recreators who travel to visit it or commuters who enjoy the view on the way to work will not be capitalized into housing values. Nor will the value of the open space as an input to the production of services (e.g., species or water quality) that are enjoyed further afield. It is therefore important to understand that valuation studies may capture only a fraction of the total value of these resources. They may capture only the benefits to particular user groups (e.g., neighboring households, hunters, and birders), and they may capture only the direct, proximate benefits of the resource.

Refer again to the following simplified depiction of an ecosystem service production system (Figure 11). Valuation studies often detect the value of the policy action to neighboring households or businesses (in the case of hedonic analysis) or recreators who travel to the site (in the case of travel cost methods). In other words, they capture a part of the on-site benefits generated by a policy action. But by themselves, such studies do not measure the full ecosystem service benefits associated with the site's role in spatial biophysical production.

Figure 11. The Limits of Certain Valuation Studies



Consider a hedonic analysis that finds a price premium for houses in proximity to a wetland. Are all of the wetlands benefits (those associated with open space, water quality improvements, and crab abundance) reflected in the hedonic premium? In the case of housing, the example system would identify the wetland's open space endpoint as being likely to matter to the utility of nearby homeowners. In effect, there is a clear linkage between the market commodity (housing) and related consumption of open space. In contrast, the other two wetland endpoints are not likely to appear in the value of the market good. This is true for several reasons. First, the role of wetlands in the production of less flashy hydrographs and crabs may not be known to homebuyers. Second, even if they are known, the benefits they produce may not be enjoyed by local households. Improved flood risk profiles, water quality, or crab abundance may occur far from the households in question (e.g., far downstream), in which case the value will not appear in home values and thus will not be detected as benefits.

To be clear, the value of off-site ecosystem endpoint improvements can be valued economically. But they must be evaluated via a production systems approach to ecosystem service analysis in which spatial biophysical production is taken into account. This is the motivation for so-called total economic valuation (TEV) assessments, which evaluate natural resources in a more comprehensive way. They are explicitly designed to value all the welfare consequences of a given policy scenario or choice. In practice, each endpoint change triggered by a given LULC scenario must be valued independently and with knowledge of the delivery of the various endpoints to often very different groups of beneficiaries. In the above example, the impact of the land cover change on local aesthetics and downstream water quality improvements—and the subsequent impact on species abundance—involves three distinct valuation exercises involving three distinct groups of beneficiaries. Stated preference studies are more amenable to the TEV approach than revealed preference studies because they present survey respondents with hypothetical choices designed to capture a wider range of ecological outcomes.

We also note the limited relevance of certain valuation studies that put a dollar value on highly aggregated ecological systems and claim to derive a value for ecosystems in a particular region, state, or country. Arguably, these are the most well-publicized "value of nature" studies, but they should be interpreted with great caution. Examples include Costanza et al. (1997, 2006), who place an economic value of $33 trillion on the world's ecosystems and $18 billion on New Jersey's ecosystems. The researchers established per-acre dollar values—using existing economic estimates from the academic literature—for a particular set of land types (wetlands, croplands, grassland, and green spaces). They derive their total value estimate by multiplying these dollar values by the total acreage of the particular land use in the relevant region. These studies are useful in their ability to capture the public's imagination, stimulate discussion, and convey the notion that nature's value can be described in monetary terms. From a scientific standpoint, however, they are not broadly accepted. First, they do not account for location-specific ecological or economic factors. Second, and more importantly, the analysis combines separately measured values for individual resources without accounting for the fact that the aggregate value of those resources is not equal to the sum of the individual parts. Third, willingness to pay for such resources is limited by people's ability to pay (Bockstael et al. 2000); notably, Costanza et al.'s measure of value exceeds global income by a wide margin. These studies illustrate the dangers of overly simplified benefit transfers for valuation and highlight the need for analysis of marginal changes in ecosystem services delivered—in other words, decision-relevant changes in ecosystem endpoints.

4.2 The Application of EBIs

Analysts can also evaluate social benefits using indicators of benefits that stop short of monetary valuation. Monetary valuation requires the use of data and methods that substantially add to the assessment burden. As noted above, analysis of the total economic value associated with ecosystem service production typically requires the application of multiple econometric studies designed to detect benefits for distinct beneficiary groups. Because of the cost of such studies, it is common to see only a single environmental benefit monetized (though stated preference experiments can get around this problem). EBIs, an alternative to monetary valuation studies, are based on existing, publicly available data that can be relatively quickly applied to ecosystem endpoint assessments.

EBIs are quantitative, countable features of the physical and social landscape that depict the ways in which ecological endpoint changes produce changes in human welfare. The relevance of EBIs to benefit assessment is motivated by the basic principles of economics (as described in Section 2). They help describe (a) the distribution of ecosystem service benefits to different populations and (b) the scale of demand for a given ecosystem endpoint. In addition, they help rank choices by describing economically relevant factors, such as the scarcity of the endpoint, substitutes for the endpoints, and goods and services that are complementary to—or necessary for—enjoyment of the service.

To illustrate the use of EBIs, refer to the endpoints associated with nitrate reductions depicted in Figure 10: greater abundance of a given species, improved subsurface drinking water quality, and reduced risk of illness from surface water contact. Although measuring changes in these endpoints is likely to be quite challenging, the benefit indicators described in Table 4 are fairly easy to derive from existing geospatial land cover and census data sets or from existing assessments (e.g., recreational usage data) and can therefore be easily applied to social assessment.

Table 4. EBIs Relevant to Benefits Associated with Endpoint Changes

Species abundance change	**Subsurface water quality change**	**Reduced risk of (surface) waterborne illness**
	Demand indicators	
Number of recreational users with access to species population Usage data (e.g., hunting licenses and parks and public lands visitation data)	Number of households drawing well water from affected aquifer	Population density in proximity to water body Recreational usage data (e.g., boating permits, fishing licenses, and beach visitation data)
	Scarcity and substitutability indicators	
Global or regional rarity/abundance of relevant species	Availability of/proximity to public (treated) water sources	Presence of other water bodies in proximity to affected site
	Complementary good indicators	
Infrastructure allowing access to species (e.g., trails, roads, docks, and boat ramps) Land uses allowing access (e.g., public lands, parks navigable waters, and beaches)		Infrastructure allowing access to waters (e.g., trails, roads, docks, and boat ramps) Land uses allowing access (e.g., public lands, parks navigable waters, and beaches)

Note that these indicators describe the relationship between an endpoint change, its spatial (social and biophysical) context, and the benefits that result from the endpoint change. Hypothetically, consider an LULC scenario or management choice that results in an increase in species abundance in either location A or B. Which location yields the greatest social benefit? EBIs help answer that question by:

- ranking the choice in terms of the number of beneficiaries affected (location A is preferred, all else being equal, if more beneficiaries are affected);
- relating the ecological change to the service's scarcity and the availability of substitute services (location A is preferred, all else being equal, if the delivered service is in shorter supply or if fewer substitutes are available); and
- relating the ecological change to the presence of complementary assets that enhance, or are necessary for, the enjoyment of the service (location A is preferred, all else being equal, if it is accompanied by more abundant complementary access infrastructure or land uses).

As should be clear from this example, EBIs do not necessarily lead to a clear choice or ranking of policy choices. Rather, they inform the choice by providing stakeholders and resource managers with data that are relevant to the benefits delivered. Linked to specific ecological endpoints, they can allow for more comprehensive evaluation of multiple goods and services given limited budgets for analysis.

Other examples of EBIs related to different ecosystem service endpoints include data related to the benefits of flood risk mitigation. For example, the benefits of endpoint changes in the probability, depth, and speed of flood events are a function of indicators such as:

- the number of housing and commercial units,
- the value of those housing and commercial units,
- the presence and value of other infrastructure subject to flood damage (e.g., roads and bridges), and
- the presence and value of crops vulnerable to flooding.

All else being equal, the greater the number and value of properties protected, the greater the value of the service delivered.

EBIs can also be used to assess the mitigation potential of a given LULC scenario. As noted earlier, an important co-effect to be analyzed is the impact of the LULC scenarios on nitrate loadings. Nutrient loads are a function of both current and historic land uses, and we note that decadal lags can occur between nutrient applications and groundwater effects. However, EBIs can nevertheless be a useful screening device to target areas in need of nutrient capture and cycling. The more impaired the received runoff (currently or historically), the greater the mitigation project's likely benefits. Location-specific EBIs can help depict the quality of waters received by a forest or wetland. Measurable indicators related to the likelihood of nitrate loadings include:

- the percentage of crop or pasture land in the vicinity of a mitigation project;
- the percentage of the source watershed in crop or pasture land; and
- the existence of specific water quality threats in the vicinity of the watershed, including concentrated animal feeding operations or landfills.

EBIs can also be used to depict the scarcity of a given function, such as nitrate retention or removal, in a given area. If nearby forested wetlands are very abundant, for example, the loss of one area may not lead to a significant loss of water quality benefits. But if wetlands are scarce, the benefits lost will tend to be more significant. Measures that speak to the role of scarcity include:

- the percentage of land cover in wetland, both locally and across the watershed, and
- the percentage of nonagricultural natural land cover in the watershed.

EBIs help illuminate the portfolio of changes associated with ecosystem service co-effects. EBI analysis fosters an appreciation of the way in which ecological functions are related to the biophysical characteristics of the larger landscape. Second, landscape analysis highlights the human dimension of the surrounding environment. Third, these kinds of landscape factors can help rank and prioritize policy choices by both extremely good and extremely poor landscape scenarios and by identifying (if not resolving) important trade-offs.

Finally, we note that when endpoints are presented along with EBI information, the social importance of ecological outcomes can be communicated in a way that is nontechnical, but nevertheless ecologically and economically substantive. Consider a set of hypothetical examples, where a proposed reforestation or afforestation change will:

- improve the aesthetic environment by adding Y square miles of forested land cover (an endpoint) viewable by X households and commuters (an EBI) in a viewshed where Z percent of the landscape is developed (an EBI);

- lead to healthier communities by reducing concentrations of coarse particulate matter (via afforestation) by Y percent (an endpoint) in an airshed with X children, seniors, and at-risk adults (an EBI);
- improve recreational opportunities by adding Y mature sport fish (an endpoint) to a watershed fished by X license holders (an EBI) in a region where it is the only fishable river (an EBI);
- lower agriculture's irrigation costs by increasing aquifer recharge volumes by Y (an endpoint) used by X acres of farmland (an EBI) producing Z million dollars of output per year (an EBI); and
- reduce expected flood damages by reducing the probability of a major flood by Y percent (an endpoint) along a river reach with X exposed residential and commercial structures (an EBI) and crops worth Z dollars (an EBI).

For policymakers and land resource managers, these kinds of quantitative outcomes can be as—if not more—powerful than monetary valuations. EBIs are not a substitute for traditional economic valuation studies, but they are a cost-effective way to inform stakeholders and decisionmakers so that socially beneficial priorities can be set and trade-offs resolved.

5. Stylized Valuation Study and Issues of Aggregation

5.1 Introduction

The previous sections in this report discuss, in general form, various economic valuation approaches that exist and have been applied successfully. At this point we ask, what would constitute a research plan to value changes in ecosystem services that result as a co-effect of carbon sequestration? Additionally, what are some of the issues associated with aggregating to a regional-scale assessment?

Of the several technical economic approaches suggested in Section 2, here we focus on a stylized framework for the stated preference approach. Why this approach? In our opinion, this technique (a) is the most robust approach for application over a variety of landscapes and beneficiaries, (b) allows for monetization, (c) enhances the policymaker's ability to make choices, and (d) serves as a valuation building block for a regional assessment. A set of consistently performed stated localized preference studies would ultimately allow for the aggregation of localized studies into regional estimates. We suggest developing a bottom-up aggregation approach for ecosystem services to be consistent with regional sequestration estimation.

Consider the conceptual issues from the previous sections.

- All ecosystem services are not appropriate for valuation; only ecological endpoints (a subset of ecosystem services) are appropriate.
- The presentation of ecological endpoints should be derived from biophysical relationships (coupled models).
- It is *changes* to ecological endpoints that are of interest; economic valuation does not value entire systems, but rather marginal changes to a system.
- The valuation effort should be localized in nature, given the uniqueness of the interrelationships among biological processes and the need for individuals to be able to address the valuation of endpoints cognitively.

5.2 A Framework: Overview

What is the value of changes in ecological endpoints if carbon sequestration is undertaken in a particular region of the United States (e.g., the Southwest)? This question assumes that values vary by region: both because biophysical production processes may vary and because values are place-dependent.

Although this question is of interest from a regional policy perspective, it remains too broad for a valuation study. As noted in the earlier sections, valuation depends on a localized set of identified changes in certain ecological endpoints. Thus, we ask the following questions.

1. What is (are) the candidate subarea(s) for carbon sequestration within the identified ecoregion?
2. Among these land units, is the best available science of the biophysical processes available in all locations, or are the scientific data of varying quality? Not only does this issue increase uncertainty in the estimation of sequestration, it also affects economic estimation.
3. Are the biophysical data transferable to other subareas in the same ecoregion?
4. As a corollary to question 3, are the subareas sufficiently similar that an aggregation effort can be made for policy purposes? Further, does this collection of subareas reflect the original ecoregion, or was the initial choice of region too large?
5. What are the relevant ecological endpoints in each subarea of a level II ecoregion?

6. In applying the stated preference approach, should the analyst use

    a. CV, which usually evaluates a single policy change for a single ecological endpoint, or

    b. A choice framework, which values multiple ecological endpoints simultaneously and provides marginal values?

7. In implementing the framework, should the survey valuation sample be only for users of that subarea, for the region as a whole, or for the nation? This question is related to the categories of benefits being evaluated. Studies focused on beneficiaries beyond resources users require consideration of option and existence values, in addition to use values.

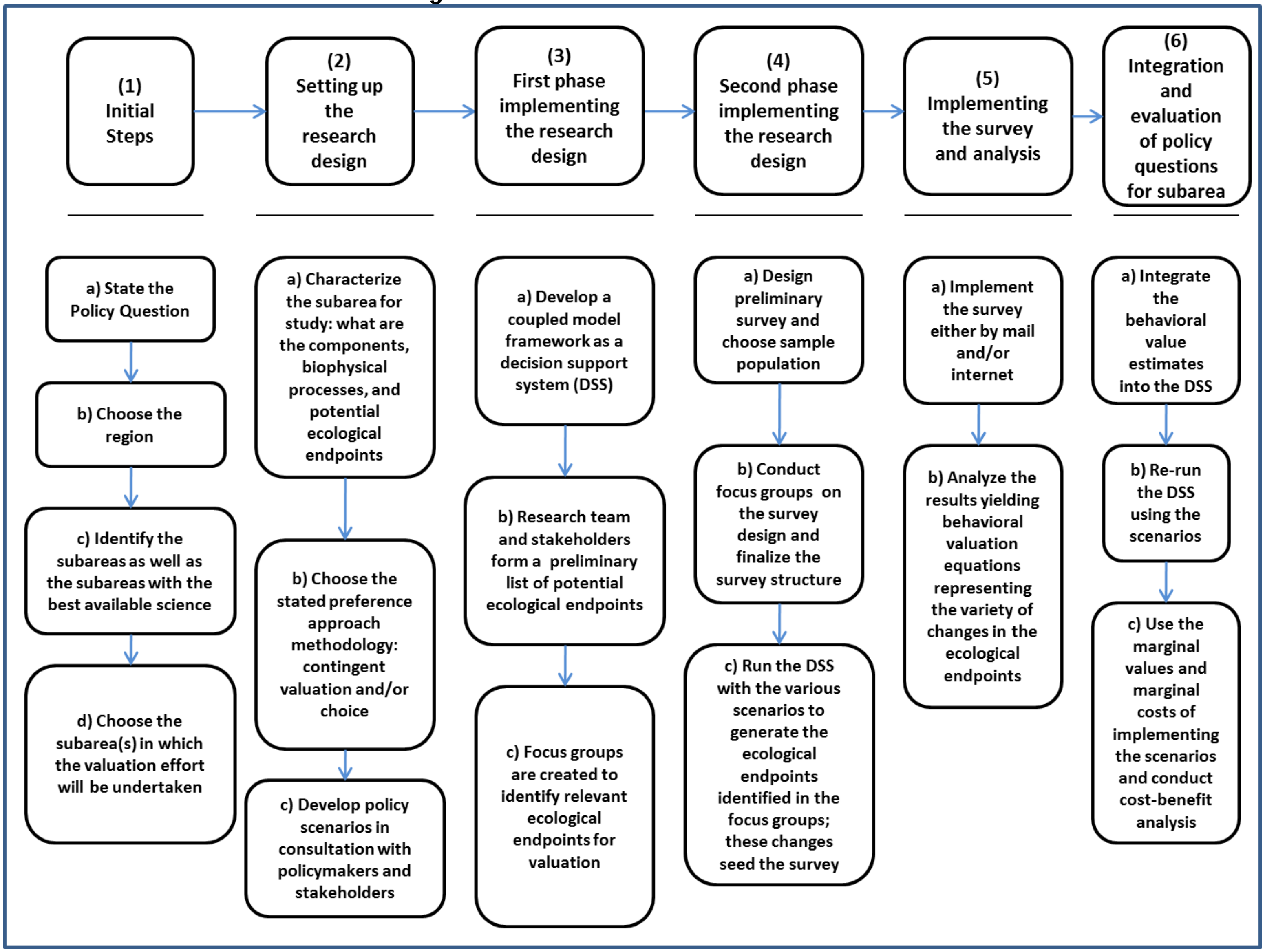
8. What is the temporal framework for changes in an ecoregion—is it months, years, decades, or a specified date, say 2050? This question reflects a need for LC to provide not only gross measures of ecological change, but rates of change, as well.

5.3 The Structure of the Framework: Coupling the Parts of a Valuation Study

We begin by adapting Figure 6 to reflect the eight questions just identified. From Figure 6, we discuss only the flows that go from left to right, passing through the monetary valuation box. We expand various parts of the figure into a complete step-wise discussion of the components of an economic valuation framework.

In Figure 12, we divide the valuation effort into six tasks (columns) with associated subtasks under each column. Although the effort and tasks could be disaggregated further, or possibly arranged differently, our approach is an attempt to be reasonably transparent in identifying the major phases of a valuation effort. The figure helps draw the broad outlines of such an effort.

Figure 12. Valuation Tasks and Subtask



Column 1: Initial Steps. Column 1 (C1) is the framing of the research. It involves the choice of the policy question and the definition of the subarea(s) for which the valuation must be undertaken.

Box C1-a: Essential to a valuation study is the statement of the policy question(s) that is (are) to be answered. This flow diagram is set up to answer the question, what are the benefits and costs of alternative carbon sequestration management regimes (see box C6-c). It is essential that the policy question be clearly defined. The source of the policy question could stem inherently from legislation and/or agency interest. Stakeholders and researchers can also contribute to a clear statement of the problem.

Box C1-b: The next step is to choose the region of study. The initial choice of a region can be rather broad, such as an ecoregion. Regions that are defined by ecological "boundaries" (rather than political or jurisdictional boundaries) are desirable. Regions defined by watershed boundaries, for example, place relatively clear, defensible bounds on the system to be analyzed. Also, watersheds are a common focus of environmental regulation, planning, and policy. Eco-regional classification systems draw ecological boundaries differently, based on – for example – land cover, climate, and habitat features. These boundaries are also useful to ecosystem service analysis because they identify ecologically distinct resources and systems. This is important to the extrapolation of results across a region. Extrapolation of results within a given eco-region is easier and more defensible than extrapolation across eco-regional types.

Box C1-c: Once the region has been chosen, subareas should be chosen because, as noted earlier in this report, economic analysis is more appropriately applied when the scale of the valuation analysis is localized. It is assumed that this step in Figure 12 includes the identification of all potential subareas with the chosen region.

How would a set of subareas be chosen? There are two criteria: discernable ecological endpoints (e.g., what a stakeholder would understand) and the understanding of the biophysical science. Regarding the first criterion, can we imagine a stakeholder understanding changes in the variety of ecosystems within the Mississippi River basin as a whole? Probably not, but from a behavioral point of view there are quite possibly stretches of the river that a stakeholder would consider a coherent whole. That is, they might visit the area and might experience the endpoints; and if the endpoints changed, they would be able to perceive and express a corresponding change in their wellbeing..

The second criterion for a candidate subarea is that the ecosystem's biophysical processes are reasonably well defined. What is the underlying scientific understanding of ecosystem processes? What natural science studies are available and, further, to what degree are they useful to the policy question being addressed? This is critical, as the valuation effort relies on, and is only as good as, the underlying science and its applicability. Further, if the goal of transferring the values to other subareas and/or aggregation is a potential consideration, then the choice of the subarea(s) with the best available science will enhance this effort by ensuring that, at least initially, the uncertainties of the science and the values are minimized to the extent possible.

Box C1-d: Two broad factors go into the final choice of the subarea(s), from the overall set, for which the valuation effort will be undertaken: the availability of the science and whether the area is known to and used by the public. The most problematic circumstance is a subarea that has the best available science yet is relatively unknown to or unappreciated by the public. The ideal combination is a subarea that has the best available science and is used, enjoyed, and appreciated by a wide range of beneficiaries.

Column 2: Setting up the Research Design. The subtasks in Column 2 lay out in broad terms the necessary decisions that must be made for the overall research design. These include characterizing the selected subarea, choosing the stated preference methodology, and finally developing policy scenarios that fall within the frame of the overall policy question (C1-a).

Box C2-a: After the subarea has been chosen, it is formally characterized and inventoried. Importantly, this box represents the initial identification of a set of ecological endpoints. In some sense, this has begun in the course of identifying the area. The biophysical data products in LC should be considered as potential ecological endpoints. The final assessment is an iterative process that should involve stakeholders.

Box C2-b: Here we consider the nonmarket valuation technique of the stated preference approach. Two broad candidates within this approach are CV and choice modeling (CM). In its simplest form, CM elicits an individual's preferences by asking the subject to consider an indication of current conditions as represented by a bundle of specific ecosystem service attributes relative to an alternative bundle. This decision process is repeated multiple times. From this information, the researcher may infer the marginal value (i.e., the value associated with the ecosystem attribute) for the various ecosystem attributes individually. CV, on the other hand, asks individuals to explicitly state their willingness to pay for a proposed change in a single ecosystem attribute. The outcome of these approaches will yield average or marginal dollar values for changes in ecological endpoints.

Box C2-c: In implementing stated preference approaches, a set of scenarios is required from LC or elsewhere. Although not simple to identify, in principal they are simply alternative future courses of management actions (e.g., sequester X tons of carbon in area Y for a period Z). Carbon sequestration management scenarios must be realistic, practicable, and acceptable to policymakers and stakeholders. The scenarios must be specific enough to generate ecosystem service changes, and thus endpoints, through a decision support system (DSS) framework. That is, a management option is undertaken at a particular time, continued for a specific period within the subarea at a certain effort level or economic scale. Additionally, the cost of the management option should be estimated or obtained from other sources for cost–benefit analysis (C6-c).

Column 3: First Phase of Implementing the Research Design. This first phase is an integration of the beginning of the research design and the initial introduction of specific scenarios to stakeholders.

Box C3-a: A DSS must be designed. Referring back to Figure 6, this step would involve an expansion of the boxes labeled "Biophysical production." Thus the "Land use and land cover changes" box is actually a series of coupled biophysical models. For river systems, these might include models of the groundwater and surface water systems, riparian vegetation, and avian species in the area. It is critically important that the models be linked or coupled, ultimately producing the endpoints delineated in Figure 6.

Box C3-b: As the DSS framework is developed, ecological endpoints are identified, defined, and quantified. Although a preliminary list was developed in C2-a, the production of ecological endpoints involves the interaction of systems that will differ across different landscapes. Thus, in the coupling of the models (e.g., LC models) the relationships between biophysical inputs and outcomes will become more apparent for a particular subarea.

Box C3-c: While the outcomes of the DSS are being characterized as LC data outputs, focus groups will evaluate the potential endpoints. This is a critical step as these endpoints are the central focus of the valuation exercise.

At this point an illustration is helpful. Consider a carbon sequestration project that alters the riparian vegetation of a river and, in so doing, alters the abundance of birds. But what part of the abundance measure is important to stakeholders? Is it the migratory, water-bound birds, and/or the nesting birds? All of these are ecological endpoints of the DSS, but not all of them may be of interest to the stakeholders. Thus, the use of focus groups narrows down the possibilities for inclusion in the survey.

Column 4: Second Phase of Implementing the Research Design: A Survey Instrument. Issues in the second phase of valuation include designing the valuation instrument, obtaining the reaction of the focus group to the preliminary design, and running the DSS to generate the chosen ecological endpoints.

Box C4-a: The survey has multiple component parts: a broad introduction to the policy question, an education component that details the underlying science of the problem in an understandable manner, a valuation section, and a socioeconomic data section.

Also, this effort includes the choice of respondents to comprise the sample. For instance, one must decide whether the sample will include only individuals who use a specific subarea, or individuals living in the region as a whole, regardless of use. The survey should be designed by the social scientist and the biophysical scientist, ideally with the contribution of a science writer. An important aspect of this process is to make sure that the survey instrument is true to the science yet understandable to the public.

Box C4-b: After the social scientist and the biophysical scientist have vetted the survey instrument, a series of focus groups should be conducted. The focus groups will help to further refine the language of the survey. As part of this process, the focus group could be asked to examine the chosen ecological endpoints (C3-c).

Box C4-c: After the survey is finalized, including the selection of the ecological endpoints, the DSS is run with the scenarios (C2-c). The result of the DSS for different scenarios and the changes in the endpoints are determined. It is these changes that are the focus of the valuation effort and are embedded in the choice question in the survey.

Column 5: Implementing the Survey and Analysis. This stage is straightforward as it involves implementation of the survey and analysis of the resulting data. It can take up to three months to implement the survey following the appropriate survey protocols.

Box C5-a: The survey instrument can be implemented by mail, Internet, or a hybrid of both. Central to the implementation is the use of an appropriate sampling methodology, such as the Dillman (2000) method. The Dillman method is an approach for contacting potential respondents and for conducting associated follow-ups to ensure an appropriate response rate.

Box C5-b: After the survey responses have been assembled into a data set, appropriate econometric tools are used to estimate the values for the changes in the ecological endpoints.

Column 6: Integration and Evaluation of Policy Questions for the Subarea. This task involves integrating into the DSS the values for the changes of the ecological endpoints, enabling a direct assessment of the policy question. Specifically, the initial configuration of the DSS does not include any relationships between the changes in the ecological endpoints and a valuation measure.

Box C6-a: The DSS now has the behavioral equations representing valuation directly introduced into the framework. This is essential for the cost–benefit analysis to follow.

Box C6-b: The DSS is then rerun with the range of scenarios, generating the marginal benefits and costs associated with each scenario. The tool is now available and can be used in an adaptive management framework.

Box C6-c: A cost–benefit analysis of the alternative scenarios can be conducted. The goal is to determine the maximum net benefits from alternative management plans for the subarea of study.

5.4 Beyond the Subareas to a Regional Assessment: Aggregation Thoughts

To achieve an ecoregional assessment, two broad steps are required. First, the DSS must be extended to other subareas. We term this step *horizontal extension*. Second, an aggregation protocol must be designed for reach the regional level, a step we term *vertical extension*.

This effort by its very nature is a bottom-up approach. Two questions are central: (a) To what degree can the science and the valuations obtained for the subarea(s) be extended to other subareas? (b) Can the subareas be aggregated into a regional assessment?

Horizontal Extension. In Task C1-b, a region of study is chosen. The next steps are to move beyond the subarea(s) in which the valuation effort has been undertaken and apply the DSS to the other subareas that were originally identified in C1-c. The goal is to create a multitude of valuations for a multitude of subareas. As we note above, this is essentially a benefit transfer exercise. But, it actually goes beyond the traditional benefit transfer methodology because the exercise involves assessing the applicability of the science models as well as the valuation models to additional subareas. Typically, benefit transfer methods assume implicitly that the biophysical science that drives the values at the study site (the site of the original valuation effort) is identical to that at the transfer site. In some cases this assumption may be reasonable, and in other cases it may not be reasonable. Essentially, one must ensure that the underlying science in the DSS is appropriate for all of the subareas identified C1-c. In addition, following benefit transfer protocols, data for the subareas must be collected.

Vertical Extension. Moving to an ecoregional assessment is not simply an exercise in summation of the various subarea valuation assessments. Here we touch on two of the issues. First, one must determine initially whether the ecoregional assessment should be representative of only the subareas or whether Task C1-c should be revisited. Second, aggregation also involves behavioral issues. For instance, if the various subareas are substitutes, one must determine whether a second study is required to determine stakeholders' views of multiple sites. We view this as an ongoing research question.

6. Conclusions

Several recommendations emerge from this evaluation. The principle recommendation is for future LC analysis to address gaps between existing outcome measures and what we have called ecological endpoints. As it stands, the ecosystem service measures proposed by LC make it difficult to clearly connect biophysical and social evaluation. Most of LC's currently proposed outcome measures *require further biophysical translation to facilitate social evaluation.* To be clear, precursor and intermediate biophysical outcome measures are an important foundation on which to build. But they thwart social evaluation because of their distance from social decisionmaking, choices, and comprehension.

Another, perhaps obvious, recommendation is to expand the set of ecological outcomes that is currently contemplated by the LC method. It is obvious to many LC audiences that land conversion will affect a range of water-related outcomes, such as aquifer depth and quality and the timing, depth, and speed of surface water flows. Land cover change will also affect air quality, fire risk, and the aesthetic features of the landscape. Going forward, development of outcome measures around these social issues will presumably be expected.

We recommend that data products be organized around the concepts of biophysical and economic production. As described in Section 2, production theory disciplines and clarifies analysis by articulating the relationships between inputs and outcomes in complex systems. Analytical confusion can arise from LC's current depiction of analysis and data products, where inputs, processes and functions, outcomes, and distinctions between natural production and social (technological) production are not made clear.

Finally, we strongly encourage the proposed development of case studies to explore a wider range of ecosystem service co-effects, develop additional biophysical production and process models, and generate outcome measures at finer spatial resolutions. Such analysis will more effectively address the needs and expectations of LC's stakeholder and policymaker audiences.

References

Arrow, Kenneth, Robert Solow, Paul Portney, E.E. Leamer, R. Radner, and H. Schuman. 1993. Report of the NOAA Panel on Contingent Valuation. *Federal Register* 58(10): 4601–4614.

Baker M.E., D.E. Weller, and T.E. Jordan. 2006. Improved Methods for Quantifying Potential Nutrient Interception by Riparian Buffers. *Landscape Ecology* 21: 1327–1345.

Barbier, Edward. 2009. Ecosystem Service Trade-offs. In *Ecosystem-Based Management for the Oceans*, edited by Karen McLeod and Heather Leslie. Washington, DC: Island Press, 129–143.

Bockstael, N.E. 1996. Modeling Economics and Ecology: The Importance of a Spatial Perspective. *American Journal of Agricultural Economics* 78: 1168.

Bockstael, Nancy, and Elena Irwin. 2000. Economics and the Land Use–Environment Link. In *The International Yearbook of Environmental and Resource Economics 2000/2001*, edited by T. Tietenberg and H. Folmer. Cheltenham, UK: Elgar Publishing, 1–54.

Boyd, J. 2007. The Endpoint Problem. *Resources* Spring: 26–28.

Boyd, J., and S. Banzhaf. 2007. What Are Ecosystem Services? *Ecological Economics* 63: 616–626.

Boyd, J., and A. Krupnick. 2009. The Definition and Choice of Environmental Commodities for Nonmarket Valuation. Discussion paper 09-35, Washington, DC: Resources for the Future.

Boyd, J., and L. Wainger. 2003. Measuring Ecosystem Service Benefits: The Use of Landscape Analysis To Evaluate Environmental Trades and Compensation. Discussion paper 02-63. Washington, DC: Resources for the Future. http://www.rff.org/documents/RFF-DP-02-63.pdf (accessed April 13, 2011).

Carson, Richard, Nicholas Flores, and Norman Meade. 2001. Contingent Valuation: Controversies and Evidence. *Environmental and Resource Economics* 19: 173–210.

Chapman, D., and W.M. Hanemann. 2001. Environmental Damages in Court: the American Trader Case. In *The Law and Economics of the Environment*, edited by Anthony Heyes. Northampton, MA: Edward Elgar Publishing, Inc., 319–367.

Correll, D.L., T.E. Jordan, and D.E. Weller. 1992. Nutrient Flux in a Landscape: Effects of Coastal Land Use and Terrestrial Community Mosaic on Nutrient Transport to Coastal Waters. *Estuaries and Coasts* 15(4): 431–442.

Costanza, Robert, Ralph D'Arge, Rudolph DeGroot, Stephen Farber, Monica Grasso, Bruce Hannon, Karin Limburg, et al. 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature* 387: 253–260.

Costanza, R., M. Wilson, A, Troy, A. Voinov, S. Liu, and J. d'Agostino. 2006. *The Value of New Jersey's Ecosystem Services and Natural Capital*. Trenton, NJ: New Jersey Department of Environmental Protection.

Daily, G., and P. Matson. 2008. Ecosystem Services: From Theory to Implementation. *Proceedings of the National Academy of Sciences of the United States* 105(28): 9455–9456.

Dillman, D. 2000. *Mail and Internet Surveys: The Tailored Design Method*. New York, NY: John Wiley and Sons.

Flather, C.H., and J.R. Sauer. 1996. Using Landscape Ecology To Test Hypotheses about Large-Scale Abundance Patterns in Migratory Birds. *Ecology* 77(1): 28–35.

Freeman, A. Myrick. 1993. *The Measurement of Environmental and Resource Values: Theory and Methods*. Washington DC: Resources for the Future.

Gardner, R.H., R.V. O'Neill, and M.G. Turner. 1993. Ecological Implications of Landscape Fragmentation. In *Humans as Components of Ecosystems: Subtle Human Effects and the Ecology of Populated Areas*, edited by S.T.A. Pickett and M.J. McDonnell. New York, NY: Springer-Verlag,

Green, A., P. Lokani, S. Sheppard, J. Almany, S. Keu, J. Aitsi, J. Warku Karvon, R. Hamilton, and G. Lipsett-Moore. 2007. *Scientific Design of a Resilient Network of Marine Protected Areas: Kimbe Bay, West New Britain, Papua New Guinea*. TNC Pacific Island Countries Report No. 2/07. South Brisbane, Australia: The Nature Conservancy, Indio-Pacific Resource Centre.

Guichard, F., and G. Peterson. 2009. Ecological Cross-Scale Interactions. In *Ecosystem-Based Management for the Oceans*, edited by Karen McLeod and Heather Leslie. Washington, DC: Island Press, 74–91.

Gustafson, E.J. 1998. Quantifying Landscape Spatial Pattern: What Is the State of the Art? *Ecosystems* 1: 143–56.

Jenkins, W., B. Murray, R. Kramer, and S. Faulkner. 2010. Valuing Ecosystem Services from Wetland Restoration in the Mississippi Alluvial Valley. Ecological Economics, 69, 1051-1061.

Kirchhoff, S., B. Colby, and T. LaFrance. 1997. Evaluating the Performance of Benefit Transfer: An Empirical Inquiry. *Journal of Environmental Economics and Management* 35: 75–93.

Kopp, Raymond J., and V. Kerry Smith (eds.). 1995. *Valuing Natural Assets.* Washington, DC: Resources for the Future.

Litke, D.W. 1999. *Review of Phosphorus Control Measures in the United States and Their Effects on Water Quality*. Water-Resources Investigations Report 99-4007. Reston, VA: U.S. Geological Survey.

Lowrance, R.R., L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, et al. 1997. Water Quality Functions of Riparian Forest Buffers in Chesapeake Bay Watersheds. *Environmental Management* 21: 687–712.

Mahan, B., S. Polasky, and R. Adams. 2000. Valuing Urban Wetlands: A Property Price Approach. *Land Economics* 76(1): 100–113.

McConnell, Kenneth. 1992. On-Site Time in the Demand for Recreation. *American Journal of Agricultural Economics* 74: 918.

Noss, R.F. 1990. Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conservation Biology* 4(4): 355–364.

Polasky, S., E. Nelson, J. Camm, B. Csuti, P. Fackler, E. Lonsdorff, C. Montgomery, et al. 2008. Where To Put Things? Spatial Land Management To Sustain Biodiversity and Economic Returns. *Biological Conservation* 141: 1505–1524.

Richards, C.L., L.B. Johnson, and G.E. Host. 1996. Landscape-Scale Influences on Stream Habitats and Biota. *Canadian Journal of Fisheries and Aquatic Science* 53(1): 295–311.

Roberts, C., B. Halpern, S. Palumbi, and R. Warner. 2001. Designing Networks of Marine Reserves: Why Small Isolated Protected Areas Are Not Enough. *Conservation* 2(3): 10–17.

Soil Conservation Service. 1975. *Urban Hydrology for Small Watersheds*. Technical Release 55. Washington, DC: U.S. Department of Agriculture.

U.S. EPA (Environmental Protection Agency) Science Advisory Board. 2009. *Valuing the Protection of Ecological Systems and Services: A Report of the EPA Science Advisory Board.* EPA-SAB-09-012. Washington DC: U.S. EPA.

Water Resources Research. 1992. Special Issue, Vol. 28.

Zhu, Zhiliang (ed.), Brian Bergamaschi, Richard Bernknopf, David Clow, Dennis Dye, Stephen Faulkner, William Forney, et al. 2010. *A Method for Assessing Carbon Stocks, Carbon Sequestration, and Greenhouse-Gas Fluxes in Ecosystems of the United States under Present Conditions and Future Scenarios*. Scientific Investigations Report 2010–5233. Reston, VA: U.S. Geological Survey.