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Synthesis Chapter

The Valuation of Ecosystem Services from Farms and Forests

*Informing a systematic approach to quantifying
benefits of conservation programs*



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Synthesis Chapter - The Valuation of Ecosystem Services from Farms and Forests: Informing a systematic approach to quantifying benefits of conservation programs

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Disclaimer

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Please note, the report and its chapters are intended to demonstrate a framework approach to ecosystem service valuation. The report or any chapter there within is not to be cited for the purpose of supporting or opposing any government or private program.

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Synthesis Chapter - The Valuation of Ecosystem Services from Farms and Forests: Informing a systematic approach to quantifying benefits of conservation programs^{1,2}

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ABSTRACT

The condition of natural resources affects human well-being in a multitude of ways, including changes to health, safety, and recreational opportunities. Valuing environmental changes, as the specific goods and services that emerge from ecosystems, has been proposed as a method for systematically including these often-ignored benefits in policy decisions. While governmental and academic studies have estimated the value of some benefit types, a comprehensive valuation has not been possible.

This project begins to address that gap by demonstrating the potential scope and methods for valuing a broad range of ecosystem service benefits generated by USDA programs. The USDA has a long history of implementing conservation programs on farm and forest lands to enhance conditions of land, water, air, wildlife, and other natural resources. These programs can potentially deliver a wide range of benefits to the public, including to landowners and communities located well outside the boundaries of the activities on farms and forests. The question addressed here is: How well can these public ecosystem service benefits be captured using available data, knowledge, and tools?

Working with interdisciplinary teams of government and academic scientists, we identify approaches that use available science to estimate values flowing from the programs, using monetary values and benefit-relevant (nonmonetary) indicators. The overarching goals are to 1) identify and demonstrate current approaches and 2) use that understanding to identify investments in data collection and modeling that would improve or broaden value estimates. Case studies of pollinator habitat, forest carbon sequestration, and water quality were chosen to reveal differences in the state of the science across natural resource areas and demonstrate alternatives to overcoming data and knowledge gaps.

The teams' findings show that a substantial subset of the salient values emanating from the conservation programs can be monetized with the types of methods and data that are routinely used in federal decision making. However, broadening the suite of measured benefits beyond those addressed in traditional cost-benefit analyses proved challenging. We offer five suggestions designed to promote a successful approach to full ecosystem service valuation for USDA conservation programs:

1. Define terms, such as ecosystem service values, but retain the flexibility to tailor metrics and methods to policies, authorities, landholder interests, and data limitations;
2. Create interdisciplinary analytic teams to enhance the credibility of all aspects of value assessment;

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² Please note, the report and its chapters are intended to demonstrate a framework approach to ecosystem service valuation. The report or any chapter there within is not to be cited for the purpose of supporting or opposing any government or private program.

3. Monetize benefits where appropriate, but use benefit-relevant indicators to complement or serve as alternatives to monetary values when significant stakeholder concerns cannot or should not be monetized;
4. Demonstrate sensitivities to assumptions and sources of error to develop a shared understanding of how results can best be used in policy; and
5. Identify opportunities for strategic investments in science that will improve the accuracy of value estimates and allow more services to be measured and valued.

PROJECT INTRODUCTION

The U.S. Department of Agriculture invests more than \$10 billion annually, using eight percent of its budget to implement conservation programs.³ The largest of these programs is the Conservation Reserve Program (CRP), which has long-term environmental goals to improve water quality, prevent soil erosion, and preserve wildlife, wetlands, and forests. Other federal programs, similarly, seek to improve the environment so that it will provide valuable ecosystem goods and services to the public and minimize environmental externalities (ancillary effects) that can result from private actions, such as downstream water pollution. Assessing the impacts of these conservation programs provides critical information that can be used to measure outcomes and improve the efficacy of meeting policy goals.

Using ecosystem services to assess policy or program impacts has been proposed to systematically include the human welfare effects of ecosystem changes in decision making (Daily 1997, OMB et al. 2015). Executive departments and agencies of the U.S. government have started expanding their environmental accounting systems to guide natural resource management decisions beyond traditional regulatory or performance targets (Schaefer et al. 2015). Of relevance to USDA conservation programs, the effects of implementing practices can be quantified in terms of the ecosystem service benefits generated from improvements in soil quality, water quality, air quality, wildlife habitat, recreation, carbon sequestration, and other impacts. Values also include any substantive unintended negative effects on ecosystem services, or disservices, such as invasive species spread. This broad focus offers some advantages for making long-term natural resource management decisions because it enables researchers to incorporate a suite of natural resource tradeoffs that might not be included in traditional cost-benefit analysis.

This report describes the results of a project to evaluate the state of the science for conducting such analyses by having teams of analysts test which values could be generated for salient ecosystem services stemming from conservation programs. The goal is to support program evaluations by developing reliable and tractable approaches that use the best available science to estimate monetary and non-monetary benefits. The *best available science* minimizes subjectivity and accurately represents limits of the methods and data (Sullivan et al. 2006; Ruhl 2004; and NRC, 2004). Further, the word *available* implies a feasible level of effort given available resources. As a result, the best available science may not represent the best possible science for minimizing error, if that approach would exceed time and analytic resource constraints. For example, the large number of sites that must be assessed for a programmatic review can make conducting original valuation surveys intractable. If that is the case, the most cost-effective methods that will serve analysis goals are substituted.

The need for this project emerged in a 2015 workshop organized by the Council on Food, Agricultural, and Resource Economics (C-FARE), and sponsored by the USDA's Office of

³ <http://www.obpa.usda.gov/budsum/fy16budsum.pdf>

Ecosystem Markets, to explore the potential of ecosystem service valuation for USDA conservation programs (The Council on Food, Agricultural and Resource Economics, 2015). That workshop discussion suggested that the impediments to ecosystem service valuation were substantial but that further work could improve our understanding of the connections between conservation practices, ecosystem effects, and social benefits.

Estimating the economic value of environmental changes is not a new endeavor; considerable effort over the past 40 years by government scientists, academic researchers, and others has been undertaken to understand the benefits of conservation programs on farms and forests. Many valuation techniques have been developed to value market and non-market goods and services, as described in numerous sources (e.g., NESP 2015, U.S. EPA SAB. 2009) and Appendix A. The economic valuation of ecosystem services has encompassed active uses, such as recreational activities, and passive non-uses, such as the pleasure derived from knowing that certain wildlife species thrive (National Research Council (NRC) 2005, Barbier 2008). For those services that are not amenable to monetization, due to missing data or incompatible social context, quantitative nonmonetary measures (e.g., biodiversity metrics) have been used to add important dimensions of intensity of concern for environmental changes and scarcity of services.

Despite the considerable scientific progress on valuation, the USDA and other federal agencies are not routinely measuring the environmental benefits of programs and may benefit from taking advantage of recent advancements in available science to measure the values of nature's benefits to humans (Guerry et al. 2016, Johnston et al. 2016). This effort was intended to use existing work to draw attention to models and data that may be useful for government analysts seeking to quantify program effects in terms of ecosystem services. This project lays the scientific and institutional groundwork to build a sound foundation for comprehensive ecosystem service valuation in USDA.

The scientific foundation of ecosystem service valuation

Before valuing ecosystem services, it is helpful to define them. Yet, the term *ecosystem services* (or *ecosystem goods and services*) has not been used consistently in the literature. While it is commonly defined as “the benefits that people obtain from ecosystems” (Millennium Ecosystem Assessment [MEA] 2005), this definition does not provide clear guidance for choosing measurement endpoints. Further, the classification system proposed by the MEA authors is often criticized for encouraging double-counting of benefits and generally lacking operational consistency (Boyd and Banzhaf 2007, Fisher et al. 2009). Other classification systems have been proposed, but no standard has been widely adopted (Nahlik et al. 2012, Haines-Young and Potschin 2013, Soba et al. 2015). While different definitions and classification systems will likely be needed for different decision contexts (Costanza 2008), some level of consistency in definitions and classifications is desirable for making comparisons across projects or programs.

For this project, we define ecosystem services as the outputs of ecosystems (including biotic and abiotic processes and their interactions) that are appreciated or used (directly or indirectly) by people. *Ecosystem service benefits* then emerge from the interaction of ecosystem service changes within the relevant social and economic context. This definition is meant to be consistent with the definition of “final ecosystem services” (Boyd and Banzhaf 2007, Johnston and Russell 2011), which requires that biophysical measures of services reflect characteristics that are important to potential beneficiaries and that valuation reflects beneficiaries' perspectives.

Consistent and rigid definitions of ecosystem service benefits create operational challenges, because benefits will depend on the specific uses and concerns of institutions and private landholders. Thus,

the essential recommendation is that valuation will be served by creating metrics of biophysical changes that clarify and simplify the translation to values. For example, quantifying changes in pollinator abundance is not necessarily correlated with benefits, since benefits or harms of added pollinators will depend on location (e.g., due to presence of crops or other plants that can benefit from wild pollinators). However, an ecosystem service metric of *yield changes due to pollinator abundance* is more closely associated with potential benefits because it relates pollinator abundance to the potential effect of lowering the costs of producing crops, which has the potential to create value for farmers and consumers. Although this ecosystem service metric is closer to a measure of benefits, many other factors will still determine the magnitude of yield changes and whether they generate net economic value.

Ecosystem service values represent changes in human well-being

The essence of ecosystem service valuation is linking changes in ecosystems to outcomes that matter to people. Economic value, as defined in economic welfare theory, reflects peoples' willingness to pay for, or otherwise trade off other goods and services for, given changes in ecosystem services (Freeman et al. 2014). In other words, value stems from how people would choose among alternative bundles of goods and services, where bundles can include everything from direct income benefits (e.g., less expensive food) to intangible psychological or spiritual benefits (e.g., satisfaction from preserving a rare species) (Wainger and Boyd 2009). Economic welfare theory further embeds the perspective that individuals are best able to judge what promotes their well-being, and that the collective good is generally served by maximizing aggregate well-being, while accounting for distributional and other effects.⁴

Applying this definition of value reveals that not all ecosystem changes will generate substantial welfare effects. Environmental changes may not change the quality or quantity of goods and services that people use and appreciate. Due to the availability of surpluses, substitutes and other site-specific conditions, not all changes in goods and services have welfare effects. Even if ecosystem service changes affect welfare, they may not be amenable to quantification, for a variety of reasons, including the difficulty of defining and measuring peoples' preferences, particularly for intangible benefits. Because of these circumstances, the initial challenge to valuing ecosystem services is to determine:

1. the type and magnitude of ecosystem or biophysical changes that result from an action;
2. whether such changes are likely to have noticeable effects on net income, asset values, consumption, leisure activities, time costs, moral satisfaction, or other components of welfare, such as shifts in their distribution; and
3. whether data and tools are available (and appropriate) to value the welfare effects.

Integrated ecological and economic models are the tools used to describe connections between actions and welfare effects. In the ecosystem services literature, analysts have used everything from simple logical relationships to complex simulation models to establish connections. The work presented in this report similarly uses a range of models. For example, the water quality team used an empirical relationship to connect changes in field-scale sheet and rill erosion to dredging costs avoided in reservoirs, while another analysis by the same team required linking multiple process-based simulation models and empirical ecological and economic models to evaluate the effects of conservation practices on fish populations and recreational fishing benefits (Wainger et al. 2017).

⁴The reader may refer to Arrow et al. (2010) for further information about efficient allocation of resources using individually established preferences and the relationship between efficiency and equitable distribution of goods and services. Further, much has been written about the impediments to applying economic welfare theory, including descriptions of cases in which individuals seemingly do not generate preferences that maximize their well-being, suggesting that it may be appropriate for institutions to supplement individual perspectives (Pearce & Turner 1990).

The type of model integration applied will be driven by the availability of analytic resources and the requirements for precision and other analytic details necessary to support the decision under consideration (see Kline et al. 2013 for further discussion).

Conceptual value diagrams align interdisciplinary research teams

Interdisciplinary teams are invaluable for accurately measuring and valuing ecosystem service changes, but must overcome differences in conceptual and analytic approaches to function effectively (National Research Council 2013; Olander et al. 2015). The teams conducting this effort were made up of a mix of biophysical and social scientists, including ecologists, hydrologists, agricultural economists, and environmental economists. To facilitate the analysis of the linkages between actions and benefits/harms among the interdisciplinary research teams, the teams in this project used *conceptual value diagrams* to blend their expertise and draft an initial valuation approach (Figure 1).

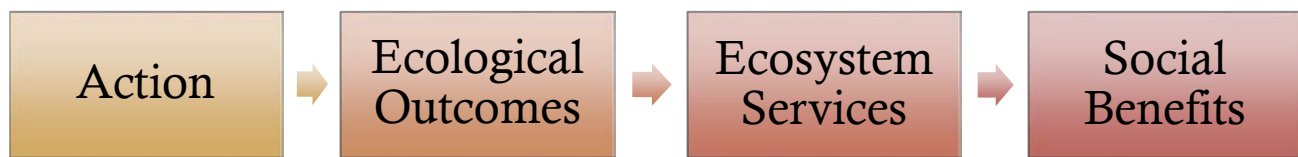


Figure 1. The conceptual value diagram links conservation actions to social benefits by illustrating the intermediate changes in biophysical and socio-economic systems that need to be quantified to be able to value benefits. To understand effects of a conservation action, a team might need to quantify the initial biophysical changes in terms of physical processes (e.g., erosion rates) or biological structures (e.g., vegetation density) and then link those initial changes to ecological outcomes recognizable as beneficial (e.g., game abundance). Next those biophysical changes are evaluated in the context of human use and appreciation (e.g., number of hunters who are likely to benefit from increased game). Finally, social benefits are evaluated in terms of willingness to pay for the change (e.g., value of additional game to likely hunters), which will incorporate supply and demand conditions that influence value by location.

The basic diagram was adapted by the teams to identify links (representing models) and endpoints (representing metrics) that established the multiple cause and effect relationships needed to connect actions to welfare (Figure 2). Such diagrams are useful for:

1. conceptualizing analytic relationships;
2. establishing feasible approaches given current data, tools, and scientific understanding;
3. communicating the scope of potential benefits; and
4. identifying the analytic components needed for valuation.

As is typical for ecosystem service valuation, these diagrams reflect a reductionist approach to isolating separable pathways of benefits to facilitate valuation of changes for specific groups of beneficiaries. The diagrams imply that the multiple benefits generated for different ecosystem services can be summed to assess the total benefits of a program, even though this may not be possible in practice. It is often the case that economic methods used to measure values of environmental changes do not isolate one type of ecosystem service benefit but rather lump different types of benefits. For example, some valuation studies enable an analyst to estimate value per household (within a watershed) for a change in water quality as a mix of use and nonuse values. In cases where benefits across ecosystem services are likely to overlap, analysts must choose whether benefit aggregation is appropriate and/or how double-counting might be minimized.

The *conceptual value diagrams* (Figures 2a-c) provide a general overview of the valuation process, but each link has the potential to be difficult and resource-intensive to model or may be impossible to measure with existing resources. Ecological and economic systems are complex and do not necessarily exhibit simple linear responses. Therefore, ecosystem service analyses have the potential to require substantial resources. In particular, many ecological process models were not designed to link to economic models, creating the need for additional, sometimes complex, models to make necessary links or new economic valuation work. Teams that do not have resources to develop new models can easily be stymied by such gaps between ecological and economic models.

The case study chapters of this report describe, and in some cases, demonstrate methods for the three resources (pollinator habitat, water quality, forest carbon sequestration). This evaluation is not exhaustive and will quickly be out of date, since the resources available to facilitate ecosystem service modeling and valuation are constantly evolving. Sources for current information include published literature, and, within the USDA, the Economic Research Service and the USDA Environmental Markets websites contain both primary data and links to additional resources. In addition, some websites catalog analytic resources or case studies, for example:

- The National Ecosystem Services Partnership
- The Ecosystem Services Partnership [International]
- The U.S. EPA EnviroAtlas
- The Natural Capital Project
- The OpenNESS project
- The Environmental Valuation Reference Inventory

From the teams' case study analyses and other work, some biophysical and ecological models are now commonly used to evaluate ecosystem service changes, and thus may serve future analyses because the results will be comparable across regions (Table 1). It is important to note is that biophysical outputs are often far from the outcomes that are needed as input to economic models. For example, although models are readily available to estimate nutrient loads, models that estimate in-situ water clarity or biotic responses are not uniform across sites. Such models are critical for making connections between biophysical and economic models and they need to be vetted among scientists and made available to all affected areas of the U.S. if they are to support regional assessment of national programs.

Table 1. Biophysical models identified for conducting nationally consistent ecosystem service assessments

Resource area	Biophysical outputs	Model	Source
<i>Pollinator habitat</i>	Crop yields	InVest – pollination module	Lonsdorf et al. 2009, Olsson et al. 2015
<i>Water quality</i>	Nutrient loads to water bodies	Agricultural Policy/Environmental eXtender (APEX) and Soil and Water Assessment Tool (SWAT)	USDA NRCS, 2016
<i>Carbon</i>	Carbon sequestration	Forest Carbon Accounting Framework	Wear and Coulston 2015, Woodall et al. 2015

The economic methods demonstrated in this report all rely on benefit transfer, an expedient valuation method frequently used by federal agencies. Benefit transfer takes many forms, but in all cases, values generated by existing (primary) valuation studies are used to estimate values for unstudied sites (Freeman et al. 2014). For example, if a valuation study has shown that people in a

neighborhood are willing to pay for increased wetland area near their homes, that value (e.g., the percentage increase in home value due to wetland area, holding all other factors constant) can be used to estimate the value of a wetland expansion for a similar but unstudied neighborhood.

Two main approaches are used to estimate values: unit value transfer and functional transfer. Unit value transfers apply a single value to the change at the unstudied site, and that single value is often an average value across multiple studies or an adjusted value, based on best professional judgment. Benefit function transfers, in contrast, calculate values using a function estimated from empirical data that allows multiple site factors, such as baseline ecological conditions and income level, to be used to adjust the study site value to the policy site (Johnston et al. 2015).

A variety of databases and models support benefit transfer (e.g., Environment Canada 2011, Rosenberger 2013, NCSE 2016). Available databases catalog primary valuation studies using standard economic valuation techniques (descriptions of primary valuation methods are available in U.S. EPA 2009, Freeman et al. 2014, and Wainger et al. 2014). Such databases can be used to conduct unit value transfer or develop functional transfers. Since functional transfers can be time-consuming to develop, existing models are often drawn from the literature. An example of applying a functional transfer is shown in Chapter 2 ([Ecosystem Service Benefits Generated by Improved Water Quality from Conservation Practices](#)) and in Johnston and Wainger (2015).

In general, primary valuation studies are considered to provide the most robust estimates of value of an environmental change, but are often expensive to conduct, particularly when many sites must be evaluated. Benefit transfer is considered less accurate than primary studies, because it cannot represent all the local conditions that determine value. However, benefit transfer can be conducted in ways that minimize the error associated with applying data measured elsewhere to a new study site (Johnston et al. 2015).

Why non-monetary measures may be necessary and desirable

A lack of primary valuation studies limits which benefits can be valued using benefit transfer methods. To be inclusive of potential welfare effects, when multiple primary studies do not exist, analysts may choose to follow the Office of Management and Budget (OMB) guidance for regulatory cost-benefit analysis, which suggests three steps:

1. monetize what can be monetized;
2. quantify what cannot be monetized;
3. describe what can be neither monetized nor quantified.⁵

Non-monetary benefit metrics or benefit-relevant indicators (BRIs), as they are referred to in this report, can effectively support cost-effectiveness analysis or alternative decision support methods in which alternative actions are compared based on quantitative metrics of probable benefits. The BRI is used in place of, or as a supplement to, monetary values. A BRI measures an outcome that is expected *to be correlated with* benefits. For example, improved crop yields can be correlated with a producer's profits.

Any non-monetary benefit metric of ecosystem service effects, whether used in cost-effectiveness analysis or benefit summaries, will be better able to support robust decisions if it explicitly incorporates beneficiary concerns (Wainger and Mazzotta 2011). Metrics suggest welfare changes

⁵ U.S. Office of Management and Budget 2003. Circular A-4. Subject: Regulatory Analysis. Washington, D.C. <https://georgewbush-whitehouse.archives.gov/omb/circulars/a004/a-4.html>

by incorporating aspects that affect what a beneficiary would be likely to pay for a change or to avoid a change. Commonly, the marginal value of an additional increment of a good will depend on the existing abundance of that good. For example, an additional fish caught will be more valuable in a case where that fish is scarce, relative to a case where that fish is abundant, all else being equal. The more scarcity or other value-relevant factors can be incorporated into a non-monetary metric, the more likely it is to correlate with a change in value to beneficiaries.

The *conceptual value diagrams* that mapped the steps necessary to value selected resources (Figure 2) also serve to identify BRIs. BRIs often support economic benefit transfer, for example, by estimating the number of potential ecosystem service users. Or, BRIs may be an end in themselves to capture evidence of potential value, when it cannot be monetized. Analysts may wish to consider a broad range of factors when designing non-monetary BRIs to represent value, including:

1. the degree to which service quality supports a beneficiary's use or appreciation;
2. availability of capital and labor necessary to enable use of ecological outputs;
3. the number and characteristics of users or beneficiaries;
4. reliability of the future stream of services; and
5. scarcity and substitutability of services.⁶

Such factors provide only general guidance. Stakeholder involvement is needed to develop the indicators that best reflect concerns relevant to the specific decision context (Ervin et al. 2014). In addition, BRIs are subject to misuse if they do not accurately reflect likely user responses, including ability and willingness to adapt. It is not sufficient to simply note that 'the quality of an ecosystem service is excellent' or that 'the number of users is high' to suggest that a beneficiary experiences a change in well-being from a change in an ecosystem service. Rather, indicators should reflect when a change is most likely to have value, such as an increase in the quantity or quality of a scarce resource.

For example, consider a case in which restoration of a tidal wetland reduces the frequency of nuisance flooding on a coastal road. The BRI could be the number of road users affected. However, a better metric would reflect the availability of substitutes for that travel route to better reflect the degree of concern. A metric that estimated the increased driving time required to bypass the flood (summed across all road users) would differentiate cases where a flooded road is easily bypassed

from cases where flooding imposes a major burden. With well-grounded assumptions based in observed human behavior, metrics can capture close to the same evidence of monetary values in terms of reflecting the intensity of user preferences for or aversions to change. However, the appropriateness of such indicators will depend on the decision context and the quality of the information available to support the assumptions used.



⁶ <https://nespguidebook.com/assessment-framework/quantifying-social-and-economic-context-in-bris/> for further discussion

PROJECT APPROACH

The development of analytic teams and advisors—with representatives from multiple federal agencies, academic departments, and non-profit organizations—was conducted by the Council on Food, Agricultural, and Resource Economics at the behest of the USDA’s Office of Environmental Markets. Natural and social scientists from the government and from universities were identified for each resource team to provide complementary expertise. The teams were given the following set of tasks as a general template to guide their valuation exercise.

Tasks

1. Identify a set of salient ecosystem services for USDA conservation and natural resource programs.
2. Review the best, available, evidence-based methods to evaluate the ecosystem service benefits due to USDA programs (relative to a baseline without the program) and for an appropriate analysis region.
3. Assess available science and data to support credible estimates of relevant *biophysical changes* due to program activities for each salient ecosystem service.
4. Assess available science and data to support credible estimates of *socio-economic benefits* for changes in each salient ecosystem service.
5. Create evidence-based estimates of values using monetary units or non-monetary benefit metrics, when existing methods and data permit.
6. Characterize the uncertainties and caveats that should accompany each benefit estimate when used by public and private organizations.
7. Describe the strengths and limitations of best methods and data to estimate service values.
8. Identify key missing information and data.
9. Write a report of the team’s findings.
10. Conduct rigorous peer review.
11. Prepare a final report.

Each team tailored their approach to a resource of concern. The integrated teams of natural and social scientists brought knowledge of a broad range of available analytic resources that they applied to designing a scientifically available analysis. The project teams first identified salient ecosystem changes based on whether they were likely to generate substantial changes in value. Government analyses will also need to consider the saliency of services as judged by beneficiary groups, government officials responsible for implementing the conservation programs, and/or and related statutory authorities. In all three resource areas, the services that were quantitatively analyzed had positive effects because potentially negative effects of conservation programs were judged to be minor when aggregated at the regional or national scale. However, regional positive values do not preclude negative local effects that may be consequential for individual stakeholders.

Teams relied exclusively on existing databases, models, and published materials to conduct the analyses. However, researchers found that they still needed to develop new analyses to manipulate existing data and conduct benefit transfer exercises. Even simple benefit transfer exercises required substantial effort to get biophysical data into the appropriate form to conduct analyses. In some cases, the published literature was insufficient to guide the teams’ analyses. In those cases, the teams had to engage with developers of the ecological and economic models being used. Modelers were invited to join the team or assist the team by providing supplemental information and confirming that appropriate data and methods were used.

STATE OF THE SCIENCE

Common themes emerged from the efforts to value ecosystem services for the three resources of concern. The chapters that follow this synthesis chapter provide more detail on the state of the science for each: pollinator habitat, water quality and forest carbon sequestration. Each team had unique opportunities for and challenges to valuing ecosystem services and made distinct choices in overcoming them. Yet, the diversity of challenges and adaptations reveals issues that are likely to arise in any federal agency effort to value ecosystem service impacts of its programs and use ecosystem service concepts in policy analysis. Three primary questions were initially asked of the project:

1. What ecosystem service changes could be feasibly monetized now with available data and understanding?
2. What could be monetized with modest investment?
3. What remains difficult to monetize for the foreseeable future, or is inappropriate to monetize, and is best addressed through quantitative, qualitative, or descriptive assessment?



What emerged from the analysis were the many issues that determine whether valuation methods are likely to be deemed acceptable and useful. These issues include the types of decisions being supported, the necessary level of precision of values, how uncertainty should be addressed, whether methods need to be consistent nationally, and the duration of benefit assessment, among others. The divergent approaches chosen suggest that no uniform template for valuation will generate sensible outcomes across natural resources or programs.

A summary of what could be valued

A modest number of ecosystem service benefits from conservation actions were monetarily valued with available data and information (Table 2), and a much larger set of potential ecosystem service benefits were quantified and described. In most cases, the effects being valued were for potential future actions rather than results of specific USDA programs. Teams were limited to analyzing effects that had previously been modeled, which were most often what-if scenarios rather than retrospective analysis.

Based on our experiences, the valuation methods are of a level of precision similar to those used to support regulatory analysis and many types of government decision-making. Therefore, monetary values generated with equivalent approaches are likely to be appropriate for reporting program performance or supporting decisions that are robust to underlying levels of uncertainty. However, imprecision in value estimates is likely to limit their ability to differentiate subtle policy alternatives (e.g., alternative geographic distributions of similar actions), since uncertainty ranges may exceed differences among alternatives.

Table 2. Monetizable values for the three natural resource areas analyzed

Resource	Ecosystem services valued (monetary units)
<i>Pollinator habitat</i>	Crop and honey yield changes* (valued as quantity * market price)
	Reduced farmer production costs*
<i>Water quality</i>	Property value support
	Recreational fishing
	Nonuse values for aquatic ecosystems
	Drinking water quality and recreational services from reservoirs
	Commercial shipping benefits from channels
<i>Carbon</i>	Carbon sequestration in forests (representing a bundle of ecosystem service benefits from increased water provision, food provision, property protection, human health and safety, etc.)

*Values not estimated by available methods were described.

Given the substantial scientific investments that have been made in ecological monitoring and modeling and in economic benefit assessment, all teams found resources to conceptually demonstrate how benefits could be assessed for alternative policies or program actions. The water quality and carbon teams generated estimates of monetary values for selected services and the pollinator team described several approaches to measuring welfare changes. The valuation approach that was identified by the pollinator team as feasible was multiplying total yield change by market price of that commodity. This practical approach to valuation works best when certain market conditions are met. Agricultural output changes can be used to estimate profit changes when an action does not impose costs to farmers and the total yield changes from a program are not large enough to affect market price. If yield changes are large, then additional models are needed—partial or general equilibrium models—to estimate market responses and resulting price changes.⁷ Such models are complex and require specialized expertise that will substantially increase the time required to generate value estimates.

The current state of data and knowledge enabled reasonably robust analysis for some portions of the benefits analysis but also required teams to make bold assumptions to bridge some data gaps. Therefore, the precision of the estimates is highly uncertain. Uncertainty arose from many sources including the need to bridge gaps among models, apply model results to new geographic or temporal extents, and from a lack of replicate valuation studies that are needed to adjust measured values to conditions at new locations. Sensitivity analyses revealed that uncertain data inputs had a large effect on estimated values. For example, variations in the social discount rate that was used to compute the present value of benefits from forest carbon sequestration caused benefit estimates to vary by an order of magnitude.

In many instances data gaps exist that cannot be overcome without more investment in primary studies. A variety of investments in ecological and economic science will be needed to increase precision. Some gaps are more likely to be bridged by coordinating with data developers and modeling analysts inside and outside the USDA. For example, agencies outside USDA may have

⁷ A preferred measure of benefits from crop yield changes is the change in producer or consumer surplus due to a change in the supply conditions (producers offer goods at a lower price). Consumer surplus represents the excess of what the consumer would have been willing to pay over the price or costs of goods, whereas producer surplus represents the amount a producer is willing to accept for the good (a value related to profit). These surpluses are considered more robust measures of benefits because they account for changes in supply and demand conditions that may affect the benefits received by consumers or producers.

complementary data and models needed to fill in pieces of the conceptual value diagram, such as effects on species valued for hunting, fishing and wildlife watching.

Easier to evaluate future alternatives rather than past actions

Long-term investments in data collection and modeling as part of the USDA Forest Service's Forest Inventory and Analysis (FIA) Program, the Natural Resources Inventory (NRI), and the Conservation Effects and Assessment Program (CEAP) were instrumental in allowing the carbon sequestration team and the water quality teams to make the first links in their *conceptual value diagrams* and connect actions to observed biophysical changes. However, models were not generally available to separate the proportion of observed biophysical change that is attributable to policy or program actions from other background trends. Instead, for all but two ecosystem service analyses, modelers—in research conducted prior to this project—used field data to estimate the effects of future land management scenarios rather than past program actions. Therefore, the resource teams' choices were constrained, in all but these two cases,⁸ to valuing future scenarios of change rather than past actions. It is important to note that understanding potential future benefits is necessary for supporting future planning, but the common choice by researchers to project future changes limits the evaluation of program benefits.

Future what-if scenarios are more commonly available because they allow modelers to control for confounding conditions such as land use changes when estimating changes due to program actions. For example, if additional farm acres are brought into production while additional conservation practices are implemented, the observed acres put into production can mask conservation practice effects. Therefore, historical analyses of programs require sufficient spatial and temporal data to allow researchers to model and control for non-program effects when constructing the baseline of comparison with program impacts. These counterfactual (unobserved) scenarios are critical to judging past program effects by controlling for external influences, but they are often time-consuming to construct.

If modelers can be given sufficiently precise information on practices and land use and management changes unrelated to program initiatives, then monitoring data and existing biophysical models may be sufficient to estimate counterfactual scenarios. However, there is also a high level of natural variability in these systems. Therefore, the magnitude of biophysical change must be large enough to overcome such variability for such models to be successful at measuring changes in ecosystem services (Sowa et al. 2016).

Analysis complexity varies widely among ecosystem services

The analyses across the three resource areas of concern demonstrated how many different models need to be linked to estimate total benefits. At one extreme, in the water quality analyses, only two models were needed to connect conservation practices for sheet and rill erosion to benefits from dredging costs avoided for reservoirs and channels. In this case, the biophysical and economic models were relatively simple empirical relationships to estimate overland soil movement and costs of sediment deposition. At the other extreme, at least four pre-existing models: Agricultural Policy / Environmental eXtender (APEX), Soil and Water Assessment Tool (SWAT), the piscivore condition model, and the recreational fishing benefit transfer meta-analysis, as well as multiple intervening calculations were needed to link scenarios of conservation practices to recreational fishing benefits.

⁸ The two exceptions were the ecosystem services analyzed by the water quality team that were generated by preventing sedimentation of reservoirs and shipping channels. That work was possible because models had been developed by researchers within the USDA Economic Research Service to examine effects of conservation practices on dredging costs.

Most valuation analyses fell in the middle, where conservation practice scenarios were analyzed using one or two models to generate changes in ecological outcomes before valuing that change using a unit value benefit transfer (as described in the Project Introduction). Two or more biophysical models can be needed to move from basic biophysical conditions or processes (e.g., nutrient loads, soil carbon content) to a more integrated ecological outcome (e.g., game fish response, water clarity, or net annual carbon sequestration) as needed to link to an economic valuation model. Models varied in complexity, so a single model may have involved numerous computations, most of which introduced an unmeasured level of uncertainty into calculations.

Benefit transfer exercises that were used to value outcomes most commonly assumed a linear relationship between ecosystem service change and benefits. Yet, the reliability of this assumption is likely to vary by context and may be inappropriate in cases in which ecosystems or species are rare or the service is irreplaceable. For example, by convention, each additional unit of carbon sequestration within a given year receives the same value, which helps to simplify what would otherwise be an analytically intractable problem of incorporating different non-linearities of multiple risks. Similarly, the pollinator habitat team proposed a linear relationship between crop harvest or honey production and relative CRP benefits, which may be a reasonable assumption. However, the value of some types of changes can depend on whether the change is sufficient to overcome a limitation to a service being delivered (Wainger and Mazzotta 2011; Sowa et al. 2016). For example, water quality changes in severely degraded streams only improve trout fishing if they overcome the biophysical (threshold) limits to trout survival or reproduction.

Benefit transfers can be adapted to reflect non-linearities either by using thresholds or by estimating values for different baseline conditions. The water team demonstrated this type of adjustment when they used a baseline sportfish catch rate to evaluate the value of additional fish caught. What was not demonstrated, due to time limitations, was that the functional transfer could have been fit multiple times to better represent incremental improvements in sport fishing. Having enough valuation studies to develop functional transfers is, then, critical to incorporating non-linearities in value changes.

Whether monetary valuation was useful varied by decision type

Analysts on different resource teams had different responses to the level of uncertainty in value estimates, seemingly due to the different decision contexts that they constructed. The pollinator habitat team envisioned a decision context in which they were aiming to differentiate site-by-site differences in ecosystem services to compare offers made to the CRP. They concluded that, based on existing science, calculating monetary values would not improve upon estimates of biophysical improvements (e.g., crop yields, honey production) and would only add additional uncertainty to these cost-effectiveness measures.

In contrast, the water team aimed to provide an order of magnitude estimate of benefits from conservation practices in a large watershed. They used the available data and models to create rough benefit estimates, but also concluded that estimates would require further refinement before being used. The carbon team was, perhaps, the most comfortable of all the teams in using the substantial amount of available data and published models to project benefit estimates for a (hypothetical) nationwide program of forest management. These choices demonstrate how decision context can influence the determination of whether an ecosystem service can or should be reliably valued.

Benefit relevant indicators were rarely quantified outside of valuation

Many federal programs already use ecological indicators as proxies for benefits, and many of these types of metrics were quantified in the course of the ecosystem service valuation analyses. However, most ecological metrics are not BRIs because they have not explicitly connected ecological changes to beneficiary concerns or preferences. In contrast, BRIs are meant to be tightly correlated with benefits by evaluating the conditions under which a change has value.

The teams identified BRIs during the conceptual process, but none of the teams quantified BRIs for services that could not be valued. For example, the pollinator habitat team specified many types of quantitative indicators to represent potential benefits (i.e., recreational enhancements due to the increased abundance or diversity of song and game bird populations) but noted that “the lack of information on alternative bird watching sites in the context of valuing competing CRP offers was likely to preclude the explicit use of such an approach”.

For this project, the teams tended to include services that could be monetarily valued and did not quantify non-monetary benefit indicators or BRIs that were not linked to monetizable benefits. The teams predominantly used BRIs only as stepping stones to valuations, rather than as a method for expanding the scope of benefits analyzed. However, it can be perfectly legitimate to include a BRI that is not appropriate for monetization to address other dimensions of value for program evaluation. Yet, the interdisciplinary teams seemed hesitant to take this step. Therefore, if stakeholders or managers want to broaden the services that are measured, they will likely need to identify specific approaches for using quantitative indicators to measure services that cannot be monetized. However, for BRIs to serve as close proxies for benefits, they must incorporate beneficiary interest. For example, rather than measuring area of pollinator habitat, analysts could estimate the area of cropland that is likely to benefit from increased abundance of pollinators. For more information, see Chapter 1: Assessing Pollinator Habitat Services to Optimize Conservation Programs.

Location-specific nature of values are challenges for program analysis

The USDA aims to have a nationally consistent process for estimating program benefits that also reflects regional differences in values. However, the carbon team was the only team that demonstrated an approach to measuring values nationwide. Their task may have been simpler than other teams because, although carbon sequestration rates vary by location, the value of carbon sequestration does not depend on local socio-economic conditions. Although climate change risks vary by location, the risks are ameliorated to the same degree by sequestering carbon anywhere, since climate change is driven by the total atmospheric gas content. Further, because forest data is collected consistently nationwide, modelers had sufficient data to incorporate many of the location-specific variables that affect risk-adjusted rates of net annual carbon sequestration such as climate, forest age, and fire risk. Therefore, once the analysts had controlled for regional differences in net annual sequestration rates, they could generate a monetary value for all regions by applying a common unit value transfer, the social cost of carbon, to estimate a value for national (potential) sequestration (U.S. Interagency Working Group on Social Cost of Carbon. 2013).

More commonly, values will vary by location in ways that create substantial challenges to national assessments, as discussed in the water quality and pollinator habitat resource areas. As a simple example, nutrient reductions to water bodies are not going to have substantial value if the water body is readily assimilating current loads (i.e., if it is not eutrophic or otherwise impaired). Even in cases in which it may be safe to assume that nutrient reductions are beneficial, challenges remain in tailoring valuation to local preferences and other factors that vary spatially or temporally (such as anglers who prefer fish that tolerate or thrive in eutrophic water). More generally, people must want

a change (i.e., have demand) for a given ecological change to have value, and the degree of demand will almost always vary spatially.

The water quality team chose a data- and model-rich regional case study area (the Western Lake Erie Basin) to demonstrate valuation methods that were tailored to many (but not all) location conditions. Similar methods could be used for any region with similar information. However, tailoring values to location-specific data is time-consuming, because many data sources are usually needed, including population data, property data, ecological monitoring data (to judge impairments), and recreational use data, among others.

Often, data are not available to support all the conditions considered necessary for an economic analysis. For example, the pollinator habitat team did not find credible data on the local links between specific forb-rich plantings on CRP lands and pollinator population abundance or health, but described the monitoring data needed to make site comparisons. Data gaps can prevent the estimation of benefits or limit the amount of location-specific adjustments that are feasible, as demonstrated in the water quality chapter.

SUGGESTED NEXT STEPS

1. Engage federal government and academic scientists

This ecosystem service valuation project demonstrated that government and academic scientists can and should be engaged as interdisciplinary teams. They bring different perspectives and different sets of knowledge and skills to the analytical process. Their joint participation is essential to assure that valuations address the pragmatic needs of the agency for effective program delivery while using the latest science to provide credible estimates. Academic scientists should be chosen not only for their expertise in the resource in question, but also their demonstrated commitment to further applied science in support of federal conservation programs.

Government scientists bring a federal perspective that few if any academic scientists possess. Because of their agency responsibilities, they have intimate familiarity with the programs being studied, including their statutory and administrative purposes and implementation details, such as priority targets and salient services. As the valuations are intended to inform national program-level actions, the need to aggregate values from the various regions into national information bases that illustrate regional tradeoffs and complementarities becomes critical for the USDA and oversight agencies. Moreover, government scientists are likely to be more familiar with regional and national databases that can be used in the analysis, and with their limitations and advantages.

Academic scientists generally add more regional knowledge of services and their use and non-use values. This may include finer spatial detail on service variation across the region and other supply-side details than national data reveal. Their knowledge and expertise also likely includes familiarity with the variations in demand for the services and trends in the demand drivers over time. Finally, academic researchers, owing to the nature of their prime responsibility to advance theory and methods, place a heavy emphasis on the frontier developments in each field, which is also important to federal scientists, but which can sometimes take a back seat to specific program needs. As federal valuations of ecosystem services expand to include all salient but less tangible benefits, use of the latest theory and method advancements will be crucial.

The challenge is that program knowledge does not exist within the purview of the researchers. Thus, it is essential that in any process, the program staff are actively engaged and included in any

valuation projects. Government program administrators and lawyers have knowledge of outcome metrics, program legal aspects, reporting requirements, and other aspects of communicating the value of ecosystems services to decision-makers both within and outside the government.

2. Broaden the set of services to be quantified

The theory and data necessary to analyze ecosystem service values that emanate from federal programs are patchy and immature for many resources of concern. Additional valuation studies will be needed to broaden valuation to new services and deepen understanding of how values differ by location. As stated at the outset, significant progress in building the science supporting ecosystem service valuation has occurred since the turn of this century (Guerry et al. 2015). Nonetheless, the application of that improved science to federal decision making has lagged (Polasky et al. 2015, Schaefer et al. 2015). Provisioning services with a high market profile, such as crops, often get major attention as constituencies argue for inclusion of their value in policy making. However, emergent science is increasing the evidence of connections for other types of flows beyond water quality, particularly for non-use services. Even if data are insufficient to value a broad set of services, rigorous qualitative assessments of important non-quantifiable effects can complement the quantitative assessments to give a fuller picture of impacts for decision makers (Ervin et al. 2014).

Two dimensions not emphasized in this description are the services that do not pass through markets, i.e., non-market services, and services that are not used in a physical or biological sense, i.e., non-use. Examples from this project include some cultural values placed by Lake Erie area residents on improving water quality by limiting nutrient and pesticide contamination and providing native pollinator habitat that increases wildlife population numbers and health.

The non-market and non-use categories are legitimate conceptual sources of value that have been documented in local and regional studies of environmental quality (NRC 2005). However, strategic investments in the underlying science and databases will be required to prepare credible regional and national estimates of their quantitative magnitudes. The advancements in science can come from USDA internal and external research funding sources, such as the National Institute for Food and Agriculture, the National Science Foundation, and the National Oceanic and Atmospheric Administration. It would be ideal, for the reasons discussed above, if such new research initiatives brought together interdisciplinary teams of government and academic scientists. One key task of the research programs will be to develop new models and databases that permit a tight assessment of how conservation program practices affect salient ecosystem service outcomes, apart from other potential influences, to isolate the effect of programs on human values.

3. Evaluating uncertainty informs appropriate application of results

As the example analyses in the resource team chapters demonstrate, uncertainty in ecosystem service values cannot be avoided. Models of socio-ecological systems can only be developed by tolerating the uncertainty that stems from missing data, incomplete and untested models and assumptions, the influence of external shocks, and other factors (Johnson et al. 2012). Ecosystem science and data will always be incomplete at any point in time. Waiting for complete information amounts to making the perfect the enemy of the good, as Voltaire admonished (Ervin et al. 2012). Furthermore, the implementation of major pieces of legislation often must begin with an incomplete base of science and then gradually fill in key gaps.

Yet uncertainty analysis is useful for developing a shared understanding of what counts as “best available science.” Further, policy makers can be informed of the nature and magnitudes of the uncertainties that bear on their decisions by analyzing the potential effects of uncertainty on their

decision outcomes (Ervin et al. 2014). Therefore, recognizing and disclosing the uncertainties encountered, and analyzing and describing the potential impacts those uncertainties could bring to bear on the decision, will build the trust of resource program managers. Recent quantitative analyses of values have taken up this challenge (e.g., Johnson et al. 2012).

If quantitative uncertainty analysis is not feasible, a qualitative assessment of uncertainty may be useful. A first step would be to list all major sources of uncertainty for the valuation analysis being conducted and provide an objective or subjective assessment of their import. The example analyses in the water quality and carbon chapters illustrate the type of sensitivity analyses that can be used to demonstrate how uncertainty can affect value estimates.

4. Improve accuracy in benefit transfer

Ecosystem service values generally derive from the underlying demand and supply conditions in a local area or region. Most relevant is that the value of a change in an ecosystem service frequently depends on its relative scarcity, which is typically defined as demand in excess of supply, at the prevailing cost to access the service. Adding more of a service where it is in short supply will be more valuable than adding it where it is plentiful, all else being equal.

Because of the need to reflect local conditions, benefit transfer accuracy is improved by the ability to tailor values to location conditions. For example, if an area places a high value on recreational activities in native prairies, such as bird watching, and little of that habitat remains, the value will be higher than under the reverse conditions. However, the ability to tailor values to local conditions comes from the capacity to conduct primary valuation to represent specifics or from having sufficient studies across a range of conditions to be able to interpolate appropriately.

Clearly, it is implausible to conduct studies for all major user groups of ecosystem services impacted by federal conservation programs. However, strategic investments in primary studies to cover a sample or gradient of supply and demand conditions could facilitate the development of generic transfer functions of the kind demonstrated for sport fishing in Chapter 2.

Three research priorities emerge related to ecosystem service valuation through benefit transfer for USDA conservation programs. First, we need more gap-filling studies of what are judged to be salient ecosystem service values for areas in which little research has been done and/or the estimated benefits of the services are likely to be significant. In other words, we need new information from these areas even to gauge whether the demand and supply conditions are close enough to transfer values without significant error. Second, new tools that improve the ability to transfer values based on existing data, such as meta-regression models, merit increased attention in the USDA and other research organizations. Finally, new modeling efforts of the likely shifts in demand for ecosystem services over space and socio-demographic groups and over time will be critical. For example, as new exurban development occurs near rural areas where conservation practices are implemented, the differences in income of the in-migrants versus the current residents will be important to understand value effects.

5. Develop analytical processes to construct national ecosystem service value assessments

A comprehensive evaluation of the benefits of USDA conservation programs not only requires improved benefit transfer capabilities, it also requires better data and models to quantify changes in relevant ecological outcomes. For example, currently, it is most common for water quality models to measure changes in loadings to waterbodies without considering whether changes have a substantial

impact on water quality or species that are consumed, viewed or just appreciated by people. Further, model coverage is not nationally consistent, which hinders national program evaluation. Thus, expanding the set of ecosystem services that can be evaluated will depend on similarly strategic investments to better capture relevant biophysical changes in a variety of settings.

Some federal databases, such as the National Resources Inventory (USDA NRCS 2016b), have sophisticated sampling schemes to assure adequate national coverage. This on-the-ground survey collects data at 10 year intervals on the status and changing conditions of the soil, water, and related resources from 300,000 sample plots and 800,000 sample points on private land in the United States. Such information is critical to building models that analyze the shifting supplies of ecosystem services from farm and forest lands. Despite the large number of sample plots and points, the data are not sufficient to analyze fine scale changes in ecosystem services without augmentation from other sources and data access requires approval. Therefore, expanded data collection with more replication and streamlined USDA data access protocols that ease use but also protect the confidentiality of the data would enable the analysis of values on a site-by-site basis. These improvements should be done in a complementary fashion with natural resource surveys by other agencies.

6. Develop methodologies for integrating information from monetary and non-monetary metrics into decision analytic frameworks

The operative question for enabling a comprehensive assessment of ecosystem services is how to compare a full set of quantified values when not all salient effects can be monetized. If analysts follow the guidance to use benefit-relevant indicators to broaden the set of ecosystem services, then ecosystem service values cannot be directly aggregated. The cost-effectiveness of management options can be compared by using stakeholder preferences to weight values that have been measured in different units. Each management option is evaluated by comparing an array of ecosystem service outcomes, including monetized and quantified (but not able to be monetized) impacts of pursuing the options that have been put into common units.

Numerous multi-criteria approaches have been devised to guide users to establish weights and apply them to support decisions (e.g., Mendoza and Martins 2006). Incorporating subjective judgments about uncertainty and personal values are considered part of modern decision science (Clemen and Reilly 2013). Considering the impacts of ecosystem services that cannot be monetized, along with monetary outcomes, has been judged good analytical practice for benefit cost analysis (U.S. Office of Management and Budget 2003).

Key elements of multi-metric approaches (often called multi-criteria analysis, among other names) are to assess all metrics that are significant to stakeholders, scale indicators, and weight outcomes by preferences. Often preferences are organized hierarchically to avoid giving more weight than is intended, simply based on the number of metrics used to represent a given type of benefit. Tables and graphics are used to conduct preference elicitation and reflect results back to participants and decision-makers for vetting and eventual use in decisions.

Conclusions and Recommendations

As our test cases made clear, the information that is currently available to identify potential ecosystem service values generated by conservation actions on farms and forests has improved in recent years. A substantial subset of those identified values could be monetized using the types of methods that are routinely used in federal decision making. However, even simple benefit transfer

exercises required considerable analytic work to match up ecological model outputs with required economic model inputs.

Results demonstrated that information available to conduct valuation varies by natural resource area. The carbon sequestration team had the most extensive and most thoroughly peer-reviewed literature to draw on for their analysis of the bundle of benefits (costs avoided) from reducing climate change risks. The water quality team used well-established ecological models and benefit transfer methods on multiple services. However, in two cases, only a single primary valuation study was available to provide region-appropriate values to transfer. The pollinator habitat team identified recent ecological research that has broadened the understanding of the types of potential benefits from pollinator habitat, such as improved honeybee health, and determined that valuation of increased crop yields would be relatively straightforward.

The teams had limited success broadening the suite of measured benefits beyond those that might be used in traditional cost-benefit analysis. The water quality team was the only team that demonstrated an example of monetizing a non-use (intangible) value, in this case, benefits from improving aquatic ecosystem condition. Further, some monetary values, while feasible to estimate, relied on a single primary valuation study, giving the analysts low confidence in the results. Additional primary valuation studies will be needed to expand the scope of services that can be valued or to enable the type of functional benefit transfers that apply data to adjust values to local conditions. It appears that traditional areas of environmental economic valuation (recreational fishing, property values) are the ones most amenable to monetization. Benefit-relevant indicators will continue to be necessary in the short term to quantify service changes that either have not yet been valued or that may not be amenable to valuation due to limits of understanding or cultural sensitivities.

The list of benefits and harms of ecosystem changes that can be valued is expanding but can never be exhaustive. Even if actions under USDA conservation programs can be linked to human welfare, not all recognizable welfare effects can be readily quantified or monetized, such as losses of culturally significant ecosystems. Furthermore, the composition of users will likely shift over time and alter the demands for certain ecosystem services that can be delivered by conservation programs. Hence, valuation analysis will need to evolve to reflect new conditions.

We offer the following conclusions and propositions from this project.

Suggestions for a systematic approach to ecosystem service valuation

To different degrees, the case studies found similar opportunities for and constraints to valuing ecosystem services. The following suggestions for promoting success emerged:

1. Define terms, such as what is meant by ecosystem service values, but retain the flexibility to tailor metrics and methods to statutory authority, local conditions, and data limitations;
2. Create interdisciplinary analytic teams of government and academic scientists to enhance the credibility of all aspects of value assessment;
3. Monetize benefits where appropriate, but use benefit relevant indicators to complement or serve as alternatives to monetary values when significant stakeholder concerns cannot be monetized;
4. Demonstrate sensitivities to assumptions and sources of error to develop a shared understanding of how results can best be used in policy; and
5. Identify opportunities for strategic investments in science that will allow more services to be measured and valued.

Caveats to using valuation to support decisions

The valuation approaches demonstrated in the three resource areas all isolate ecosystem service value flows to specific stakeholders (Figure 2). This approach to simplifying analysis has advantages and disadvantages. Deconstructing complex ecological and economic systems into specific pathways of benefits often enables empirical valuation with clear assumptions, explicit error handling, and parsimonious models. However, the reductionist approach may ignore benefits that only emerge from a systemic and holistic evaluation of multiple functions and processes, such as the benefits of increased resilience to multiple ecosystem services (Wainger & Boyd 2009).

Omitted information only becomes a major problem if it creates systematic bias in decision making or disenfranchises groups. For example, a common issue in ecosystem service valuation is that non-use values are missing. Omitting these intangible benefits that people derive from preserving ecological elements that they appreciate but do not use will create a systemic bias because use values are often high near dense populations, whereas non-use values are often high in remote locations.⁹ Remote locations are more likely to have high non-use values due to higher levels of ecosystem function that contribute to such outcomes as superior habitat for rare species. Therefore, omitting non-use values will tend to result in a consistently higher value for changes in and around urban areas, relative to rural areas.

Achieving next steps

Achieving the goal of increased accuracy and broadening the range of services that can be feasibly valued by government analysts will require new scientific investments. Future research needs suggested by this project (addressed within or external to the USDA) include:

1. Continue supporting monitoring programs that enable the outcomes of program actions to be distinguished from those likely to occur under a without program baseline and to understand the geographic variability in program outcomes at meaningful ecological scales (e.g., Conservation Effects Assessment Program; Forest Inventory Analysis).
2. Develop streamlined National Resources Inventory and Conservation Effects Assessment Project cropland data access protocols that also protect data confidentiality.
3. Catalog relevant models (biophysical and economic) and provide metadata on the appropriate use of model results in the same or new locations.
4. Analyze the temporal patterns in program outcomes and ecosystem service values under uncertain biophysical and socioeconomic trends.
5. Create additional benefit transfer functions (e.g., meta-regression analysis) to enhance the replicability of value estimates across program locations.
6. Conduct additional primary valuation studies for ecosystem services from farm and forest lands that have been valued in some areas but that lack sufficient studies for creating benefit transfer functions.
7. Conduct primary valuation studies for ecosystem services that have not been valued.
8. Develop peer-reviewed and replicable quantitative methods for measuring benefit relevant indicators for services that will not be valued.

These recommendations reflect the need to do more to enable valuation by making data and tools more accessible and by working to expand knowledge on both the ecological and socio-economic fronts. We believe that these research needs can best be filled through interdisciplinary approaches that integrate diverse knowledge bases. Three types of actions could help fill the critical gaps. As a

⁹ This bias can be exacerbated when federal guidance is to avoid stated preference studies since such studies are the only method available for monetizing non-use values (Freeman et al. 2014).

first step, agencies within the USDA that have responsibilities for the administration and assessment of the conservation programs could coordinate their data collection and analytical efforts on common resource issues. Second, the USDA can join forces with other federal agencies that have overlapping responsibilities regarding the natural resources of concern to expand the science and to continue identifying needs for future research. This finding applies to all three resource issues analyzed in this prototype analysis, as the impacts of pollinator habitat, water quality, and forest carbon sequestration extend beyond USDA jurisdiction. Finally, intramural and extramural research initiatives and funding can be established to target new resources to priority areas needed to advance scientific theory, methods, and database development. These research initiatives could be informed by regular reviews of the state of the science on valuing ecosystem service values by agency and academic scientists. It is imperative that these research priorities are formed with interdisciplinary teams of natural and social scientists from inside and outside the federal government who have extensive knowledge of the priority natural resource issues.

Figure 2 (a-c). Conceptual value diagrams created by natural resources teams. (View following pages.) Each graphic documents how the teams conceptualized the valuation challenge in linking actions to ecological changes and eventually to benefits or harms. Figures demonstrate the potential structure of benefit-relevant indicators (services in Figure a; benefit-relevant indicators in Figure b), which are outputs that can be used or appreciated by people. They differ from basic ecological structure or processes (process column in Figure a; features and processes in Figure b), such as bird abundance, and differ from monetized values (shown in value or benefits columns/boxes).

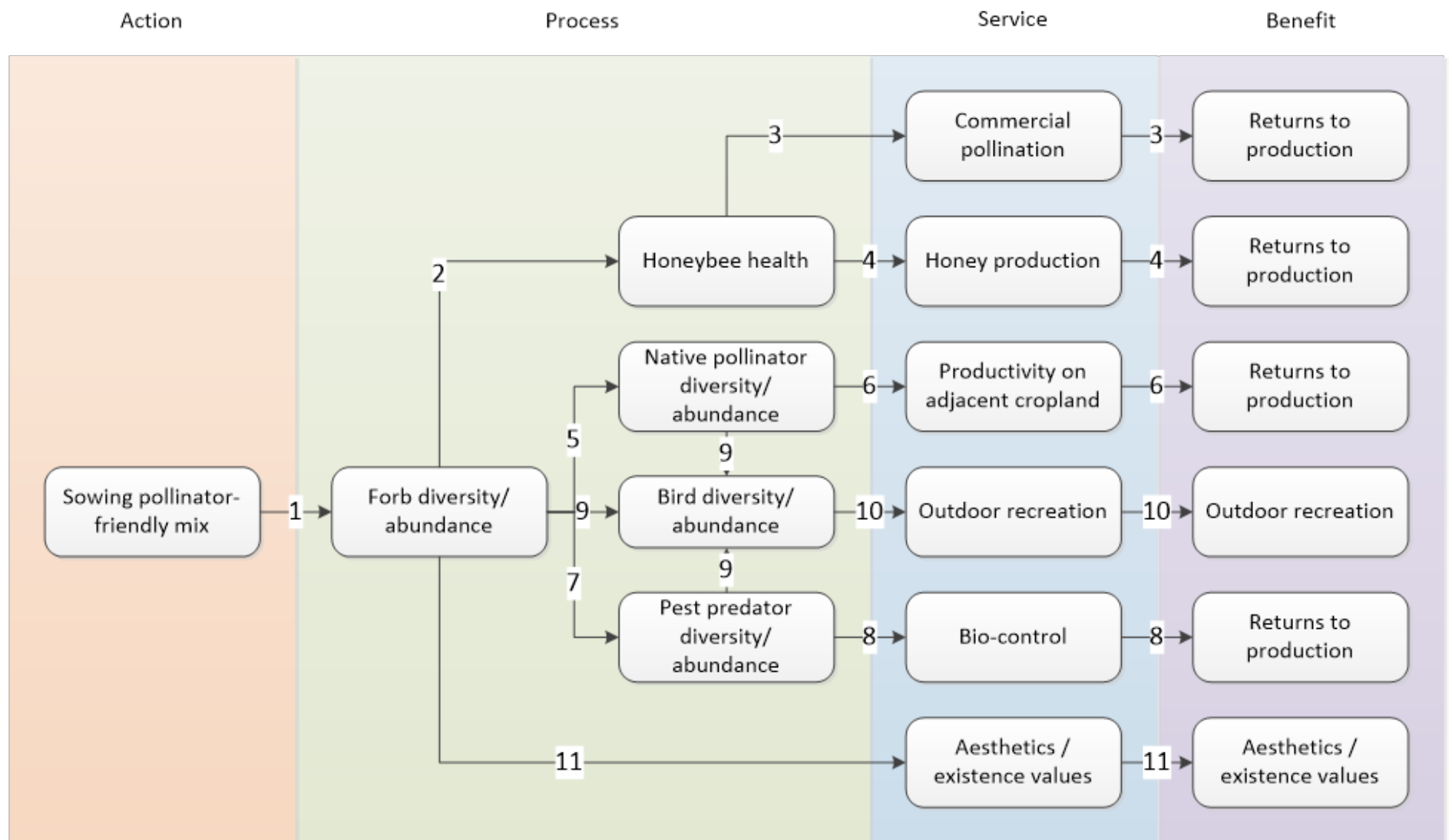


Figure 2a. Pollinator habitat: Linking forb-rich habitat on agricultural conservation lands to ecological processes, ecosystem services, and potential human values.

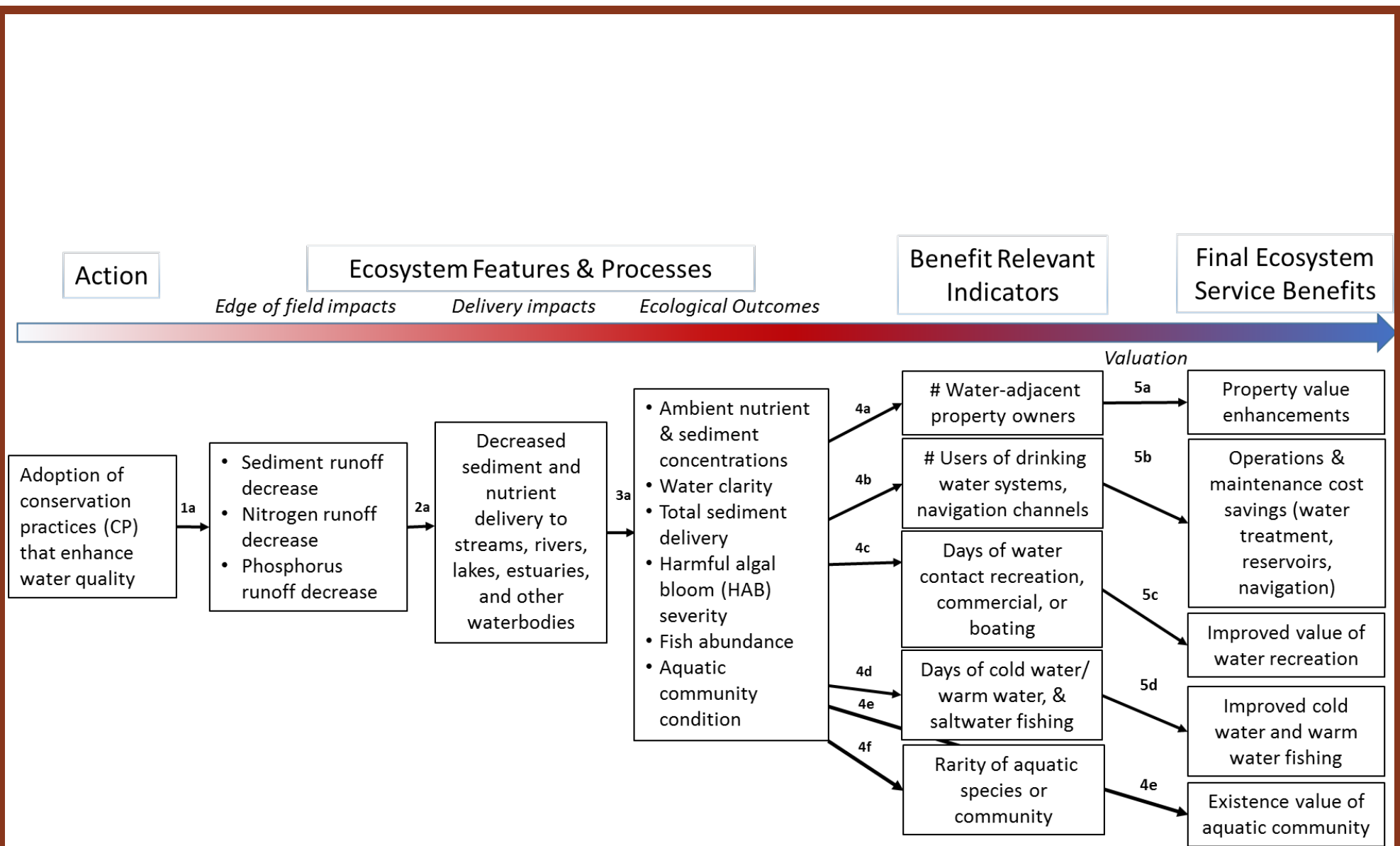


Figure 2b. Water quality: Tracing the flow of biophysical effects from USDA conservation practices to ecological outcomes, benefit-relevant indicators, and ecosystem service values for users and non-users.

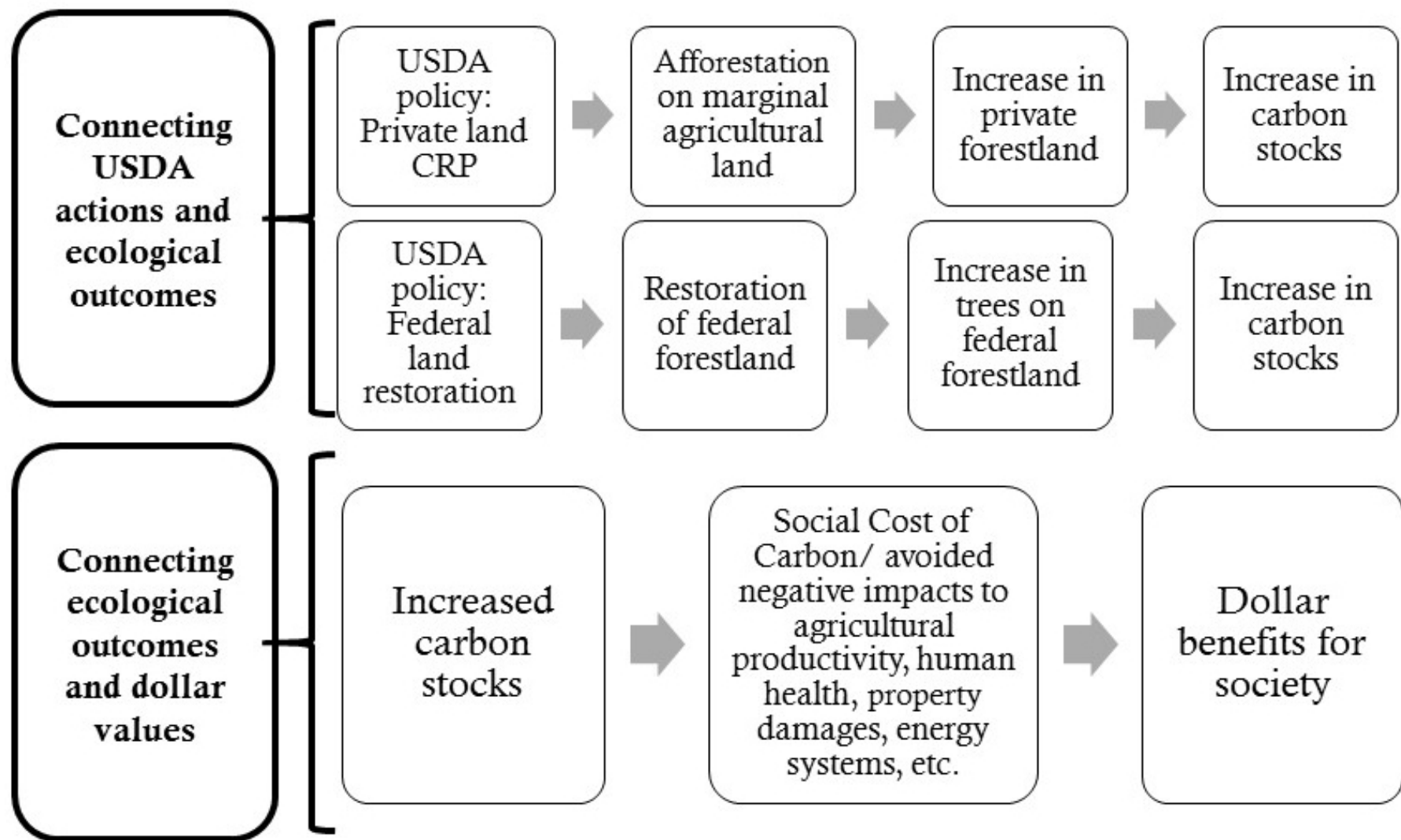


Figure 2c. Carbon sequestration: Connecting USDA conservation programs for afforestation and reforestation to carbon sequestration and values to humans of avoided damages.

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Appendix A. Primary Valuation Methods Applied to Ecosystem Goods and Services*

	Valuation Method	Description	Examples of Ecosystem Services Valued
Market Valuation	Market analysis and transactions	Derives value from household's or firm's inverse demand function based on observations of use	Fish Timber Water Other raw goods
	Production function	Derives value based on the contribution of an ecosystem to the production of marketed goods	Crop production (contributions from pollination, natural pest control) Fish production (contributions from wetlands, seagrass, coral)
Revealed Preferences	Hedonic price method	Derives an implicit value for an ES from market prices of goods	Aesthetics (from air and water quality, natural lands) Health benefits (from air quality)
	Recreation demand methods	Derives an implicit value of an on-site activity based on observed travel behavior	Recreation value (contributions from: Water quality and quantity Fish and bird communities Landscape configuration Air quality)
Stated Preference	Contingent valuation (open-ended and discrete choice)	Creates a hypothetical market by asking survey respondents to state their willingness-to-pay or willingness-to-accept payment for an outcome (open-ended), or by asking them whether they would vote for or choose particular actions or policies with given outcomes and costs (discrete choice).	Non-use values (species and ecosystem protection), recreation aesthetics
	Choice modeling / experiments	Creates a hypothetical market by asking survey respondents to choose among multi-attribute bundles of goods and derives value using statistical models.	Non-use values (species and ecosystem protection), recreation aesthetics
Defensive and Damage Costs Avoided**	Damage costs avoided	Value is inferred from the direct and indirect expenses incurred as a result of damage to the built environment or to people.	Flood protection (costs of rebuilding homes) Health and safety benefits (treatment costs)
	Averting behavior / defensive expenditures	Value is inferred from costs and expenditures incurred in mitigating or avoiding damages	Health and safety benefits (e.g., cost of an installed air filtration system suggests a minimum willingness-to-pay to avoid discomfort or illness from polluted air)
	Replacement / restoration cost	Value is inferred from potential expenditures incurred from replacing or restoring an ES.	Drinking water quality (treatment costs avoided) Fire management
	Public pricing	Public investment serves as a surrogate for market transactions (e.g., government money spent on purchasing easements).	Non-use values (species and ecosystem protection) Open space Recreation

* Adapted from Turner et al. (2008), Table 4.8 and reproduced from the NESP guidebook.

** These methods are frequently used for expediency, but are also more weakly grounded in economic theory than other approaches and can be misused. See discussion in NESP guidebook and in U.S. EPA 2009 and NRC 2005.

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