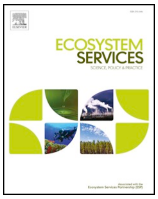
Ecosystem Services 43 (2020) 101099

Contents lists available at ScienceDirect

Ecosystem Services

journal homepage: www.elsevier.com/locate/ecoser



Testing ecosystem accounting in the United States: A case study for the Southeast

Katherine J.D. Warnell[a], Marc Russell[b], Charles Rhodes[c], Kenneth J. Bagstad[d,*], Lydia P. Olander[a], David J. Nowak[e], Rajendra Poudel[f], Pierre D. Glynn[g], Julie L. Hass[h], Satoshi Hirabayashi[i], Jane Carter Ingram[j], John Matuszak[k], Kirsten L.L. Oleson[l], Stephen M. Posner[m], Ferdinando Villa[n]

[a] Nicholas Institute for Environmental Policy Solutions, Duke University, United States
[b] U.S. Environmental Protection Agency, United States
[c] TBD Economics, United States
[d] U.S. Geological Survey, Geosciences & Environmental Change Science Center, United States
[e] U.S. Forest Service, United States
[f] National Oceanic and Atmospheric Administration, United States
[g] U.S. Geological Survey, Water Cycle Branch, United States
[h] Bureau of Economic Analysis, United States
[i] The Davey Institute, United States
[j] Ernst & Young, United States
[k] National Council for Science and Environment, United States
[l] University of Hawai'i Mānoa, United States
[m] COMPASS, United States
[n] Basque Center for Climate Change and IKERBASQUE, Basque Foundation for Science, Spain

ARTICLE INFO

Keywords:
Experimental ecosystem accounting
System of environmental-economic accounts
National Ecosystem Services Classification System
Pollination
Air quality regulation
Recreational birding

ABSTRACT

Ecosystem accounts, as formalized by the System of Environmental-Economic Accounting Experimental Ecosystem Accounts (SEEA EEA), have been compiled in a number of countries, yet there have been few attempts to develop them for the U.S. We explore the potential for U.S. ecosystem accounting by compiling ecosystem extent, condition, and ecosystem services supply and use accounts for a 10-state region in the Southeast. The pilot accounts address air quality, water quality, biodiversity, carbon storage, recreation, and pollination for selected years from 2001 to 2015. Results illustrate how information from ecosystem accounts can contribute to policy and decision making. Using an example from Atlanta, we also show how ecosystem accounts can be considered alongside other SEEA accounts to give a more complete picture of a local area's environmental-economic trends. The process by which we determined where to place metrics within the accounting framework, which was strongly informed by the National Ecosystem Services Classification System (NESCS), can provide guidance for future ecosystem accounts in the U.S. and other countries. Finally, we identify knowledge gaps that limit the inclusion of certain ecosystem services in the accounts and suggest future research that can close these gaps and improve future U.S. ecosystem accounts.

1. Introduction

Natural capital accounting is a method of assessing the contributions of ecosystems to the economy consistent with the System of National Accounts, which governments use to measure economic activity (Guerry et al., 2015; WAVES, 2012). Examples include the contributions of recreation (BEA, 2018), land (Wentland et al., this issue), or water (Bagstad et al., this issue) to the U.S. economy and accounts developed for ecosystem services in other nations (see Heris et al., this issue, examples for Europe in this issue and Section 1.1). The information contained in natural capital accounts highlights the connections between ecosystems and economic systems, and can give governments, businesses, and other resource managers a better understanding of (1) economies' reliance on ecosystems and (2) the effects of

---

[*] Corresponding author at: P.O. Box 25046, MS 980, Denver, CO 80225, United States.
E-mail address: kjbagstad@usgs.gov (K.J. Bagstad).

https://doi.org/10.1016/j.ecoser.2020.101099
Received 21 June 2019; Accepted 24 March 2020

2212-0416/ Published by Elsevier B.V.

economic and policy choices on ecosystems. The accounts track changes in ecosystems that have implications for various industries and user groups, and can support natural resource and ecosystem management to sustain economic benefits into the future.

Ecosystems yield complex flows of value that older iterations of national and corporate accounts have hidden or ignored, such as the protection of lives and property from flooding, crop pest control by wildlife, and recreational value associated with wildlife and landscapes (Boyd et al., 2018). Certain ecosystem assets (represented as stocks in ecosystem accounts), ecosystem service flows from those assets, and wealth are hidden or omitted by previous accounting practices, preventing informed decision making (Hein et al., 2015, 2016). Ecosystem accounting is a practical attempt to build data and accounting structures to fill many of these gaps, thereby improving asset and income management.

Since the early 1990s, the United Nations and partner organizations have developed a standard framework for natural capital accounting called the System of Environmental-Economic Accounting, or SEEA. The core of SEEA is the Central Framework, which quantifies environmental goods and their contributions to the economy, including land, water, minerals, and a number of other resources such as timber and fish (U.N. et al., 2014a). Accounts for several components of the SEEA Central Framework (SEEA CF) have been developed recently for the U.S. (Bagstad et al., this issue; Wentland et al., this issue). Complementing the Central Framework, the SEEA Experimental Ecosystem Accounts (SEEA EEA) track the extent and condition of ecosystem assets (e.g., stocks of forests, wetlands, cropland) and the flows of various ecosystem services they provide to people and to the economy (U.N. et al., 2014b).[1]

For the SEEA Central Framework accounts, there is general agreement about what should be measured. However, as reflected in the SEEA EEA's still-experimental status, defining what should be included in ecosystem accounts is more problematic. Difficulties of theory, definition, and scale arise when attempting to link environmental data, models, and economic measures (U.N., 2017). While some other countries have developed ecosystem accounts for key resources, we know of only two other examples for the U.S.: ecosystem extent, condition, and supply and use tables for Long Island's South Shore Bays (Dvarskas, 2019) and national-scale urban ecosystem accounts (Heris et al., this issue). This paper demonstrates how ecosystem accounts can be developed for the U.S. at a regional scale in alignment with the SEEA EEA framework and the latest U.S. Environmental Protection Agency efforts to classify final ecosystem services in a consistent, systematic, and intentional manner (National Ecosystem Services Classification System, NESCS, U.S. EPA, 2015).

1.1. Recent applications of ecosystem accounts

Ecosystem accounts following the SEEA EEA framework have been compiled at both national and subnational scales to track environmental-economic trends in a number of developing and industrialized countries. Examples include Canada (Statistics Canada, 2013), Australia (Australian Bureau of Statistics, 2017; Eigenraam et al., 2013, 2016; Keith et al., 2017), the Netherlands (Remme et al., 2018), and Peru (Conservation International, 2016a; 2016b). The World Bank's Wealth Accounting and Valuation of Ecosystem Services (WAVES) program has also published ecosystem accounts for the Philippines (Losada et al., 2017; WAVES, 2016a, 2016b, 2017). Substantial efforts are underway to construct ecosystem accounts for member nations of the European Union through the European Commission (European Commission, n.d. and European papers, this issue). In addition, the U.N. Statistics Division is leading work in Brazil, China, India, Mexico, and South Africa (System of Environmental Economic Accounting, n.d.). The United Kingdom began work on natural capital accounts in 2011 and has developed accounts for many ecosystems and sectors, including valuation of some ecosystem services (Bright et al., 2019).

Ecosystem accounts enable natural resource trends and tradeoffs to be identified more clearly by aggregating and presenting data consistently. Since they track ecosystem service trends retrospectively, ecosystem accounts can highlight otherwise hidden environmental degradation, natural resource depletion, or unsustainable use patterns. Their retrospective trends also provide a baseline for considering alternative future policy actions and understanding their impacts on specific economic units (i.e., households, individual industries or groups of industries, or government). Despite this potential, political and institutional obstacles hinder the use of information from natural capital accounts more broadly in policymaking (Ruijs et al., 2019), as do technical obstacles in terms of the data and expertise required to compile accounts and evolving best practices for the SEEA EEA (U.N. et al., 2014b; U.N., 2017, 2018). While natural capital accounts are typically seen as credible and trustworthy, it often takes time and experience to gain this political acceptance, which is aided by its understanding and support from high-level government agencies. Natural capital accounts also gain from cooperation, data sharing, and trust among agencies housing the data and with expertise to produce them, which often comes from interagency working groups similar to the U.S. group preparing this and other accounts (Boyd et al., 2018; Ruijs et al., 2019). A 2014 international survey of the use of natural capital accounting for policymaking found a few examples, including setting policy targets for acid rain and eutrophication in the Netherlands (Virto et al., 2018). Forest accounts compiled for Guatemala drew attention to the extent of forest loss, due in large part to unregulated extraction of timber and other forest products, and found that the total contribution of Guatemala's forests to its economy is much higher than recorded in the standard national accounts (FAO, 2017; WAVES, 2014). Guatemala's forest accounts spurred policy initiatives providing incentives for forest protection and restoration targeted toward conservation, stabilization of the fuelwood and timber supply, and job creation. They were also used in Guatemala's 2014 national development plan (Guatemala Consejo Nacional de Desarrollo Urbano y Rural, 2014; WAVES, 2014), which aims at more data-driven decision making (Castaneda et al., 2017).

Regular monitoring of ecosystem services is already having impacts on local and regional policy in the U.S., although it is not always referred to as ecosystem accounting. For example, urban foresters and the mayor's office for Tampa, Florida, have been inventorying their urban forest with data stretching back to 1975, using methods similar to ours for air pollution removal (Campbell and Landry, 1999). This has demonstrated the value of the city's natural capital to politicians and the public. The inventory of ecosystem services provided by urban forests was repeated in 2006 (Andreu et al., 2008), leading to adoption of the City of Tampa tree ordinance (Ord. No. 2006-74, § 9, 3-23-06), which requires a re-inventory of Tampa's urban forest every five years. The monitoring program aims to develop a science-based, publicly supported, fiscally responsible Urban Forest Management Plan based on shared vision and goals (Mayor's Steering Committee on Urban Forest Sustainability, 2009). The first 5-year re-inventory was completed in 2011 (Landry et al., 2013) and was followed by enactment of an Urban Forest Management Plan (Northrop et al., 2013) setting policy and criteria for monitoring the success of management alternatives. Those criteria were applied in the 2016 report (Landry et al., 2018) and will continue to be tracked in 2021 and subsequent years. While not organized using the SEEA EEA structure, their tables similarly categorize the condition of the urban forest and ecosystem services provided to residents of Tampa and surrounding areas, using physical and monetary measures. The regular repetition and growing policy relevance of this program show how ecosystem accounts can similarly be both practical

---

[1] Throughout this paper, "ecosystem accounts" refers to those developed using the SEEA EEA framework while "natural capital accounts" refer to SEEA CF accounts or combined applications of SEEA CF and SEEA EEA.

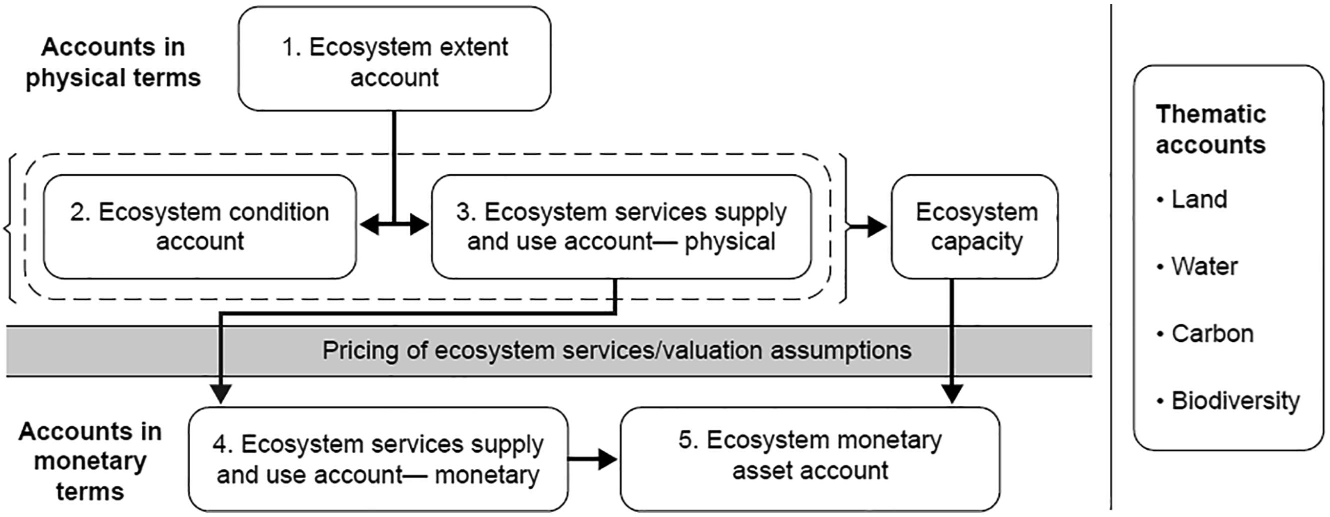


Fig. 1. Connections between ecosystem and related accounts (adapted from U.N., 2017).

to implement and influential to decision makers. While Tampa's example is an impressive one of a bottom-up effort, it is limited to one large city in the U.S. Southeast. Pairing large-scale (national or subnational) ecosystem accounts with citizens and decision makers could provide data for improved decision making in locations lacking a history of urban forest accounting like Tampa's.

1.2. SEEA EEA account structure

The SEEA EEA was originally released in 2014, with supplemental technical recommendations published in 2017, and additional revisions currently underway (Fig. 1; U.N. et al., 2014b; U.N., 2017, 2018). The SEEA EEA framework links physical and monetary accounts, which, in principle, allow ecosystem accounts to be integrated with System of National Accounts and SEEA Central Framework accounts. The functional relationships roughly follow a stocks-and-flows design, where natural capital includes stocks and ecosystem services are flows.

Five primary ecosystem accounts are delineated in the SEEA EEA framework. The first is an area-based account of ecosystem extent (Fig. 1, box 1). As productivity within any area is contingent on the integrity of ecological characteristics and processes, the second account is an ecosystem condition account (box 2). Combining information from the extent and condition accounts informs the physical ecosystem services supply and use accounts (box 3). These first three accounts quantify biophysical characteristics and matter, energy, or information flows, while the fourth and fifth accounts focus on monetary value. With appropriate valuation methods, physical ecosystem services supply and use data can be used to generate monetary accounts estimating the value of each ecosystem service flow for the accounting period (typically annually; box 4). Finally, a monetary ecosystem asset account (box 5) can be created by estimating the net present value of all accounted services provided by each ecosystem. The monetary ecosystem asset account also relies on information from the extent account and corrections related to each ecosystem's capacity, which addresses its ability to provide services in the future, addressing ecosystem conversion, recovery, degradation, or enhancement (Hein et al., 2016).

Ecosystem account developers and users have recognized that it is useful to track a number of crosscutting elements in thematic accounts that complement the core set of ecosystem accounts. Thematic accounts have been proposed for four elements: land and water accounts (from the SEEA Central Framework), plus biodiversity and carbon accounts that have particular relevance for climate and conservation policy (U.N., 2017). Some of these accounts, like biodiversity, may underpin other ecosystem services but are not used directly by households, industries, or governments as final ecosystem services (see Section 2.5).

1.3. Objectives

This paper sets the stage for ecosystem accounting in the U.S. by evaluating the SEEA EEA account structure and core terminology in the context of U.S. data availability and NESCS. The recent SEEA EEA Technical Recommendations note that "it will be necessary to consider the different merits and roles that might be played by the different classifications," (U.N., 2017, ¶ 5.68), which we contribute to by crosswalking SEEA EEA and NESCS in a subnational ecosystem account to develop more theoretically robust accounts. NESCS (U.S. EPA, 2015) identifies and classifies final ecosystem services according to the environmental 'supplier' and human 'user' of the service, enabling the more consistent identification of where certain metrics best fit within the ecosystem accounting structure—particularly in ecosystem condition versus supply and use tables.

We assess the suitability of U.S. data and models for ecosystem accounting, and develop general strategies and specific approaches for populating the accounts using selected data and models. We chose metrics for the pilot accounts for a 10-state region of the Southeast U.S. to populate different parts of the SEEA EEA framework representing diverse interest areas (water purification, air quality regulation, biodiversity, carbon storage, recreation, and agriculture). Our assessment is a scoping and exploratory effort, and is not meant to be comprehensive. Finally, we explore spatiotemporal trends that can be gleaned from our pilot ecosystem accounts to understand their potential use in decision making. Our ecosystem accounting metrics record a first set of values that, like all environmental-economic accounts, should eventually become a recorded time series. These metrics could also be expanded geographically to a national scale (Heris et al., this issue).

We did not attempt to estimate monetary value of either ecosystem service flows (account 4) or ecosystem assets (account 5) because we lacked the necessary data to develop comprehensive monetary accounts for the selected services. U.S. land accounts provide an initial compilation of land cover and use that future work could further adapt into ecosystem extent accounts (account 1; Wentland et al., this issue); we summarize land-cover changes in the results and supplementary materials to aid in the interpretation of condition and supply and use accounts. The pilot ecosystem accounts for the Southeast thus focus on ecosystem condition and physical supply and use for a selected list of ecosystem services that are of interest for natural resource management decisions across the region and for which data were available.

2. Methods

2.1. Working group process

A working group comprising federal government agencies, academic and non-governmental organizations, and the private sector engaged in a series of five in-person meetings from 2016 to 2019 to develop initial natural capital accounts for the U.S. In an iterative process, the group explored the SEEA EEA framework, relevant U.S. data sources, and how these data would best fit into the SEEA EEA, according to the data considerations described in Section 2.3. We used NESCS to select metrics appropriate for populating ecosystem condition and supply and use accounts, and the underlying data and models needed to quantify them, as described in Section 2.5. We selected a set of ecosystem services and condition metrics to include in the pilot accounts to test that process and examine the results. We made choices of which ES and metrics to include primarily based on data availability (see Section 2.3) and not necessarily on management importance; they show a broad range of ES and metrics suitable for inclusion in ecosystem condition and supply and use tables. These pilot accounts contain only a subset of potential ecosystem services and metrics that could be included in U.S. ecosystem accounts; others could feasibly be included in the future and extended to cover larger parts of the U.S. (e.g., Heris et al., this issue).

2.2. Study area

Pilot ecosystem accounts were developed for 10 states in the Southeast U.S. (Fig. 2). These states cover 1.37 million square kilometers, or approximately 17.8% of the land area of the conterminous U.S. (i.e., excluding Alaska and Hawaii). This region accounted for 19.2% of national GDP and 22.7% of U.S. population in 2010, and had 13.3% population growth from 2000-2010, as compared to 9.7% for the U.S. as a whole (BEA, 2019; Mackun and Wilson, 2011). In terms of land cover change, the Southeast includes eight of the nine states with the greatest rates of land cover change from 2001 to 2011; its overall rate of land cover change over this decade was substantially greater than the average for the conterminous U.S. (8% vs. 2.96%) (Homer et al., 2015). This rapid change makes the region an interesting subnational case study for a SEEA EEA pilot, since because of it we would expect to see faster change in ecosystem services here than elsewhere.

2.3. Data considerations

We used three criteria to select data and modeling approaches for this study:

- Data and methods must be publicly accessible;
- Data must be available at a broad spatial scale (ideally the entire U.S., but at minimum covering the 10-state region); and
- Data must be available for multiple years so that a time series can be constructed, and likely to be collected and available into the future so that the accounts can be updated. Most metrics in the pilot accounts were calculated for 2001, 2006, and 2011, corresponding with the National Land Cover Database (NLCD, Homer et al., 2015), since land cover is an input for most metrics.

2.4. Measurement and aggregation

The SEEA EEA uses three levels of spatial organization: ecosystem accounting areas, ecosystem assets, and basic spatial units.[2]

The ecosystem accounting areas in this study were the 10-state study region (Fig. 2) plus 10 individual state-level accounts. These were produced to facilitate cross-state comparisons and provide state-level information relevant for ecosystem management (see Section 3.2.2).

Ecosystem assets are contiguous areas covered by a certain type of ecosystem, while ecosystem types include all assets of the same kind of ecosystem (e.g., deciduous forest or woody wetland); all locations have a single associated ecosystem type. Ecosystem assets and types ideally account for ecological characteristics like “vegetation structure and type, species composition, ecological processes, climate, hydrology, soil characteristics, and topography” (U.N., 2017, ¶ 3.16). However, since ecosystem type coverage can change over time, data used to delineate ecosystem types must exist for multiple time periods and be updatable. With no dataset available that met these criteria to allow for the delineation of actual ecosystem types, we used the NLCD as a proxy for ecosystem type in our accounts. Several other published ecosystem accounts have used a similar approach (Bright et al., 2019; Remme et al., 2014; Statistics Canada, 2013; WAVES, 2016b). We recognize that land cover alone does not fully differentiate between ecosystem types and do not recommend that it be used as a standard for ecosystem accounting in the U.S. Our choice is a placeholder until better options become available, possibly through spatial areas classification work associated with the SEEA EEA revision (U.N., 2018).

Basic spatial units are the smallest spatial area to which ecosystem accounting data can be attributed; each has attributes including an ecosystem type and ecosystem accounting area. Like most ecosystem accounts, we used a raster-based approach for delineating basic spatial units to accommodate information from spatial datasets with different resolutions while avoiding information loss from the use of a single reference grid.[3] This approach facilitates re-aggregation of the results for each metric on a finer level than the current region and state-level accounting tables show (e.g., by county or watershed). The spatial resolution of each metric included in the accounting tables is determined by its input data; most of our metrics are at 30-meter resolution (see detailed methods in supplementary materials).

We incorporate non-raster based data for recreational birding and carbon storage in the pilot accounts. For the recreational birding analysis, we totaled birding observation points by state and ecosystem type to match the accounting tables’ aggregation levels. The carbon storage analysis used a look-up table of carbon storage per unit area by a combination of ecosystem type, ecoregion, and land protection status. Such integration is commonly required in the development of ecosystem accounts. The use of basic spatial units that enable aggregation to consistent ecosystem types and ecosystem accounting areas is important for coherence across different metrics and accounts within a given application of the SEEA EEA.

2.5. Using SEEA EEA and NESCS to place metrics in the ecosystem accounting structure

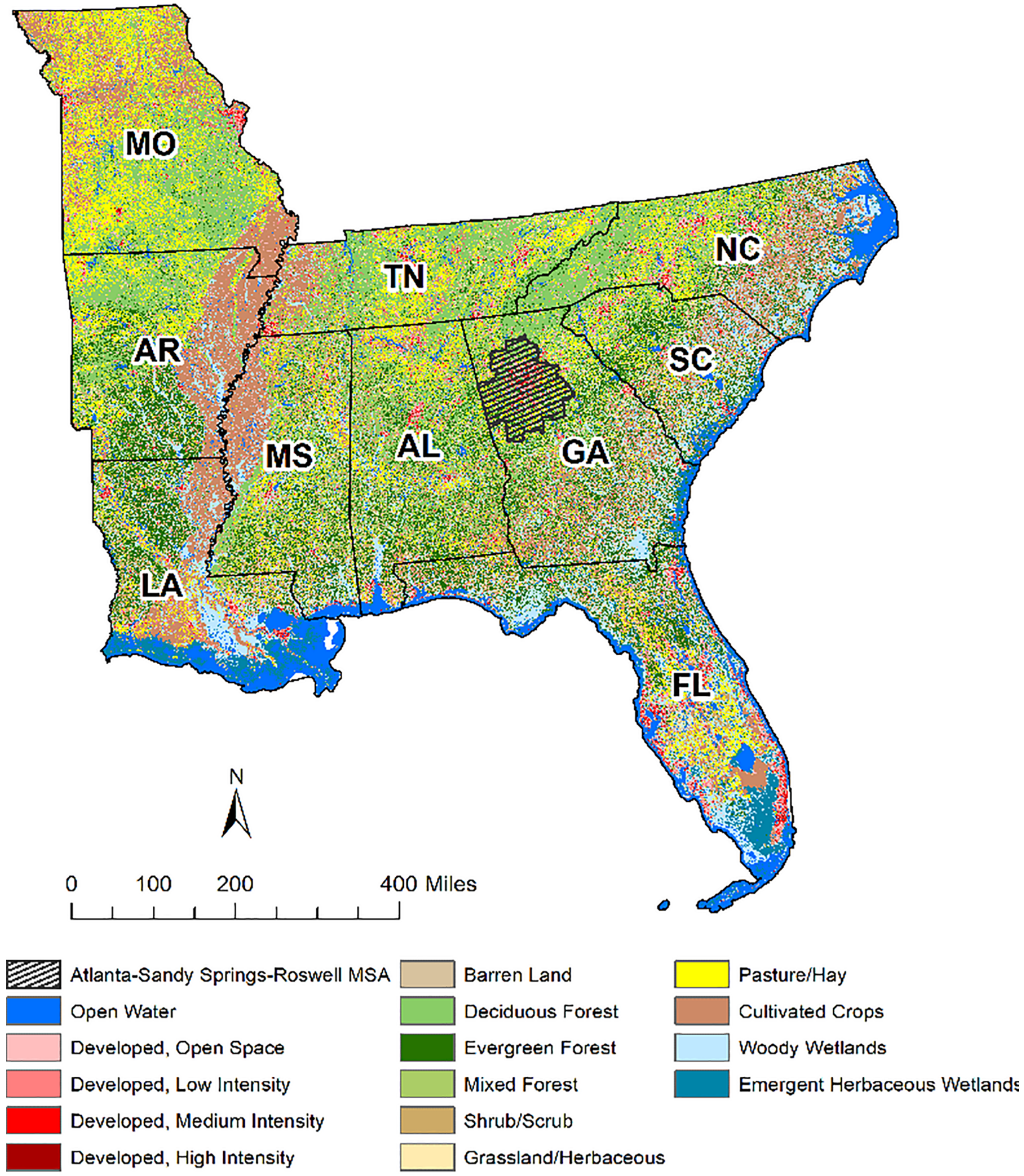
The information included in each ecosystem account is determined by SEEA EEA account definitions and the above-mentioned data considerations. We used the NESCS (U.S. EPA, 2015) to help distinguish between ecosystem services eligible for inclusion in supply and use accounts and information better suited to condition accounts, acknowledging that the SEEA EEA recognizes ecosystem services supply and use to be a function ecosystem condition and extent (Fig. 1). NESCS provides strict guidelines for what is considered a service, as opposed to the state or condition of the ecosystem that generates it.

Supply and use accounts represent transactions. Each number in a given cell quantifies an exchange between an ecosystem asset (supplier)

---

[2] The terminology for spatial units varies from the SEEA EEA to the SEEA EEA Technical Recommendations (TR). We are using the terminology from the SEEA EEA TR (U.N., 2017).

[3] The SEEA EEA suggests that raster grid cells or small polygons can function as BSUs; functionally the two would of course be similar the more closely small polygons resembled a fine-resolution, square grid (U.N., 2017, ¶ 3.14).

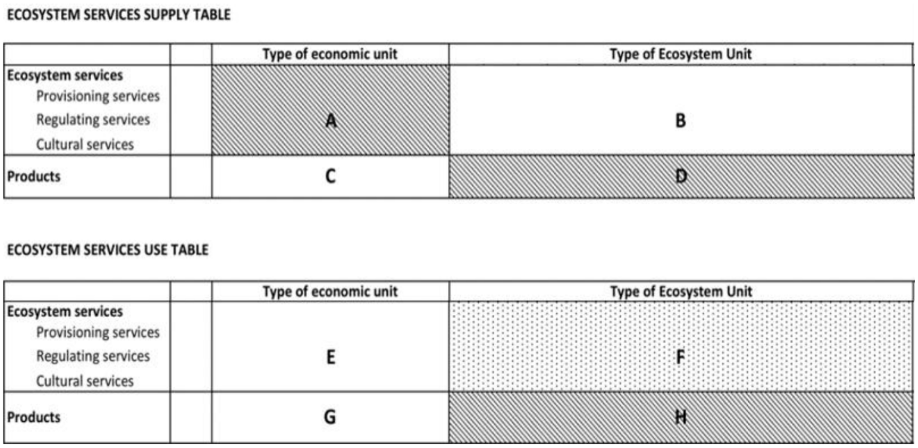


Source: National Land Cover Dataset, 2011

Fig. 2. Land cover (2011) for states included in the Southeast U.S. pilot ecosystem accounts and location of the Atlanta-Sandy Springs-Roswell Metropolitan Statistical Area. Note: State abbreviations: Alabama (AL), Arkansas (AR), Florida (FL), Georgia (GA), Louisiana (LA), Mississippi (MS), Missouri (MO), North Carolina (NC), South Carolina (SC), and Tennessee (TN).

and an individual or economic entity (user) (Fig. 3a, final ecosystem services represented in boxes B and E). Intermediate ecosystem services are recorded in box F as services supplied from one ecosystem unit to another, rather than to an economic unit (U.N., 2017, ¶ 5.11). Therefore, in physical terms, the total amount of ecological end-products (biophysical elements created by ecosystem processes) supplied by ecosystem assets must equal the total amount of ecological end-products used by economic units (i.e., households, industries, government, plus imports/exports); this transaction constitutes the final ecosystem service received by the user. In other words, summed values for each ecosystem service in boxes B and E must be equal. This transaction, recorded in the supply and use tables, is ideally a physical quantity of used ecological end-product, although proxies can be used for ecosystem services where such quantification is complex or direct measurement is infeasible. Transactions occur at a location that may involve movement of both people (e.g., recreationists to a park) and/or matter, energy, or information provided or mitigated by ecosystems (e.g., movement and mitigation of floodwater). In the latter case, NESCS does not require the ecological end-product and user to be physically co-located (particularly for regulating services that frequently are underpinned by flows of matter, energy, or information, Bagstad et al., 2013). Currently, non-use values are typically excluded from SEEA EEA accounts as their measurement and particularly their valuation are challenging to reconcile with the SEEA and System of

(a) SEEA EEA ecosystem services supply and use table structure (adapted from December 2015 draft of U.N., 2017)



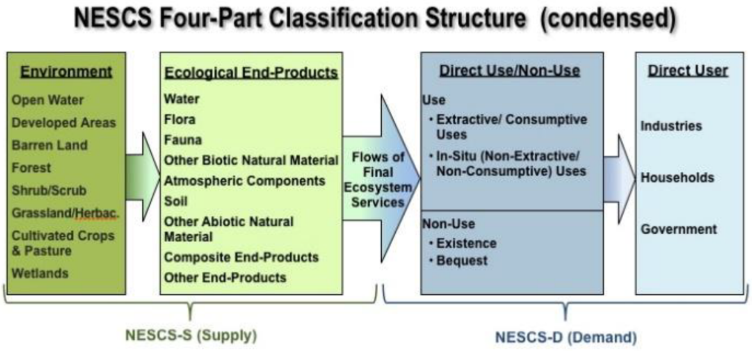
ECOSYSTEM SERVICES SUPPLY TABLE

		Type of economic unit	Type of Ecosystem Unit
Ecosystem services Provisioning services Regulating services Cultural services		A	B
Products		C	D

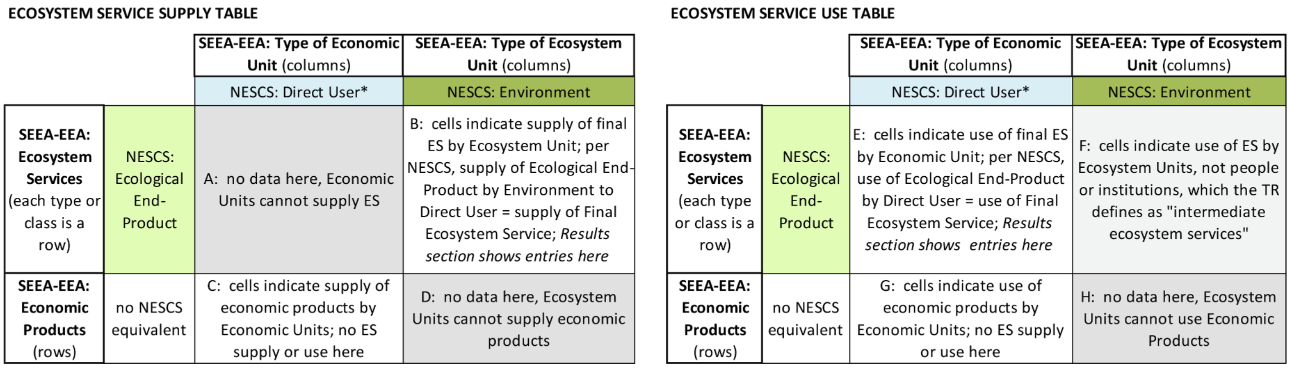
ECOSYSTEM SERVICES USE TABLE

		Type of economic unit	Type of Ecosystem Unit
Ecosystem services Provisioning services Regulating services Cultural services		E	F
Products		G	H

(b) National Ecosystem Services Classification System structure (adapted from U.S. EPA, 2015)



(c) NESCS structure (b) superimposed on SEEA EEA ecosystem services supply and use table structure (a) to show alignment



ECOSYSTEM SERVICE SUPPLY TABLE

		SEEA-EEA: Type of Economic Unit (columns)	**SEEA-EEA: Type of Ecosystem Unit** (columns)
		NESCS: Direct User*	NESCS: Environment
SEEA-EEA: Ecosystem Services (each type or class is a row)	NESCS: Ecological End-Product	A: no data here, Economic Units cannot supply ES	B: cells indicate supply of final ES by Ecosystem Unit; per NESCS, supply of Ecological End-Product by Environment to Direct User = supply of Final Ecosystem Service; *Results section shows entries here*
SEEA-EEA: Economic Products (rows)	no NESCS equivalent	C: cells indicate supply of economic products by Economic Units; no ES supply or use here	D: no data here, Ecosystem Units cannot supply economic products

ECOSYSTEM SERVICE USE TABLE

		SEEA-EEA: Type of Economic Unit (columns)	**SEEA-EEA: Type of Ecosystem Unit** (columns)
		NESCS: Direct User*	NESCS: Environment
SEEA-EEA: Ecosystem Services (each type or class is a row)	NESCS: Ecological End-Product	E: cells indicate use of final ES by Economic Unit; per NESCS, use of Ecological End-Product by Direct User = use of Final Ecosystem Service; *Results section shows entries here*	F: cells indicate use of ES by Ecosystem Units, not people or institutions, which the TR defines as "intermediate ecosystem services"
SEEA-EEA: Economic Products (rows)	no NESCS equivalent	G: cells indicate use of economic products by Economic Units; no ES supply or use here	H: no data here, Ecosystem Units cannot use Economic Products

Fig. 3. Alignment of SEEA EEA ecosystem services supply and use tables with the National Ecosystem Services Classification System. (a) SEEA EEA ecosystem services supply and use table structure (adapted from December 2015 draft of U.N., 2017). (b) National Ecosystem Services Classification System structure (adapted from U.S. EPA, 2015). (c) NESCS structure (b) superimposed on SEEA EEA ecosystem services supply and use table structure (a) to show alignment. *Direct Users could be associated with Direct Uses by nesting or dividing columns under Direct User (Economic Unit), so an Industry, Household, or Government might have Extractive or in situ uses of an Ecological End-Product. Under Direct Use, the Non-Use paths are not relevant in SEEA EEA terms, because there is no direct transaction between an Environment and an Economic Unit.

National Accounts' valuation framework (U.N., 2017, ¶ 6.48).

NESCS defines ecosystem service flows by bringing together an ecosystem-specific location, ecological end-products, and use of these elements by specific users (Fig. 3b). These four parts (environmental location, end product, use, and user) correspond to rows and columns in the SEEA EEA supply and use tables to represent final ecosystem services (Fig. 3c). The intersection of two NESCS elements defines each box to be populated within the supply and use table. For instance, the NESCS environment and ecological end-product elements define box B of the supply table, while the NESCS ecological end-product and direct user elements define box E of the use table. NESCS' approach to describing ecosystem services contrasts with the Common International Classification of Ecosystem Services (CICES, Haines-Young and Potschin, 2018), which offers a hierarchically defined list of ecosystem services (i.e., rows in the supply and use tables). As we demonstrate in this paper, NESCS thus offers a new view on ecosystem service definition and metric placement within the SEEA EEA, which we believe improves consistency in ecosystem accounting. Once metrics are systematically defined by their various NESCS elements and placed in the supply and use tables (Fig. 3c), the relevant ecosystem service flows can be easily labeled using CICES terminology, if desired (maintaining added NESCS-provided detail for Users/Beneficiaries.

By viewing the SEEA EEA framework through the NESCS lens, we separated ecosystem characteristics and processes (appropriate for inclusion in SEEA EEA condition tables) that some have perhaps erroneously called "ecosystem services" (like habitat and biodiversity) from final ecosystem services allowable in supply and use accounts (with finality proven by direct use; Boyd and Banzhaf, 2007; Landers and Nahlik, 2013; Landers, 2015; U.N., 2017; U.S. EPA, 2015). NESCS and accounting rules also help differentiate products jointly produced by economic and ecological inputs (e.g., crops, livestock), and proxy measures like visitor count from the actual final ES being transacted. Any aspect of ecosystem condition that is relevant to the production of ecological end-products used by people, informative about the ecosystem's persistence into the future, or otherwise of interest to managers and decision makers may be a good candidate for inclusion in the condition table. We identified two types of information that would be useful to track in the condition account: (1) metrics related to the ecosystem's ability to generate ecological end-products ("condition for ecosystem services") and (2) metrics providing information about the likelihood that the ecosystem will continue to exist into the future in a sufficient condition for its intrinsic survival ("condition for maintenance," e.g., Mace, 2019). Our examples for the U.S. Southeast were all of the first type of condition account metric, which was not an intentional, up-front choice but the product of our evaluation of candidate metrics for inclusion in the supply and use tables and their assignment to supply and use or condition accounts (see Section 2.6.1). This method exposed, in a way not previously evident, that it helps to identify as ecosystem condition metrics a handful of characteristics and processes related to ecosystem service flows but that are not the thing that is directly used and *valued in a final ecosystem service*. These are thus not part of final ecosystem services, but are important enough to measure and report as metrics in the condition account, which is more flexibly defined in the SEEA EEA than the supply and use accounts.

2.5.1. *Identification of ecological end-products for regulating services*

Regulating services are challenging to conceptualize and quantify, as compared to provisioning or cultural ecosystem services (Sutherland et al., 2018). Ecological end-products (Fig. 3b; the thing in the ecosystem that is being used) are fairly easy to identify for provisioning services, since the end-product is always a physical object with which humans directly interact or experience, either by extracting or using a stock of things in situ. It is also possible to identify a physical thing that humans "use" as they experience nature and derive benefits from cultural services, such as scenic viewsheds or the cultural value of foods harvested by indigenous groups (independent from the provisioning service, i.e., nutrition that is concurrently provided). The quantification of ecological end-products for provisioning and cultural services typically entails assessment of the portion of a stock of things used and thus directly valued in a particular time period.

It is more difficult to identify the ecological end-products for regulating ecosystem services that reduce the amounts of physical things experienced directly with our senses, such as floodwaters or high temperatures in urban heat islands. For example, while a homeowner may experience floodwater in their house, it is not beneficial in any way and so cannot, by definition, be an ecological end-product because it is not used to derive a benefit. The end-product used and valued by the homeowner in this case is a *complex upstream ecosystem (e.g., soil, plants, topography)* that together retains water that would otherwise flood the house. The associated reduction in floodwater depth is the benefit of the ecosystem service, not the ecosystem providing the benefit. Similarly, vegetation in cities can remove or filter air pollutants. While the remaining pollutants are carried by wind and inhaled by people (directly experienced), pollutant concentrations are not the ecological end-product since they do not provide any benefit when used. Vegetation is the ecological end-product in this case and "produces" fewer pollutants to breathe in. For regulating services, the ecological end-product is doing something for the user when it is "used" so should be quantified as a rate rather than a stock. In both of these cases, movement of matter or energy means that users and ecological end-products are not physically co-located but are connected by flows that can be quantified using models (Bagstad et al., 2013).

2.6. *Quantifying metrics for the pilot ecosystem accounts*

Based on the above considerations for data, models, and placement of metrics, we defined a set of five final ecosystem services for inclusion in our pilot accounts, as well as their associated metrics in supply and use or condition accounts (Table 1). Full methods are described as supplementary materials.

2.6.1. *Ecosystem extent account*

We include land-cover change data as an initial ecosystem extent account for the years 2001, 2006, and 2011, with state-level results included in the supplementary materials. This information can aid in the interpretation of both ecosystem condition and supply and use tables, and is presented and described more comprehensively by Wentland et al. (this issue).

2.6.2. *Ecosystem condition account*

We chose metrics for five ecosystem processes and characteristics to include in the ecosystem condition accounts: wild pollinator habitat, water purification, air purification, bird species richness, and carbon storage. Wild pollinator habitat and water purification were chosen because they are related to economically important ecosystem services that cannot currently be included in the supply and use account because data about their use are unavailable. We included air pollutant concentrations to demonstrate how a human influenced biophysical factor in the condition table can directly relate to an ecosystem service metric (air quality regulation) in the supply and use table. We selected bird species richness to illustrate the use of a biodiversity metric and because it is related to a metric in the supply and use table (recreational birding days), since recreational birdwatchers place a value on viewing diverse species (Kolstoe and Cameron, 2017; Loomis et al., 2018).

Wild pollinator habitat contributes to the ecosystem service "wild pollination," which increases yields when habitat is located near pollinator-dependent crops. Visits by wild insect pollinators have been shown to increase fruit set in crops even when managed honeybee pollinator visitation is high (Garibaldi et al., 2013). At regional to national scales, there are insufficient data and models to estimate the amount of wild pollination occurring. Instead, we mapped potential pollinator habitat and pollinator-dependent crops and calculated

Table 1

Final ecosystem services represented in the pilot ecosystem accounts by proxies for the final ecosystem service (in the supply and use account) or by metrics related to the ecosystem's ability to generate the ecological end-products used in the service (in the condition account).

Final ecosystem service	Ecological end-product (NESCS element b)	User(s) (NESCS element d)	Benefit to user	Related metrics in pilot accounts (account type)
Ecological structures and fauna that are valued and used by recreational birders	Composite product including bird biodiversity, bird population, and other components that attract birders	Households	Value to birder of interaction	Recreational birding days (supply and use account) Bird species richness (condition account)
Reduction of air pollutants	Ecological structures (i.e., vegetation) responsible for removing air pollutants	Households	Reduced risk of adverse health outcomes	Removal of target air pollutants (supply and use account) Concentrations of target air pollutants, weather conditions, and percent canopy cover (condition account)
Reduction of water pollutants	Ecological structures (i.e., vegetation, soil structure and biota) responsible for removing water pollutants	Industries, households, government	Reduced risk of adverse health outcomes, industrial production that uses water (e.g., paper, food processing, etc.)	Area of purifying land cover types between nonpoint-source pollutant sources and waterways (condition account) % of flowpath between nonpoint-source pollution sources and waterways that is in purifying land cover types (condition account)
Wild pollination	Wild pollinators	Farmer, gardener, land manager	Increased crop yield attributable to wild pollination	Area of wild pollinator habitat near pollinator-dependent crops (condition account) Area of wild pollinator-dependent crops near pollinator habitat (condition account) Ratio of wild pollinator habitat to pollinator-dependent crops (condition account)
Carbon storage is not considered an ecosystem service in the NESCS framework, but is included in the pilot accounts because it is commonly included in other early ecosystem accounting efforts.				Carbon storage (condition account)

several metrics related to their proximity and relative area (Table 2; Olander et al., 2017b). Together, these metrics provide information on different aspects of ecosystems' ability to provide relevant wild pollinator populations, i.e., where there are crops that benefit from pollination.

Water purification supports a variety of ecosystem services (e.g., water of sufficient quality for recreation, drinking, commercially harvested aquatic organisms, and agricultural and industrial uses). We mapped potential water pollution nonpoint sources (urban and agricultural land) and ecosystems that may purify overland flow as it moves toward waterways (e.g., wetlands, forests, and grasslands; Baker et al., 2006; Olander et al., 2017b); neither nonpoint sources nor ecosystems were weighted for the quantity of pollutant or "purification potential." This relatively simple analysis assumes that water flows follow surface topography and does not include managed stormwater drainage systems, subsurface flow, or resuspension of pollutants. From these maps, we developed several metrics related to ecosystems' ability to purify water before it reaches streams; these metrics might explain observed changes or indicate potential future changes in these services (Table 2).

Air purification occurs when air pollutants settle onto plant surfaces or are absorbed by plants' leaves and are directly removed from the air or washed away during rainfall events. Left in the air, these pollutants can affect human cardiovascular, pulmonary, and neurological health. Air pollutant removal is a function of atmospheric pollutant concentrations and net deposition rates, which depend on wind, temperature, precipitation events, and the amount and type of vegetation present in an area. Differences in growing season can also affect removal rates due to changes in the leaf area of deciduous versus evergreen species (Nowak et al., 2014). Nowak's model:

1) Estimates pollution removal by trees (Flux or removal = deposition velocity × concentration);
2) Converts that removal to change in pollution concentration based on boundary layer heights; and
3) Uses change in concentration and human population demographics to estimate health impacts and values due to pollution removal by trees.

For the condition table, we included pollution concentrations since pollutant removal rates by vegetation depend on ambient air pollution levels. The vegetation used by humans for air pollutant removal is the end-product supplied by the ecosystem. See supplementary materials for additional discussion of the placement of air pollution removal metrics. We modeled hourly removal of six pollutants related to human health (CO, NO_2, O_3, PM_{10}, $PM_{2.5}$, and SO_2) using atmospheric concentrations and weather data for 2010 and 2015. Due to a lack of updated data, we were limited to using 2011 NLCD tree canopy data for both years. Thus, in this pilot account, changes in removal rates between 2010 and 2015 result from weather and air pollutant concentration differences between the two years, not vegetation changes (which would be more representative of the ecosystem changes underlying the supply and use of this service).

Bird species richness influences the suitability of a given location for recreational birdwatching and can provide information relevant to this service and potentially others (e.g., pest control). We estimated bird species richness with a generalized joint attribute model that models species in a community instead of individually, allowing species to be influenced by other species as well as the environment (Clark et al., 2017). Species-level bird counts from the Breeding Bird Survey, a survey conducted annually in North America since 1966 (Pardieck et al., 2018), were used in the model. Bird counts were modeled based on 18 predictor variables including climate, soil, topography, and land cover (see Table S4 for details), on a 0.5-degree grid. This corresponds with the spatial resolution of Breeding Bird Survey routes, which we summarize by state and ecosystem type for inclusion in the condition account.

Table 2
Condition metrics included in the pilot ecosystem condition account.

Ecosystem process or characteristic	Metric	Description
Wild pollinator habitat	Pollinator habitat near dependent crops	Area of pollinator habitat within pollinator flight distance (1308 meters) of pollinator-dependent crops
	Dependent crops near pollinator habitat	Area of pollinator-dependent crops within pollinator flight distance (1308 meters) of pollinator habitat
	Ratio of pollinator habitat to dependent crops	‘Pollinator habitat near dependent crops’ metric divided by ‘dependent crops near pollinator habitat’ metric
Water purification	Purifying land cover in flowpath	Area of purifying land cover types in the flowpath between nonpoint-source pollutant sources and waterways
	% of flowpath in purifying land cover	‘Purifying land cover in flowpath’ metric divided by total area of the flowpath between nonpoint-source pollutant sources and waterways, expressed as percentage
Air quality regulation	Weather conditions (temperature, wind speed, precipitation)	Average values used to drive the model; weather conditions affect air pollutant deposition and removal rates
	Tree canopy cover	Percentage of tree canopy cover, which affects air pollution removal rates by vegetation
	Pollutant concentrations: CO, NO_2, O_3, PM_{10}, $PM_{2.5}$, and SO_2	Concentration of air pollutants that can be removed
Bird biodiversity	Bird species richness	Mean number of bird species (out of 160 species modeled) predicted to occur

Finally, we estimated terrestrial carbon storage from data provided by Sleeter et al. (2018a), who used a semi-spatial state-and-transition simulation model to estimate changes in terrestrial carbon storage (above- and belowground biomass, woody debris, and soil) based on four attributes—ecoregion, land use, land protection status, and events such as wildfire and forest harvesting. Data provided by the authors of that study included the total area of land and estimated carbon stored in each unique attribute combination for each year from 1973 through 2010, i.e., accounting for changes to stocks over time due to carbon sequestration, harvest, fire, and other land-cover change. The model addresses the annual agricultural crop harvest cycle with parameters for grain, straw, and litter biomass, the latter of which is an input to the soil while the first two are assumed to eventually be emitted back to the atmosphere. We used spatial overlays of Omernik Level III ecoregions, protected land (PAD-US; USGS, 2016), and land cover classes (NLCD, 2001, 2006, 2011) (Homer et al., 2015) to estimate the total area for each unique attribute combination in each state and focal year. Since Sleeter et al. (2018a) provide a limited accounting for carbon storage in wetlands, we added wetland soil carbon storage estimates from Nahlik and Fennessy (2016) based on NLCD wetland cover and wetland location in three regions of the Southeast–the Coastal Plains, Eastern Mountains, and Interior Plains. Our analysis did not assess carbon storage in aquatic systems.

2.6.3. *Physical supply and use accounts*

We included two metrics in the pilot physical supply and use accounts: recreational birdwatching (measured in birding days) and air quality regulation (air pollutant removal by vegetation). As none of these metrics is directly measured at the spatial and temporal scales represented in the accounts (e.g., regional, state, county), we estimated them based on summation of a variety of input data.

To estimate the number of birding days, we combined birding estimates from the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (NSFHWAR; U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau, 2011) with spatiotemporally explicit birding activity data from eBird, a citizen-science-based repository of bird sightings that includes information on when and where each participant went birding (“eBird Basic Dataset”, n.d.). We obtained estimates of the total number of birding days away from home for each state in the study area from the 2001, 2006, and 2011 NSFHWAR reports. We then totaled the number of eBird observations made in each state during each study year and calculated a conversion factor between NSFHWAR birding days and eBird observations. To estimate birding days taking place within each ecosystem type in each state, we overlaid the appropriate year’s NLCD data on the eBird observation points, totaled the number of eBird observations by ecosystem type, and used the conversion factor to translate those totals into birding days by state and ecosystem type.

We estimated air quality regulation based on previous work by the USDA Forest Service, which quantified the removal of air pollutants by vegetation at the county scale for the continental U.S. (Nowak et al., 2014). The metric used for our supply and use table are the removal rates, leaving the mean concentration of each air pollutant, annual weather, and tree canopy cover metrics for the condition table as driving factors responsible for determining air pollutant removal rates. Monitoring data on hourly pollution concentrations from the U.S. EPA’s Air Quality System national database (U.S. EPA, 2013) served as one input into Nowak et al.’s (2014) modeling results and are summarized here for the Southeast for 2010 and 2015. While the majority of human interaction with air pollutants, and thus the use of air pollution reduction, happens in developed areas where people are concentrated, we had to distribute this metric across all land cover classes by county in the supply side of the table. This is necessary since our data are spatially aggregated at the county scale and not associated with specific land-cover types such as developed land. Since the model considers benefits that accrue to users within each county who are connected to pollutant-removing ecosystems by air currents (particularly during the valuation step that we did not conduct in this paper), its estimates do not overestimate the quantity of this service used by people (Nowak et al., 2014). In the use table, air pollutant concentrations are assigned to the household user.

3. Results

3.1. *Pilot ecosystem accounts*

3.1.1. *Extent account*

The ecosystem extent account (Table 3) shows changes in 15 land cover types over the 10-year period that corresponds to most of the metrics in our ecosystem condition and supply and use tables. The most notable changes during this time are gains in herbaceous, scrub/shrub, developed land cover, and cultivated crops, and declines in pasture/hay and deciduous forests (largely reflecting continuing regional trends dating to the 1970s (Sleeter et al., 2018b)). Since ecosystem change underlies the provision of ES, these changes are important to interpret the trends identified in the ecosystem condition and supply and use accounts.

3.1.2. *Condition account*

The condition account (Table 4) includes a variety of condition metrics (rows), grouped by the ecosystem characteristic or ecological process to which they relate (far left); columns indicate the ecosystem

Table 3
Ecosystem extent table for the Southeast U.S. for the years 2001, 2006, and 2011 (km^2).

		Ecosystem Types (Land Cover)														
		Open Water	Developed - Open	Developed - Low	Developed - Medium	Developed - High	Barren	Deciduous Forest	Evergreen Forest	Mixed Forest	Shrub/Scrub	Herbaceous	Pasture/Hay	Cultivated Crops	Woody Wetlands	Emergent Herbaceous Wetlands
Area (sq km)	2001	31,227.7	45,306.1	21,326.0	6,284.0	2,332.8	2,306.9	164,751.8	141,820.3	64,591.5	25,228.0	22,317.5	142,087.8	120,350.0	122,409.9	26,445.0
	2006	31,433.6	46,403.9	22,241.8	7,256.9	2,672.9	2,213.2	159,661.3	138,959.3	64,623.2	29,205.4	28,959.4	135,479.4	121,008.6	120,850.0	27,816.2
	2011	31,399.4	46,868.5	22,736.6	7,879.7	2,943.3	2,294.8	157,360.8	142,371.0	64,419.8	29,871.3	28,516.8	131,512.1	122,097.0	121,670.8	26,843.2
% change, 2001 to 2011		0.5%	3.4%	6.6%	25.4%	26.2%	-0.5%	-4.5%	0.4%	-0.3%	18.4%	27.8%	-7.4%	1.5%	-0.6%	1.5%

Table 4
Condition table for the Southeast U.S., selected years between 2001 and 2015.

				Ecosystem Types (Land Cover)																	
				Offshore	Open Water - non-freshwater	Open Water - freshwater	Developed - Open	Developed - Low	Developed - Medium	Developed - High	Barren	Deciduous Forest	Evergreen Forest	Mixed Forest	Shrub/Scrub	Grassland/Herbaceous	Pasture/Hay	Cultivated Crops	Woody Wetlands	Emergent Herbaceous Wetlands	**TOTAL**
Wild pollination*	Area of pollinator habitat in flight range of pollinator-dependent crops (sq km)		2001									5,471	2,516	1,336	1,290	165			7,061	172	18,011
			2006									4,152	2,125	1,459	2,191	423			11,539	371	22,259
			2011									53,679	30,441	6,670	18,388	9,314			43,104	3,354	164,951
	Area of pollinator-dependent crops in flight range of pollinator habitat (sq km)		2001															11,182			11,182
			2006															21,581			21,581
			2011															65,818			65,818
	Ratio of pollinator habitat to pollinator dependent crops		2001															1.66			
			2006															1.05			
			2011															2.55			
Water purification	Area of purifying land cover types between NPS sources and waterways (sq km)		2001									31,542	20,238	6,959		5,385			25,463	3,379	92,966
			2006									31,453	19,780	6,678		5,997			25,427	3,504	92,840
			2011									31,005	19,330	6,353		6,192			25,151	3,789	91,820
	% of flowpath between NPS sources and waterways in purifying land cover types		2001			30.6%															
			2006			30.4%															
			2011			29.9%															
Bird biodiversity	Mean bird species richness (out of 160 species modeled)		2001	142.1	141.6		145.0	149.0				147.5	148.3				147.5	148.0	145.5	127.9	
			2006	142.5	142.0		146.0	150.0				147.6	148.3		145.0		147.5	148.3	145.8	129.0	
			2011	142.4	142.5		144.5	150.0				147.5	148.3		144.0		148.0	147.6	145.9	129.4	
Air quality regulation	Wind Speed (m/s)		2010	2.42																	
			2015	2.54																	
	Temperature (°C)		2010	17.06																	
			2015	17.38																	
	Precipitation (mm/yr)		2010	962																	
			2015	1344																	
	Canopy cover (%)		2010	58%																	
	Air pollutant concentrations (annual mean, ppb except for PM (µg/m3))	CO	2010	314.6																	
			2015	290.1																	
		NO_2	2010	7.3																	
			2015	7.0																	
		O_3	2010	30.6																	
			2015	27.9																	
		PM_{10}	2010	9.5																	
			2015	9.5																	
		$PM_{2.5}$	2010	10.9																	
			2015	10.4																	
		SO_2	2010	2.0																	
			2015	1.0																	
Carbon	Carbon storage (kilotons of C)		2001	0	0	0	307,170			0	11,039,035			1,211,205	601,250	4,941,118		3,276,189	741,315	22,117,283	44,234,565
			2006	0	0	0	361,629			0	11,074,236			1,282,818	713,026	4,897,046		3,260,913	748,974	22,338,642	44,677,284
			2010	0	0	0	384,934			0	10,935,461			1,464,086	740,303	4,887,533		3,225,475	782,570	22,420,361	44,840,723

*The metrics related to wild pollination are not directly comparable across years due to changes in the geographic extent of the Cropland Data Layer available for each year. The Cropland Data Layer is available nationally starting in 2008; in 2006 it was available for Arkansas, Louisiana, and Mississippi, and in 2001 it was available for Arkansas and Mississippi, enabling the analysis of full time trends for those states (see supplementary materials).

Table 5
Supply table for the Southeast U.S., selected years between 2001 and 2015.

			Ecosystem Types (Land Cover)																
			Offshore	Open Water	Developed - Open	Developed - Low	Developed - Medium	Developed - High	Barren	Deciduous Forest	Evergreen Forest	Mixed Forest	Shrub/Scrub	Grassland/Herbaceous	Pasture/Hay	Cultivated Crops	Woody Wetlands	Emergent Herbaceous Wetlands	**Total**
Recreational birding (thousands of birding days)		2001	2,015	8,471	6,935	5,897	1,850	978	416	6,586	3,441	365	1,075	1,498	2,285	4,614	7,106	3,343	56,874
		2006	518	4,418	8,552	9,451	4,368	1,129	780	6,273	3,433	531	2,208	2,808	2,833	3,658	6,196	2,204	59,360
		2011	1,236	5,207	10,022	7,420	3,553	1,046	1,408	7,173	3,816	692	1,966	1,833	4,050	2,634	4,964	3,695	60,715
Pollution removal (tonnes/year)	CO	2010	98,690																98,690
		2015	92,583																92,583
	NO_2	2010	438,139																438,139
		2015	494,268																494,268
	O_3	2010	4,531,927																4,531,927
		2015	4,258,878																4,258,878
	PM_{10}	2010	1,327,037																1,327,037
		2015	1,205,268																1,205,268
	$PM_{2.5}$	2010	220,218																220,218
		2015	257,912																257,912
	SO_2	2010	329,580																329,580
		2015	176,681																176,681

type associated with the relevant metric. For example, the ‘area of pollinator habitat in flight range of pollinator-dependent crops’ metric is divided among the ecosystem types that provide wild pollinator habitat. The ‘ratio of pollinator habitat to pollinator-dependent crops’ metric is shown in the ‘cultivated crops’ column since it is indicative of wild pollinator activity on cropland. The air purification metric ‘pollution removal’ takes place across all ecosystem types although it is most relevant to human health in developed areas, where people are concentrated. The three weather variables associated with air purification are not associated with any particular ecosystem type because they are annual averages for the entire study area. State-level tables (supplementary materials) allow for comparison across states and years.

3.1.3. *Supply and use account*

The supply table quantifies the provision of each ecological end-product by ecosystem type across multiple years (Table 5), while the use table quantifies the users of each ecological end-product (Table 6). In accounting, an ecosystem service is by definition used by some economic unit or individual, so the total supply and use of a given ecosystem service in a given year must balance. The column in which each supply metric is placed indicates the location where the ecological end-product is used by a user (i.e., where the ecosystem service transaction occurs). State-level tables (see supplementary materials) allow for comparison across individual states within the region.

Table 6
Use table for the Southeast U.S., selected years between 2001 and 2015.

			Economic units			Total
			Industry	Government	Households	
Recreational birding (thousands of birding days)		2001	0	0	56,874	56,874
		2006	0	0	59,360	59,360
		2011	0	0	60,715	60,715
Pollution removal (tonnes/year)	CO	2010				98,690
		2015				92,583
	NO_2	2010				438,139
		2015				494,268
	O_3	2010				4,531,927
		2015				4,258,878
	PM_{10}	2010				1,327,037
		2015				1,205,268
	$PM_{2.5}$	2010				220,218
		2015				257,912
	SO_2	2010				329,580
		2015				176,681

3.2. *Assessment of pilot accounts*

3.2.1. *Regional-scale results*

The total amount of recreational birding in the Southeast U.S. increased by 6.8% from 2001 to 2011, a slightly slower rate than the region’s 13.3% population growth from 2000 to 2010 (Tables 5–6). In 2011, the Developed–Open and Developed–Low-intensity land cover classes were the most commonly used ecosystem types for recreational birding. Together, all developed land cover classes provided more than one-third of all recreational birding days in the region in 2006 and 2011. Forests and open water were also popular for recreational birding throughout the study period. Offshore and open-water birding both decreased substantially from 2001 to 2006 and partly recovered by 2011; the cause of these changes is not clear from the regional account.

With the exception of PM_{10}, which remained steady, air pollutant concentrations decreased between 2010 and 2015 throughout the Southeast U.S. (Table 4). We might consider this as an indicator that the provision of clean air is increasing, but the concentrations reflect the regional balance between emissions and removal rates. Pollutant removal, which is based on pollutant concentrations, the amount of vegetation, and prevailing weather patterns, decreased along with the respective concentrations for all pollutants other than NO_2 and $PM_{2.5}$, but this pattern was not consistent across states within the Southeast due to differences in concentration changes relative to weather and tree canopy conditions (Tables 4–6).

Carbon stored in ecosystems in the Southeast U.S. increased by approximately 1.4% from 2001 to 2010 (Table 4). This increase did not occur uniformly across ecosystem types; the largest net gain occurred in shrub/scrub ecosystems, while the largest net loss occurred in forests. Agricultural ecosystems and woody wetlands also stored less carbon in 2010 than in 2001, while developed areas, grasslands, and emergent herbaceous wetlands stored more carbon. These trends result from changes in the extent of each ecosystem type (Table 3) and in carbon storage related to forest succession. For example, while forest area in the Southeast declined (by 1.9% from 2001 to 2011), forest carbon storage increased from 29.74 to 30.03 kT/km^2, due to forest succession. Similarly, developed and agricultural lands stored more carbon on a per-area basis in 2010 than in 2001, which offset declines in the area of wetlands and agricultural lands. Despite the relatively large carbon storage losses in forest ecosystems, forests continued to provide more

Table 7
State-scale differences in air pollutant concentrations, removal rates, and weather between years 2010 and 2015. Positive values represent an increase over time.

		Alabama	Arkansas	Florida	Georgia	Louisiana	Mississippi	Missouri	North Carolina	South Carolina	Tennessee
Mean Concentration Difference (ppb except for PM (μg/m3))	CO	39.01	−85.14	−66.14	5.80	57.31	−24.81	−13.19	−94.66	−37.99	−30.17
	NO_2	3.19	−1.65	−0.25	−0.63	−0.52	0.04	0.08	−1.72	0.12	−0.27
	O_3	−2.70	−3.13	−2.33	−0.86	−3.43	−2.39	−6.65	0.38	−0.54	−5.16
	PM_{10}	−4.52	0.90	−3.17	−0.52	−3.06	0.11	3.21	0.36	0.51	3.62
	$PM_{2.5}$	2.42	−3.14	0.81	−1.71	0.86	1.91	−1.48	−0.79	−2.27	−0.41
	SO_2	−0.56	−0.72	−0.25	−0.53	−1.13	−1.42	−1.75	−0.81	−0.78	−1.36
Removal Rate Difference (tonnes/yr)	CO	2,183	−3,402	−2,752	450	720	591	635	−3,250	−808	−432
	NO_2	31,893	−7,017	4,123	1,624	−497	12,949	10,013	−7,997	4,322	6,883
	O_3	−33,689	−35,481	−33,375	−23,662	−45,865	−21,391	−35,537	−3,142	−16,702	−23,643
	PM_{10}	−90,228	11,956	−55,737	−12,394	−42,296	2,303	20,018	9,947	3,002	31,771
	$PM_{2.5}$	11,017	−3,514	8,356	1,299	4,933	6,793	1,777	3,592	−117	3,575
	SO_2	−8,997	−6,613	−1,063	−12,234	−11,184	−24,847	− −31,844	-20,556	−10,831	−11,496
Weather Difference	Wind Sp. (m/s)	0.05	0.07	0.08	0.06	−0.01	0.24	0.21	−0.03	0.17	0.39
	Temp. (°C)	1.22	−0.29	1.79	0.96	1.02	−1.03	−0.17	0.51	1.16	−1.19
	Precip. (mm/yr)	697.92	573.10	193.34	373.46	563.49	390.06	291.27	255.37	329.58	290.25

than half of all carbon storage in the Southeast U.S. over the study period.

3.2.2. State-level results

When little change is seen at the regional scale, or the regional accounts raise unanswered questions, state-level analysis may be informative. For example, the state-level recreational birding accounts show that the observed decline from 2001 to 2006 in offshore birding in the Southeast was driven by declines in North Carolina, Florida, and Alabama (supplementary materials). From 2001 to 2006, North Carolina and Florida's offshore birding decreased by more than 60%, while Alabama's decreased by 100%. By 2011, offshore birding in Florida had almost completely recovered to 2001 levels, North Carolina was at about half of its 2001 level, and Alabama still had zero recorded offshore birding. While it is possible that these patterns are caused by inconsistencies in eBird use across those states, external factors may also have influenced offshore birding over the study period. Since offshore birding is often done with tour groups (many offshore eBird observations list group sizes of 10 or greater; this is rare in land-based eBird observations); these results could indicate a change in the number of tours going out, possibly due to impacts from extreme weather events or other disasters (e.g., Hurricane Katrina in 2005 or the Deepwater Horizon oil spill in 2010). Follow-up with the tour boat industry in these states could help to explain these changes in offshore birding over time.

While regional air pollutant concentrations generally decreased between 2010 and 2015, all states had at least one pollutant concentration that increased (Table 7). Conversely, removal rates generally decreased, with some notable exceptions such as Alabama, which had relatively large increases in NO_2 and $PM_{2.5}$ concentrations and removal rates. Located next to Alabama, Mississippi also had a relatively large increase in $PM_{2.5}$ concentration and removal. Missouri and Tennessee had relatively large increases in PM_{10} concentration and removal. While NO_2 concentrations decreased in Tennessee, Georgia, and Florida, removal rates increased due to differences in concentration changes relative to weather and tree canopy conditions between 2010 and 2015. Regional precipitation was 40% greater in 2015 than 2010, while temperatures showed state-level differences and wind speeds remained similar or increased.

3.2.3. Local-scale results

In some cases, even state-level analysis is too broad to see meaningful changes over a study period. For example, the water purification condition metric "percent of flowpath in purifying land cover types" had an absolute change of less than 1% in the Southeast and less than 2% in any individual state from 2001 to 2011. At these aggregation levels, substantial local changes are counteracted by changes in the opposite direction in other parts of a state or region. For example, the 29 counties in the Atlanta-Sandy Springs-Roswell Metropolitan Statistical Area show much sharper declines in this metric than either the Southeast U.S. or Georgia (Fig. 4); all but three counties had changes exceeding 2%, and 6 of the 29 counties had changes of more than 5% from 2001 to 2011. This condition metric could indicate substantial changes in water quality, likely driven by development and potentially affecting water treatment costs or water quality for recreational uses. Additionally, this metric does not include the effect of engineered drainage systems in urban areas, which can directly connect pollution sources to waterways. Therefore, these results likely underestimate the decline in water purification resulting from large-scale land development.

4. Discussion

4.1. Alignment with SEEA EEA and NESCS

The decisions reflected in the design of these pilot ecosystem accounts respect the overarching SEEA EEA framework while accommodating data and conceptual issues that arose as we developed them. NESCS offered a practical and flexible structure and set of rules for naming final ecosystem service flows as the central object of measure in the supply and use account. Its specificity helped to narrow our conception of appropriate metrics.

For example, agricultural crops or livestock are sometimes characterized as a final ecosystem service and included in SEEA EEA supply and use tables (Remme et al., 2018; Vallecillo et al., 2019). From the NESCS perspective, agricultural crops are not an ecosystem service because their production results from both ecosystem processes and components (e.g., fertile soil, wild pollination) and human inputs (e.g., fertilizer, irrigation, weed control). Each ecosystem process or component contributing to crop production is itself an ecosystem service, such as wild pollination, as considered in our accounts. Another aspect of food production, wild foods, are considered ecosystem services within the NESCS system because they are generated by the ecosystem without human inputs aside from their direct collection and use by people.

Our team asked questions common for groups starting the development of ecosystem accounts, but by carefully applying the NESCS method (Section 2.5), we structured our accounts differently than previous efforts in other countries by classifying as condition indicators all metrics that fail the test of being final, unique ecological end-products used in ES transactions that benefit users (i.e., all four boxes of

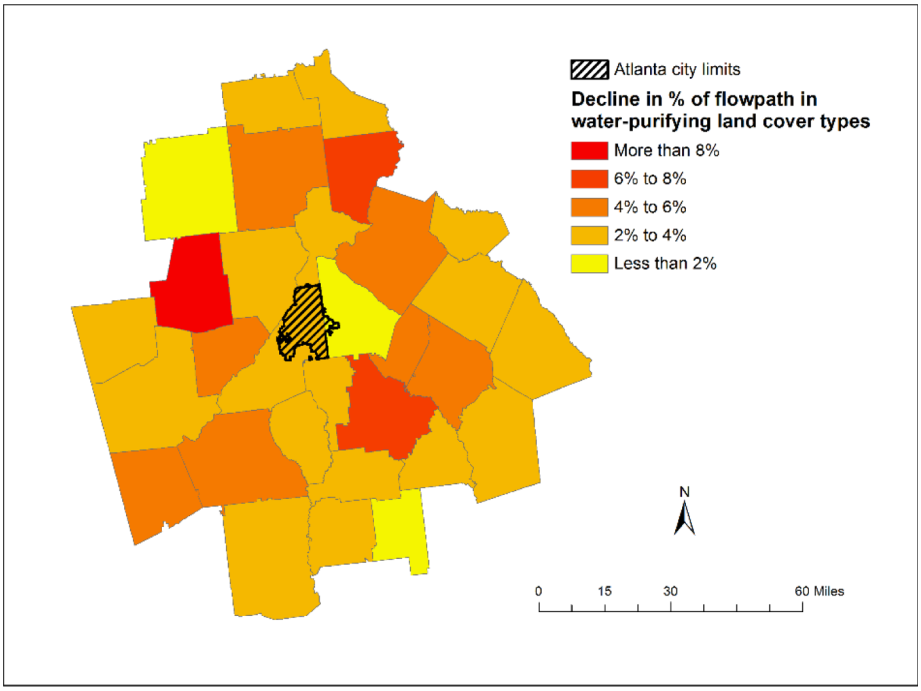


Fig. 4. Absolute change in water purification condition metric "percent of flowpath in purifying land cover types" in the 29-county Atlanta-Sandy Springs-Roswell Metropolitan Statistical Area, from 2001 to 2011.

Fig. 3b). Biodiversity is classified as a characteristic of ecosystems, and crops or livestock are joint products of ecological and economic inputs. Including either in an SUT invites double counting. Visitor counts can quantify recreational use, but the human- and ecosystem-provided aspects of recreation are not always fully addressed in ecosystem accounts. Tables 1, 2, 4–6 all had to pass through many discussions to reach consensus on NESCS and SEEA EEA definitions for use in supply and use tables, leaving a number of remaining metrics that populate the ecosystem condition tables.

If an ecological structure, function, or process is clearly important, but not allowable as a final ecosystem service in the supply and use tables, then should we put it in an account, and if so, where? One early discussion centered on what to do when a formally defined ecosystem service flow, actual wild pollination of crops (exclusive of paid pollination services from honeybees), is much harder to measure than a relevant indicator like pollinator habitat. Our team decided that naming a discrete set of ecosystem characteristics and processes within the ecosystem condition table could give weight to these in a formal account. This solution extends the concept of the condition account beyond a list of general but pertinent variables or indices, but was not precluded by the SEEA EEA (our approach differed from that of Maes et al., 2018, who used expert groups to define a comprehensive set of ecosystem condition indicators for Europe using ecosystems rather than ecosystem services as their organizing unit). These rows in ecosystem condition accounts provide data that will be useful to parameterize models that inform supply and use tables when additional needed data become available to quantify their direct use by economic units. This may increase the useful information in the accounts, without yielding tables with an excessive number of rows. Additionally, ecosystem condition metrics can themselves be useful for decision making, as they can highlight environmental trends over time and space (e.g., increasing pollination dependence for crops grown in Arkansas and Mississippi, Table S6 and S10, or changes in air or water pollutants). While such information may be provided elsewhere (e.g., air or water quality reports), its inclusion in the SEEA EEA typically places multiple metrics alongside each other to facilitate analysis. The key difference between ecosystem condition and supply and use metrics is of course that condition accounts do not measure final ecosystem services, i.e., lack explicit links to economic units (users). Finally, thematic accounts for carbon, land, water, and biodiversity have been developed to increase the visibility of these important metrics for use in decision making (U.N. et al., 2014a; U.N., 2017).

4.2. *Integrating ecosystem accounts with other accounts*

Ecosystem accounts are one part of the larger set of SEEA accounts, and are most useful when considered in this context. For example, the pilot ecosystem accounts presented here complement recent U.S. land and water accounts (Bagstad et al., this issue; Wentland et al., this issue).

The land accounts show that from 2001 to 2011, states in the Southeast had the highest rates of land cover change in the U.S. This trend was driven by high rates of urban growth (especially in Florida and Georgia, and to a lesser extent in South Carolina, North Carolina, and Tennessee), forest loss (notably in North Carolina, Georgia, and Alabama), and farmland loss (particularly in Georgia, Alabama, and Mississippi). This information can facilitate interpretation of the ecosystem accounts; for example, the decline in the ratio of pollinator habitat to dependent crops in Mississippi from 2001 to 2011 could be related to the loss of small farms, leaving larger areas of farmland (USDA, 2014) that are may be isolated from ecosystems that provide pollinator habitat.

Similarly, information from the land and water accounts can be considered alongside relevant metrics from the ecosystem condition and supply and use accounts to give a more complete picture of environmental-economic trends in a given area. In the 29-county Atlanta-Sandy Springs-Roswell Metropolitan Statistical Area, ecosystem accounts can be combined with information from traditional economic accounts and the water and land accounts (Bagstad et al., this issue; Wentland et al., this issue) to make connections between changes in human population, land cover, and ecosystem services (Table 8). Similar to an analysis by Sun et al. (2018) for Atlanta, our results show substantial changes in ecosystem condition and services accompanying rapid urbanization, but also from changes in weather and atmospheric

Table 8
Changes in land, water, ecosystem, and economic accounts for the 29-county Atlanta-Sandy Springs-Roswell Metropolitan Statistical Area.

Account	Metric		% change, 2001–2011
Land accounts[1]	Developed land cover		17.2%
	Agricultural land cover		−6.3%
	Forested land cover		−9.3%
	Other land cover		18.6%
Water accounts	Total water use (million gallons/day, 2000–2010)[2]		−57.8%
	Water productivity ($/100 gallons water use, 2000–2010)[3]		153.3%
	% of water-quality monitoring sites reporting significant declines, 2002–2012[4]	Nitrate (n = 7)	57%
		Specific conductance (n = 6)	67%
		Total suspended solids (n = 4)	25%
Ecosystem accounts[5]	% of flowpath in purifying land cover		−18.2%
	Mean annual concentration, CO (2010–2015)		21.3%
	Mean annual concentration, NO$_2$ (2010–2015)		−0.8%
	Mean annual concentration, O$_3$ (2010–2015)		−2.7%
	Mean annual concentration, PM$_{10}$ (2010–2015)		−18.2%
	Mean annual concentration, PM$_{2.5}$ (2010–2015)		−10.2%
	Mean annual concentration, SO$_2$ (2010–2015)		−57.0%
	Mean annual removal rates, CO (2010–2015)		25.3%
	Mean annual removal rates, NO$_2$ (2010–2015)		9.1%
	Mean annual removal rates, O$_3$ (2010–2015)		−2.7%
	Mean annual removal rates, PM$_{10}$ (2010–2015)		−20.5%
	Mean annual removal rates, PM$_{2.5}$ (2010–2015)		11.0%
	Mean annual removal rates, SO$_2$ (2010–2015)		−49.2%
	Total precipitation		31.9%
	Temperature		6.9%
	Recreational birding-days		209.6%
	Carbon storage (2001–2010)		−1.6%
Urban ecosystem accounts[6]	Energy savings due to cooling effect of urban trees		2%
	Rainfall intercepted by urban trees		−8%
Economic accounts[7]	GDP, all industries		8.8%
Population (2000–2010)[8]			24.0%

[1] National Land Cover Database, 2011 (Homer et al., 2015).
[2] Hutson et al., 2004; Maupin et al., 2014.
[3] Hutson et al., 2004; Maupin et al., 2014; Bureau of Economic Analysis, Gross Domestic Product by Metropolitan Area, 2017, https://www.bea.gov/data/gdp/gdp-metropolitan-area. GDP data are for 2001 and 2010, as GDP by metropolitan area estimates are not available prior to 2001.
4Oelsner et al., 2017.
5 This paper.
[6] Heris et al., this issue.
[7] Bureau of Economic Analysis, Gross Domestic Product by Metropolitan Area, 2017, https://www.bea.gov/data/gdp/gdp-metropolitan-area.
[8] U.S. Census Bureau, Population Change for Counties in the United States and Municipios in Puerto Rico, 2000 to 2010, https://www.census.gov/data/tables/time-series/dec/cph-series/cph-t/cph-t-1.html.

pollutant concentrations. Increases in GDP and population occur alongside substantial increases in recreational birding days, yet forest cover, which might be important for birds and other types of biodiversity, decreases. In addition, we see declines in total water use, but simultaneous decreases in the amount/ability of ecosystems to purify water (% of flowpath in purifying land cover), which could signal a potential quality issue even if less water is being used.

While this example has not yet been used to support decision making, the completion of pilot land, water, and ecosystem accounts for the U.S. (Bagstad et al., this issue; Heris et al., this issue; Wentland et al., this issue) provides strong potential to do so (see e.g., Tampa example in Section 1.1). In the U.S., decision making relevant to natural resources takes place at all levels of government and the private sector (Boyd et al., 2018); within the Federal government, numerous agencies have an interest in natural resources related to ecosystem accounts. Bagstad et al. (this issue) provide for a summary of linkages between SEEA water accounts and decision making in the U.S.; a similar summary for SEEA EEA accounts would be beneficial in aiding their further adoption and use.

4.3. Data gaps and research needs

The SEEA EEA Technical Recommendations state that the goal of ecosystem accounting is “to measure the supply of ecosystem services at a broad landscape scale (ideally up to national level) and also over a series of accounting periods” (U.N., 2017, ¶ 5.88). To fulfill this goal, ecosystem services data must be available at a broad spatial scale for multiple periods. Results should also be attributable to specific spatial subunits such as ecosystem types and able to be aggregated to administrative or physical boundaries of interest. Because environmental data have rarely been collected for the specific purpose of building ecosystem accounts, creating the accounts often highlights gaps where data do not exist or are inadequate for accounting purposes (U.N., 2017).

This is especially true for supply and use tables, which have a narrower scope than ecosystem condition or extent tables. Supply and use tables must contain measures of final ecosystem services or the monetary value that they add to the economy (for physical and monetary supply and use tables, respectively). These measures and values must be assigned not only to ecosystem types and accounting areas, but also to the economic units that use the services. Direct measurement of ecosystem service supply and use that aligns with ecosystem accounting definitions and fulfills the required spatio-temporal criteria is rare, and for many ecosystem services is not yet possible.

When direct measurements are not possible, modeling can provide an estimate of the supply and use of a service. The quality of available models varies for different ecosystem services; many models are developed for particular planning contexts and are not generalizable (Olander et al., 2017a). When an appropriate model does exist, its input data must meet the spatiotemporal requirements of ecosystem accounting. Models should represent current scientific understanding of the processes involved in generating the ecosystem service, and their limitations and uncertainties should be clearly described. In some cases, relevant models exist, but are not always useful for ecosystem accounting due to their cost, data needs, or computing requirements. For example, models have been used to estimate the avoided property damage from coastal storms provided by coastal habitats (Narayan et al., 2017), but they are extremely data intensive and include proprietary components, making them impractical to run regularly for public purposes, and to update in the future. Generalized models like Artificial Intelligence for Ecosystem Services (ARIES, Villa et al., 2014) and Integrated Valuation of Ecosystem Services Tradeoffs (InVEST, Sharp et al., 2018) face challenges in their application to contexts like the United States. While such applications are computationally feasible, their parameterization and calibration across large, heterogeneous environments remains challenging (e.g., Scordo et al., 2018). However, given that the European Union uses the very similar Ecosystem Services Mapping Tool (ESTIMAP, Maes et al., 2015) in its ecosystem accounts, generalized modeling tools like ARIES and InVEST may still have a useful place in future U.S. ecosystem accounts.

Because data and model requirements are less strict for condition tables than supply and use tables, many possible condition metrics exist for each ecosystem asset and final ecosystem service, each with its own

Table 9
Key data and research gaps for the ecosystem services evaluated in the pilot ecosystem accounts for the Southeast U.S.

Ecosystem service	Ideal measure for supply and use table	Key questions related to data gaps
Wild pollination	Wild pollination of pollinator-dependent plants, pollinator visits/flower (physical supply and use table) Additional revenue attributable to wild pollination (monetary supply and use table)	What are the most important wild pollinators? What is the relationship between key habitat characteristics and wild pollinator abundance? What is the relationship between wild pollinator activity on crop fields and crop yield response?
Reduction of water pollutants	Amount of water used by individuals or economic units, cubic meters (physical supply and use table) Avoided water treatment cost due to water purification by ecosystems (monetary supply and use table)	How much water pollution (e.g., sediment, nitrate) do various ecosystems remove, and how does this translate to reduced concentrations in source water bodies? What pollutant concentrations are required for various uses? What is the relationship between pollutant concentrations in source water and water treatment costs?
Reduction of air pollutants	Reduced exposure to air pollutants (physical supply and use table) Number of hospitalizations and healthcare costs avoided due to air pollutant removal (monetary supply and use table)	What effect would updated vegetation cover data including corrections for urban landscapes have on our trend analysis? This would provide tighter linkages between condition and supply and use. How to spatially separate exposure areas from areas of non-use? More frequent updates to demographic information on where people are located would be helpful.
Ecological structures and fauna that are valued and used by recreational birders	Number of birding days (physical supply and use table) Amount spent on equipment and travel for recreational birding (monetary supply and use table)	What factors drive use of recreational birding sites? (Some research related to particular bird species has been done in certain locations, see Kolstoe and Cameron, 2017)

data considerations, assumptions, and limitations that must be evaluated. For example, while there are models to predict relative wild pollinator abundance and activity levels in habitat and agricultural areas, respectively (Koh et al., 2016; Sharp et al., 2018), we have insufficient understanding of the important wild pollinator species, their habitat requirements, and their movement patterns to be confident in the accuracy of those estimates, especially across very large geographic areas. Therefore, we did not include wild pollination in a supply and use table as others have done (Remme et al., 2018; Vallecillo et al., 2019), nor use the indices of wild pollinator population size or activity level as condition metrics. Instead, we used the amount of wild pollinator habitat in proximity to pollinator-dependent crops as a "condition for service" metric.

Even when modeling metrics for condition tables, such as bird species richness, available data may limit the metrics' usefulness in the accounting context. Our bird species richness metric, for example, shows little variation across ecosystem types at the regional scale (Table 4). One reason for this is the coarse resolution of the bird occurrence data from the Breeding Bird Survey, which consists of species-level bird counts by 24.5-mile-long route. Modeling and aggregation at this coarse resolution obscures differences across ecosystem types and precludes us from assessing bird species richness for some ecosystem types that occur in smaller patches. While the state-level accounting tables (supplementary materials) are more informative than the regional-scale table in showing variation among ecosystem types, the ability to model bird species distribution at higher resolution would substantially increase the utility of the bird species richness metric in the accounting tables. Planned updates to the Breeding Bird Survey methodology may enable higher-resolution modeling in future updates to the accounts (K. Pardieck, personal communication, January 2018).

Data gaps draw attention to opportunities for additional data collection to provide relevant information, while gaps in understanding can direct future scientific research to answer questions about the ecosystem components and processes that create ecological end-products used in final ecosystem services. Enhanced knowledge and more complete data can be used to improve the models underlying ecosystem accounts in an iterative process—particularly for the United States, which is at an early stage in its development of ecosystem accounts. Data gaps and research needs are unique to the context in which ecosystem accounts are built and ecosystem services considered (e.g., Table 9, for the Southeast U.S.).

4.4. Next steps for ecosystem accounting in the United States

The next steps toward establishing ecosystem accounts for the U.S. are to expand the pilot accounts presented here in geographic extent, number of ecosystem services, and related condition metrics. Ideally, the next version of these accounts should cover at least the entire continental U.S. (Heris et al., this issue). To ensure that future accounts are aligned with manager and decision maker needs, input from both groups should be sought about what services and metrics would be most useful to include, and what analysis scales and disaggregation levels are most useful for decision making. The accounts should also be updated to incorporate new research or more detailed data that address the gaps described above, and to include new data when these become available (for example, the 2016 NLCD, released in spring 2019, covers additional years beyond those available for our analysis and would allow us to generate accounts extending to 2016, as in Heris et al., this issue).

5. Conclusion

Developers of ecosystem accounts face several important tradeoffs—particularly in choosing which ecosystem services to include in the accounts and how strictly to define ecosystem services within the SEEA EEA framework. In building a pilot set of ecosystem accounts for a 10-state region of the U.S., we inevitably could not be comprehensive, and used data availability to guide the selection of what to quantify first, prioritizing feasibility and quantitative and conceptual rigor. An initial focus on ecosystem services of greatest interest to decision makers is of course beneficial, as decision relevance is critically important for all natural capital accounts (Vardon et al., 2016; Virto et al., 2018). However, for large jurisdictions like the United States, the ecosystem services considered to be of greatest importance are likely to vary even across a 10-state subnational region. Our approach to defining ecosystem services using NESCS is an admittedly conservative one that reflects the fact that ecosystem services—particularly strictly defined, final ecosystem services—are generally more difficult to measure than related aspects of ecosystem condition. A less strict approach could rely on weaker or incomplete data and proxy measures, which are indeed sometimes used in ecosystem accounts. Our work plots one course toward rigorous ecosystem accounts, with the assumption that (1) well-quantified metrics of what we ultimately want to measure in ecosystem accounts are better than proxy measures or poorly defined measures that may double count benefits and (2) data gaps are useful in highlighting paths forward for the research and accounting communities as we aim toward next-generation ecosystem accounts, particularly in countries like the U.S. that are in the early stages of developing ecosystem accounts.

Our work to test the application of NESCS to the process of SEEA EEA accounts development follows calls to do so by the recent SEEA

EEA Technical Recommendations (U.N., 2017, ¶ 5.68), and we believe the approach has added value to pilot ecosystem accounts for the U.S. True to ecosystem accounts' current experimental nature, further testing of the NESCS approach in additional countries and to additional ecosystem services will help to show the relative value of this approach as compared to the current *status quo*.

Ecosystem accounts for the U.S. can help to make clear the many connections between ecosystems and economic systems, providing a consistent structure and platform for collecting and tracking such data in a time series. Many conceptual challenges, data limitations, and institutional and political obstacles remain to compiling ecosystem accounts and using them in decision making (Virto et al., 2018). Despite these hurdles, we see great potential for ecosystem accounting as an input for better-informed decision making at local, state, and national scales (each of which sets policies relevant for resources related to SEEA EEA; Boyd et al., 2018). Our experience constructing pilot ecosystem accounts for the Southeast U.S. provides both theoretical and methodological findings and guidance for selecting and organizing data using the SEEA EEA framework as ecosystem accounts for the U.S. are expanded, both geographically and topically, in the near future.

Acknowledgments

We would like to thank all of the members of the "Accounting for U.S. Ecosystem Services at National and Subnational Scales" working group for engaging in many discussions about ecosystem accounting and for their feedback on this paper. In particular, C. Frank Casey, Cliff Duke, Glenn-Marie Lange, François Soulard, and Michael Vardon provided helpful comments, questions, and guidance. Thanks to Ben Sleeter for allowing us to use his data on terrestrial carbon storage. Zach Ancona developed estimates of wetland soil carbon storage. Jim Clark, Amanda Schwantes, and Christoph Hellmayr provided theoretical guidance and technical expertise for the bird species richness analysis. Anthony Dvarskas provided constructive feedback on an earlier draft of this paper.

This work was conducted as a part of the "Accounting for U.S. Ecosystem Services at National and Subnational Scales" working group supported by the National Socio-Environmental Synthesis Center (SESYNC) under funding received from the National Science Foundation (grant DBI-1052875) and the U.S. Geological Survey John Wesley Powell Center for Analysis and Synthesis (grant GX16EW00ECSV00). This research was also funded by the Department of the Interior Southeast Climate Adaptation Science Center (Grant No. G17AC00204 from the U.S. Geological Survey). The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the Bureau of Economic Analysis, National Oceanic and Atmospheric Administration, USDA Forest Service, or U.S. Environmental Protection Agency. This manuscript is submitted for publication with the understanding that the U.S. Government is authorized to reproduce and distribute reprints for Governmental purposes.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2020.101099.

References

Andreu, M.G., Friedman, M.H., Landry, S.M., Northrop, R.J., 2008. City of Tampa Urban Ecological Analysis 2006-2007. Final Report to the City of Tampa, April 24, 2008. City of Tampa, Florida. https://waterinstitute.usf.edu/upload/documents/TampaUEA2006-7_FinalReport.pdf. Accessed 19 September 2019.

Australian Bureau of Statistics, 2017. Experimental Environmental-Economic Accounts for the Great Barrier Reef. Retrieved from https://www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/4680.0Main+Features12017. Accessed 13 May 2019.

Bagstad, K.J., Ancona, Z.H., Hass, J., Glynn, P.D., Wentland, S., Vardon, M., Fay, J., This issue. Integrating physical and economic data into experimental water accounts for the United States: Lessons and opportunities. Ecosyst. Serv.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: a comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125.

Baker, M.E., Weller, D.E., Jordan, T.E., 2006. Improved methods for quantifying potential nutrient interception by riparian buffers. Landscape Ecol. 21 (8), 1327–1345. https://doi.org/10.1007/s10980-006-0020-0.

Boyd, J.W., Bagstad, K.J., Ingram, J.C., Shapiro, C.D., Adkins, J.E., Casey, C.F., Wentland, S., 2018. The natural capital accounting opportunity: let's really do the numbers. Bioscience 68 (12), 940–943. https://doi.org/10.1093/biosci/biy135.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63 (2–3), 616–626. https://doi.org/10.1016/j.ecolecon.2007.01.002.

Bright, G., Connors, E., Grice, J., 2019. Measuring natural capital: Towards accounts for the UK and a basis for improved decision-making. Oxford Rev. Econ. Policy 35 (1), 88–108. https://doi.org/10.1093/oxrep/gry022.

Bureau of Economic Analysis, 2018. Outdoor recreation satellite account. Retrieved from https://www.bea.gov/data/special-topics/outdoor-recreation. Accessed 13 May 2019.

Bureau of Economic Analysis, 2019. Gross Domestic Product by State: Third Quarter 2018. Retrieved from https://www.bea.gov/data/gdp/gdp-state. Accessed 13 May 2019.

Campbell, K.N., Landry, S.M., 1999. City of Tampa Urban Ecological Analysis. Florida Center for Community Design and Research, University of South Florida: Tampa. https://waterinstitute.usf.edu/upload/documents/TampaUEA.pdf. Accessed 19 September 2019.

Castaneda, J-P., Castillo, F.C., Matias, I., 2017. Implementing natural capital accounting in developing countries: public academic partnerships and policy uptake in Guatemala. In: Vardon, M., Bass, S., Ahlroth, S., Ruijs, A. (Eds.), Better Policy through Natural Capital Accounting: Stocktake and Ways forward, WAVES World Bank, Washington D.C., pp. 103–113. https://www.wavespartnership.org/en/knowledge-center/forum-natural-capital-accounting-better-policy-decisions-taking-stock-and-moving. Accessed 13 May 2019.

Clark, J.S., Nemergut, D., Seyednasrollah, B., Turner, P., Zhang, S., 2017. Generalized joint attribute modeling for biodiversity analysis: median-zero, multivariate, multifarious data. Ecol. Monogr. 87, 34–56. https://doi.org/10.1002/ecm.1241.

Conservation International, 2016a. Pilot Experimental Ecosystem Accounts for San Martin, Peru. Accessed 13 May 2019. https://www.wavespartnership.org/es/knowledge-center/las-cuentas-experimentales-de-los-ecosistemas-en-san-martin-peru.

Conservation International, 2016b. Indicators and Other Methods Used in the Experimental Ecosystem Accounts for San Martin, Peru. Accessed 13 May 2019. https://www.wavespartnership.org/es/knowledge-center/indicadores-y-otros-m%C3%A9todos-usados-en-las-cuentas-experimentales-de-ecosistemas-en.

Dvarskas, A., 2019. Experimental ecosystem accounting for coastal and marine areas: a pilot application of the SEEA-EEA in Long Island coastal bays. Marine Policy 100, 141–151. https://doi.org/10.1016/j.marpol.2018.11.017.

eBird Basic Dataset, n.d. Retrieved from https://ebird.org/ebird/data/download. Accessed 13 May 2019.

Eigenraam, M., Chua, J., Hasker, J., 2013. Environmental-Economic Accounting: Victorian Experimental Ecosystem Accounts, Version 1.0. Department of Sustainability and Environment, State of Victoria. Retrieved from https://unstats.un.org/unsd/envaccounting/londongroup/meeting19/LG19_16_5.pdf. Accessed 13 May 2019.

Eigenraam, M., McCormick, F., Contreras, Z., 2016. Marine and Coastal Ecosystem Accounting: Port Phillip Bay. Report to the Commissioner for Environmental Sustainability. Victoria Department of Environment, Land, Water and Planning, Melbourne. Retrieved from https://www.environment.vic.gov.au/__data/assets/pdf_file/0025/49813/Marine-and-Coastal-Ecosystem-Accounting-Port-Phillip-Bay.pdf. Accessed 13 May 2019.

European Commission, n.d. Natural Capital Accounting - Environment. Retrieved from http://ec.europa.eu/environment/nature/capital_accounting/index_en.htm. Accessed 30 April 2019.

Food and Agriculture Organization of the United Nations, 2017. Guatemala to boost its forest governance and legal timber trade. Accessed 13 May 2019. http://www.fao.org/in-action/eu-fao-flegt-programme/news-events/news-details/en/c/472394/.

Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Klein, A.M., 2013. Wild pollinators enhance fruit set of crops regardless of honey bee abundance. Science 339, 1608–1611. https://doi.org/10.1126/science.1230200.

Guatemala Consejo Nacional de Desarrollo Urbano y Rural, 2014. Plan Nacional de Desarrollo K'atun: nuestra Guatemala 2032. Retrieved from http://www.segeplan.gob.gt/downloads/Plan_Nacional_de_Desarrollo_Katun_2032.pdf. Accessed 13 May 2019.

Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, et al., 2015. Natural capital and ecosystem services informing decisions: from promise to practice. Proc. National Acad. Sci. 112, 7348–7355. https://doi.org/10.1073/pnas.1503751112.

Haines-Young, R., Potschin, M., 2018. Common International Classification of Ecosystem SERvices (CICES) v5.1: Guidance on the Application of the Revised Structure. Fabis Consulting, Nottingham, U.K. https://cices.eu/content/uploads/sites/8/2018/01/Guidance-V51-01012018.pdf. Accessed 20 September 2019.

Hein, L., Bagstad, K., Edens, B., Obst, C., de Jong, R., Lesschen, J.P., 2016. Defining ecosystem assets for natural capital accounting. PLoS One 11, e0164460. https://doi.org/10.1371/journal.pone.0164460.

Hein, L., Obst, C., Edens, B., Remme, R.P., 2015. Progress and challenges in the development of ecosystem accounting as a tool to analyse ecosystem capital. Curr. Opinion

Environ. Sustain. 14, 86–92. https://doi.org/10.1016/j.cosust.2015.04.002.

Heris, M., Bagstad, K.J., Rhodes, C., Troy, A., Middel, A., Matuszak, J., This issue. Piloting urban ecosystem accounting for the U.S. Ecosyst. Serv.

Homer, C., Dewitz, J., Yang, L., Jin, S., Danielson, P., 2015. Completion of the 2011 national land cover database for the conterminous United States – representing a decade of land cover change information. Photogram. Eng. Rem. Sens. 81, 343–354. https://doi.org/10.14358/PERS.81.5.345.

Hutson, S.S., Barber, N.L., Kenny, J.F., Linsey, K.S., Lumia, D.S., Maupin, M.A., 2004. Estimated use of water in the United States in 2000. U.S. Geological Survey Circular 1268. Retrieved from https://pubs.usgs.gov/circ/2004/circ1268/. Accessed 13 May 2019.

Keith, H., Vardon, M., Stein, John, Stein, Janet, Lindenmayer, D., 2017. Ecosystem accounts define explicit and spatial trade-offs for managing natural resources. Nat. Ecol. Evol. 1, 1683–1692. https://doi.org/10.1038/s41559-017-0309-1.

Koh, I., Lonsdorf, E., Williams, N., Brittain, C., Isaacs, R., Gibbs, J., Ricketts, T., 2016. Modeling the status, trends, and impacts of wild bee abundance in the United States. Proc. Natl. Acad. Sci. 113 (1), 140–145. https://doi.org/10.1073/pnas.1517685113.

Kolstoe, S., Cameron, T.A., 2017. The non-market value of birding sites and the marginal value of additional species: biodiversity in a random utility model of site choice by eBird members. Ecol. Econ. 137, 1–12. https://doi.org/10.1016/j.ecolecon.2017.02.013.

Landers, D., 2015. National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application. U.S. Environmental Protection Agency, Washington, DC, EPA/800/R-15/002. https://www.epa.gov/sites/production/files/2015-12/documents/110915_nescs_final_report_-_compliant_1.pdf. Accessed 13 May 2019.

Landers, D., Nahlik, A., 2013. Final Ecosystem Goods and Services Classification System (FEGS-CS). U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-13/ORD_004914. Retrieved from https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NHEERL&dirEntryId=257922&CFID=139720405&CFTOKEN=59303549&jsessionid=cc303f3c529fb2342dd56e52d7c80353e2a2. Accessed 13 May 2019.

Landry, S.M., Koeser, A.K., Northrop, R.J., McLean, D., Donovan, G. Andreu, M.G., Hilbert, D., 2018. City of Tampa Tree Canopy and Urban Forest Analysis 2016. University of South Florida School of Geosciences Faculty and Staff Publications 1268: Tampa. https://scholarcommons.usf.edu/geo_facpub/1268. Accessed 19 September 2019.

Landry, S., Northrop, R.J., Andreu, M., Rhodes, C.C., 2013. City of Tampa 2011 Urban Forest Analysis: The Structure, Composition, Function and Economic Benefits of Trees and the Urban Forest. Final Report to the City of Tampa, September 2013. City of Tampa, Florida. https://waterinstitute.usf.edu/upload/projects/TampaUEA/Tampa_2011_UrbanForestAnalysis.pdf. Accessed 19 September 2019.

Loomis, J., Haefele, M., Dubovsky, J., Lien, A.M., Thogmartin, W.E., Diffendorfer, J., et al., 2018. Do economic values and expenditures for viewing waterfowl in the U.S. differ among species? Hum. Dimensions Wildlife 23 (6), 587–596. https://doi.org/10.1080/10871209.2018.1496371.

Losada, I.J., Beck, M., Menendez, P., Espejo, A., Torres, S., Diaz-Simal, P., et al., 2017. Valuing Protective Services of Mangroves in the Philippines: Technical report. World Bank. https://doi.org/10.1596/27666.

Mace, G., 2019. The ecology of natural capital accounting. Oxford Rev. Econ. Policy 35 (1), 54–67. https://doi.org/10.1093/oxrep/gry023.

Mackun, P., Wilson, S., 2011. Population distribution and change: 2000 to 2010. In: 2010 Census Briefs. C2010BR-01. https://www.census.gov/prod/cen2010/briefs/c2010br-01.pdf. Accessed 13 May 2019.

Maes, J., Fabrega, N., Zulian, G., Barbosa, A., Vizcaino, P., Ivits, E., et al., 2015. Mapping and Assessment of Ecosystems and their Services; Trends in ecosystems and ecosystem services in the European Union between 2000 and 2010. Publications Office of the European Union, Luxembourg. https://doi.org/10.2788/341839. Accessed 9 September 2019.

Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., et al., 2018. Mapping and Assessment of Ecosystems and their Services: An Analytical Framework for Ecosystem Condition. Publications Office of the European Union, Luxembourg. http://catalogue.biodiversity.europa.eu/uploads/document/file/1673/5th_MAES_report.pdf. Accessed 13 May 2019.

Maupin, M.A., Kenny, J.F., Hutson, S.S., Lovelace, J.K., Barber, N.L., Linsey, K.S., 2014. Estimated use of water in the United States in 2010. U.S. Geological Survey Circular 1405. https://pubs.usgs.gov/circ/1405/pdf/circ1405.pdf. Accessed 13 May 2019.

Mayor's Steering Committee on Urban Forest Sustainability, 2009. Final vision, goals, and strategies. City of Tampa, Florida. https://www.tampagov.net/sites/default/files/planning/files/UCAM-2010-2012/tampauf_vision_goals_strategies.pdf. Accessed 19 September 2019.

Nahlik, A.M., Fennessy, M.S., 2016. Carbon storage in US wetlands. Nat. Commun. 7, 13835. https://doi.org/10.1038/ncomms13835.

Narayan, S., Beck, M.W., Wilson, P., Thomas, C.J., Guerrero, A., Shepard, C.C., Trespalacios, D., 2017. The value of coastal wetlands for flood damage reduction in the northeastern USA. Sci. Rep. 7. https://doi.org/10.1038/s41598-017-09269-z.

Northrop, R.J., Beck, K. Irving, R., Landry S.M., Andreu, M.G., 2013. City of Tampa Urban Forest Management Plan. November 2013. City of Tampa, Florida. https://waterinstitute.usf.edu/upload/documents/Tampa_UrbanForestManagementPlan_Nov2013_web.pdf. Accessed 19 September 2019.

Nowak, D.J., Hirabayashi, S., Bodine, A., Greenfield, E., 2014. Tree and forest effects on air quality and human health in the United States. Environ. Pollut. 193, 119–129. https://doi.org/10.1016/j.envpol.2014.05.028.

Oelsner, G.P., Sprague, L.A., Murphy, J.C., Zuellig, R.E., Johnson, H.M., Ryberg, K.R., et al., 2017. Water-Quality Trends in the Nation's Rivers and Streams, 1972-2012 - Data Preparation, Statistical Methods, and Trend Results. Scientific Investigations Report. U.S. Department of the Interior, U.S. Geological Survey. https://doi.org/10.3133/sir20175006.

Olander, L., Bagstad, K., Characklis, G. W., Comer, P., Effron, M., Gunn, J., et al., 2017a. Data and Modeling Infrastructure for National Integration of Ecosystem Services into Decision Making: Expert Summaries. NESP WP 1602. Durham: National Ecosystem Services Partnership. https://nicholasinstitute.duke.edu/publications/data-and-modeling-infrastructure-national-integration-ecosystem-services-decision. Accessed 13 May 2019.

Olander, L., Mason, S., Locklier, K., Urban, D., Ihlo, C., et al., 2017b. Mapping ecosystem services for the Gulf Coastal Plains & Ozarks Landscape Conservation Cooperative. Nicholas Institute for Environmental Policy Solutions, Duke University. https://nicholasinstitute.duke.edu/sites/default/files/GCPO_Ecosystem_Service_Mapping.pdf. Accessed 13 May 2019.

Pardieck, K.L., Ziolkowski Jr., D.J., Lutmerding, M. Hudson, M.A.R., 2018. North American Breeding Bird Survey Dataset 1966 - 2017, version 2017.0. U.S. Geological Survey, Patuxent Wildlife Research Center. https://doi.org/10.5066/F76972V8.

Remme, R., Lof, M., de Jongh, L., Hein, L., Schenau, S., de Jong, R., Bogaart, P., 2018. The SEEA EEA biophysical ecosystem service supply-use account for the Netherlands. Central Bureau of Statistics and Wageningen University. https://www.cbs.nl/-/media/_pdf/2018/23/psu_ess_nl.pdf. Accessed 18 September 2019.

Remme, R.P., Schröter, M., Hein, L., 2014. Developing spatial biophysical accounting for multiple ecosystem services. Ecosyst. Serv. 10, 6–18. https://doi.org/10.1016/j.ecoser.2014.07.006.

Ruijs, A., Vardon, M., Bass, S., Ahlroth, S., 2019. Natural capital accounting for better policy. Ambio 48, 714–725. https://doi.org/10.1007/s13280-018-1107-y.

Scordo, F., Lavender, T.M., Seitz, C., Perillo, V.L., Rusak, J.A., Piccolo, M.C., Perillo, G.M.E., 2018. Modeling water yield: assessing the role of site and region-specific attributes in determining model performance of the InVEST Seasonal Water Yield model. Water 10 (11), 1496. https://doi.org/10.3390/w10111496.

Sharp, R., Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, et al., 2018. InVEST User's Guide. The Natural Capital Project, Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund. http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/. Accessed 13 May 2019.

Sleeter, B.M., Liu, J., Daniel, C., Rayfield, B., Sherba, J., Hawbaker, T.J., et al., 2018a. Effects of contemporary land-use and land-cover change on the carbon balance of terrestrial ecosystems in the United States. Environ. Res. Lett. 13, 045006. https://doi.org/10.1088/1748-9326/aab540.

Sleeter, B.M., Loveland, T., Domke, G., Herold, N., Wickham, J., Wood, N., 2018b. Land cover and land-use change. In: In: Reidmiller, D.R., Avery, C.W., Easterling, D.R., Kunkel, K.E., Lewis, L.L.M., Maycock, T.K., Stewart, B.C. (Eds.), Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment Volume II. U.S. Global Change Research Program, Washington, DC, pp. 202–231. https://doi.org/10.7930/NCA4.2018.CH5.

Statistics Canada, 2013. Human activity and the environment. Can. Pub. Policy / Analyse de Politiques 4, 587. https://doi.org/10.2307/3549992.

Sun, X., Crittenden, J.C., Li, F., Lu, Z., Dou, X., 2018. Urban expansion simulation and spatio-temporal changes of ecosystem services, a case study in Atlanta Metropolitan area, USA. Sci. Total Environ. 622–623, 974–987. https://doi.org/10.1016/j.scitotenv.2017.12.062.

Sutherland, I.J., Villamagna, A.M., Dallaire, C.O., Bennett, E.M., Chin, A.T.M., Yueng, A.C.Y., Lamothe, K.A., Tomscha, S.A., Cormier, R., 2018. Undervalued and under pressure: a plea for greater attention toward regulating services. Ecol. Ind. 94, 23–32. https://doi.org/10.1016/j.ecolind.2017.06.047.

System of Environmental Economic Accounting, n.d. Natural Capital Accounting Project: Natural Capital Accounting and Valuation of Ecosystem Services. Retrieved from https://seea.un.org/home/Natural-Capital-Accounting-Project. Accessed 30 April 2019.

United Nations, European Commission, U.N. Food and Agriculture Organization, Organization for Economic Cooperation and Development, and World Bank, 2014a. System of Environmental-Economic Accounting 2012: Experimental Ecosystem Accounting. https://seea.un.org/sites/seea.un.org/files/seea_eea_final_en_1.pdf. Accessed 13 May 2019.

United Nations, European Commission, U.N. Food and Agriculture Organization, International Monetary Fund, Organization for Economic Cooperation and Development, and World Bank, 2014b. System of Environmental-Economic Accounting 2012: Central Framework. https://seea.un.org/sites/seea.un.org/files/seea_cf_final_en.pdf. Accessed 13 May 2019.

United Nations, 2017. SEEA Experimental Ecosystem Accounting: Technical Recommendations. https://seea.un.org/sites/seea.un.org/files/technical_recommendations_in_support_of_the_seea_eea_final_white_cover.pdf. Accessed 13 May 2019.

United Nations, 2018. SEEA Experimental Ecosystem Accounting Revision 2020: Revision Issues Note – Final. Retrieved from https://seea.un.org/sites/seea.un.org/files/documents/EEA/eea_2020_revision_issue_final_jul2018.pdf. Accessed 30 April 2019.

U.S. Department of Agriculture, 2014. 2012 Census of Agriculture: Mississippi, State and County Data, vol. 1, Geographic Area Series Part 24 AC-12-A-24. USDA, Washington, DC. https://www.nass.usda.gov/Publications/AgCensus/2012/Full_Report/Volume_1,_Chapter_1_State_Level/Mississippi/msv1.pdf. Accessed 23 August 2019.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau, 2011. 2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation. https://www.census.gov/prod/2012pubs/fhw11-nat.pdf. Accessed 13 May 2019.

U.S. Environmental Protection Agency, 2013. Technology Transfer Network (TTN), Air Quality System (AQS), Download Detailed AQS Data. http://www.epa.gov/ttn/airs/airsaqs/detaildata/downloadaqsdata.htm. Accessed 02 November 2013.

U.S. Environmental Protection Agency, 2015. National Ecosystem Services Classification

System (NESCS): Framework Design and Policy Application. EPA-800-R-15-002. US EPA, Washington, DC. https://www.epa.gov/sites/production/files/2015-12/documents/110915_nescs_final_report_-_compliant_1.pdf. Accessed 13 May 2019.

U.S. Geological Survey, Gap Analysis Program (GAP), 2016. Protected Areas Database of the United States (PAD-US), version 1.4. https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/pad-us-data-download?qt-science_center_objects=0#qt-science_center_objects. Accessed 13 May 2019.

Vallecillo, S., La Notte, A., Kakoulaki, G., Kamberaj, J., Robert, N., Dottori, F., Feyen, L., Rega, C., Maes, J., 2019. Ecosystem services accounting Part II: Pilot accounts for crop and timber provision, global climate regulation, and flood control. Publications Office of the European Union: Luxembourg. https://doi.org/10.2760/631588.

Vardon, M., Burnett, P., Dovers, S., 2016. The accounting push and the policy pull: balancing environmental and economic decisions. Ecol. Econ. 124, 145–152. https://doi.org/10.1016/j.ecolecon.2016.01.021.

Villa, F., Bagstad, K.J., Voigt, B., Johnson, G.W., Portela, R., Honzak, M., Batker, D., 2014. A methodology for adaptable and robust ecosystem services assessment. PLOS One 9 (3). https://doi.org/10.1371/journal.pone.0091001.

Virto, L.R., Weber, J., Jeantil, M., 2018. Natural capital accounts and public policy decisions: findings from a survey. Ecol. Econ. 144, 244–259. https://doi.org/10.1016/j.ecolecon.2017.08.011.

WAVES, 2012. Moving Beyond GDP: How to factor natural capital into economic decision making. World Bank. Retrieved from http://documents.worldbank.org/curated/en/724461468157521765/pdf/696120WP0Movin00606020120Box369282B.pdf. Accessed 30 April 2019.

WAVES, 2014. Natural Capital Accounting in Action: Guatemala's forest accounts link forest resources with the economy. World Bank. Retrieved from https://www.wavespartnership.org/sites/waves/files/images/NCA%20in%20Action_Guatemala%20forests.pdf. Accessed 30 April 2019.

WAVES, 2016a. Pilot Ecosystem Account for Laguna de Bay Basin. World Bank. Retrieved from https://www.wavespartnership.org/sites/waves/files/kc/LdeBay%20FINAL%20LowRes%20Dec15%202016.pdf. Accessed 30 April 2019.

WAVES, 2016b. Pilot Ecosystem Account for Southern Palawan. World Bank. Retrieved from https://www.wavespartnership.org/sites/waves/files/kc/WB_Southern%20Palawan%20Tech%20Report_FINAL_Nov%202016.pdf. Accessed 30 April 2019.

WAVES, 2017. Natural Capital Accounting and Policy: The Philippines. World Bank. Retrieved from https://www.wavespartnership.org/sites/waves/files/kc/Philippines%20offer%20doc_FINAL.pdf. Accessed 30 April 2019.

Wentland, S.A., Hass, J.L., Ancona, Z., Bagstad, K.J., Boyd, J., Gindelsky, M., Moulton, J. G., This issue. Accounting for land in the United States: integrating physical land cover, land use, and monetary valuation. Ecosyst. Serv.