United Nations | United Nations Economist Network (UNEN)

## Thematic Brief

# Nature-based Solutions

*Author: David Simpson, American University School of International Services, USA*
*Report Director: Pushpam Kumar, United Nations Environment Programme*

*Editorial Assistants: Amelia Holmes, and Samia Islam, United Nations Environment Programme*



All queries should be addressed to DESA-UNEN@un.org.

# Economics of Nature-based Solutions: Current Status and Future Priorities

**The Economics of Nature-Based Solutions: Current Status and Future Priorities**



**Author:** David Simpson, American University School of International Services, USA

**Report Director:** Pushpam Kumar, United Nations Environment Programme

**Technical Editor:** Amelia Holmes, United Nations Environment Programme

**Reviewers:**

Edward Barbier, Colorado State University, United States of America

Francois Cohen, Oxford University, United Kingdom

Eric Mungatana, University of Pretoria, South Africa

Priya Shyamsundar, The Nature Conservancy, United States of America

Shun Managi, Kyushu University, Japan

**Suggested citation:**
United Nations Environment Programme (2020). *The Economics of Nature-based Solutions: Current Status and Future Priorities.* United Nations Environment Programme Nairobi.

**United Nations Environment Programme**
United Nations Avenue, Gigiri

PO BOX 30552, 00100

Nairobi, Kenya

Tel: +254 (0) 207621234

Email: unep-publications@un.org

Cover photos: ©
Back photo: ©Shutterstock/Quick Shot
Design and layout: UNON Publishing Section
IBSN:

All publications with an ISBN should feature the UNEP logo on the front and the back cover. It is recommended to use the logo on the front cover - ideally on the top right corner, and on the back cover - ideally at the bottom left corner.

# Table of Contents

# Executive Summary

There has been a great deal of recent interest in Nature-based Solutions (NbS) as an approach to tackle climate change with socio-economic and environmental co-benefits. Aspects of NbS include storing carbon to mitigate climate change, maintaining areas of vegetation to protect against higher temperatures, preventing further climate-driven increases in storm severity and frequency and the providing of a range of other services, such as pollination and treatment of pollution. In essence, NbS are an ecological approach to climate change action, whilst also enhancing the resilience of natural and managed ecosystems and the human settlements that adjoin them.

While a number of definitions of NbS have been proposed, none have been universally adopted. The lack of unanimity on a definition may not be surprising, given the range of objectives included in the approach. Because there is not yet a clear consensus as to what constitutes an NbS, there is sometimes disagreement among advocates as to what deserves the designation, and which approaches are best in particular circumstances.

This report considers the economic attributes of NbS for addressing three aspects: mitigating climate change, adapting to climate change, and providing other ecosystem services. The second and third categories are discussed together, as the characteristics of NbS in adapting to climate change and providing other ecosystem services are similar. For both climate change adaptation and ecosystem service flows, i) geographical context is critical, ii) the principle of diminishing returns often implies that the marginal value of services declines quickly in the geographical scale of the ecosystem providing them, and iii) mature ecosystems are likely to be most effective in generating benefits. This is in contrast to NbS for carbon storage, where values are largely independent of the location at which carbon is sequestered, the contributions of NbS are not of a large enough magnitude relative to the need to induce sharply diminishing returns, and the incremental yearly contributions of mature forests, grasslands, or other ecosystems decline to zero as the natural system approaches a carbon-neutral steady state.

There is widespread agreement that undisturbed forests and grasslands, among other ecosystems should be maintained as part of NbS. The value of the carbon they sequester will generally exceed the earnings that could be realized from converting them to other uses, and they often provide many additional ecosystem services. While mature ecosystems may provide little additional carbon sequestration, their disturbance would cause such large immediate emission releases that the “carbon debt” imposed could not be compensated with subsequent regrowth. Moreover, remnant areas of relatively undisturbed ecosystems are largely found at the extensive margin and therefore have not yet been converted to other uses because it is not profitable to do so. The economic case for preserving such areas is strong, then.

If existing natural areas should be maintained, should some areas that once supported forests, natural grasslands, or wetlands be restored to their previous status? Some undoubtedly should, but restoration in general involves complicated trade-offs. While a mature natural system may provide a myriad of ecological and economic benefits, it may take years or decades of growth for a newly reestablished area to fully realize the same benefits. Natural systems such as forests provide different benefits at different times. For instance, as a growing forest reaches maturity, it nears a balance between additional carbon gains and losses, and so contributes little more to storage. Conversely, climate adaptation and other ecosystem benefits may increase to their maximum values as the forest reaches maturity.

Difficult choices follow regarding priorities: should more land be devoted to restoring diverse natural ecosystems, or should currently degraded or cleared areas be managed more intensively for carbon storage or other objectives? There is not usually a clear answer, and the question is made more difficult by the prospect of leakage, whereby forgoing land conversion to agriculture at one place may increase the demand for farmland and pasture elsewhere.

If it is not clear that degraded areas or areas managed for agriculture should be restored to some semblance of their original land cover, how should they be managed? There are several possibilities. Agricultural lands might be managed to maximize the amount of carbon that can be embodied in their products. An extreme example of this approach would be to manage for bioenergy with carbon capture and storage (BECCS). Alternatively, a variety of agricultural practices may help to increase carbon storage in soils, which could prove valuable in terms of climate mitigation, on-farm productivity enhancements, and provision of ecosystem services such as erosion protection and nutrient pollution abatement. The leakage effects of such farm-based carbon soil practices are uncertain, however, including a potentially virtuous cycle of increased yields and reduced land use to a more conventional concern that increasing costs of agricultural production in some areas could lead to expansion of less benign practices to other areas.

NbS could generate a wide range of benefits in adapting to the higher temperatures, more severe storms, and other consequences of climate change. They may also provide a range of additional benefits, such as pollination, pollution control, and other ecosystem services. While such benefits may be important, they tend to share a number of characteristics that constrain their values. One is that values are very specific to context. Coastal storm protection services may, for example, be very valuable in regions with large populations and costly infrastructure at risk, but less valuable for sparsely settled coastlines. Similarly, pollination services are valuable in regions in which crops rely on them, but not as much elsewhere. Pollution treatment may be valuable when it is provided by an ecosystem situated between a source of pollution and a population vulnerable to it, but not where neither is in place. Adaptation values and other ecosystem services also tend to display rapidly diminishing returns: a narrow strip of coastal vegetation or a small population of pollinators may generate substantial value, but expansive areas or very high numbers of pollinators may not provide significantly higher values.

There has been a great deal written on the economics of NbS, but not all existing work conforms with received principles of economic analysis. Moreover, widespread adoption of

NbS would involve a number of complexities and uncertainties that are not well understood. Several recommendations are offered to make the concept of NbS more useful and guide its application.

1. *Adopt clear definitions, objectives, and scope*. Perhaps more important than achieving an agreement among all parties using the term "Nature-based Solutions" is that proposals for NbS be clear and explicit about their objectives. Some objectives inevitably conflict, so clarifying and prioritizing them is essential if NbS is to be an operationally useful concept. This is particularly important because many NbS have different effects at different spatial and temporal scales. Global implications and long-term effects should be considered whenever feasible, and choices to restrict attention to local and temporary effects should be explicitly recognized and justified. Additionally, NbS should be recognised as a powerful entry point for supporting the livelihoods of both men and women, boys and girls, such that synergistic progress is made on closing the gaps in gender inequalities.

2. *Curate existing research.* There has been a tremendous amount of research undertaken on NbS and their economic values and implications. Yet it is not clear which findings in the literature are credible or provide the most useful guidance for application in new settings. A "retroactive peer review" of existing studies could i) validate studies that may be used as exemplars for new work and/or employed to extrapolate benefits in other contexts; ii) identify common conceptual errors and warn researchers against them; and iii) recommend best practices.

3. *Assess the ability of complex models to make useful predictions at relevant spatial and temporal scales*. Issues of spatial and temporal scale are critical to consider in NbS. A number of complex modeling platforms have been developed to predict the spatial and temporal extent of different policies. An assessment could identify which model(s) might be best used for particular purposes, and what further improvements might make complex models more accurate and useful.

4. *Identify effective policy instruments for implementing NbS*. Regulations, taxes, subsidies, changes in land tenure, communal resource management, monitoring and enforcement of conservation measures, and other policy approaches have been proposed for implementing NbS. It will be helpful to determine which have proved most effective under what circumstances.

5. *Adopt adaptive management plans*. Given the uncertainties inherent in NbS, as well as the severity of the problems against which they are deployed, it is important to adapt to new information as it becomes available. Protocols should be defined for amending NbS plans in advance of implementing changes.

## Acknowledgments

The author thanks Edward Barbier, Francois Cohen, Shun Managi, Eric Mungatana, and Priya Shyamsundar for generous and constructive suggestions on an earlier draft. Conversations and correspondence with Simon Nicholson, Brent Sohngen, and Jeffrey Vincent were also helpful. Direction and guidance from Pushpam Kumar was very much appreciated. Responsibility for any errors and all opinions rests with the author alone, however.

# 1. Nature-based Solutions

In recent decades "Nature-based Solutions" (NbS) has been gaining traction as a preferred approach for addressing imminent environmental and societal challenges. While NbS have been suggested to serve a number of purposes, the appeal of the concept stems largely from two concerns. One is climate change. The earth may be warming at rates rarely seen in the geological record due to the rapid accumulation of carbon dioxide and other greenhouse gases in the atmosphere. Climate change creates a need for measures both to mitigate and to adapt to it. The second concern is that the human impact on the planet is also reflected in losses of biological diversity. Current losses may be approaching rates that are only found during a handful of cataclysmic events in the fossil record. While climate change is implicated in biodiversity loss, the larger culprit is often identified as loss of natural habitats that maintain diverse assemblages of organisms. There is, then, tremendous appeal in approaches that would use the preservation of natural systems and the diversity they support to mitigate and adapt to climate change.

## 1.1 The potential of NbS

To give some sense of the potential of NbS, it is useful to consider a recent, comprehensive, and widely cited[1] example. While the efficacy and desirability of NbS could be approached from different societal, ecological, and perhaps other perspectives, the focus in this paper will be on their economic attributes. A recent paper by Griscom et al. (2017; see also Busch et al. 2019 for an alternative approach to cost estimation yielding similar results, and Fuss et al. 2018 for cost calculations for a broader set of negative emissions technologies including biological, artificial, and hybrid approaches) illustrates both the economic potential for employing NbS and some of the issues in evaluating the extent to which that potential might be realized. The authors

[1] While it was only published a little over two years ago, Google Scholar reports that (Griscom et al. 2017) has already been cited in over 300 other works.

review literature on 20 nature-based solutions for climate mitigation. They group the 20 practices into three broad categories: six focused on forestry, ten on agriculture, and four on wetlands. The total net mitigation potential of all 20 measures might exceed 23 thousand Tg $CO_{2e}$ per year.[2] To put this in perspective, it is estimated that approximately 30 thousand Tg $CO_{2e}$ of mitigation would be required per year between now and 2030 to maintain a reasonable chance of not exceeding more than 2 degrees Celsius mean global warming. The 23 thousand Tg $CO_{2e}$ per year is an upper bound on what might be accomplished without reducing food security or increasing biodiversity loss.

A figure of 100 USD per ton of $CO_{2e}$ has been offered as an estimate of the "social cost of carbon" – societal willingness to pay to mitigate an incremental ton of carbon removal (Dietz and Stern 2015; Stiglitz et al. 2017a).[3] Griscom, et al., estimate that NbS could mitigate over 11,000 Tg $CO_{2e}$ per year at a cost of less than 100 USD per ton. Over 4,000 Tg $CO_{2e}$ could be mitigated at less than a tenth of that cost, 10 USD per ton. Again, to put such figures in context, this would more than offset the approximate 7,000 Tg $CO_{2e}$ of yearly emissions of the transport sector globally.

While the estimates of Griscom, et al. highlight the potential of NbS, they also underscore some of the issues in determining whether and how that potential might be realized. The forest sector accounts for almost two-thirds of the cost-effective (less than 100 USD per ton), and a little over half of the low-cost (less than 10 USD per ton) mitigation potential (see Table 1). Substantial investments would need to be made in reforestation, particularly in the tropics (see also Lewis et al. 2019), as well as in forestry management. The costs of such investments are highly variable, particularly as the initial costs of establishment may need to be augmented by monitoring and enforcement efforts in later years (Cohen-Shacham et al. 2019; Fuss et al. 2018). The uncertainty in Griscom, et al.'s estimates for areas that can be managed cost-

[2] "Teragrams of carbon dioxide equivalent." A teragram is a million tons. Carbon dioxide is the most common greenhouse gas, and emissions of other gases, such as methane, are converted into their equivalent, in terms of warming potential, of carbon dioxide.

[3] In a somewhat different formulation, (Dietz et al. 2018) report that a marginal cost of at least 100 USD per ton would need to be incurred to hold global warming to 1.5 degrees Celsius.

# Table 1

## Possibilities for cost-effective and low-cost NbS for climate mitigation

Cost-effective: ≤ 100 USD per Mg $CO_{2e}$ mitigated

Low-cost: ≤ 10 USD per Mg $CO_{2e}$ mitigated

Saturation: Minimum number of years by which full mitigation benefits are achieved and little further sequestration is realized

Figures in parentheses:

For mitigation, percentage of mitigation required to stay on likely no-more-than 2 degree C warming trajectory (30,000 Tg $CO_{2e}$)

For land area affected, percentage of ice-free terrestrial surface area of Earth (13,100 M Ha)

Source: (Griscom et al. 2017)

| Land use | Cost-effective mitigation (Tg CO2e) | Low-cost mitigation (Tg CO2e) | Land area affected to achieve cost-effective mitigation (M Ha) | Saturation (years) |
|---|---|---|---|---|
| Forests | 7,320 | 2,257 | 2,849 | 25 + |
| | (24.4) | (7.5) | (21.7) | |
| | | | | |
| Agriculture | 2,456 | 1,095 | 1,555 | 50 + |
| | (8.2) | (3.7) | (11.9) | |
| | | | | |
| Wetlands | 1,546 | 784 | 75 | 20 + |
| | (5.2) | (2.6) | (0.6) | |
| | | | | |
| Total | 11,355 | 4,147 | 4,513 | |
| | (37.8) | (13.8) | (34.4) | |

effectively is considerably greater for the forest sector than for agriculture and wetlands but underscores the point that any such projections are necessarily highly speculative.

This is particularly true because of the scale of the proposed interventions. All told, the land use and land management changes called for in (Griscom et al. 2017) would affect more than a third of the ice-free terrestrial area of the earth. Land use changes on this scale would have profound economic, as well as environmental and societal effects.

Areas that are reforested, farmed less intensively, or otherwise maintained in a more natural state may provide other goods and services. Some of the services may be more valuable as temperatures and sea levels rise and storms intensify under climate change. These include protection against storms and erosion and relief from heat. In addition, natural assets maintained to mitigate and adapt to climate change can also provide other benefits. These may include pollution treatment, habitat for imperiled biodiversity, pollinators for crops, and a host of others. Table 2 lists some NbS, the types of ecosystems that provide them, and the nature of benefits provided. The second column from the left is color-coded from dark ("more natural") to lighter ("more intensively managed) green to represent a qualitative assessment of the "extent of modification".

The economic values estimated for the services afforded by NbS may vary greatly, depending on the nature of the service, the system providing it, and the characteristics of the communities benefiting from them, as well as the methods employed in estimation. While they do not provide comprehensive estimates of the co-benefits that might arise from the NbS for climate mitigation they consider, Griscom, et al. (2017), report wastewater treatment values of between 785 and 34,700 USD per hectare. Similar ranges have been found for other services. To give a few examples from among many, Costanza et al. (2008) report coastal storm protection values on the East Coast of the United States ranging between 250 and 51,000 USD per hectare per year, while Huxham et al. (2015) cite figures ranging from hundreds to ten thousand or more USD for coastal protection in Southeast Asia and Africa. Ricketts and Lonsdorf (2013) estimate pollination values of up to 700 USD per hectare of forest preserved in agricultural regions of Costa Rica. The co-benefits arising from NbS adopted primarily for

Table 2

| System | Extent of Modification | Nature-based solution | Societal benefit |
|---|---|---|---|
| Forest | | Preservation | Carbon sequestration in biomass in vegetation and soils; biodiversity protection; flooding, drought, and erosion protection, recreation and tourism, water infiltration and storage |
| | | Restoration | |
| | | Enhanced management for woodfuel harvest | Carbon sequestration in biomass in vegetation and soils; provision of fuel and forest products to local users; flooding, drought, and erosion protection |
| | | Production | Carbon sequestration in standing biomass and harvested products; sustainable income; water infiltration and storage; reduced pressure on natural forests |
| Grassland / Savanna | | Preservation | Carbon sequestration in biomass in vegetation and soils; biodiversity protection; slope stabilization |
| | | Restoration | |
| | | Grazing management | Carbon sequestration in biomass in vegetation and soils; slope stabilization |
| Coastal/ riparian | | Preservation | Protecting lives and property from storms and flooding; carbon sequestration; enhancement of biodiversity and fisheries production |
| | | Restoration | |
| | | Maintenance of slope vegetation | Reduced erosion and slope stabilization |
| | | Maintenance of coastal, floodplain and riverine vegetation | Protecting lives and property from storms and flooding; carbon sequestration |
| Agriculture | | Agroforestry | Carbon sequestration in soils and biomass; reduced erosion; maintenance of soil fertility; pollinator habitat; storm protection; shading |
| | | Reduce tillage and carbon restoration practices | Carbon sequestration in soils; maintenance of soil fertility |
| | | Agricultural intensification | Enhanced food security; reduced pressure for conversion of other areas. |
| Urban | | Urban forests and green spaces | Carbon sequestration in biomass in vegetation; shading; stormwater disposal and flood protection; recreation |
| | | Green roofs | Cooling; stormwater control; pollution reduction; carbon sequestration |

climate mitigation may substantially defray their cost (just as the climate mitigation co-benefits of NbS adopted primarily for climate *adaptation* or other reasons may substantially defray *their* costs). Many of these co-benefits will vary dramatically with the scale of deployment, however.

Just as the co-benefits of NbS adopted primarily to mitigate climate change may differ across space and with the social and ecological context in which they are generated, the climate change mitigation benefits of NbS often vary across time. The right-most column of Table 1 indicates the number of years before the climate change mitigation benefits reach saturation. Plants and animals store carbon while they are growing, but a mature ecosystem nears a steady state in which its carbon storage and releases are in balance. The time required to reach such a steady state varies from as little as 20 years in restored peatlands to a century or more in pastoral systems that continue to build up soil carbon. Again, it is interesting to note that the tropical forests that may provide the best hope for rapid sequestration are also among the options that saturate most quickly, in as little as 25 years.

Moreover, Griscom, et al. (2017) constrain their estimates of the effectiveness of NbS for climate mitigation by restrictions that not only food production, but also biodiversity not be reduced by their implementation. The authors find that greater quantities of carbon might be stored by adopting more intensive land use practices in some areas. Not all "natural" approaches to environmental protection may protect broad assemblages of "nature".

The message of work such as that of Griscom, et al., is mixed. It is certainly true that NbS afford some exceptional bargains in climate mitigation, especially when the many additional benefits NbS provide in terms of climate adaptation and ancillary ecosystem services are also factored in. By the same token, however, not all NbS for climate mitigation provide large or, in some instances, even positive, benefits in other dimensions. It is also important to consider both the temporal and spatial patterns over which NbS benefits are provided.

This report will consider the benefits and costs of NbS for climate mitigation, adaptation, and other ecosystem services. The ways in which benefits and costs differ with temporal and spatial scale will also be evaluated. It is also important to consider not simply the monetary or physical magnitude of effects, but also their distribution over the segments of society they affect. All of these concerns will be addressed in this report, but it will be useful to start first by clarifying the use of some terms and concepts.

## 1.2 Definitions and context

Nature-based solutions are being explored by international organizations such as the International Union for the Conservation of Nature (IUCN 2019) World Bank (MacKinnon and Sobrevila 2008), European Commission (EC, n.d.), and United Nations Environmental Program (Kumar 2019), as well as private actors such as the Nature Conservancy (Conti 2019). Literature surveys have documented an explosion of work on NbS and its component elements in the past three decades (Cohen-Shacham et al. 2019; see also Minx et al. 2018).

The current interest in NbS follows in the tradition of efforts directed toward interdisciplinary studies of biodiversity (Pearce and Moran 1994; Wilson 1988), motivating the preservation of ecosystem services (Daily 1997; MEA 2005; TEEB 2009), appreciating nature's contributions to people (Díaz et al. 2018), ecosystem-based adaptation (Reid et al. 2019), and a host of similar concepts (Cohen-Shacham et al. 2019 survey a number of related approaches).

The coining of a new term that might encompass a number of existing approaches has occasioned some skepticism. Does the focus on NbS represent a step forward, or simply a repackaging of familiar concepts under what advocates hope will prove a more effective slogan? Even a sympathetic editorial in *Nature* on the emergence of the term characterizes Nature-based Solutions as the "latest green jargon". The editorialist argues NbS are intended both to encompass and improve upon terms such as "ecosystem services," "natural capital," and "green infrastructure," which in the writer's judgment, "set few hearts aflutter" (*Nature* 2017).

Other commentators ask if NbS are:

> intended to re-package the demand for sustainable development and nature conservation in a way *biodiversity* and *ecosystem services* do not? Does it represent an approach to policy and management distinctly different than those already being

applied? It is not altogether clear that it does. (Potschin et al. 2016; *emphases in original*)

Whether NbS represent an original development or simply a repackaging of existing approaches, the new designation may differ in ambition and scale from previous efforts. A number of authors have characterized NbS as an “umbrella” concept encompassing a variety of other approaches (see, e. g, Lafortezza et al. 2018; Nesshöver et al. 2017; Walters 2016). The broader a concept is, however, the more difficult it may be not only to define it, but also to operationalize it in ways that advance some objectives without compromising the achievement of others.

A number of definitions have been proposed for NbS (see, e. g., Cohen-Shacham et al. 2019; Nesshöver et al. 2017; see also Sekulova and Anguelovski 2017; Maes and Jacobs 2017). The International Union for the Conservation of Nature (IUCN) defines Nature-Based Solutions as “actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits” (Walters 2016). The European Commission defines them as “solutions to societal challenges that are inspired and supported by nature, which are cost-effective, provide simultaneous environmental, social and economic benefits, and help build resilience” (EC n.d.). For The Nature Conservancy, NbS are “project solutions that are motivated and supported by nature and that may also offer environmental, economic, and social benefits, while increasing resiliency” (Conti 2019).

The terminology of nature-based services gives rise to a couple of fundamental questions. The first is “How ‘natural’ must a solution be if it is deemed ‘nature-based’?” (Nesshöver et al. 2017; see also Sekulova and Anguelovski 2017) write that

> A central challenge for an ‘umbrella concept’ like NBS . . . is where to draw the line as to what is considered as ‘nature’ or ‘natural’. Many interventions may involve specific uses or manipulations of organisms and ecosystem processes; hence requiring decisions about acceptable levels of human intervention.

The definitions proposed do not offer clear guidance. The IUCN refers to “natural *or modified* ecosystems”, the European Commission refers to “solutions . . . that are *inspired* and *supported* by nature”, and the Nature Conservancy echoes the latter in calling for “solutions that are *motivated* and *supported* by nature” [all emphases added]. Nesshöver et al. (2017) ask whether genetically modified organisms would qualify as NbS? Would systems that *mimic* nature without any longer being *derived from* natural sources? How about oil palm plantations? Wind or solar power installations (Walters 2016)? Such questions are not just hypothetical. Among existing practices that have been described by some authors as NbS are breeding of new crop varieties (Reid et al. 2019), dredging and damming to create artificial lakes, selective planting and management in areas designated for “rewilding” to prevent an excess of trees from depleting water (Keesstra et al. 2018), and planting non-native species to conserve water use in landscaping (Conti 2019).

A second fundamental question concerns the problems to which nature-based approaches are “solutions”. NbS are proposed for climate change mitigation and adaptation and the preservation of biodiversity, and it may seem that there should be substantial synergies to be exploited by addressing climate and biodiversity goals by maintaining or restoring natural systems (Maes and Jacobs 2017; Walters 2016). One of the recurrent themes that arises in reviewing the literature, however, is that tradeoffs between the achievement of these objectives may be as common as synergies. Some ecosystems may sequester more carbon at the expense of reduced protection against climate change impacts or preservation of indigenous biodiversity. Others may have the opposite effect. Moreover, effects often differ across temporal and spatial scales, More carbon will be sequestered in a growing than in a mature ecosystem, for example, while the provision of ecosystem services such as groundwater recharge, pollination, and storm protection may be best provided by ecosystems that have matured to the point that they approach a steady state.

Just as there may be intertemporal tradeoffs, there may be tradeoffs across geographical locations. Native biodiversity may be suppressed in areas devoted to the most profitable agricultural or forestry production, for example, but the output provided by such intensively managed lands may be high enough to obviate the need to convert more areas from native forests, grasslands, or wetlands for production elsewhere (Martin et al. 2018).

Neither of the two fundamental questions has an easy answer. Perhaps the best approach to deciding what qualifies as "natural" is to put the question in relative rather than absolute terms.[4] Maes and Jacobs (2017) suggest distinguishing a NbS as a "*transition* to a use of ecosystem services with *decreased* input of non-renewable natural capital and *increased* investment in renewable natural processes" [emphases added]. Eggermont et al. (2015) make a similar point, suggesting that NbS be characterized by, among other attributes, the extent of human intervention in the structure and function of the ecosystems providing them.

An NbS might, then, be defined as an approach that creates or maintains systems that are "more natural" in some quantifiable sense than they are under the *status quo* or would be under a "business as usual" projection, or are "more natural" than whatever alternative measures might be enacted to address a need. The latter comparison is often expressed as one between "green" and "grey infrastructures" (Wild, Henneberry, and Gill 2017; Lafortezza et al. 2018; Cohen-Shacham et al. 2019; Seddon et al. 2019; Walters 2016).

The degree to which a particular approach solves a particular set of problems will depend on the weights an observer puts on the different elements that comprise the problems. Economics provides one prescription for determining these weights. This paper will be largely devoted to the economic analysis of NbS and will turn next to a review of a number of applications of such analysis. The literature on the economic analysis of NbS is vast and varied, with the variation extending over the thematic and geographical areas to which it has been applied, the methods employed, and the specificity and credibility of the analyses. As a comprehensive survey is not possible here, examples are chosen to illustrate common topics of application and important conceptional points.

---

[4] This may be particularly true in that archaeological evidence is increasingly demonstrating that many landscapes long presumed to have been pristine were, in fact, under human cultivation and management before the depopulation induced by epidemic diseases introduced by European colonists (Mann 2006).

## 2. Economic analysis of nature-based solutions

Nature-based solutions are diverse, so it would be difficult to fully catalogue all of the possible objectives to which they might be directed. Three general categories might be proposed, however:

- Mitigation of climate change, largely through the sequestration of carbon by embodiment in natural organisms or their products.
- Adaptation to the effects of climate change, by providing protection against sea level rise, more powerful and frequent storms, higher temperatures, and more severe droughts.
- Provision of a host of other ecosystem services, such as pollination, wastewater treatment, and groundwater recharge, as a result of maintaining or restoring natural systems.

While the second and third categories are distinguished by whether the services provided under NbS are rendered more valuable by changes in climate, the modes of analysis are similar between climate adaptation and provision of other ecosystem services. It is convenient, then, to group the adaptation and other ecosystem service provision categories together.

This division between the two types of NbS makes sense because they are characterized by different physical and economic considerations. These considerations are summarized in Table 3. The value of NbS for climate mitigation, unlike that of adaptation, is largely independent of where the mitigation is performed. Generally speaking, economic value reflects relative scarcity, and scarcity depends crucially on context. Something that is relatively abundant, and hence of little incremental value at one time and place, may be relatively scarce, and hence of substantial economic value, at another. Carbon sequestration is an exception to this rule. Because Earth's climate depends on the chemical composition of its atmosphere as a whole, the insertion of atmospheric carbon dioxide anywhere around the world has essentially the same effect as would additional $CO_2$ emissions anywhere else. This is in contrast to climate adaptation benefits and the provision of other ecosystem services, whose values depend critically on where they are provided and the areas and populations to which they are provided.

## Table 3

### Characteristics of nature-based solutions for climate mitigation vs. adaptation

| | Mitigation | Adaptation |
|---|---|---|
| Marginal value of service | Elastic: changes in provision of mitigation are unlikely greatly to affect the social cost of carbon | Generally inelastic: diminishing returns may arise even at modest local scale. |
| Geographical context | Largely irrelevant with regard to benefits: $CO_2$ emissions and mitigation has essentially the same effect regardless of where in the world they occur.<br><br>Opportunity costs of sequestration may differ dramatically from place to place. | Context specific: protection afforded depends on lives and assets at risk in the area protected, and opportunity costs of ecological preservation or restoration also vary greatly. |
| Time scale | Transient: May store carbon at relatively rapid rates during periods of growth but, unless harvested and renewed, may reach saturation quickly. | Durable: Services provided may be minimal during early periods, but once established, may provide services indefinitely |

A second distinguishing feature of NbS for climate mitigation concerns diminishing returns. Economic value arises from scarcity, scarcity is related to context, and generally speaking as the quantity in which something is being provided increases, its marginal value declines. This is also true of climate mitigation and reductions in the emissions – or, equivalently, increases in the sequestration – of greenhouse gases. However, the scale of the climate change problem is so large that the effects of even relatively large NbS may have little effect on the "social cost of carbon" – the value attached to incremental reductions (Stiglitz et al. 2017b; Dietz and Stern 2015; USIWG 2010). Mitigation through NbS may be an important component of an overall climate strategy, but must be only one element of a broader suite of actions (Griscom et al. 2019; Creutzig et al. 2019). This may be in marked contrast to a variety of ecosystem services associated with climate adaptation, such as flood prevention, shading, and coastal protection, as well as other services such as pollution treatment and pollination afforded by areas of natural habitat, which may quickly exhibit diminishing returns, or be obviated entirely in some instances by artificial interventions.

The opportunity costs of habitat preservation or restoration also vary greatly between different locations. This is, in fact, one of the most attractive aspects of forest preservation and re- and afforestation programs for climate mitigation: substantial variations in opportunity costs between locations constitute opportunities for more cost-effective allocation of the burden of reductions. At the same time, however, more intensive management of landscapes for carbon reductions may be necessary in areas with higher opportunity costs (Favero, Mendelsohn, and Sohngen 2017).

Finally, while NbS for climate mitigation may not exhibit quickly diminishing returns in the physical scale of the area providing them, they may exhibit markedly declining effectiveness over time. The ability of an area of, say, restored coastal forest to provide services such as storm protection may grow slowly over time, and reach its maximum when the forest reaches a mature state after years or decades of growth. Once such an ecosystem has reached its mature steady state, however, it may sustainably provide coastal protection services indefinitely. In contrast, a mature forest or other ecosystem may store little, if any, additional carbon (Seddon et al. 2019; Houghton and Nassikas 2018; Hausfather 2018). Moreover, more mature systems may be more likely to release stored carbon, either rapidly in fires, or more gradually with the death and decomposition of older organisms (Baldocchi and Penuelas 2019).

The sometimes long wait to realize the full potential of mature systems to adapt to threats arising from climate change or to provide other NbS may make their economics less favorable in relation to artificial alternatives (Reid et al. 2019). In contrast, NbS for climate mitigation may be valuable precisely because they are changing as they grow; biological growth is closely linked with carbon sequestration (Garzuglia and Saket 2005).

## 2.1 Climate mitigation

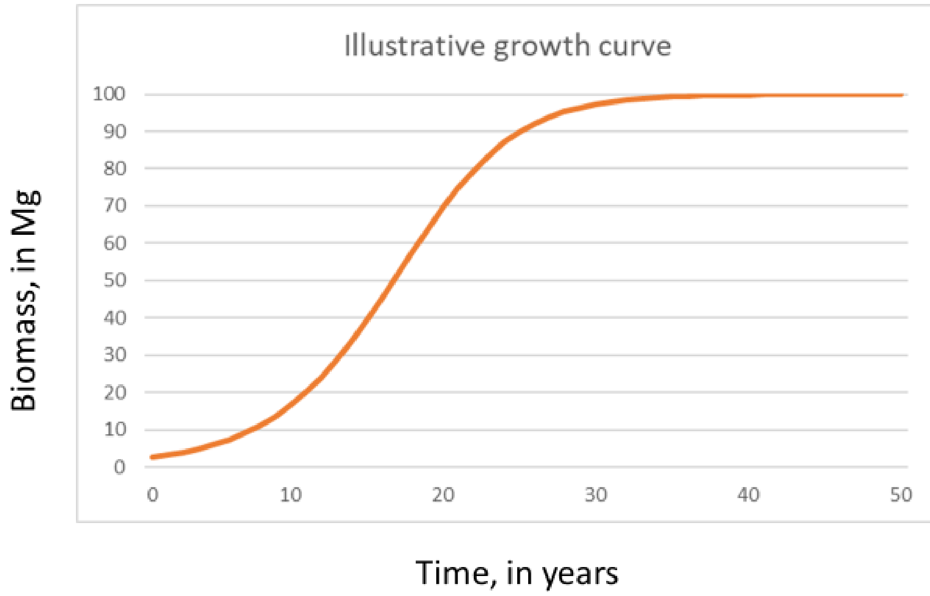
While nature-based solutions have been proposed for a number of problems, they have received a great deal of attention as potential solutions to climate change. NbS featured prominently at the 2019 COP 25 meeting in Madrid (IUCN 2020), have motivated influential contributions to the academic and policy literatures (Seddon et al. 2020; Kumar 2019; Seddon et al. 2019; Walters 2016; Cohen-Shacham et al. 2019), and have motivated major initiatives for their implementation (Monbiot 2020; UNFCCC 2019; IUCN 2020).

Figure 1 shows a growth curve for a hypothetical forest area. While biomass in such an area might grow at approximately exponential rates in the first few years after a forest is established, after 25 years this hypothetical area would have reached almost 90 percent of its long-term carbon storage potential (recall Table 1, summarizing Griscom et al. 2017's findings, and noting that some tropical forests can reach saturation of their carbon storage potential on this time scale).[5]

The possibility that the climate mitigation benefits of NbS would decline as the natural systems providing them mature raises a number of issues in the evaluation of NbS for mitigation. Three different approaches to NbS for climate mitigation are considered next: maintaining existing natural areas, growing or regrowing new ones, and integrating carbon storage in the management of working landscapes.

[5] Figure 1 adopts the commonly employed logistic growth function by which biomass in year $t + 1$, $B_{t+1}$, is assumed to be $B_t + gB_t(1 - B_t/B_\square)$, where $g$ is a parameter representing the rate at which biomass accumulates at very low levels, and $B_\square$ is the maximum level of accumulation, that will be approached over many, many years. For this example $g$ is taken to be 0.25, $B_\square$ 100 Mg, and it is assumed that the initial stock of biomass in year zero is 2.5 Mg. Different assumptions will, of course, generate curves of different shapes, but all will share the general sigmoid ("*S*-shaped") form of Figure 1, characterized by relatively rapid initial growth followed by progressively slower growth as the upper limit is approached.

Figure 1




### 2.1.1 Maintenance

About half a million Tg of carbon is maintained in earth's vegetation; most of this is stored in forests (Garzuglia and Saket 2005). This forest carbon stock represents close to 50 years' worth of $CO_2$ emissions at current levels.

Maintaining mature natural forests or other ecosystems without disturbance prevents the release of carbon dioxide from their vegetative biomass and, perhaps equally or more importantly, it retains carbon in their soils (Seddon et al. 2019; Garzuglia and Saket 2005). Even if such areas were eventually to be devoted to the alternative practices described below after they had been cleared, the carbon released in the process of the initial forest clearing could incur a "carbon debt" that could take a century or more to offset (Fargione et al. 2008; Turner et al. 2018)

The economic value of carbon stored in existing forests or other ecosystems may be substantial. Houghton and Nassikas (2018) provide estimates of the amount of carbon stored in various types of forest ecosystems (see Table 4). By multiplying these figures by a value of 100 USD per ton of $CO_{2e}$ a rough estimate may be derived of the value that would be *lost* if a hectare of forest of a given type were cleared and the carbon it stores released to the environment (see Table 4 for details of calculations). Such values can be substantial; the carbon storage potential of existing forests exceeds 50,000 USD per hectare in many forest biomes.

Timber values alone would generally not justify clearing mature forest land. While there is significant geographical variation, a rough estimate of the merchantable timber volume from a hectare of forest land might be on the order of 100 cubic meters per hectare (Garzuglia and Saket 2005). In order, then, to justify clearing land valued at 50,000 USD per hectare for its carbon storage, timber would have to command a price of at least 500 USD per cubic meter. Reported stumpage prices (that is, the price for lumber *in situ*, before costs of harvesting, transporting, milling, etc., are incurred) are generally considerably less than $100 per cubic meter (Hancock Timber Resources Group 2017).

Table 4

| Latitude | Forest type | Median carbon density of primary vegetation (MgC/ha) | Carbon density of undisturbed soils | Value of carbon storage per hectare (USD) |
|---|---|---|---|---|
| Tropical | Rain | 190 | 120 | 84,300 |
| | Moist deciduous | 78 | 100 | 40,700 |
| | Dry | 39 | 40 | 19,100 |
| | Mountain | 62 | 75 | 31,900 |
| Subtropical | Humid | 148 | 120 | 68,900 |
| | Dry | 57 | 80 | 30,800 |
| | Mountain | 80 | 120 | 44,000 |
| Temperate | Oceanic | 252 | 220 | 119,200 |

| | | | | |
|---|---|---|---|---|
| | Continental | 150 | 200 | 79,600 |
| | Mountain | 101 | 150 | 55,400 |
| Boreal | Coniferous | 67 | 206 | 49,900 |
| | Mountain | 46 | 206 | 42,200 |

Source: (Houghton and Nassikas 2018) and author's calculations based on values of 100 USD per Mg $CO_{2e}$, and assuming all primary vegetation is removed and one-third of soil carbon released.

To the extent that timber harvests do continue to occur in areas of mature forest, it is because the climate benefits of carbon sequestration are economic externalities in the absence of effective regulation or carbon pricing (see, e. g., Lin et al. 2013). Carbon sequestration benefits accrue to people around the globe in the form of forestalling climate change, while the opportunity costs of providing them, forgoing timber harvest and alternative land use, are borne by the much smaller groups of people, particularly the most vulnerable groups including women and girls, on whose land forests stand.

Lands cleared of trees might also be put to other purposes: to food production or residential use. Most forest clearing is being done at the "extensive margin," however; that is, where the private economic benefits of land clearing just balance the private costs of conversion to alternative use. Inasmuch as there are much higher *societal* costs of lost carbon storage, again it seems likely that the economic case for preserving mature forests is strong.

The extensive margin may be marching outward in parts of the world, however, and over time the balance perceived between the benefits of preservation and the needs of development could change. If human populations continue to grow, their tastes in food do not change significantly, and agricultural productivity does not grow enough to compensate, the opportunity cost of maintaining natural forest land may increase over time. Conversely, if these factors move in the other direction, more forests may be allowed to evolve to maturity (and, in fact, in parts of the world, they are; see Houghton and Nassikas 2018; Rudel et al. 2005).

Some other issues and values associated with retaining mature forests might also be flagged. There are also other benefits associated with maintaining natural ecosystems. The climate adaptation – as distinct from mitigation – benefits of maintaining natural areas will be considered below, but it bears mentioning that the same systems may both prevent the occurrence of still more dramatic climate change than might already be expected to occur and provide some protection against some of the effects of the climate change that does occur. Similarly, areas of extant ecosystems provide a suite of ancillary services that may be valuable. These other services will also be considered below. To preview that discussion, location and context are critical in the determination of many ecosystem services values. The services provided by ostensibly similar biological assets may have very different values in one place than they do in another.

Large areas of extant natural habitat often also have substantial biodiversity conservation values. Since at least the time of the pioneering ecological work of MacArthur and Wilson (1967; see also the retrospective essays in Losos and Ricklefs 2009), it has been well-established that larger areas of habitat shelter more species diversity. The relationship between area and species diversity is generally found to be highly non-linear, with species numbers increasing less-than-proportionately with the area set aside to harbor them.[6] Larger expanses of habitat will also support greater diversity because large predators need large prey populations (Terborgh 1999). Interconnected areas of diverse habitat may also facilitate seasonal movements of migratory species, as well as providing conduits for movement in response to climate change.

Preserved natural habitats may, then, also generate important conservation benefits by protecting biodiversity. There may also be a virtuous feedback loop; evidence suggests that systems with greater diversity are more biologically productive (Tilman, Isbelll, and Cowles 2014). Recently researchers have suggested a related link: that the presence of large

---

[6] A related point is that even if habitats are fragmented into unconnected parcels, the larger the number of parcels there are, the more likely it is that a population of any given particular species will find a refuge in at least one (Camm et al. 2002). Again, however the expected number of species protected increases less-than-proportionately with the number of parcels set aside.

herbivores, including such “charismatic megafauna” as elephants and rhinoceroses, may promote carbon storage by seed distribution (Bello et al. 2015), fire suppression (Waldram, Bond, and Stock 2008), altering albedo (Cromsigt et al. 2018), and other mechanisms. This has led some to call for “trophic rewilding” – reestablishing the original suite of species on a landscape – as a carbon mitigation strategy (Cromsigt et al. 2018).

There may, then, be positive connections between species diversity in general, as well as the survival or reestablishment of particular species, and the amounts of carbon their biomes can store. It is less clear how quantitatively important such connections are. The natural science work has not advanced to the point that additional economic benefits of trophic rewilding can be estimated. Maintaining large, diverse, and connected parcels also has costs: they may impede transportation and large animals occasionally trample crops and people. An analyst would want to weigh such costs against benefits before deciding on a protected area strategy intended to store carbon and conserve biodiversity. While the *carbon* benefits of maintaining larger areas in native cover may scale roughly linearly with their size, it seems likely that the biodiversity benefits, including potential feedbacks to carbon storage, would be subject to more sharply diminishing returns.

### 2.1.2 Restoration

Since mature natural forests can store large quantities of carbon, another NbS for climate mitigation may be the regrowth of such systems. Houghton and Nassikas (2018) estimate that gross carbon accumulations in forests regrowing in former farming areas may exceed 4,000 Tg per year (there are *gross* figures, as forests are still being felled in some other areas). If the international community compensated landowners for this accumulation at the rate of 100 USD per ton of $CO_2$ that has been suggested by several commentators, the yearly income generated globally would be greater than the GDP of nations such as Indonesia and Mexico. Houghton and Nassikas (2018) estimate that some 130,000 Tg of carbon could eventually be stored in regrown forests.

Reforestation is one of the most often mentioned and advocated NbS (Lewis et al. 2019; UNFCCC 2019; Parrotta, Wildburger, and Mansourian 2012; Seddon et al. 2019; Griscom et al. 2017). In many areas reforestation is cost-competitive with alternative climate mitigation strategies (Lewis et al. 2019; Griscom et al. 2017).

There are some concerns that should be noted with restoration of natural ecosystems as an NbS for climate mitigation, however. The scale at which restoration would need to occur in order to make a significant contribution to climate mitigation could be immense (Baldocchi and Penuelas 2019; Griscom et al. 2017). The additional land area devoted to new or restored forests to achieve the 130,000 Tg of additional carbon storage Houghton and Nassikas (2018) suggest is feasible could comprise some five percent of dry land on earth – an area larger than India.[7]

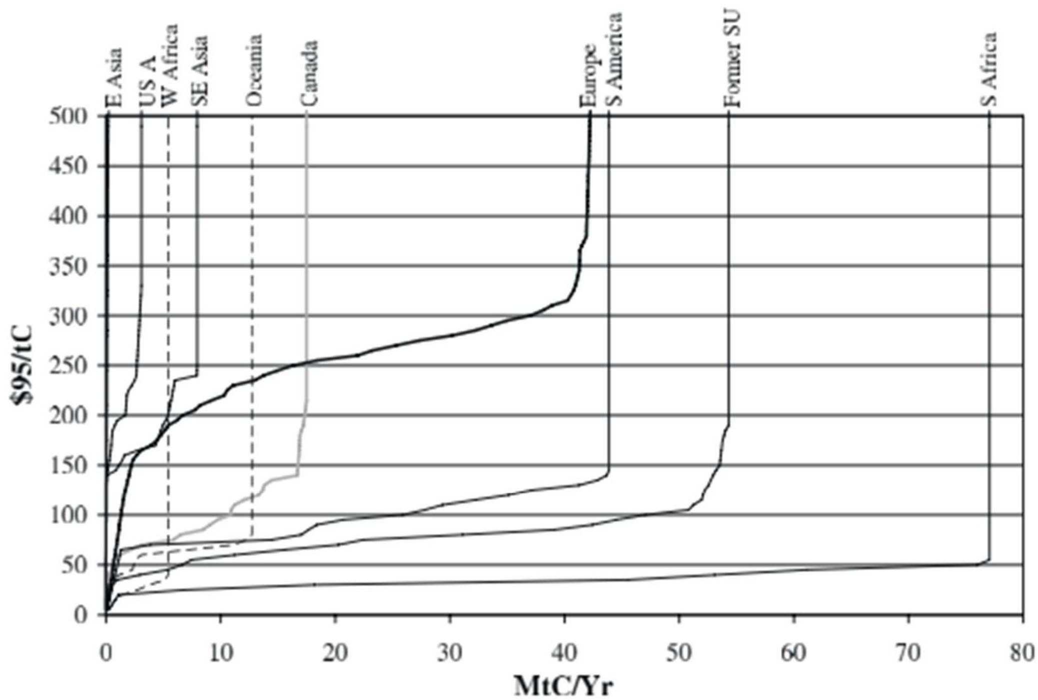
There could be a number of both economic and ecological limitations on a program on such a scale. As Griscom et al. (2017) have documented, something in excess of 2,250 Tg of $CO_2$ might be sequestered in forests at a cost of less than 10 USD per ton. Increasing the price to 100 USD per ton would increase the supply to about 7,320 Tg. The tenfold increase in price would elicit only a bit more than a tripling in supply. Figure 2 is reproduced by permission from Strengers, Van Minnen, and Eickhout (2008), which was the source for some of the estimates reported in Griscom et al. (2017). The figure shows that the supply of further carbon storage eventually becomes completely inelastic (i. e., no more carbon would be stored regardless of how high the price offered is) in some regions at prices not much higher than 100 USD.

While figures reported in Griscom, et al. (2017), and based on earlier work by Strengers, Van Minnen, and Eickhout (2008), have been emphasized, a variety of other studies, often employing different methods, offer similar depictions of forest carbon supply curves (Busch et al. 2019; Enkvist, Nauclér, and Rosander 2007). These studies generally show that some areas afford *some* opportunities for low-cost carbon sequestration, but that such opportunities are

[7] If we take 200 Mg per hectare as a generous figure for additional carbon stored in new or restored forest areas, 130,000 Tg/200 Mg/ha = 650 million ha. The land area of the Earth is about 14 billion ha. India's land area is about 328 million ha.

generally exhausted well before the storage potential they offer would offset a majority of global emissions: there is only a limited supply of land that could be reforested before the opportunity cost of diverting it from other uses becomes prohibitive.

*Figure 2*

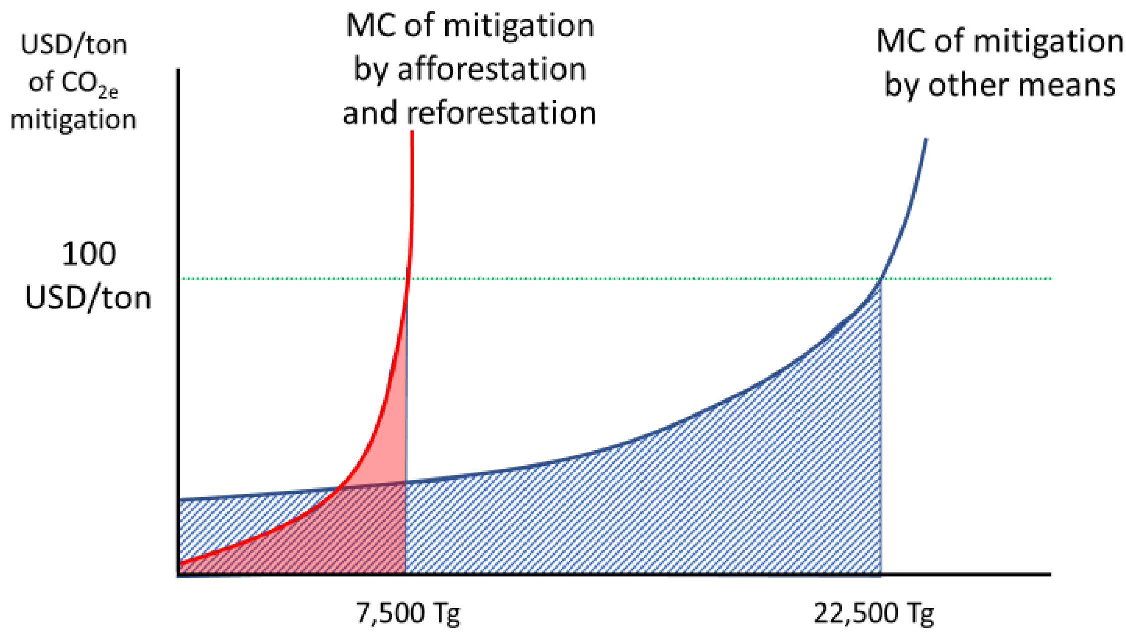



Source: Strengers, Van Minnen, and Eickhout 2008. © Springer Nature. Reprinted by kind permission from *Climatic Change*.

It is important to appreciate the implications of this finding. Some authors (see e. g. Monbiot 2020; Barbier et al. 2020) have noted that only a few percent of funds allocated for mitigating climate change have so far been devoted to NbS such as reforestation. Given the bargains some NbS present, it is natural to ask why more funding is not devoted to these low-cost options. While low-cost mitigation options afforded by NbS have not yet been exploited as broadly as they should be, it is also not clear that a large fraction of mitigation costs incurred ought to involve NbS. This point is made in Figure 3, where two hypothetical marginal cost curves are depicted. While the example is illustrative, the shape of the marginal cost curve for reforestation (the red curve) is a stylized representation of those found in the empirical studies cited above. Small amounts of climate mitigation through carbon storage may be achieved at relatively low costs, but beyond a certain point, the marginal cost of further mitigation increases very rapidly. The other, blue, curve, representing the marginal cost of mitigation by other means, is hypothetical, but might be thought of as a stylized representation of other mitigation options (see, e. g., (Enkvist, Nauclér, and Rosander 2007). The figure shows a situation in which the overall mitigation required to prevent drastic climate change, 30,000 Tg, could be most cost-effectively achieved by imposing a price on carbon emissions or, equivalently, offering a payments for carbon storage, of 100 USD per ton of $CO_{2e}$. Under this scenario 7,500 Tg of $CO_{2e}$ mitigation would be realized by afforestation and reforestation, with the remaining 22,500 Tg achieved by reducing other sources or increasing other sinks. The *total* cost of carbon mitigation through forestry (the area under the marginal cost curve for forestry practices shaded in red) is much lower than is the *total* cost of carbon mitigation through other means (the area under the marginal cost curve for other practices shaded in blue). This does not mean, however, that more reliance should be placed on forestry measures. A cost-effective division of mitigation is one in which the *marginal* costs of mitigation are equal across sources.

Another economic limitation on NbS for carbon storage is leakage. The author of one recent paper on NbS asks "Is there any reason why reducing deforestation in Brazil could not be pursued at the same time as regrowing forests in India or restoring mangroves in Indonesia?" (Turner 2018). While the question was likely intended to be rhetorical, an answer could be offered: when deforestation is reduced in Brazil, demand for the products no longer derived from Brazilian forests may be transferred to countries such as India or Indonesia. When economic production is constrained in one area, the price of the good produced increases in response to the reduction in supply, creating an incentive to expand production in other areas. Such "leakage" has been extensively studied in the environmental and resource economics literature (see, e. g., Parrotta, Wildburger, and Mansourian 2012; Murray et al. 2002; Bode et al. 2014; Hertel 2018; Wu 2000)

Figure 3




Leakage may be particularly relevant to discussions of the efficacy of restoration strategies. Re-establishing natural ecosystems in areas from which they had earlier been converted necessarily implies forgoing whatever use to which they had earlier been converted. In some instances, reforestation or other restoration may occur in response to changing economic circumstances under which production is some areas is no longer profitable. There is a substantial literature on "forest transitions" documenting and explaining why many parts of the world, such as the New England states of the U. S. (Rudel, Perez-Lugo, and Zichal 2000; Meyfroidt, Rudel, and Lambin 2010) have been reforested after land was initially cleared for farming. In many such instances the explanation has not been that the demand for new farmland has declined globally, but rather, that more fertile lands have been identified elsewhere (Mather 1997). There is, then, a danger that reclaiming land to restore natural ecosystems in some areas will displace land-clearing efforts to others.

Empirical estimates of the magnitude of leakage effects vary over place and time. Murray et al. (2002) found that for every ton of carbon sequestration achieved by providing incentives for forest owners in the U. S. in one place, between 100 and 900 kg of carbon were released as a result of land use change in another. Wu (2000), also working in the U. S., studied the Conservation Reserve Program (CRP), which compensated landowners for taking ecologically sensitive farmland out of production. For every five acres enrolled in the CRP, one hectare of land entered into production elsewhere. Recent results from Pendrill et al. (2019) find that for every three hectares reforested in the world, a hectare is lost in tropic or subtropical forests. A more pessimistic assessment is offered by Haya (2019), who reviews literature suggesting that forest carbon policies adopted in California could generate leakage of 80% or more elsewhere; every ton of carbon sequestration realized in California could induce more than 800 kg of carbon releases elsewhere. While the extent of leakage found varies considerably between circumstances and studies, the literature has established that leakage is an important concern and its effects can be substantial.

Leakage concerns could be obviated by instituting comprehensive global land use regulation which would restrict land clearing in one area in response to reforestation in another. International institutions for enacting such policies are lacking, though. Even if leakage could be prevented, the consequence would be that less land would be available for growing food and other purposes, with consequent effects on nutrition and poverty (see, e. g., Baldocchi and Penuelas 2019; Creutzig et al. 2019; Burns and Nicholson 2017). These effects could, in turn, be countered by deploying technologies to enhance agricultural yields (Tilman et al. 2011; Martin et al. 2018; Houghton and Nassikas 2018; Lin and Huang 2019) or appealing to consumers to make lifestyle changes, such as adopting less carnivorous diets (Bertram et al. 2018).

It is clear from these observations that a full consideration of the economic and societal implications of afforestation and reforestation policies would require a very comprehensive and detailed approach. Gender-sensitive approaches, for example, should be mainstreamed in NbS frameworks to ensure that specific issues that affect women and men are part of the analysis, project planning, and implementation of NbS projects (See Box 1 for further elaboration). Hertel 2018 discusses considerations in determining the complexity of requried modeling; Bertram et al. 2018 and Schmitz et al. 2012 are examples of complex computable general equilibrium models applied to issues of conservation and agricultural policy; Hertel, West, and Villoria 2019 review and compare a number of such models).

Ecological limitations also affect the efficacy of NbS for carbon storage. Growing more trees may necessitate increased diversion and use of water in the production of biomass (Burns and Nicholson 2017; Baldocchi and Penuelas 2019). Forest cover may also sometimes have counterintuitive local climate effects. The reduced albedo (reflection of incident radiation) of forests at higher latitudes relative to other forms of cover may increase local temperatures by more than the carbon dioxide removal they accomplish reduces them (Baldocchi and Penuelas 2019; Parrotta, Wildburger, and Mansourian 2012). Finally, natural storage of carbon has limitations, in that the rate of carbon retention decreases as the systems sequestering it mature, and mature systems may be susceptible to carbon re-release through fire or death and decomposition (Baldocchi and Penuelas 2019; Griscom et al. 2017). Fuss et al. (2018) suggest that afforestation and reforestation, along with increased soil carbon storage, may be " '21st century N[*egative*]E[*mission*]T[*echnologie*]s': promising stop-gaps, but limited in long-term potential".

The argument that (re-) establishing forests may be a temporary and insufficient stop-gap underscores another sometimes controversial point. One criticism of nature-based solutions for climate mitigation, as of a variety of negative emissions technologies and geoengineering approaches more broadly,[8] is that they may generate a moral hazard.[9] If growing forests promises to remove some of the carbon dioxide emitted from burning fossil fuels, the need to reduce dependence on fossil fuels may be seen as less urgent. Virtually all commentators recognize that, at best, NbS could reverse only a fraction of the anticipated accumulation of greenhouse gases in the atmosphere – and, by extension, of the anticipated increase in global

[8] Geoengineering may include injecting reflective aerosols into the upper atmosphere, putting shades or reflectors into earth orbit, fertilizing the oceans to accelerate algal growth, or direct capture of carbon dioxide from the atmosphere. Some NbS might be regarded as geoengineering under such a definition.

[9] "Moral hazard" occurs when an agent is less likely to take steps to avoid the occurrence of an event against which they are indemnified. A driver may not drive as cautiously, for example, if their insurance covers them for expenses incurred in the event of accident.

temperature – over the coming decades. NbS are, then, best seen as a part of a broader set of initiatives that must be undertaken to control climate change that should be implemented in conjunction with, rather than as a substitute for, emissions reductions (Baldocchi and Penuelas 2019; Anderson et al. 2019; Creutzig et al. 2019).

**Box 1: Gendered Lens Necessary for Successful NbS Analysis and Implementation**

When discussing the Economics of Nature-Based Solutions, it is critical to recognise discussion and analysis of economic empowerment and opportunity necessarily entails a gender dimension. A gender approach redefines an environmental situation through the lens of social relationships and their reflection in human-environment interactions, instead of defining the state of the environment primarily in its physical or ecological forms. Guidance and analysis of NbS policy and implementation would certainly be enriched by demonstrating the different pathways in which NBS initiatives can impact the livelihoods the men and women, and girls and boys within communities. There is an acknowledge lack of disaggregated data by gender on this topic and it is pertinent that this data gap be addressed when going forward with NbS framework development.

Gender parity should also be taken into account in all conservation and ecosystem rehabilitation activities, which form the crux of successful NbS implementation. Women hold key roles across society that influence how to produce, consume and market sustainable solutions and thus should be placed in central roles for decision-making on water availability considerations. Sustainable solutions call for the consideration of various stakeholders' needs and importantly, the communities that will be directly impacted based on their gendered needs and uses. There are synergistic gender approaches between NbS analysis and implementation and the Global Biodiversity Framework. These include:

- Enhance women's agency and promote their effective participation and leadership in biodiversity conservation;
- Promote and protect women's rights and access to and control of resources;
- Enhance and ensure equitable benefits and human well-being;
- Include a specific gender target;

Ultimately, mainstreaming gender in policy creation can improve both developmental and environmental outcomes. Programmes formed using the NbS approach that empower women in the forest sector are an essential prerequisite for building economies based on social justice and environmental improvement. To enhance this linkage, women's work in forestry (both paid and unpaid) should be captured in national statistics, as well as generally moving towards gender-disaggregated data for the forest sector, acknowledging women's specific and valuable experience-led knowledge, and achieving gender balance in decision-making in the relevant groups and associations.

### 2.1.3 Rotational harvesting and agriculture

Maintaining forests prevents the reserves of carbon stored in their biomass and soils from being released into the atmosphere. Reestablishing natural forests where they had once stood could result in substantial removals of carbon dioxide from the atmosphere. However, forests eventually reach a state in which they sequester little additional carbon and are at increasing risk of massive carbon release through burning. These observations beg the question of whether forests and other carbon-storing ecosystems should be allowed to grow to maturity or should, instead, be periodically harvested. If one of the main purposes for which an ecosystem is maintained or restored is to produce a product – in this case, "carbon storage" – it also raises the question of how intensively the ecosystem should be managed to maximize the production of the intended commodity at the expense of forgoing other possible benefits.

Dating from the pioneering work of Faustmann (1849) forestry economists have developed models of the optimal harvest rotation.[10] Hartman (1976) demonstrated how the Faustmann formula should be amended to account for ecological and other values conferred by standing forests. A number of authors have since considered how carbon sequestration values should be included in forest management policy (an important early contribution is van Kooten, Binkley, and Delcourt 1995; see also Hoel, Holtsmark, and Holtsmark 2014, Favero, Mendelsohn, and Sohngen 2017, and, for an authoritative survey of both economic and policy issues, van Kooten and Johnston 2016).

A crucial concern in the analysis of forest carbon storage is what van Kooten, Binkley, and Delcourt (1995) have dubbed the "pickling rate": the faction of carbon contained in harvested materials that may be permanently removed through storage. If this fraction is high enough, rotational harvesting might be preferred to a policy by which areas are re- or afforested and then preserved indefinitely. It is worth underscoring here that these considerations could prevent the realization of synergies between carbon storage and the provision of other

[10] Faustmann demonstrated that, in an environment with constant prices and unchanging biological conditions over time, trees should be harvested when their rate of growth is equal to the discount rate times the sum of the revenue from harvest and value of land.

ecosystem services that some commentators have emphasized (Maes and Jacobs 2017; Walters 2016). Such synergies might not be consistent with the realization of maximum value from carbon storage in combination with the provision of other ecosystem services, if realizing high values from the former is inconsistent with generating high values from the latter (Pannell et al. 2006; Fuss et al. 2018).

Harvested forest products might be embodied in slow-to-decay forms such as building materials, or burned for energy. Other plant-based materials might also be harvested for biofuels. When biofuels are burned they return carbon dioxide to the atmosphere. To the extent that biofuels replace fossil fuels, however, their advocates represent them as a "carbon neutral" energy source: carbon would be endlessly recycled from the atmosphere, to plant growth, to fuel, to the atmosphere, and back into plant growth. Biofuels have earned a poor reputation in some instances, however. Many commentators have noted that the carbon releases arising from initially felling natural forests or converting land from other uses may take decades to offset by subsequent displacement of fossil fuel combustion (Fargione et al. 2008). Moreover, the amount of carbon retained in the types of plantations that might be established to grow biofuels is far less than that in natural forests (Lewis et al. 2019). It seems reasonable, then, to conclude that existing forests should generally not be felled to grow energy crops (Turner et al. 2018).

Plantations might be more attractive, however, if they could be established in areas that are currently not forested. They would be still more attractive if carbon were not simply recycled from the atmosphere to biomass and then back to the atmosphere again via combustion or decomposition. Bioenergy with carbon capture and storage (BECCS) has been suggested as a superior approach. Despite the technology not yet having been demonstrated at scale (Hausfather 2018), BECCS now figures prominently in many plans for meeting Paris climate agreement goals (Burns and Nicholson 2017; Creutzig et al. 2019). Some authors suggest that, at different times and places, forests might both be maintained to sequester carbon in place and harvested to feed BECCS facilities (Favero, Mendelsohn, and Sohngen 2017).

The example of BECCS underscores the questions raised above concerning what constitutes a nature-based solution, and how the answer might depend on the scale of analysis. Biomass grown for energy and whose carbon content is destined for sequestration would be, essentially, an agricultural product. Many agricultural commodities are most profitably grown in intensive monocultures. It does not seem unreasonable to suppose that crops grown to provide biologically based energy from whose production and/or consumption carbon would be captured and stored would also be highly specialized. This may exacerbate other environmental and resource problems, such as water shortage and fertilizer use, as well as creating new pressures on food supply through competition for land and other inputs (Burns and Nicholson 2017; Minx et al. 2018; Dietz et al. 2018).

While planting monocultures of rapidly growing species is obviously not consistent with the maintenance of biodiversity, such practices may prove more profitable than less intensive management of more diverse landscapes (Lin et al. 2013; Pannell et al. 2006; Fuss et al. 2018).[11] The opportunity cost of sacrificing carbon storage potential in order to achieve other objectives such as sustaining biodiversity increases in the value attached to carbon storage relative to those other objectives. In the absence of additional payments explicitly for the preservation of biodiversity, landowners are likely to trade off diversity for greater carbon storage (Bryan et al. 2016). Similarly, the division of forestland between areas left uncut to sequester carbon in place and others devoted to rotational harvest for biofuels with carbon capture and storage will favor the latter over the former the greater is the value assigned to climate change mitigation (Favero, Mendelsohn, and Sohngen 2017).

BECCS contemplates the capture of carbon dioxide, either from combustion or in the process of fuel production, and its injection in permanent storage (Burns and Nicholson 2017; Turner et al. 2018). Carbon might also be stored in agricultural and other soils. Soil carbon sequestration (hereinafter, SCS) may be augmented through a variety of practices involving tillage, mulching,

---

[11] It is interesting to note one of the reasons Lin et al. (2013) cite for this conclusion: that it may be easier for certifying authorities to verify the carbon storage performance of monoculture plantations.

erosion control, fertilization, and farm management (Fuss et al. 2018; Keesstra et al. 2018; MacKinnon and Sobrevila 2008; Lal 2009; Ontl 2018).

One crucial distinction between BECCS and SCS are the capacity and permanence of the carbon storage they offer. Carbon dioxide stored in geologically stable formations might reasonably be regarded as permanent. Moreover, the storage capacity is large relative to the rate at which $CO_2$ might be generated from biofuel production and/or consumption. A potential drawback, though, is that the location of stable geological formations for storage may not be convenient to the areas in which biomass is harvested and $CO_2$ captured (Creutzig et al. 2019).

Conversely, local sources can be drawn upon to store carbon in SCS. The drawback in these cases are limitations on storage. While soil carbon uptake might be on the order of 500 kg of carbon per hectare of land per year in the early years of efforts to restore carbon to depleted soils (Lal 2009), soil storage potential might be reached in a matter of decades (Lal 2009; Fuss et al. 2018). SCS may, then, be among the bridging strategies that could be effective in mitigating atmospheric $CO_2$ accumulation while emissions from other sources are being reduced, but not available indefinitely (Minx et al. 2018).

If the figure of 500 kg C $ha^{-1}$ $yr^{-1}$ is adopted for potential carbon uptake by SCS, then applying a price of 100 USD per ton of $CO_2$ and using the factor 3.67 to convert carbon to carbon dioxide, annual payments of about 185 USD per hectare of land on which SCS practices are being implemented would be justified. Some authors question whether payments should be needed at all to motivate farmers to adopt SCS; SCS is sometimes given as an example of a "negative cost" intervention (Minx et al. 2018). Increasing soil carbon content provides a number of agricultural benefits, including increased retention of water and fertilizers, enhanced diversity of beneficial soil organisms, and reduced erosion (Lal 2009). These may result in both increased crop yields and reduced costs of irrigation and fertilization. Moreover, over and above the global benefits resulting from reduced atmospheric carbon dioxide, SCS may also confer local environmental benefits by reducing eutrophication associated with fertilizer runoff and downstream deposition of eroded soils (Lal 2009).

The leakage effects of measures intended to increase carbon storage in agricultural soils may be difficult to predict. Under some circumstances, they might generate “negative leakage”. If SCS practices could be instituted at negative costs, it might imply that the same amount of food could be grown on less land, freeing up more areas for reforestation or restoration to other native ecosystems. This may not be as simple as it first appears, however. Negative cost practices would also make agricultural products cheaper. This could potentially induce a “rebound effect” through increased demand that could, in theory, lead to even more demand for agricultural land. While such perverse results may be unlikely, the size and direction of actual effects is an empirical question (see, e. g., Byerlee, Stevenson, and Villoria 2014).

Substitution of services provided by preserved landscape features, such as hedgerows or trees interplanted with crops in agroforestry, might both reduce the costs of other purchased inputs and reduce yield per unit of total land area devoted to agricultural systems (Simpson 2014). This could induce an expansion of overall land area in farms, although the diversity supported within farms might be higher. Finally, if policies imposed to increase soil carbon sequestration on farms increase both costs and productivity, but make farming less profitable overall, the net effect could be to induce leakage to other areas not covered by such policies.

While many of the environmental effects of agricultural measures adopted to increase soil carbon storage are beneficial, there are also some potential drawbacks. Increasing carbon storage in soils requires that adequate sources of other nutrients be available. Adding a ton of carbon to soils requires the addition of 80 kg of nitrogen, as well as substantial quantities of phosphorus and potassium (Fuss et al. 2018). This may result in water pollution from runoff of fertilizers that are not taken up. Carbon-rich soils may also be the source of methane and nitrous oxide, both potent greenhouse gases, though both might be minimized with appropriate management (Fuss et al. 2018). Finally, storage of carbon in soils, much like in mature forests, can be precarious. Land disturbances can results in carbon that had built up in soils over decades being released on a much shorter time scale (Fuss et al. 2018).

## 2.2 Climate adaptation and other ecosystem services

Efforts to reduce greenhouse gas emissions and sequester carbon dioxide already in the atmosphere may serve to slow the occurrence and reduce the ultimate severity of climate change.  Regrettably, however, the concentrations of greenhouse gases in the atmosphere likely commit the planet to substantial warming regardless of mitigation measures that may be taken.

NbS have, then, been proposed not only to mitigate climate change, but also to adapt to it (Keesstra et al. 2018; Griscom et al. 2019; Seddon et al. 2019; IUCN 2020).[12] A changing climate is predicted to have many effects including rising sea levels, increased frequency and severity of tropical storms, more intense heat and droughts, changes in seasonal precipitation patterns, and, in consequence of these factors, greater erosion and increased likelihood of landslides.

There has been a great deal of interest in the role of NbS in adapting to these consequences of climate change, and many studies have been conducted of them.  In addition, there is a parallel literature on a variety of other services provided by preserved or restored natural ecosystems. Examples include pollution treatment, groundwater recharge, and pollination.  As the economic analysis of such ancillary ecosystem services is similar in many respects to that of nature-based solutions for climate adaptation, they will be treated together here.

The range of climate change and other challenges that could be addressed with nature-based solutions is too broad to attempt a comprehensive categorization here, but a number of examples will be presented to give a sense of the modes of analysis that might be employed in their evaluation and the challenges that arise in employing them.

---

[12] The term "ecosystem-based adaptation" has been used to describe many of these measures Seddon et al., (n.d.) and Reid et al. (2019) adopt the term in their titles, and the keywords "ecosystem-based management" or "ecosystem-based adaptation" appear in many other papers.

### 2.2.1 Coastal protection

Both tropical and temperate regions may benefit from the coastal protection afforded by forests, wetlands, or other vegetated areas. These values have been the subject of considerable attention. With rising sea levels and more powerful storms predicted to occur as the climate warms, coastal protection values may become increasingly important. A number of researchers have estimated in both wealthy and developing countries (see, e. g., Barbier et al. 2008; Huxham et al. 2015; Das and Vincent 2009).

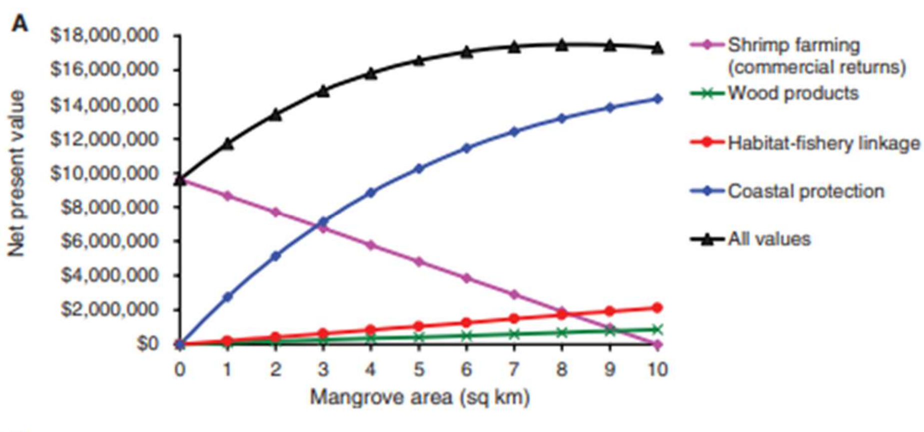
A widely cited report (Narayan et al. 2017) estimates that coastal wetlands prevented US$625 million in property damage from flooding when Hurricane Sandy struck the northeastern United States in 2012. While this is figure is substantial, it does not necessarily tell an analyst enough to make an informed choice about how large an area of coastal wetlands should be preserved in a more natural condition to protect against storms. That decision would need to be informed by a few additional factors. One is the opportunity cost of the land. While coastal habitats may protect against storms, they are also often extremely valuable for real estate development, even if they are at greater risk of damage. Another factor concerns the likelihood that property would be at risk from a storm. Total property losses in Hurricane Sandy were about US$ 50 billion, but Hurricane Sandy was, by historical standards, an unusual event.[13] The value of property protected in any given storm event would have to be weighted by the probability of occurrence of such an event. An earlier study conducted such an analysis and estimated an average value of about US$ 33,000 per hectare of coastal wetland maintained (Costanza et al. 2008).

This approach is also not fully satisfactory, though, as it did not relate expected losses to the area of coastal areas maintained, and so did not provide information with which to calculate incremental values. This latter information may need to be derived from natural science models. Barbier et al. (2008) review some relevant models. Their findings underscore an

[13] Of course a concern that arises with climate change is that historical patterns may not provide very useful guidance as to future events; the frequency and severity of storms may increase, and sea-level rise may make coastal areas more vulnerable.

important principle:  while coastal wetlands and forests can provide substantial protection against storms, the value of such protections exhibits diminishing returns in the area devoted to providing it.  Barbier et al. (2008) considered both benefits and costs of maintaining coastal wetlands.  While the coastal protection afforded by coastal mangrove forests was valuable in the region of Thailand they used as an example, the value varied nonlinearly with area of mangroves maintained.  The value of coastal vegetation in providing *some* coastal protection exceeded the opportunity cost of forgoing shrimp farming in ponds established in areas cleared of mangroves.  The coastal forests provided not only storm protection, but also fishery and local wood product benefits. When a strip of forest about 800 m in width is maintained , however, marginal benefits of widening the strip further were not sufficient to compensate for the lost value from forgone shrimp ponds (see Figure 4).

*Figure 4*




Source:  (Barbier et al. 2008).  Reproduced from *Science* by kind permission of American Association for the Advancement of Science.

Ideally, a researcher would have the information with which to estimate relationships such as those illustrated above.  This can be time-consuming and expensive, however.  Consequently, researchers sometimes resort to the expedient of estimating the value of storm protection services by the cost of the measures that would have to be implemented to provide them in the absence of the mangroves. This procedure is problematic for a couple of reasons. First, the cost of replacing coastal protection (or any other) services is only a valid measure of their value if it would, in fact, be worth the cost to provide the protection by another means. In many places it would not. Coastal protection services would be of little economic value, for example if there were few people or structures at risk of damage. Second, the cost of completely replacing mangroves with seawalls or other protective measures may do little to inform the decision of how much of the mangrove barrier to maintain.

In their study of the value of mangroves in providing protection along the Kenyan coastline Huxham et al. (2015) use the cost of building seawalls for protection as a proxy for the value of mangroves. They follow conservative procedures in estimating these values, applying them only where there is evidence that structures would actually be in harm's way in the event of a coastal storm. As the authors note, others who have adopted less conservative assumptions have often arrived at much higher – and less plausible – figures. While the replacement cost method would not be preferred when other approaches are feasible, it may provide some useful information when its limitations are borne in mind. Following their procedure Huxham, et al., estimated a value of about 500 USD $ha^{-1}$ $yr^{-1}$ for coastal mangroves. The authors also incorporate carbon sequestration, fishery, and other benefits in the analysis, arriving at an overall figure of 1166 USD $ha^{-1}yr^{-1}$ for mangrove preservation.

### 2.2.2 Cooling

Another NbS that is often cited for adaptation to climate change is maintaining vegetation to cool urban areas (see, e. g., Eggermont et al. 2015; Seddon et al., n.d.; Walters 2016). Cooling provides an interesting example because it illustrates the many different perspectives from which NbS can be considered. The same service, temperature regulation, can be provided by a variety of natural assets, as well as by artificial alternatives. Conversely, a single natural asset can provide a range of services. Another dimension is the nature of the value realized from cooling: effects could be measured in the value of cooling reflected in home prices, in expenditures on alternative cooling approaches or technologies, or in the costs of consequences of overheating, such as adverse health outcomes. While such measures may be interrelated (the price of a home should, for example, reflect the costs of keeping its occupants comfortable and healthy), they represent different empirical approaches to valuation, and may be expected to give different answers depending on how they are implemented.

There are at least three separate, though not mutually exclusive, ways in which vegetation reduces temperatures. One is simply by shading. A second is through evapotranspiration: plants draw water from the ground and release vapor into the air (Fuss et al. 2018). Finally, to the extent that vegetation may provide cooling, expenditures on, and emissions from the energy required to generate, air conditioning may be reduced (Raymond et al. 2017). The literature provides examples of services provided by areas of natural vegetation, such as parks, maintained in cities, as well as of planted and managed areas. In the context of cooling the most common example of the latter are “green roofs” established and maintained on top of buildings.

There have been several efforts to quantify nature-based cooling benefits, though many are incomplete or problematic. A widely cited meta-analysis found that urban green areas generate a 0.94 C reduction in urban temperatures on average (Bowler et al. 2010). This study combined evidence from forests, parks of various sizes, and ground and roof-top vegetation, so not only the size of effects but the mechanisms for achieving them varied considerably. More recent work has begun to relate the amount of cooling afforded by urban green spaces to the size of the spaces and their distance from structures (Ziter et al. 2019). Deriving the often-nonlinear relationships between areas generating services and the extent of the services provided is key in assessing what the appropriate scale for them should be.

Few researchers have yet taken the next step of quantifying the costs and benefits of green spaces providing relief from increasing urban heat. In an extensive review of literature on managing climate change in cities Hobbie and Grimm (2020) list improving cost benefit analyses among the top research priorities. Such economic studies as have been done are suggestive, though they also illustrate the challenges of constructing monetary estimates. Becker et al. (2019) investigate the relationship between urban greenspace and Medicare expenditures.

While heat effects are just one of the possible nexuses between green spaces and health (Kabisch, van den Bosch, and Lafortezza 2017), they may explain in part why Becker et al. (2019) find a small but significant inverse correlation between green space and health expenditures. Becker and colleagues controlled for a number of potential confounding factors, but it could be very difficult to correct for potential selection bias: healthier people may be more likely to live in areas with more outdoor recreational opportunities (Chiabai et al. 2020 give an example of procedures to correct for such biases for other landscape-related services).

A more promising approach for economic valuation may be to look at cooling costs that may be avoided by plantings. Measuring the cost of providing a service by alternative means is not, in general, a valid approach to estimating economic value, because it is not always the case that such costs would be incurred. In the case of cooling residential and commercial spaces, though, building occupants are generally willing to pay to maintain temperatures within relatively narrow ranges. One way of moderating building temperatures is to establish roof-top gardens and plantings often described as "green roofs" (see, e. g., Stagakis, Somarakis, and Chrysoulakis 2019).

A couple of studies have attempted carefully to measure the cost savings of green roofs. William et al. (2016) find that the cooling savings afforded by green roofs justify their costs relative to conventional roofs. Clark, Adriaens, and Talbot (2008) come to similar conclusions: the life cycle costs of green roof are between 20 and 25% less than those of conventional roofing when heating and cooling cost savings are factored in. They note further that consideration of air quality and stormwater retention benefits could increase the differential to approximately 40% in favor of green roofs. Air quality and stormwater retention benefits are, generally speaking, externalities; they benefit the public at large, but the property owners providing them are not compensated (although builders in some areas are now required to account for the stormwater runoff generated by the impervious surfaces they construct).

While Clark, Adriaens, and Talbot's (2008) analysis indicates that green roofs are less costly in net present value terms than conventional alternatives, their findings are based on a five percent discount rate. One might suppose that a higher rate might have been in order over the period they were considering (they based their analysis on data for 2006), and this would, in turn, have increased the cost of green relative to conventional roofing.[14] In any event, it would seem that the choice of adopting the nature-based solution depends in this instance, as in many others, on the ability to align private incentives that may not always make a compelling case for adoption relative to alternatives, with public incentives that will require more active public policy to effectuate but could provide the extra margin necessary to make a nature-based approach more cost-effective than an artificial alternative.

### 2.2.3 Pollination

The idea of nature-based solutions encompasses extensive earlier work on ecosystem services. While the focus of this report has largely been on climate change mitigation and adaptation, maintaining natural systems that store carbon and help in adaptation to climate change often confers ancillary benefits. Alternatively, it might be said that natural systems whose maintenance has often been motivated by other ecosystem services also store carbon and protect against climate change. Many of these ecosystem service benefits have been the subject of extensive study.

One such service is pollination. Many important food crops depend on insects and other animals to pollinate them. These pollinators depend, in turn, on natural environments for refuge from predators and the elements, breeding areas, and alternative sources of fodder. The example of pollination illustrates some of the important caveats that must be observed when estimating economic values associated with NbS, however. Pollinators are valuable in that they enhance the productivity of agriculture, and retained areas of natural habitat are valuable in that they sustain larger pollinator populations. Diminishing returns characterize both relationships, however. Moreover, while many crops require insect pollinators, many

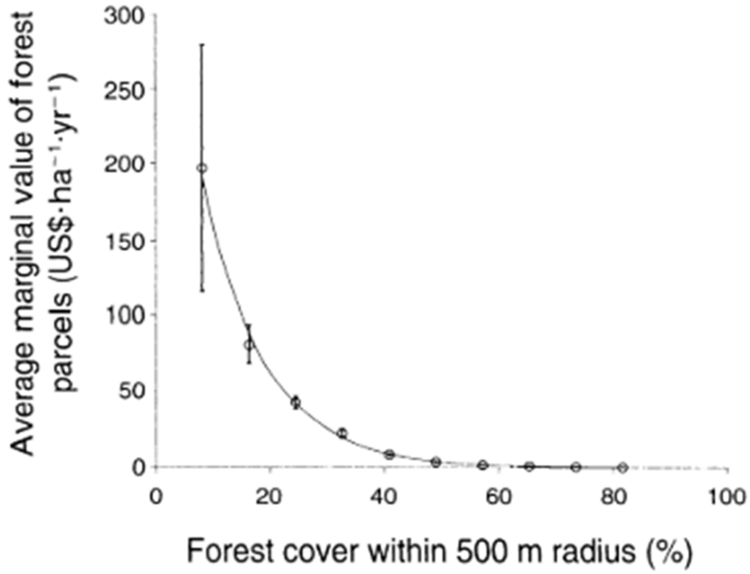
[14] While green roofs are more expensive to install initially relative to conventional roofs, they offset their higher installation costs over time with net savings in operating costs. The higher is the discount rate, the more important are the up-front costs relative to the future cost savings.

others do not, and so the value of pollination services is necessarily constrained if farmers are able to switch to crops that do not require insect pollination (Ghazoul 2005; Macaulay 2006).

A number of authors have studied the value of pollinators to the fertilization of crops and, by extension, the value of areas of natural habitat set aside for the protection of pollinators (for a recent critical review of studies on the value of pollination, see Breeze et al. 2016). Economic values are based on the incremental contributions, and this has not been reflected consistently in studies of the value of pollination. A notable exception is work by Ricketts and Lonsdorf (2013; see also Simpson 2019) on the value of services provided by wild pollinators harbored in areas of remnant forest. The authors note two sources of diminishing returns in the provision of natural pollination services. First, increasing the size of an area of forest will provide less-than-proportional additional numbers of pollinators. Second, the pollination function saturates: regardless of the number of pollinators available to serve a crop, its yield is bounded by the maximum amount of product – in Ricketts and Lonsdorf's example, coffee beans – that can be produced on a fixed number of plants.

The combination of these factors leads Ricketts and Lonsdorf to find that, while the marginal value of habitat sheltering pollinators could be very high when such habitat is scarce and pollinators are in high demand, it quickly drops to near zero as habitat becomes abundant relative to the local need. This pattern is illustrated in Figure 5, reproduced with permission from Ricketts and Lonsdorf (2013).

*Figure 5*




Source: Ricketts and Lonsdorf (2013) © Ecological Society of America; reproduced by kind permission.

These findings might be related to some results noted earlier. Griscom et al. (2017) reported that it might be cost-effective to reforest substantial areas of the globe at a carbon price of 100 USD Mg $CO_{2e} \cdot y^{-1}$. This translates to a value of about 1,720 USD $h^{-1} \cdot y^{-1}$ for potential forest land in the tropics.[15] If additional pollination benefits of several hundred dollars per hectare could be added to this amount, the incentive to regrow forests to provide both carbon storage and pollinator habitat might be considerably stronger. As the figure shows, though, the relationship saturates quickly (in the *spatial*, as opposed to *temporal*, domain in this instance). If a substantial area of land were already being set aside to sequester carbon, it might well already

[15] Griscom, et al., report that forest land in the tropics can sequester about 4.7 Mg of carbon per year. This corresponds to removal of about 17.2 Mg of $CO_2$ per year. Valued at 100 USD per Mg, the resultant per-hectare value is as indicated.

be providing enough pollination services as to make any benefits from further enhancing them negligible.

This example suggests some important caveats to bear in mind when thinking about the valuation of NbS. While a single area maintained to provide a nature-based solution to one problem might also contribute to the solution of many others, care must be taken in adding values. Some NbS are mutually exclusive; the same area of land cannot be used at the same time to both maintain a standing forest and grow new biomass, for example (Minx et al. 2018). It is also important to appreciate that values calculated in one context cannot generally be extrapolated to another, at least not without making allowances for aspects that differ between times and places. As noted above, for example, an area of restored forest may be generating the largest carbon storage value when it is growing most rapidly, but might have the greatest value as pollinator habitat or resistance to erosion when the forest has reached maturity. In the spatial dimension, carbon storage values may scale linearly with the size of a forest area, while, as Figure 5 shows, marginal pollination values may decline very rapidly with the area providing them.

### 2.2.4 Pollution treatment

Another commonly cited ecosystem service and, hence, NbS, underscores some of the points just made concerning diminishing returns. One of the services areas of natural forest, strips of vegetation along rivers, and wetlands may perform is the retention and treatment of pollutants. Examples of water purification in the Catskills watershed that supplies New York City's water (Chichilnisky and Heal 1998) and the source from which the Perrier Company draws its water (Perrot-Maitre 2006) have often been cited to demonstrate the values nature can provide. Such natural purification might be particularly important in developing countries, where financial resources to invest in wastewater treatment facilities are often unavailable (see, e. g. Emerton et al. 1999's study of the value of the Nakivubo wetlands near Kampala, Uganda for purifying water).

As with pollination, coastal protection, and other services, however, pollution treatment often exhibits sharply diminishing returns. The value of a wetland or riparian buffer for the treatment of pollution depends on its ability to capture and neutralize pollutants. The more pollution can be captured in the first hectare of such a system, however, the less remains to be retained by the next, and so the marginal product of additional areas necessarily declines. This has been substantiated in a number of natural science studies of riparian buffer strips and wetlands (Mayer et al. 2007; Mander 2008).

Context is also critical; if natural waste-treatment services are to be valuable, the system providing them must be located between an upstream source of pollutants and a downstream population that is vulnerable to contamination (Plummer 2009). The capability to treat pollution is of little value unless there are pollutants to be treated and a compelling reason to treat them.

Natural assets must also be cost-competitive with artificial alternatives if preservation of natural assets is to be justified by their value in providing waste-treatment services. While New York City continues to pay for watershed protection, it has also invested some $200 million in wastewater treatment plants (Hu 2018). Similarly, Uganda has constructed a large plant to treat wastewater discharged *into* the Navikubo Wetland. While such investments do not obviate the importance of natural treatment, other things being equal, they may diminish the incremental value of natural treatment.

### 2.2.5 Biodiversity

More diverse ecosystems may be more biologically productive, and, hence, increase the storage of carbon. They may also provide greater protection against storms and floods, resistance to erosion, and other adaptive responses to climate change. Biological diversity itself may also be a valuable attribute of natural ecosystems.

Willingness to pay for natural diversity may be manifested in different ways. Tourists may pay to visit areas in which they can see profuse and/or unusual wildlife. This demand can be measured using methods to infer values from the expenses tourists incur to visit sites (see, e. g., Kuriyama, Shoji, & Tsuge, 2012). While tourists may pay more to visit more biologically diverse, or biologically unique, sites, it may be difficult to infer how their willingness to pay to view diverse or unusual assemblages of species translates into a value for the habitats that sustain them. There may be little additional value attached to areas of habitat larger than those required to maintain viable populations of "charismatic mega-vertebrates" (Terborgh 1999). Even highly modified habitats may be managed for selected wildlife by providing them shelter and fodder; the zoological park is the extreme case. While larger areas of maintained habitats can be expected to support broader assemblages of species, the relationship may be highly nonlinear, with diversity increasing much more slowly than habitat area (MacArthur and Wilson 1967; Losos and Ricklefs 2009).

Biological diversity may also have a value as a source of new products. Genetic combinations found in nature have proved to be the source of new commercial and pharmaceutical products and, of course, virtually all agricultural products had their origins in natural antecedents. Wild genetic variation may still be a source of disease resistance or yield increases. Yet the enthusiasm for commercialization of wild genetic resources has elicited conflicting academic views as to the likely value they imply for the areas harboring them, with some authors optimistic about the economic prospects (see, e. g., Rausser and Small 2000), while others argue they are likely to be negligible (see, e. g., Simpson et al. 1996; Costello and Ward 2006).

Real-world experience with “bioprospecting” for commercially valuable genetic resources has often proved disappointing (Conniff 2012).

Over and above whatever values biological diversity might afford as a magnet for tourists or a source of new products, there are “existence values” arising from the ethical obligation many people feel to maintain Earth’s diversity of life. Inasmuch as such values cannot be tied to the purchase or consumption of any marketed product, they cannot be estimated by common economic methods. Such values, if they are to be estimated at all, can only be inferred from survey results derived using “stated preference” methods (Pascual et al. 2012). While a great deal of research continues to be done on these methods, they remain controversial among many economists (Hausman 2012; Desvousges, Mathews, and Train 2015).

Perhaps most importantly from the present perspective, an argument that nature’s values could largely be ascribed to the intangible preferences of people the world over seems to circle back to the starting point that an appeal to NbS may have been intended to avoid. Nature-based solutions are often posed as the ways in which nature might provide solutions to tangible and pragmatic problems, as an alternative to more abstract arguments for maintaining diversity that have not gained traction (Armsworth et al. 2007). The problem to which a nature-based solution is posited is generally not “the decline of nature” *per se*.

## 3. Some overarching issues

The previous section has provided illustrative examples of the application of economic analyses to nature-based solutions. It has not been possible to review the full range of applications of NbS that might be explored from an economic perspective, let alone to survey the vast literatures that are relevant to such issues. The examples do, however, point to several important concerns with the application of economics to NbS. Three are considered here: tradeoffs *between* environmental and ecological objectives, as opposed to tradeoffs between environmental and ecological objectives on one hand, and other societal interests, on the other; common problems and misconceptions in economic valuation of ecological assets; and the unavoidable uncertainty inherent in contemplating extensive changes from the *status quo*.

## 3.1 Tradeoffs

In a paper comparing strategies for absorbing atmospheric $CO_2$ Williamson (2016) writes that widespread adoption of BECCS to constrain mean global temperature change to less than 2 °C would require appropriation of vast areas of forest and grasslands. Such a loss of habitat could cause greater losses of terrestrial species than would the temperature increase it was intended to forestall. From the perspective of biodiversity protection, at least, Williamson suggests that the cure might prove worse than the disease.

BECCS may be an extreme example. Perhaps regularly harvesting managed plantations to produce biofuels, capturing the carbon they embody, and injecting it in deep storage is not a "real" a nature-based solution (Hausfather 2018; Lewis et al. 2019). Yet by the definitions reviewed above, it is not clear that BECCS, which arguably is "inspired" (EC, n. d.) and "motivated and supported" by nature (Conti 2019) would not qualify. Moreover, BECCS could offer additional carbon storage long after forests being replanted now have reached maturity and, hence, would store little more carbon (Minx et al. 2018). BECCS might comprise a more "natural" approach than, say, direct air carbon capture and storage through electro-mechanical means (Fuss et al. 2018) under the comparative standard Maes and Jacobs (2017) propose to distinguish NbS from artificial alternatives.

BECCS may provide an extreme example of how a more a tenuously nature-based solution might work at cross-purposes with a less ambiguously "natural" alternative, but there are many other instances in which alternative nature-based solutions are likely to "solve" different problems to different degrees. For example, monoculture plantings have been disparaged by commentators who decry the loss of biodiversity they may cause (see, e. g., Lewis et al. 2019). Yet others argue that monoculture plantations may offer the most rapid path for carbon sequestration (see, e. g., Lin et al. 2013).

The issues of leakage discussed above may also be interpreted as a question of tradeoffs. Is it better to regrow forests in some places if doing so will result in more deforestation elsewhere?

The answer depends on the relative values attached to the carbon sequestration *and* biodiversity values (and potentially others) in both the places where forest areas will be gained and those from which they may be lost. Similar issues arise with respect to the intensity of agriculture. It is possible grow to food in, for example, mixed agroforestry systems that may shelter more local biodiversity and fix more soil carbon (Stagakis, Somarakis, and Chrysoulakis 2019). If these benefits are achieved at the expense of reducing yield per hectare, however, they may involve employing larger areas less intensively, as opposed to smaller areas more intensively (Simpson 2014). This, in turn, may involve reducing areas of still relatively pristine ecosystems.

These concerns recall debates that have been occurring in conservation biology for many years between advocates of preservation of pristine systems (see Soulé 2013) and those who would integrate the remnants of natural systems in working landscapes in order to preserve them (see Kareiva and Marvier 2012). Findings from both natural and social sciences may be helpful in predicting the sometimes unanticipated effects of alternative policies, such as leakage. More fundamental questions involve values in the philosophical, rather than economic, sense of the term: what *should* the objectives of policies to implement nature-based solutions or alternative approaches be? Economic analysis cannot help with these fundamental choices. Once they have been made, however, economic approaches may be helpful in determining how best to achieve objectives determined by other means.

The more general point is that in order to decide how best to achieve an objective, it is essential first to be clear about what the objective *is*. This is an ongoing problem with nature-based solutions. NbS may address climate change mitigation and adaptation as well as the provision of other environmental and ecological benefits, all while enhancing resiliency, human well-being and societal equity (Walters 2016; Conti 2019; EC n. d.). Different approaches will achieve different objectives to different degrees, and few if any are likely to promote them all. As some approaches are mutually exclusive – the same land cannot be used for different purposes at the same time – it is crucial that objectives be clarified if NbS are to be made operational (Cohen-Shacham et al. 2019).

## 3.2 Challenges in economic valuation

There has been a great deal of work in recent decades on the economics of biodiversity, ecosystem services and, more recently, NbS. This includes several comprehensive studies (see, e. g. Pearce and Moran 1994; TEEB 2009; Daily 1997; Millennium Ecosystem Assessment 2005; Díaz et al. 2018; Bateman et al. 2014) and literally thousands of case studies. A reader might wonder, then, what remains to be done.

There is often less to ostensibly "economic" studies than there may seem, however. Studies that may seem to address economic values sometimes provide only partial or suggestive evidence. A survey of nearly 400 studies on water-based ecosystem services found that the majority "failed to adequately link changes in environmental conditions to human well-being, instead stopping at the point of suggesting that one was connected to the other" (Brauman 2015). Another survey of a broader set of ecosystem service valuation studies found that less than a third "provided a sound basis for their conclusions" (Seppelt et al. 2011). Recent work on the value and effectiveness of NbS reveals similar shortcomings. Reed et al. (2017)'s survey of values arising from maintained natural forests includes several studies (Asase et al. 2008; Renard, Rhemtull, and Bennett 2015; Chauhan et al. 2010) that point to forests' contributions to food production and livelihoods, but do not quantify economic values and/or relate them to costs of their provision. Reid et al. (2019) review 13 studies offering economic analyses of ecosystem-based adaptation to climate change. In only 4 of 13 cases do they find results they deem to be supported by quantitative evidence that ecosystem-based adaptation measures are more cost effective than alternatives.[16]

Studies that do report economic values of NbS and their associated ecosystem services are also often flawed. Many do not conform with best practice for economic valuation. The economic value of a good or service must reflect actual willingness to pay, as opposed simply to the cost

[16] In two instances the authors report either that the evidence did not support a conclusion that ecosystem-based adaptation was cost-effective, or that a determination could not be made, and in seven they report that participants *perceived* ecosystem-based adaptation measures to be more cost-effective than alternatives, but could not substantiate their impressions with hard data.

of replacing it without reference to whether such a replacement would be cost effective. Moreover, economic values are determined for incremental comparisons, not on the basis of total effects or extrapolation of averages effects (see, e. g., Freeman, Herriges, and Kling 2014).

Valuation of ecosystem services is an inherently multidisciplinary undertaking, but the economic aspect of the work is not always undertaken by economists. Torres and Hanley (2017) survey nonmarket valuation studies of coastal and marine ecosystem services, and report that many of the studies they identified were published in natural science, rather than economic journals. Other large compilations of valuation studies reveal similar patterns. The Ecosystem Services Valuation Database (Van der Ploeg and de Groot 2010) contains over 1,300 value estimates and has been used in such major undertakings as the project on *The Economics of Ecosystems and Biodiversity* (TEEB 2009) and the United Kingdoms' *National Ecosystem Assessment* (Bateman et al. 2014). It too assembles estimates of economic values from a variety of sources. This is not to say that natural scientists cannot perform credible economic analyses, nor that economists do not publish in natural science journals. When Blomqvist and Simpson (2017) chose 28 value estimates at random from those collected in the ESVD to conduct a careful review of their methods, however, their findings were troubling. Over half used replacement cost inappropriately as a valuation method or confused total and incremental benefits.

Such errors are particularly problematic in that much of the work on NbS is done by compiling the results of earlier studies. The practice of "benefit transfer," or applying the results of a valuation study done at one time or place to another, after adjusting for differences between contexts (see Johnston 2017), is increasingly adopted as an expedient when not enough time or budget is available to conduct an original valuation study (U.S. EPA 2014; Siikamäki, Santiago-Ávila, and Vail 2015). Context is critical in valuation; goods are economically valuable to the extent that they are scarce. A value estimate, even if it is computed properly, may not be useful in a benefit transfer if does not relate value to the quantity of the good or service being provided, as that quantity will likely vary from setting to setting. Of course, a value estimate that is *not* computed properly may not be useful in any setting.[17]

## 3.3 Large changes and pervasive complexity

Adopting nature-based solutions to climate change will require changes in land use over very large areas. While NbS for coastal storm protection, temperature moderation, or provision of other more local ecosystem services might not involve as much land use change in a single place, they might have significant effects on the economies of particular communities, as well as substantial effects on regional or national economies if adopted in a large number of areas.

Economic valuation is based on *incremental* comparisons. The value placed on an area of forest for the carbon storage or pollination services it supplies is not determined by what society would be willing to pay for a stable climate or an adequate food supply as opposed to the complete absence of either. Rather, the economic value placed on maintenance or restoration of an additional area of forest is determined by what incremental benefit society would realize from the change in temperature or increase in food supply it would afford. It is a truism in economics that "value is determined on the margin": by how much a little more of something is worth. This leads to a sort of paradox. Estimates of economic value are based on willingness to pay for small increments, but large changes may be required to address important challenges.[18] This raises the question of how large changes in land use and other inputs underlying NbS should be.

As noted above, large changes in land use in one area are likely to induce offsetting changes in other areas through leakage. Leakage may be controlled by instituting coordinated policies

[17] It may be useful to distinguish between unavoidable stochastic variation in the estimation of values and bias. The former merely means that estimates are uncertain, but that such uncertainties should be reduced over repeated samples. Bias is more problematic. Values estimated on the basis of replacement costs or that result from the confusion of total or average for marginal values will generally be too high, and the bias will not be reduced by including more flawed studies in a meta-analysis.

[18] The problem of inferring the value of large changes has long been recognized. Alfred Marshall (1842 – 1924) is credited with introducing the notion of externalities in economics, but also noted that speculations regarding willingness to pay is "highly conjectural except in the neighborhood of the customary price".

across very broad landscapes. These in turn may induce changes in the price of food, the incomes of farmers and other workers, and ripple effects through the entire economy. To understand the implications of large scale NbS, then, a model of the entire economy might be required.

Such computable general equilibrium models have been used to predict the effects of climate change and policies to control it. However, tremendous uncertainties characterize such efforts (Gillingham et al. 2016), and some commentators suggest that many models are artefacts of the assumptions on which they are based, and consequently not reliable enough to provide very much practical assistance in planning (Pindyck 2013). Complex and wide-ranging models of land use have been or could be used to predict the effects of NbS (Hertel, West, and Villoria 2019 review more than a dozen large-scale land-use models). Similar concerns might be raised about their reliability.

George E. P. Box's statement that "all models are wrong, but some are useful" is often repeated. One of the challenges in implementing nature-based solutions is to determine which of the models that can be used to guide them are useful.

# 4. Recommendations

In conclusion, a number of recommendations are offered for better understanding and advancing work on the economic analysis of nature-based solutions.

## Recommendation 1: Adopt clear definitions, objectives, and scope

Ongoing work on nature-based solutions might adopt one of the many existing definitions of the term, combine elements of two or more, or develop a new definition. What is more important than the definition *per se* is that the objectives of NbS be clarified. What are the problems nature-based solutions are intended to solve? Merely listing desiderata – that NbS should mitigate and adapt to climate change, preserve biodiversity, provide local ecosystem services, promote resilient ecological and human communities, enhance equity and participation, etc. – is not sufficient, as some of these criteria inevitably come into conflict. There may be no strictly scientific way in which to reconcile competing objectives, but clarifying and prioritizing objectives is essential if NbS is to be an operationally useful concept (Cohen-Shacham et al. 2019; Conti 2019).

This is particularly true in light of the fact that many NbS have different effects on different spatial and temporal scales. Ideally, policies would be adopted with an appreciation for their global implications and an understanding of how they might need to be modified or supplemented over time. This may not always be possible, but when a choice is made to restrict attention to local and temporary effects, it should be explicitly recognized and justified.

## Recommendation 2: Curate existing research

There has been a tremendous amount of research undertaken on nature-based solutions and their economic values and implications. At this time it may be as helpful to carefully evaluate what has been done to date as it would be to initiate a large number of new projects. While much – though by no means all – of the work on NbS that is widely cited and has proved influential has appeared in peer reviewed publications, journal editors have not necessarily assigned reviewers across the broad range of natural and social science disciplines that would be required for full evaluation. Consequently, it is not clear that findings from much of the literature are credible or provide useful guidance for application in new settings.

A "retroactive peer review" of existing studies could

i. Validate studies that may be used as exemplars for new work and/or employed in benefit transfer studies;
ii. Identify common conceptual errors and warn researchers against them; and
iii. Recommend best practices.

Such a review could be accomplished by developing a protocol for sampling from existing work, and from the resultant sample identifying the prevalence of both problematic and replicable procedures.

## Recommendation 3: Assess the ability of complex models to make useful predictions at relevant spatial and temporal scales

Issues of spatial and temporal scale are critical in considering NbS. The effects of implementing a policy at one location cannot be fully evaluated without understanding how adverse consequences might "leak" into other areas. Similarly, it must be known how the effects of NbS vary over time, so that stop-gap measures can be implemented over any periods before they become fully effective, and subsequent interventions planned if they will later become ineffective.

There is a multitude of complex modeling platforms that have been used to predict the spatial and temporal extent of different policies (Hertel, West, and Villoria 2019; Baldos and Hertel 2012; Searchinger et al. 2018; Lanz, Dietz, and Swanson 2018; Schmitz et al. 2012, Bertram et al. 2018) Yet much more work remains to be done to enhance their applicability and predictive capabilities (Byerlee, Stevenson, and Villoria 2014). An assessment could be made of the performance of such models by using them to "predict" known outcomes based on earlier data. Models might also be compared in the precision with which they can predict the spatial and temporal effects of NbS and their sensitivity and robustness to parameter choices. Such exercises might identify which model(s) might be best used for particular purposes, and what further improvements might make complex models more accurate and useful.

### Recommendation 4: Identify effective policy instruments for implementing NbS

NbS often induce effects over a range of places and times. They are also introduced in an effort to achieve a variety of societal and ecological ends. It is generally necessary to tailor and coordinate a suite of policies if a policy maker wishes simultaneously to achieve multiple objectives (Tinbergen 1952). While several authors have noted the importance of coordinating policy measures (see, e. g., DeFries and Rosenzweig 2010), careful investigations of the interactions between policy measures are less common (see, e. g. Bode et al. 2014; Schmitz et al. 2012; Bertram et al. 2018). Additionally, policy actions can and must redress the disproportionate impact on women and girls of economic, social and environmental shocks and stresses.

Many different policy approaches have been recommended for implementing NbS and fine-tuning their effects: regulations, taxes, subsidies, changes in land tenure, communal resource management, monitoring and enforcement of conservation measures, and a host of others. Surveys of instruments and incentives now being offered to landowners and other stakeholders, as well as the interaction between policies, could identify which have proved most effective under what circumstances and identify problems that may not have been foreseen when they were first implemented.

### Recommendation 5: Adopt adaptive management plans

The effectiveness of NbS is likely to be extremely uncertain. Projections concerning, for example, how much carbon a growing forest can store, or how much protection the same forest might provide against erosion are necessarily imprecise. Moreover, they are made against the backdrop of a changing environment. Carbon fertilization may enhance forest growth, or perhaps drought will slow it. Erosion may be more or less severe depending on the frequency and intensity of precipitation in a changing climate. On top of uncertainties on questions of natural science, there are also societal uncertainties. How quickly will populations grow, how far will people migrate, and how will they respond to incentives?

Given the uncertainties inherent in reliance on NbS, as well as the severity of the problems against which they are deployed, it is important to adapt to new understandings of the effectiveness of NbS. It is reasonable to suggest that NbS be employed aggressively as part of a broad-based program to prevent and adapt to climate change. Because the need to address climate change and biodiversity loss is growing urgent, it is reasonable to suggest that *all* feasible approaches be employed aggressively. By the same token, though, it is also advisable to be prepared to amend plans based on emerging evidence: to expand on the use of those measures that are shown to be most effective, while reducing dependence on others that are not performing as well.

Protocols should be defined for amending NbS plans in advance of the need to implement changes. Deciding on what modifications to make in response to which signals of interim success or failure will enable more effective adaptation to circumstances that will inevitably change.

REFERENCES


Armsworth, P. R., K. M.A. Chan, G. C. Daily, P. R. Ehrlich, C. Kremen, T. H. Ricketts, and M. A. Sanjayan. 2007. “Ecosystem-Service Science and the Way Forward for Conservation.” *Conservation Biology*. https://doi.org/10.1111/j.1523-1739.2007.00821.x.

Asase, a., S.a. Wade, K. Ofori-Frimpong, P. Hadley, and K. Norris. 2008. *Carbon Storage and the Health of Cocoa Agroforestry Ecosystems in South-Eastern Ghana*. *Africa and the Carbon Cycle: World Soil Resources Reports*.

Baldocchi, Dennis, and Josep Penuelas. 2019. “The Physics and Ecology of Mining Carbon Dioxide from the Atmosphere by Ecosystems.” *Global Change Biology*. https://doi.org/10.1111/gcb.14559.

Baldos, U, and TW Hertel. 2012. “SIMPLE: A Simplified International Model of Agricultural Prices, Land Use and the Environment.” *… Analysis, Department of Agricultural*.

Barbier, Edward B., Ricardo Lozano, Carlos Manuel Rodríguez, and Sebastian Troëng. 2020. “Adopt a Carbon Tax to Protect Tropical Forests.” *Nature* 578: 213–16.

Barbier, Edward B, Evamaria W Koch, Brian R Silliman, Sally D Hacker, Eric Wolanski, Jurgenne Primavera, Elise F Granek, et al. 2008. “Nonlinear Ecological Functions and Values.” *Science*. https://doi.org/10.1126/science.1150349.

Bateman, Ian J., B Day, M Agarwala, P Bacon, T Badura, A Binner, A De-Gol, et al. 2014. “UK National Ecosystem Assessment Follow-on. Work Package Report 3: Economic Value of Ecosystem Services,” no. February: 1–246.

Becker, Douglas A., Matthew H.E.M. Browning, Ming Kuo, and Stephen K. Van Den Eeden. 2019. “Is Green Land Cover Associated with Less Health Care Spending? Promising Findings from County-Level Medicare Spending in the Continental United States.” *Urban Forestry and Urban Greening*. https://doi.org/10.1016/j.ufug.2019.02.012.

Bello, Carolina, Mauro Galetti, Marco A. Pizo, Luiz Fernando S. Magnago, Mariana F. Rocha, Renato A.F. Lima, Carlos A. Peres, Otso Ovaskainen, and Pedro Jordano. 2015.

“Defaunation Affects Carbon Storage in Tropical Forests.” *Science Advances* 1 (11): 1–11. https://doi.org/10.1126/sciadv.1501105.

Bertram, Christoph, Gunnar Luderer, Alexander Popp, Jan Christoph Minx, William F. Lamb, Miodrag Stevanović, Florian Humpenöder, Anastasis Giannousakis, and Elmar Kriegler. 2018. “Targeted Policies Can Compensate Most of the Increased Sustainability Risks in 1.5 °c Mitigation Scenarios.” *Environmental Research Letters* 13 (6). https://doi.org/10.1088/1748-9326/aac3ec.

Blomqvist, Linus, and R. David Simpson. 2017. “The Value of Ecosystem Services: What Is the Evidence?” In *Effective Conservation Science: Data Not Dogma*. https://doi.org/10.1093/oso/9780198808978.003.0003.

Bode, Michael, Ayesha I. T. Tulloch, Morena Mills, Oscar Venter, and Amy W. Ando. 2014. “A Conservation Planning Approach to Mitigate the Impacts of Leakage from Protected Area Networks.” *Conservation Biology* 00 (0): n/a-n/a. https://doi.org/10.1111/cobi.12434.

Bowler, Diana E., Lisette Buyung-Ali, Teri M. Knight, and Andrew S. Pullin. 2010. “Urban Greening to Cool Towns and Cities: A Systematic Review of the Empirical Evidence.” *Landscape and Urban Planning*. https://doi.org/10.1016/j.landurbplan.2010.05.006.

Brauman, Kate A. 2015. “Hydrologic Ecosystem Services: Linking Ecohydrologic Processes to Human Well-Being in Water Research and Watershed Management.” *Wiley Interdisciplinary Reviews: Water*. https://doi.org/10.1002/wat2.1081.

Breeze, Tom D., Nicola Gallai, Lucas A. Garibaldi, and Xui S. Li. 2016. “Economic Measures of Pollination Services: Shortcomings and Future Directions.” *Trends in Ecology and Evolution*. https://doi.org/10.1016/j.tree.2016.09.002.

Bryan, Brett A., Rebecca K. Runting, Tim Capon, Michael P. Perring, Shaun C. Cunningham, Marit E. Kragt, Martin Nolan, et al. 2016. “Designer Policy for Carbon and Biodiversity Co-Benefits under Global Change.” *Nature Climate Change* 6 (3): 301–5. https://doi.org/10.1038/nclimate2874.

Burns, Wil, and Simon Nicholson. 2017. "Bioenergy and Carbon Capture with Storage (BECCS): The Prospects and Challenges of an Emerging Climate Policy Response." *Journal of Environmental Studies and Sciences* 7 (4): 527–34. https://doi.org/10.1007/s13412-017-0445-6.

Busch, Jonah, Jens Engelmann, Susan C. Cook-Patton, Bronson W. Griscom, Timm Kroeger, Hugh Possingham, and Priya Shyamsundar. 2019. "Potential for Low-Cost Carbon Dioxide Removal through Tropical Reforestation." *Nature Climate Change*. https://doi.org/10.1038/s41558-019-0485-x.

Byerlee, Derek, James Stevenson, and Nelson Villoria. 2014. "Does Intensification Slow Crop Land Expansion or Encourage Deforestation?" *Global Food Security* 3 (2): 92–98. https://doi.org/10.1016/j.gfs.2014.04.001.

Camm, Jeffrey D., Susan K. Norman, Stephen Polasky, and Andrew R. Solow. 2002. "Nature Reserve Site Selection to Maximize Expected Species Covered." *Operations Research* 50 (6): 946-955+1076. https://doi.org/10.1287/opre.50.6.946.351.

Chauhan, S.K., S.C Sharma, V. Beri, S. Yadav, and N Gupta. 2010. "YIELD AND CARBON SEQUESTRATION IN WHEAT-POPLAR INTERCROPPING." *Indian Journal of Agricultural Sciences*.

Chiabai, Aline, Sonia Quiroga, Pablo Martinez-Juarez, Cristina Suárez, Silvestre García de Jalón, and Tim Taylor. 2020. "Exposure to Green Areas: Modelling Health Benefits in a Context of Study Heterogeneity." *Ecological Economics*. https://doi.org/10.1016/j.ecolecon.2019.106401.

Chichilnisky, G., and G. Heal. 1998. "Economic Returns from the Biosphere." *Nature*. https://doi.org/10.1038/35481.

Clark, Corrie, Peter Adriaens, and F. Brian Talbot. 2008. "Green Roof Valuation: A Probabilistic Economic Analysis of Environmental Benefits." *Environmental Science and Technology*. https://doi.org/10.1021/es0706652.

Cohen-Shacham, Emmanuelle, Angela Andrade, James Dalton, Nigel Dudley, Mike Jones, Chetan Kumar, Stewart Maginnis, et al. 2019. "Core Principles for Successfully Implementing and Upscaling Nature-Based Solutions." *Environmental Science and Policy* 98 (April): 20–29. https://doi.org/10.1016/j.envsci.2019.04.014.

Conniff, Richard. 2012. "A Bitter Pill." *Conservation*.

Conti, Jeff. 2019. "Strategies for Operationalizing Nature-Based Solutions in the Private Sector." https://www.nature.org/content/dam/tnc/nature/en/documents/NBSWhitePaper.pdf.

Costanza, Robert, Octavio Pérez-Maqueo, Paul Sutton, Kenneth Mulder, M. Luisa Martinez, and Sharolyn J. Anderson. 2008. "The Value of Coastal Wetlands for Hurricane Protection." *AMBIO: A Journal of the Human Environment*. https://doi.org/10.1579/0044-7447(2008)37[241:tvocwf]2.0.co;2.

Costello, Christopher, and Michael Ward. 2006. "Search, Bioprospecting and Biodiversity Conservation." *Journal of Environmental Economics and Management*. https://doi.org/10.1016/j.jeem.2006.04.001.

Creutzig, Felix, Christian Breyer, Jérôme Hilaire, Jan Minx, Glen P. Peters, and Robert Socolow. 2019. "The Mutual Dependence of Negative Emission Technologies and Energy Systems." *Energy and Environmental Science* 12 (6): 1805–17. https://doi.org/10.1039/c8ee03682a.

Cromsigt, Joris P.G.M., Mariska Te Beest, Graham I.H. Kerley, Marietjie Landman, Elizabeth Le Roux, and Felisa A. Smith. 2018. "Trophic Rewilding as a Climate Change Mitigation Strategy?" *Philosophical Transactions of the Royal Society B: Biological Sciences*. https://doi.org/10.1098/rstb.2017.0440.

Daily, Gretchen C. 1997. "Nature's Services : Societal Dependence on Natural Ecosystems / Edited by Gretchen C. Daily." *Nature's Services: Societal Dependence On Natural Ecosystems*. https://doi.org/10.1023/a:1023307309124.

Das, S., and J. R. Vincent. 2009. "Mangroves Protected Villages and Reduced Death Toll during Indian Super Cyclone." *Proceedings of the National Academy of Sciences*.

https://doi.org/10.1073/pnas.0810440106.

DeFries, Ruth S, and C Rosenzweig. 2010. “Toward a Whole-Landscape Approach for Sustainable Land Use in the Tropics.” *Proceedings of the National Academy of Sciences of the United States of America* 107 (46): 19627–32. https://doi.org/10.1073/pnas.1011163107.

Desvousges, William, Kristy Mathews, and Kenneth Train. 2015. “From Curious to Pragmatically Curious: Comment on ‘from Hopeless to Curious?: Thoughts on Hausman’s “dubious to Hopeless” Critique of Contingent Valuation.’” *Applied Economic Perspectives and Policy*. https://doi.org/10.1093/aepp/ppv023.

Díaz, Sandra, Unai Pascual, Marie Stenseke, Berta Martín-López, Robert T. Watson, Zsolt Molnár, Rosemary Hill, et al. 2018. “Assessing Nature’s Contributions to People.” *Science* 359 (6373): 270–72. https://doi.org/10.1126/science.aap8826.

Dietz, Simon, Alex Bowen, Baran Doda, Ajay Gambhir, and Rachel Warren. 2018. “The Economics of 1.5◦C Climate Change.” *Annual Review OfEnvironment and Resources* 43: 455–80.

Dietz, Simon, and Nicholas Stern. 2015. “Endogenous Growth, Convexity of Damage and Climate Risk: How Nordhaus’ Framework Supports Deep Cuts in Carbon Emissions.” *Economic Journal*. https://doi.org/10.1111/ecoj.12188.

Eggermont, Hilde, Estelle Balian, José Manuel N. Azevedo, Victor Beumer, Tomas Brodin, Joachim Claudet, Bruno Fady, et al. 2015. “Nature-Based Solutions: New Influence for Environmental Management and Research in Europe.” *GAIA*. https://doi.org/10.14512/gaia.24.4.9.

Emerton, Lucy, Lucy Iyango, Phoebe Luwum, and Andrew Malinga. 1999. “The Present Economic Value of Nakivubo Urban Wetland, Uganda.” *And National Wetlands*, no. September: 1–30. https://doi.org/10.2307/2234529.

Enkvist, Per Anders, Tomas Nauclér, and Jerker Rosander. 2007. “A Cost Curve for Greenhouse

Gas Reduction." *McKinsey Quarterly*, no. 1: 34–45.

European Commission. n.d. "Nature-Based Solutions." https://ec.europa.eu/research/environment/index.cfm?pg=nbs.

Fargione, Joseph, Jason Hill, David Tilman, Stephen Polasky, and Peter Hawthorne. 2008. "Land Clearing and the Biofuel Carbon Debt." *Science* 319 (5867): 1235–38. https://doi.org/10.1126/science.1152747.

Faustmann, Martin. 1849. "Calculation of the Value Which Forest Land and Immature Stands Possess for Forestry."

Favero, Alice, Robert Mendelsohn, and Brent Sohngen. 2017. "Using Forests for Climate Mitigation: Sequester Carbon or Produce Woody Biomass?" *Climatic Change* 144 (2): 195–206. https://doi.org/10.1007/s10584-017-2034-9.

Freeman, Myrick A, Joseph A Herriges, and Catherine L Kling. 2014. *The Measurement of Environmental Resource Values*.

Fuss, Sabine, William F. Lamb, Max W. Callaghan, Jérôme Hilaire, Felix Creutzig, Thorben Amann, Tim Beringer, et al. 2018. "Negative Emissions - Part 2: Costs, Potentials and Side Effects." *Environmental Research Letters* 13 (6). https://doi.org/10.1088/1748-9326/aabf9f.

Garzuglia, Monica, and Mohamed Saket. 2005. "Wood Volume and Woody Biomass: Review of FRA 2000 Estimates." 68. Forest Resources Assessment. Rome.

Ghazoul, Jaboury. 2005. "Buzziness as Usual? Questioning the Global Pollination Crisis." *Trends in Ecology and Evolution* 20 (7): 367–73. https://doi.org/10.1016/j.tree.2005.04.026.

Gillingham, Kenneth, William D. Nordhaus, David Anthoff, Valentina Bosetti, Haewon McJeon, Geoffrey J. Blanford, Peter Christensen, John M. Reilly, and Paul Sztorc. 2016. "Modeling Uncertainty in Climate Change: A Multi-Model Comparison." *SSRN Electronic Journal*. https://doi.org/10.2139/ssrn.2743137.

Griscom, Bronson W., Justin Adams, Peter W. Ellis, Richard A. Houghton, Guy Lomax, Daniela A.

Miteva, William H. Schlesinger, et al. 2017. “Natural Climate Solutions.” *Proceedings of the National Academy of Sciences of the United States of America* 114 (44): 11645–50. https://doi.org/10.1073/pnas.1710465114.

Griscom, Bronson W., Guy Lomax, Timm Kroeger, Joseph E. Fargione, Justin Adams, Lucy Almond, Deborah Bossio, et al. 2019. “We Need Both Natural and Energy Solutions to Stabilize Our Climate.” *Global Change Biology*. https://doi.org/10.1111/gcb.14612.

Hancock Timber Resources Group. 2017. “2016 Timberland Investment Performance.”

Hartman, Richard. 1976. “The Harvesting Decision When a Standing Forest Has Value.” *Economic Inquiry*. https://doi.org/10.1111/j.1465-7295.1976.tb00377.x.

Hausfather, Zeke. 2018. “Analysis: How ‘Natural Climate Solutions’ Can Reduce the Need for BECCS.” *Carbon Brief*, 2018. https://www.carbonbrief.org/analysis-how-natural-climate-solutions-can-reduce-the-need-for-beccs.

Hausman, Jerry. 2012. “Contingent Valuation: From Dubious to Hopeless.” *Journal of Economic Perspectives*. https://doi.org/10.1257/jep.26.4.43.

Haya, Barbara. 2019. “POLICY BRIEF: The California Air Resources Board’s U.S. Forest Offset Protocol Underestimates Leakage,” 1–7. https://gspp.berkeley.edu/assets/uploads/research/pdf/Policy_Brief-US_Forest_Projects-Leakage-Haya_4.pdf.

Hertel, T.W., T.A.P. West, and N.B. Villoria. 2019. “A Review of Global-Local-Global Linkages in Economic Land-Use/Cover Change Models.” *Environmental Research Letters*.

Hertel, Thomas W. 2018. “Economic Perspectives on Land Use Change and Leakage.” *Environ. Res. Lett* 13 (7): 75012. https://doi.org/10.1088/1748-9326/aad2a4.

Hobbie, Sarah E., and Nancy B. Grimm. 2020. “Nature-Based Approaches to Managing Climate Change Impacts in Cities.” *Philosophical Transactions of the Royal Society B: Biological Sciences*. https://doi.org/10.1098/rstb.2019.0124.

Hoel, Michael, Bjart Holtsmark, and Katinka Holtsmark. 2014. “Faustmann and the Climate.”

*Journal of Forest Economics*. https://doi.org/10.1016/j.jfe.2014.04.003.

Houghton, Richard A, and Alexander A Nassikas. 2018. "Negative Emissions from Stopping Deforestation and Forest Degradation , Globally," no. May 2017: 350–59. https://doi.org/10.1111/gcb.13876.

Hu, Winnie. 2018. "A Billion-Dollar Investment in New York's Water." *New York Times*, 2018.

Huxham, Mark, Lucy Emerton, James Kairo, Fridah Munyi, Hassan Abdirizak, Tabitha Muriuki, Fiona Nunan, and Robert A. Briers. 2015. "Applying Climate Compatible Development and Economic Valuation to Coastal Management: A Case Study of Kenya's Mangrove Forests." *Journal of Environmental Management* 157 (April 2015): 168–81. https://doi.org/10.1016/j.jenvman.2015.04.018.

Interagency Working Group on Social Cost of Carbon, United States Government. 2010. "Technical Support Document: ? Social Cost of Carbon for Regulatory Impact Analysis ? Under Executive Order 12866."

International Union for Conservation of Nature and Natual Resources (IUCN). 2019. "NBS : Concept, Practices and Benefits."

———. 2020. "IUCN ' s Key Messages for UNFCCC COP25." In .

Johnston, Robert J. 2017. *Benefit Transfer of Environmental and Resource Values A Guide for Researchers and Practitioner*.

K. MacKinnon, C. Sobrevila, V. Hickey. 2008. "Biodiversity, Climate Change and Adaptation," 1–112.

Kabisch, Nadja, Matilda van den Bosch, and Raffaele Lafortezza. 2017. "The Health Benefits of Nature-Based Solutions to Urbanization Challenges for Children and the Elderly – A Systematic Review." *Environmental Research*. https://doi.org/10.1016/j.envres.2017.08.004.

Kareiva, Peter, and Michelle Marvier. 2012. "What Is Conservation Science?" *BioScience*. https://doi.org/10.1525/bio.2012.62.11.5.

Keesstra, Saskia, Joao Nunes, Agata Novara, David Finger, David Avelar, Zahra Kalantari, and Artemi Cerdà. 2018. “The Superior Effect of Nature Based Solutions in Land Management for Enhancing Ecosystem Services.” *Science of the Total Environment* 610–611: 997–1009. https://doi.org/10.1016/j.scitotenv.2017.08.077.

Kooten, G. Cornelis Van, and Craig M.T. Johnston. 2016. “The Economics of Forest Carbon Offsets.” *Annual Review of Resource Economics* 8 (1): 227–46. https://doi.org/10.1146/annurev-resource-100815-095548.

Kooten, G. Cornells van, Clark S. Binkley, and Gregg Delcourt. 1995. “Effect of Carbon Taxes and Subsidies on Optimal Forest Rotation Age and Supply of Carbon Services.” *American Journal of Agricultural Economics* 77 (2): 363–72. https://doi.org/10.2307/1243546.

Kumar, Pushpam. 2019. “Nature-Based Solutions: The Missing Ingredient in Climate Adaptation?” State of the Planet: Columbia University Earth Institute. 2019. https://www.preventionweb.net/news/view/66286?&a=email&utm_source=pw_email.

KURIYAMA, OICHI, YASUSHI SHOJI, and TAKAHIRO TSUGE. 2012. “The Value of Biodiversity and Recreation Demand Models: A Spatial Kuhn–Tucker Model.” In *The Economics of Biodiversity and Ecosystem Services*.

Lafortezza, Raffaele, Jiquan Chen, Cecil Konijnendijk van den Bosch, and Thomas B. Randrup. 2018. “Nature-Based Solutions for Resilient Landscapes and Cities.” *Environmental Research* 165: 431–41. https://doi.org/10.1016/j.envres.2017.11.038.

Lal, Rattan. 2009. “Soil Carbon Sequestration: Land and Water Use Options for Climate Change Adaptation and Mitigation in Agriculture.” *SOLAW Background Thematic Report – TRO4B*, 37. https://doi.org/10.1016/j.geoderma.2004.01.032.

Lanz, Bruno, Simon Dietz, and Tim Swanson. 2018. “The Expansion of Modern Agriculture and Global Biodiversity Decline: An Integrated Assessment.” *Ecological Economics*. https://doi.org/10.1016/j.ecolecon.2017.07.018.

Lewis, Simon L., Charlotte E. Wheeler, Edward T.A. Mitchard, and Alexander Koch. 2019.

“Restoring Natural Forests Is the Best Way to Remove Atmospheric Carbon.” *Nature* 568 (7750): 25–28. https://doi.org/10.1038/d41586-019-01026-8.

Lin, Brenda B., Sarina Macfadyen, Anna R. Renwick, Saul A. Cunningham, and Nancy A. Schellhorn. 2013. “Maximizing the Environmental Benefits of Carbon Farming through Ecosystem Service Delivery.” *BioScience*. https://doi.org/10.1525/bio.2013.63.10.6.

Lin, Meimei, and Qiping Huang. 2019. “Exploring the Relationship between Agricultural Intensification and Changes in Cropland Areas in the US.” *Agriculture, Ecosystems and Environment* 274 (March): 33–40. https://doi.org/10.1016/j.agee.2018.12.019.

Losos, Jonathan B., and Robert E. Ricklefs. 2009. *The Theory of Island Biogeography Revisited*. *The Theory of Island Biogeography Revisited*. Princeton University Press. https://doi.org/10.1111/j.1442-9993.2010.02226.x.

MacArthur, Robert, and Edward O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton University Press.

Macaulay, Douglas. 2006. “Selling out Nature.” *Nature*.

Maes, Joachim, and Sander Jacobs. 2017. “Nature-Based Solutions for Europe’s Sustainable Development.” *Conservation Letters* 10 (1): 121–24. https://doi.org/10.1111/conl.12216.

Mander. 2008. “Watershed Management.” In *Encyclopedia of Ecology, Five-Volume Set*. https://doi.org/10.1016/B978-008045405-4.00082-3.

Mann, Charles C. 2006. *1491: New Revelations of the Americas Before Columbus*.

Martin, Adrian, Brendan Coolsaet, Esteve Corbera, Neil Dawson, Janet Fisher, Phil Franks, Ole Mertz, Unai Pascual, Laura Vang Rasmussen, and Casey Ryan. 2018. “Land Use Intensification: The Promise of Sustainability and the Reality of Trade-Offs.” In *Ecosystem Services and Poverty Alleviation: Trade-Offs and Governance*. https://doi.org/10.4324/9780429507090.

Mather, A S. 1997. “The Forest Transition : A Theoretical Basis,” no. 1 998.

Mayer, Paul M., Steven K. Reynolds, Marshall D. McCutchen, and Timothy J. Canfield. 2007. "Meta-Analysis of Nitrogen Removal in Riparian Buffers." *Journal of Environmental Quality*. https://doi.org/10.2134/jeq2006.0462.

Meyfroidt, Patrick, Thomas K Rudel, and Eric F Lambin. 2010. "Forest Transitions, Trade, and the Global Displacement of Land Use." *Proceedings of the National Academy of Sciences of the United States of America* 107 (49): 20917–22. https://doi.org/10.1073/pnas.1014773107.

Millennium Ecosystem Assessment. 2005. "Ecosystem and Human Well-Being." *Ecosystems and Human Well-Being: Synthesis*. https://doi.org/http,//www.maweb.org/.

Minx, Jan C., William F. Lamb, Max W. Callaghan, Sabine Fuss, Jérôme Hilaire, Felix Creutzig, Thorben Amann, et al. 2018. "Negative Emissions - Part 1: Research Landscape and Synthesis." *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/aabf9b.

Monbiot, George. 2020. "Averting Climate Breakdown by Restoring Ecosystems A Call to Action." 2020.

Murray, Brian C BC, Bruce A Mccarl Texas A, Heng-chi Lee Texas A, BA McCarl, and HC Lee. 2002. "Estimating Leakage From Forest Carbon Sequestration Programs." *Land Economics* 80 (May): 109–24. http://le.uwpress.org/content/80/1/109.short.

Narayan, Siddharth, Michael W. Beck, Paul Wilson, Christopher J. Thomas, Alexandra Guerrero, Christine C. Shepard, Borja G. Reguero, Guillermo Franco, Jane Carter Ingram, and Dania Trespalacios. 2017. "The Value of Coastal Wetlands for Flood Damage Reduction in the Northeastern USA." *Scientific Reports* 7 (1): 1–12. https://doi.org/10.1038/s41598-017-09269-z.

Nature. 2017. "Nature-Based Solutions' Is the Latest Green Jargon That Means More than You Might Think." *Nature*. https://doi.org/10.1038/541133b.

Nesshöver, Carsten, Timo Assmuth, Katherine N Irvine, Graciela M Rusch, Kerry A Waylen, Ben Delbaere, Dagmar Haase, et al. 2017. "Science of the Total Environment The Science , Policy and Practice of Nature-Based Solutions : An Interdisciplinary Perspective." *Science of* *the Total Environment* 579: 1215–27. https://doi.org/10.1016/j.scitotenv.2016.11.106.

Ontl, Todd. 2018. "Soil Carbon Storage." *Soil Carbon Storage*, no. January 2012. https://doi.org/10.1016/c2016-0-03949-9.

Pannell, D. J., G. R. Marshall, N. Barr, A. Curtis, F. Vanclay, and R. Wilkinson. 2006. "Understanding and Promoting Adoption of Conservation Practices by Rural Landholders." *Australian Journal of Experimental Agriculture*. CSIRO. https://doi.org/10.1071/EA05037.

Parrotta, John A, Christoph Wildburger, and Stephanie Mansourian. 2012. *Understanding Relationships between Biodiversity, Carbon, Forests and People: The Key to Achieving REDD+ Objectives. A Global Assessment Report Prepared by the Global Forest Expert Panel on Biodiversity, Forest Management and REDD+*. *IUFRO World Series*. Vol. 31. https://doi.org/1016-3263.

Pascual, Unai, Roldan Muradian, Luke Brander, Erik Gómez-Baggethun, Berta Martín-López, Madhu Verma, Paul Armsworth, et al. 2012. "The Economics of Valuing Ecosystem Services and Biodiversity." In *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. https://doi.org/10.4324/9781849775489.

Pearce, David, and Dominic Moran. 1994. "The Economic Value of Biodiversity. IUCN Report." *Diversity*.

Pendrill, Florence, U. Martin Persson, Javier Godar, and Thomas Kastner. 2019. "Deforestation Displaced: Trade in Forest-Risk Commodities and the Prospects for a Global Forest Transition." *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/ab0d41.

Perrot-Maitre, D. 2006. "Microsoft Word - Vittel Web Version 1-12-06.Doc." *Microsoft Word - Vittel Web Version 1-12-06.Doc*, no. September: 24. http://pubs.iied.org/pdfs/G00388.pdf%0Ahttp://scholar.google.com/scholar?q=Microsoft Word - Vittel web version 1-12-06.doc&btnG=&hl=en&num=20&as_sdt=0%2C22 VN - readcube.com.

Pindyck, Robert S. 2013. "Climate Change Policy: What Do the Models Tell Us?" *Journal of*

*Economic Literature*. https://doi.org/10.1257/jel.51.3.860.

Ploeg, S. Van der, and R. S. de Groot. 2010. "Ecosystem Service Valuation Database." 2010. https://www.es-partnership.org/services/data-knowledge-sharing/ecosystem-service-valuation-database/.

Plummer, Mark L. 2009. "Assessing Benefit Transfer for the Valuation of Ecosystem Services." *Frontiers in Ecology and the Environment* 7 (1): 38–45. https://doi.org/10.1890/080091.

Potschin, Marion, Conor Kretsch, Roy Haines-Young, Eeva Furman, and Baró Francesc. 2016. "Nature-Based Solutions. In: Potschin, M. and K. Jax (Eds): OpenNESS Ecosystem Services Reference Book.," 1–5. http://www.openness-project.eu/sites/default/files/SP_Nature-based-solutions.pdf.

Rausser, Gordon C., and Arthur A. Small. 2000. "Valuing Research Leads: Bioprospecting and the Conservation of Genetic Resources." *Journal of Political Economy*. https://doi.org/10.1086/262115.

Raymond, Christopher M., Berry Pam, Margaretha Breil, Mihai R. Nita, Nadja Kabisch, Mark de Bel, Vera Enzi, et al. 2017. *An Impact Evaluation Framework to Support Planning and Evaluation of Nature-Based Solutions Projects. EKLIPSE Expert Working Group Report*. https://doi.org/10.13140/RG.2.2.18682.08643.

Reed, James, Josh van Vianen, Samson Foli, Jessica Clendenning, Kevin Yang, Margaret MacDonald, Gillian Petrokofsky, Christine Padoch, and Terry Sunderland. 2017. "Trees for Life: The Ecosystem Service Contribution of Trees to Food Production and Livelihoods in the Tropics." *Forest Policy and Economics* 84: 62–71. https://doi.org/10.1016/j.forpol.2017.01.012.

Reid, Hannah, Xiaoting Hou Jones, Ina Porras, Charlotte Hicks, Sylvia Wicander, Nathalie Seddon, Val Kapos, Ali R Rizvi, and Dilys Roe. 2019. *Is Ecosystem-Based Adaptation Effective ?*

Renard, Delphine, Jeanine M. Rhemtull, and Elena M. Bennett. 2015. "Historical Dynamics in

Ecosystem Service Bundles." *Proceedings of the National Academy of Sciences of the United States of America*. https://doi.org/10.1073/pnas.1502565112.

Ricketts, Taylor H., and Eric Lonsdorf. 2013. "Mapping the Margin: Comparing Marginal Values of Tropical Forest Remnants for Pollination Services." *Ecological Applications* 23 (5): 1113–23. https://doi.org/10.1890/12-1600.1.

Rudel, Thomas K., Oliver T. Coomes, Emilio Moran, Frederic Achard, Arild Angelsen, Jianchu Xu, and Eric Lambin. 2005. "Forest Transitions: Towards a Global Understanding of Land Use Change." *Global Environmental Change Part A* 15 (1): 23–31. https://doi.org/10.1016/j.gloenvcha.2004.11.001.

Rudel, Thomas K, Marla Perez-Lugo, and Heather Zichal. 2000. "When Fields Revert to Forest: Development and Spontaneous Reforestation in Post-War Puerto Rico." *The Professional Geographer* 52 (3): 386–97. https://doi.org/10.1111/0033-0124.00233.

Schmitz, M.F., D.G.G. Matos, I. De Aranzabal, D. Ruiz-Labourdette, and F.D. Pineda. 2012. "Effects of a Protected Area on Land-Use Dynamics and Socioeconomic Development of Local Populations." *Biological Conservation* 149 (February): 122–35. https://doi.org/10.1016/j.biocon.2012.01.043.

Searchinger, Tim, Richard Waite, Tim Beringer, Agneta Forslund, Hervé Guyomard, Chantal Le Mouël, Stéphane Manceron, and Elodie Marajo-Petitzon. 2018. *Creating a Sustainable Food Future*. Washington, D.C.: World Resources Institute. https://www.wri.org/our-work/project/world-resources-report/publications.

Seddon, Nathalie, Alexandre Chausson, Pam Berry, and Cécile A J Girardin. 2019. "Understanding the Value and Limits of Nature-Based Solutions to Climate Change and Other Global Challenges."

Seddon, Nathalie, Hannah Reid, Edmund Barrow, Charlotte Hicks, Xiaoting Hou-Jones, Val Kapos, Ali Raza Rizvi, Dilys Roe, Unep-Wcmc Ali, and Raza Rizvi. n.d. "Ecosystem-Based Approaches to Adaptation: Strengthening the Evidence and Informing Policy Research Overview and Overarching Questions." www.iied.org%5Cnwww.iied.org.

Seddon, Nathalie, Sandeep Sengupta, Irina Hauler, and Ali Raza Rizvi. 2020. "Nature-Based Solutions in Nationally Determined Contributions."

Sekulova, Filka, and Isabelle Anguelovski. 2017. "The Governance and Politics of Nature-Based Solutions," no. May. https://doi.org/RBM EXP. No. 980177.

Seppelt, Ralf, Carsten F. Dormann, Florian V. Eppink, Sven Lautenbach, and Stefan Schmidt. 2011. "A Quantitative Review of Ecosystem Service Studies: Approaches, Shortcomings and the Road Ahead." *Journal of Applied Ecology*. https://doi.org/10.1111/j.1365-2664.2010.01952.x.

Siikamäki, Juha, Francisco J Santiago-Ávila, and Peter Vail. 2015. "Global Assessment of Nonwood Forest Ecosystem Services:  Spatially Explicit Meta-Analysis and Benefit Transfer to Improve the World Bank's Forest Wealth Database," 97. www.profor.info/knowledge.

Simpson, David, Roger A Sedjo, John W Reid, R David Simpson, Roger A Sedjo, and John W Reid. 1996. "Valuing Biodiveristy for Use in Pharmaceutical Research." *The Journal of Political Economy*. https://doi.org/10.1086/262021.

Simpson, R. David. 2014. "Ecosystem Services as Substitute Inputs: Basic Results and Important Implications for Conservation Policy." *Ecological Economics*. https://doi.org/10.1016/j.ecolecon.2013.12.012.

———. 2019. "Conservation Incentives from an Ecosystem Service: How Much Farmland Might Be Devoted to Native Pollinators?" *Environmental and Resource Economics* 73 (2): 661–78.

Soulé, Michael. 2013. "The 'New Conservation.'" *Conservation Biology*. https://doi.org/10.1111/cobi.12147.

Stagakis, Stavros, Giorgos Somarakis, and Naktarios Chrysoulakis. 2019. "ThinkNature Nature Based Solutions Handbook." *ThinkNature Project Funded by the EU Horizon 2020 Research and Innovation Programme*, no. 730338: 1–226.

Stiglitz, Joseph, Nicholas Stern, Maosheng Duan, Ottmar Edenhofer, Gaël Giraud, Geoffrey Heal, Emilio Lèbre la Rovere, et al. 2017a. "Report of the High-Level Commission on Carbon

Prices." *Ipcc*, 68. https://doi.org/10.4103/2231-4040.101015.

———. 2017b. "Report of the High-Level Commission on Carbon Prices." *Ipcc*. https://doi.org/10.4103/2231-4040.101015.

Strengers, Bart J., Jelle G. Van Minnen, and Bas Eickhout. 2008. "The Role of Carbon Plantations in Mitigating Climate Change: Potentials and Costs." *Climatic Change*. https://doi.org/10.1007/s10584-007-9334-4.

Teeb. 2009. *The Economics of Ecosystems & Biodiversity*. *Teeb*. https://doi.org/ISBN 978-3-9813410-0-3.

Terborgh, John. 1999. *Requiem for Nature*. Washington: Island Press.

Tilman, David, Christian Balzer, Jason Hill, and Belinda L. Befort. 2011. "Global Food Demand and the Sustainable Intensification of Agriculture." *Proceedings of the National Academy of Sciences* 108 (50): 20260–64. https://doi.org/10.1073/PNAS.1116437108.

Tilman, David, Forest Isbelll, and Jane M. Cowles. 2014. "Biodiversity and Ecosystem Functioning." *Annual Review of Ecology, Evolution, and Systematics* 45 (1): 471–93.

Tinbergen, Jan. 1952. *On the Theory of Economic Policy*. New York: North-Holland.

Turner, P. A., K. J. Mach, D. B. Lobell, S. M. Benson, E. Baik, D. L. Sanchez, and C. B. Field. 2018. "The Global Overlap of Bioenergy and Carbon Sequestration Potential." *Climatic Change* 148 (1–2): 1–10. https://doi.org/10.1007/s10584-018-2189-z.

Turner, Will R. 2018. "Looking to Nature for Solutions." *Nature Climate Change*. https://doi.org/10.1038/s41558-017-0048-y.

U.S. EPA. 2014. "Guidelines for Preparing Economic Analyses, National Center for Environmental Economics Office of Policy - U.S. Environmental Protection Agency." *National Center for Environmental Economics Office of Policy - U.S. Environmental Protection Agency* 2010 (May): 1–302.

UNFCCC. 2019. "The Nature-Based Solutions for Climate Manifesto," no. August: 1–2.

Waldram, M.S., W.J. Bond, and W.D. Stock. 2008. “Ecological Engineering by a Mega-Grazer: White Rhino Impacts on a South African Savanna.” *Ecosystems* 11: 101–12.

Walters, Gretchen Marie. 2016. *Nature-Based Solutions to Address Global Societal Challenges*. *Nature-Based Solutions to Address Global Societal Challenges*. https://doi.org/10.2305/iucn.ch.2016.13.en.

Wheeler, Charlotte E. n.d. “Regenerate Natural Forests to Store Carbon,” 3–6.

Wild, T. C., J. Henneberry, and L. Gill. 2017. “Comprehending the Multiple ‘Values’ of Green Infrastructure – Valuing Nature-Based Solutions for Urban Water Management from Multiple Perspectives.” *Environmental Research* 158 (May): 179–87. https://doi.org/10.1016/j.envres.2017.05.043.

William, Reshmina, Allison Goodwell, Meredith Richardson, Phong V.V. Le, Praveen Kumar, and Ashlynn S. Stillwell. 2016. “An Environmental Cost-Benefit Analysis of Alternative Green Roofing Strategies.” *Ecological Engineering*. https://doi.org/10.1016/j.ecoleng.2016.06.091.

Williamson, Phil. 2016. “Emissions Reduction: Scrutinize CO2 Removal Methods.” *Nature* 530 (7589): 153–55. https://doi.org/10.1038/530153a.

Wilson, E. O. 1988. *Biodiversity*. National Academies Press (U. S.).

Wu, Jun Jie. 2000. “Slippage Effects of the Conservation Reserve Program.” *American Journal of Agricultural Economics*. https://doi.org/10.1111/0002-9092.00096.

Ziter, Carly D., Eric J. Pedersen, Christopher J. Kucharik, and Monica G. Turner. 2019. “Scale-Dependent Interactions between Tree Canopy Cover and Impervious Surfaces Reduce Daytime Urban Heat during Summer.” *Proceedings of the National Academy of Sciences of the United States of America* 116 (15): 7575–80. https://doi.org/10.1073/pnas.1817561116.