Chapter 9: Ecosystem Services and Wealth Accounting

Inclusive Wealth Report

Sponsors: UNU-IHDP
UNEP
UN-Water Decade Programme on Capacity Development (UNW-DPC)
Natural Capital Project, Stanford University.

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January 15, 2012

Abstract

The aim of this chapter is to examine the methodology and the challenges of including ecosystem services in a wealth accounting framework. The chapter draws on the general inclusive wealth framework of treating nature as a capital asset as discussed by Dasgupta (2009 and 2012), and elaborated for biodiversity and ecosystem services by Perrings (2012). As both authors emphasize, the main challenges in incorporating ecosystem services in wealth accounting is in determining the ecological production of ecosystem goods and services, or benefits, and the appropriate values, or shadow prices, attributed to such benefits, which are often non-marketed. Barbier (2008 and 2011a) also suggests that, in order to treat ecosystems as a form of wealth generating a flow of benefits, one first has to adopt a useful "metric" for measuring ecosystems as a form of natural capital, as well as consider ecological collapse and resilience as potential influences on this stock of wealth. Overcoming these issues and challenges is the main theme of this chapter.

Introduction

The growing scarcity of ecosystem goods and services, or *ecological scarcity*, indicates that an important source of economic wealth, the world's ecosystems, is being irreversibly lost or degraded (Barbier 2011a). Over the past 50 years, ecosystems have been modified more rapidly and extensively than in any comparable period in human history, largely to meet rapidly growing demands for food, fresh water, timber, fiber and fuel. The result has been a substantial and largely irreversible loss in biological diversity, ecosystems and ecological services that they

[1] I am grateful to Pablo Munoz and two anonymous referees for helpful comments on an earlier draft of this chapter.

provide. Approximately 15 out of 24 major global ecosystem services have been degraded or used unsustainably, including freshwater, capture fisheries, air and water purification, and the regulation of regional and local climate, natural hazards, and pests (MA 2005). Over the next 50 years, the rate of biodiversity loss is also expected to accelerate, leading to the extinction of at least 500 or the 1,192 currently threatened bird species and 565 of the 1,137 mammal species (Dirzo and Raven 2003).

An important contribution of the Millennium Ecosystem Assessment was to define ecosystem goods and services as valuable "benefits" to humans and to highlight the deteriorating state of many global ecosystems and their key services (MA 2005). As a US National Research Council Report points out, "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values" (NRC 2005, p. 2). The main reason for this challenge is the "lack of multiproduct, ecological production functions to quantitatively map ecosystem structure and function to a flow of services that can then be valued" (Polasky and Segerson 2009, p. 422).

Despite these valuation problems, the consensus in the literature is that ecosystems are assets that produce a flow of beneficial goods and services over time.[2] For example, as Daily et al. (2000, p. 395) state, "the world's ecosystems are capital assets. If properly managed, they yield a flow of vital services, including the production of goods (such as seafood and timber), life support processes (such as pollination and water purification), and life-fulfilling conditions (such as beauty and serenity)." Ecosystems should therefore be treated as an important asset in an economy, and in principle, ecosystem services should be valued in a similar manner as any form of wealth. That is, regardless of whether or not there exists a market for the goods and services produced by ecosystems, they make contributions to current and future wellbeing. The importance of this economic contribution of ecosystems has become the focal point of recent international and national studies, such as the Economics of Ecosystems and Biodiversity (TEEB 2010) and the UK National Ecosystem Assessment (2011).

As chapters in this report have stressed, accounting for the depreciation of ecological assets is essential to any inclusive wealth accounting framework (See, especially, Dasgupta 2012; Perrings 2012; Tallis and Polasky 2012). However, a major difficulty arising in treating ecosystems as economic assets is in quantifying this form of capital and in measuring the valuable benefits that it produces (Barbier 2008, 2011a and 2011b; Mäler et al. 2008; Polasky and Segerson 2009). The valuation challenge is further exacerbated by the difficulty in determining the ecological production of many ecosystem goods and services and in observing values for the myriad economic benefits, many of which are non-marketed. The purpose of this chapter is to review progress in economics and ecology in assessing ecosystem services and their values, and to discuss the resulting implications for including such services in a wealth accounting framework.

The next section provides an overview of the wealth accounting method suggested for natural capital by Dasgupta (2009 and 2012) and Perrings (2012), and discusses how ecosystem services can be incorporated into this framework. The subsequent section examines further how ecosystems can be characterized as economic by adopting ecological landscape, or land area, as

[2] See, for example, Barbier (2008) and (2011a); Daily et al. (2000); EPA (2009); MA (2005); NRC (2005); Polasky and Segerson (2009); TEEB (2010); WRI (2001).

the basic measuring unit. As the next section explains, understanding the relationship between ecosystems, their structure and functions, and the ecological services they generate is essential to determining how the structure and functions of an ecosystem provide valuable goods and services to humans. The section that follows discusses how the economic concept of a "benefit" should be applied to ecosystem goods and services as a guide to their correct economic valuation through integrating the "ecological production" of ecosystem goods and services with "economic valuation" of these benefits. The subsequent section overviews the substantial progress that has been made by economists working with ecologists and other natural scientists on this "fundamental challenge" to improve the application of environmental valuation methodologies to nonmarket ecosystem services. To illustrate, the next section provides an example of inclusive wealth accounting with the example of the case study from Thailand involving mangrove loss, based on Barbier (2007). The penultimate section discusses uncertainty, irreversible loss, resilience and other important issues to consider in valuing ecosystem services. The conclusion to this chapter offers some final remarks on how incorporating ecosystem services in wealth accounting can be further improved.

Wealth accounting and ecosystem services

Following the framework developed by Dasgupta (2009), previous chapters in this report suggest a common wealth accounting methodology for natural capital, including ecosystems (Dasgupta 2012 and Perrings 2012). Such an accounting framework defines the aggregate wealth as the shadow value of the stocks of all the assets of an economy, and suggests that ecosystems should be included as an important form of "natural capital" in this wealth. Moreover, the aggregate wealth of the economy will increase over time only if current consumption is less than the net national (or domestic) product, provided that the latter correctly accounts for the economic contributions of all capital, including ecosystems. The following section summarizes the basic wealth accounting principles arising from this methodology, extends it to incorporate ecosystems and their valuable goods and services, and thus constructs an adjusted measure of net domestic product (NDP) that accounts for the additional contributions of ecological capital.

For most economies, the standard measure of economic progress is real per capita gross domestic product (GDP), the market value of all final goods and services produced within the economy.[3] The problem with GDP as an economic indicator, however, is that it does not reflect changes in the capital stock underlying the production of goods and services.[4] GDP accounts for gross investment in an economy but for any depreciation in existing capital. Since the purpose of new investment is to increase the quantity and quality of the economy's total capital stock, or wealth, adjusting GDP for depreciation in this stock would measure more accurately whether net

[3] The alternative measure to gross domestic product is gross national product (GNP), which is the GDP of an economy plus net income from abroad. The latter net payments consist of the income that people and organizations resident in the domestic economy receive from abroad on account of property and other assets which they own in foreign economies less the income paid to non-residents from their holdings of property and assets in the domestic economy. Thus, GDP is the total income of an economy produced domestically, whereas GNP is the total gross income received by the residents of an economy.

[4] For further discussion of the limitations of GDP as an economic indicator, see the chapter "Beyond GDP" in this report by Kumar and Dasgupta (2012).

additions to capital are occurring. And, as has been demonstrated, economic development is sustained if and only if such investment in overall wealth is non-negative over time (Dasgupta 2009; Dasgupta and Mäler 2000; Hamilton and Clemens 1999).

The idea of deducting capital depreciation from GDP to obtain a "net" domestic product (NDP) measure is not new. Lindahl (1933) first provided the justification by suggesting that an economy's income should exceed current consumption, including any consumption of existing capital, to prevent the economy's total wealth from declining. However, the aggregate stock of economic assets should be much broader than conventional reproducible (or fixed) assets, such as roads, buildings, machinery and factories. Investments in human capital, such as education and skills training, are also essential to sustaining development. Similarly, an economy's endowment of natural resources is an important form of "natural wealth". Thus, a better indicator of an economy's progress would be an expanded measure of NDP that is "adjusted" for real depreciation in reproducible and natural capital, as well as any net additions to human capital, such as through real education, health and training expenditures in the economy.[5]

In economics, and in systems of national accounts, "capital" is conventionally defined as *reproducible real assets*, which includes roads, railways, buildings, private dwellings, factories, machinery, equipment and other human-manufactured fixed assets. Thus, *investment* in the economy, or *gross capital formation*, is conventionally measured as outlays or additions to these reproducible assets plus net changes in the level of inventories and valuables. If allowance is made for any capital consumption, or depreciation, then the net changes in reproducible assets represent net investment in the economy.

However, the economy does not just depend on reproducible assets but also human and natural capital. Traditionally, investment in human capital, which can be thought of the education, skills and health per person, are not included in the national accounts. Similarly, additions to and depreciation of natural capital are excluded. In a true wealth accounting framework to estimated the NDP of an economy, both of these omissions need correcting. That is, the three basic assets comprising the overall wealth of an economy are reproducible, human and natural capital.

More formally, assume a closed economy with a constant population that is normalized to one.[6] At time t, let $K(t)$ be a numerical index of the economy's stock of reproducible capital assets, and $H(t)$ be a numerical index of the total quantity of human capital, i.e. the level of health, education and skills per person. Reproducible capital depreciates at the constant rate $\omega > 0$, and assume that $E(t)$ is investment in human capital (e.g. current education, health and training expenditures). Denoting the real GDP of the economy at time t as $Y(t)$ and aggregate consumption of goods and services as $C(t)$, then net accumulation of reproducible capital is

$$\dot{K} = Y(t) - C(t) - \omega K(t) - E(t), \quad \dot{K} = dK(t)/dt \,. \tag{1}$$

[5] See, for example, Aronsson and Löfgren (1996); Dasgupta (2009) and (2012); Hamilton and Clemens (1999); Hartwick (1990); Mäler (1991); Pearce and Barbier (2000); and Perrings (2012).

[6] As shown by Arrow et al. (2003) and Dasgupta (2009), the following analysis could accommodate population growth, but it is conceptually more difficult to do so.

Following Hamilton and Clemens (1999), letting $h\left(E\left(t\right)\right)$ represent the rate at which education, health and training investments are transformed into human capital, then the latter accumulates according to

$$\dot{H} = h\left(E\left(t\right)\right), \quad h' > 0, \quad \dot{H} = dH\left(t\right)/dt \tag{2}$$

Along with human and reproducible capital, the aggregate stock of natural capital available at time *t* is also important to the economy. Clearly, this stock must include those conventional natural resources that are the source of raw material, land and energy inputs to the economy, such as fossil fuels, minerals, metals, forest resources and arable land. If we represent these natural resource stocks as $S(t)$, and let

$$\dot{S} = G\left(S\left(t\right)\right) - R\left(t\right), \quad \dot{S} = dS\left(t\right)/dt \tag{3}$$

where the function G represents the natural growth rate for any renewable resources, and $R(t)$ is the use of any natural resource inputs by the economy.

But, in addition to $S(t)$, natural capital should include those ecosystems that through their natural functioning and habitats provide important goods and services to the economy. As suggested by Barbier (2007), these benefits are wide-ranging, which in economics would normally be classified under three different categories:

(i) "goods" (e.g., products obtained from ecosystems, such as resource harvests, water and genetic material),

(ii) "services" (e.g., recreational and tourism benefits or certain ecological regulatory and habitat functions, such as water purification, climate regulation, erosion control and habitat provision), and

(iii) cultural benefits (e.g., spiritual and religious beliefs, heritage values).

It is clear that some of these ecosystem goods and services contribute *directly* to human well-being, e.g. through enhancing recreation and other direct enjoyment of the environment, augmenting our current and future natural heritage or by reducing harmful pollution and assimilating waste. But some services, either on their own or combined with human inputs, also contribute *indirectly* to human welfare by supporting economic production (e.g. raw materials, food and other harvested inputs; provision of freshwater, watershed protection, coastal habitats for off-shore fisheries) or by protecting production activities, property and lives (e.g. flood control, storm protection, managing climate). In other words, "ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations" (EPA 2009, p. 12).

However, as noted in the introduction, ecosystems globally are under threat from degradation and loss. Global land use change has been a major cause of the alteration and loss of terrestrial ecosystems, especially in developing economies and tropical regions (Barbier 2011a; Dirzo and Raven 2003; FAO 2006; MA 2005). Coastal and marine ecosystems are also some of the most heavily used and threatened natural systems globally, such that 50% of salt marshes, 35% of mangroves, 30% of coral reefs and 29% of seagrasses are either converted or degraded worldwide (FAO 2007b; MA 2005; Orth et al. 2006; UNEP 2006; Valiela et al. 2001; Waycott et al. 2009). The major reason for this loss is land conversion, such as the transformation of forests and wetlands to crop and grazing land, expansion of aquaculture and agriculture in coastal areas, and the demand for land for urban and commercial development. In national accounting terms, the implication is that the depreciation of an important natural asset – ecosystems – is partly compensated for the appreciation of another asset – more land for economic production and development. As Hartwick (1992) has illustrated with the example of agricultural conversion of tropical forests, such changes in the stock of an economy's wealth must be included as capital value adjustments in an accounting framework. In effect, the opportunity cost of holding on to ecosystems as natural capital is the foregone benefits of economic development based on converting ecological landscape (Barbier 2008 and 2011a).

In sum, ecosystems can be considered a component of natural capital - or *ecological capital* for short - that affects current economic well-being, either directly or indirectly through supporting production and protecting human lives and property. However, ecological capital is unlikely to be intact, as in most economies ecosystems continue to be converted to land for economic development and production. It follows that the aggregate stock of developed land, $D(t)$, increases at the expense of ecological capital $N(t)$

$$\dot{D} = c(t) = -\dot{N}, \quad \dot{D} = dD(t)/dt, \quad \dot{N} = dN(t)/dt \tag{4}$$

where $c(t) \geq 0$ represents any ecosystem conversion to developed land at time t.

Following Dasgupta (2009), let $A(t)$ be a combined index of publicly known ideas and the effectiveness of the economies institutions, which can be interpreted as *total factor productivity* in the economy at time t. Given equations (1)-(4), the economy's real GDP, denoted as $Y(t)$, can be stated as

$$Y(t) = A(t)F\left(K(t), H(t), R(t), D(t), N(t)\right), \tag{5}$$

where F is a non-decreasing and twice differentiable function, and $F = 0$ if any of its arguments are zero.[7] Note that the production function of the economy should include ecological capital, $N(t)$, given that many ecosystem services support and protection production activities.

Letting $V(t)$ denote intergenerational well-being at time t, which takes the form

$$V(t) = \int_t^{\infty} U\left(C(\tau), N(\tau)\right) e^{-\delta(\tau - t)} d\tau \tag{6}$$

where $\delta > 0$ is the social rate of discount. Note that inter-generational welfare depends not only on aggregate consumption but also on the direct benefits of ecosystems, which are represented by the inclusion of $N(t)$ in the function for instantaneous well-being, or "utility", $U(t)$. It is assumed that the latter function is twice differentiable, additively separable and concave with respect to its two arguments.

As Dasgupta (2009) proves, regardless of whether or not the resource allocation mechanism of the economy is optimal or even efficient, given (1)-(6), for any such mechanism it is possible to define a set of shadow prices at time t for the various assets of the economy

$$v^i(t) = \partial V(t)/\partial i(t), \quad i = K,\ H,\ S, N, D. \tag{7}$$

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[7] As Dasgupta (2009) points out, unlike a standard neoclassical production function, F is not necessarily concave.

Given these shadow prices, the economy's aggregate, or *inclusive*, wealth $W(t)$ and investment $I(t)$ at time t are, respectively

$$W(t) = v^K(t)K(t) + v^H H(t) + v^S S(t) + v^N N(t) + v^D D(t) \tag{8}$$

and $$I(t) = v^K(t)\dot{K} + v^H \dot{H} + v^S \dot{S} + v^N \dot{N} + v^D \dot{D} \quad . \tag{9}$$

The current-value Hamiltonian that ensures intergenerational wellbeing (6) is at a maximum for any given resource allocation mechanism of the economy is therefore

$$\mathrm{H}(t) = U(C(t), N(t)) + I(t) = \delta V(t) \tag{10}$$

The current-value Hamiltonian as specified in (10) is therefore an indicator of the return on intergenerational well-being, regardless of whether or not the resource allocation mechanism of the economy is efficient or optimal.[8]

By expressing the utility function $U(t)$ as $U(C(t), N(t)) = U_C C(t) + U_N N(t)$, equation (10) can be used to define aggregate or inclusive NDP of the economy at time t in "utils"

$$NDP(t) = U_C C(t) + U_N N(t) + I(t). \tag{11}$$

Equation (11) depicts NDP as the sum of investment in the aggregate capital stocks of an economy plus the value of consumption and ecosystem goods and services. Following an approach analogous to Dasgupta (2009), NDP as defined by (11) can also be used as an indicator for measuring whether intergenerational wellbeing in an economy is improving or not over time.

Differentiating (6) with respect to time yields $dV(t)/dt = \delta V(t) - U(C(t), N(t))$. Using the latter expression in (10), one obtains

$$\frac{dV(t)}{dt} = I(t). \tag{12}$$

Condition (12) states that investment in the aggregate capital stock of an economy determines changes in intergenerational wellbeing over time, and as a result, NDP as defined by (11) is an exact measure of these welfare changes. That is, (11) and (12) yield a condition akin to Proposition 9 in Dasgupta (2009): $dV(t)/dt \geq 0$ if and only if $NDP(t) \geq U_C C(t) + U_N N(t)$. As long as NDP exceeds the value of consumption and ecosystem goods and services, intergenerational welfare will not decline.

This condition has an important economic interpretation, given that non-declining welfare is the crucial criterion defining sustainable development of an economy.[9] As $dV(t)/dt \geq 0$ also implies $I(t) \geq 0$, then it follows from (8) that sustainable economic development will occur at time t if the aggregate wealth of the economy $W(t)$ does not decline.

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[8] $\mathrm{H}(t) = \delta V(t)$ can be found by integrating the current-value Hamiltonian $\mathrm{H}(t) = U(C(t),N(t)) + I(t)$. See Dasgupta (2009).

[9] For example, Pearce et al. (1989, p. 32) state: "the wellbeing of a defined population should be at least constant over time and, preferably, increasing for there to be sustainable development."

Thus, the sustainability criterion that "welfare does not decline over time" essentially "requires managing and enhancing a portfolio of economic assets, the *total capital stock*, such that its aggregate value does not decline over time", but only if it is recognized that "the total stock of the economy available to the economy for producing goods and services, and ultimately well-being, consists not just of human and physical capital but also of natural capital" (Pearce and Barbier 2000, pp. 20-21).

To understand the importance of measuring explicitly the contributions of natural capital, and especially that of ecological capital, it is necessary to decompose NDP as defined by (11). Using the first-order conditions for maximizing the current-value Hamiltonian (10) with respect to $C(t)$ and $E(t)$, (11) can be rewritten (suppressing the time arguments) as

$$\begin{aligned} NDP &= v^K\left[C+\dot{K}+\frac{h(E)}{h'}\right]+U_N N+v^S\dot{S}+v^N\dot{N}+v^D\dot{D} \\ &= v^K\left[Y-\omega K\right]+v^K\left(\frac{h(E)}{h'}-E\right)+v^K AF_R\left[G(S)-R\right]+U_N N+\left(v^D-v^N\right)c \end{aligned} \tag{13}$$

In (13), the expression $v^K(t)\left[Y(t)-\omega K(t)\right]$ is conventionally defined net domestic product, i.e. the gross domestic product of the economy less any depreciation (in value terms) of previously accumulated reproducible capital. This is NDP as currently measured in most national accounts of economies, although of course it is usually valued at market prices rather than in terms of the shadow price of reproducible capital. It is clear from (13) that, if NDP is to serve as a true measure of the changes in an economy's wealth, it must include any appreciation or depreciation to human and natural capital as well. For instance, $v^K(t)\left(h\left(E(t)\right)/h'-E(t)\right)$ is the net appreciation (in value terms) in human capital, and $v^K(t)A(t)F_R\left[G\left(S(t)\right)-R(t)\right]$ represents the net changes (in value terms) in natural resource stocks.[10] In the case of non-renewable resources, such as fossil fuels and minerals, $G(S)=0$ and so $-v^K AF_R R$ measures the deduction from NDP of resource depletion. For renewable resources, such as forests and fisheries, NDP must include any depreciation in natural resource stocks if $G(S)<R$. The expression $U_N N(t)+\left[v^D(t)-v^N(t)\right]c(t)$ includes both the benefits to current well-being provided by ecosystems, $U_N N$, and any capital revaluation that occurs as ecosystems are converted by land use change for development, $\left(v^D-v^N\right)c$.[11] To interpret the latter term, it is

[10] In (13), it is assumed that v^S accounts for the marginal cost of resource extraction or harvesting. For example, suppose that such costs can be represented by the function $f(R),\ f_R>0$, which are in turn paid out of an economy's gross domestic product, *Y*. It follows from the first-order condition for maximizing the current-value Hamiltonian (10) $\partial \mathrm{H}/\partial R=0$ that $v^S=v^K\left[AF_R-f_r\right]$, or equivalently, $v^K AF_R=v^S+v^K f_r$.

[11] In (13), it is assumed that v^D accounts for the marginal costs of converting ecosystems to land for development. For example, if such costs are represented by $g(c),\ g_c>0$ and deducted from the economy's gross domestic product, *Y*, then it follows from the first-order condition of maximizing (10) $\partial \mathrm{H}/\partial c=0$ that $v^D=v^N+v^K g_c$, or equivalently $v^D-v^N=v^K g_c$. However, as will be discussed presently, as the resource allocation mechanism of the economy may

helpful to explore further the shadow value of ecological capital $v^N(t)$ and developed land $v^D(t)$, respectively.

By definition, from (10), $v^N(t) = \int_t^\infty \frac{\partial H}{\partial N}(\tau) e^{-\delta(\tau - t)} d\tau$ and $v^D(t) = \int_t^\infty \frac{\partial H}{\partial D}(\tau) e^{-\delta(\tau - t)} d\tau$. It follows that

$$v^D(t) - v^N(t) = \int_t^\infty e^{-\delta(\tau-t)} v^K(\tau) A(\tau) F_D(\tau) d\tau - \int_t^\infty e^{-\delta(\tau-t)} \left[U_N(\tau) + v^K(\tau) A(\tau) F_N(\tau) \right] d\tau \quad . \quad (14)$$

Thus, $v^D(t)$ is the present value of any additional production resulting from any increase in land for economic development land, whereas $v^N(t)$ is the present value of any additional ecosystem benefits due to increases in ecosystem land. That is, $v^D(t)$ and $v^N(t)$ are the capitalized values, or *prices*, of development and ecosystem land, respectively. As ecosystems are converted by land use change for development, $(v^D - v^N)c$ is the capital appreciation (depreciation) in land that occurs if $v^D > v^N$ $(v^D < v^N)$. As land is a durable and capital good, condition (13) indicates that NDP must be adjusted for any such capital revaluation.

To summarize, although conditions (13) and (14) seem complicated, they help clarify how we should value and include changes in ecological capital in wealth accounting. First, we should adjust the NDP of the economy to include two contributions due to ecological capital:

- the value of the *direct benefits* provided by the current stock of ecosystems, $U_N N$, and
- any capital revaluation as a result of conversion of ecosystems to other land uses, $(v^D - v^N)c$, with the price of changes in ecological capital, $v^N(t)$, reflecting the present value of the future *direct and indirect benefits* of ecosystems.[12]

As discussed previously, the direct ecosystem benefits might include the value of ecosystems in providing recreational, educational and scientific benefits, their value in terms of natural heritage or bequests to future generation or the value of ecosystems in reducing harmful pollution, assimilating waste and managing climate. In addition, ecological capital protects or supports economic activity, property and human lives. These indirect ecosystem benefits are broad ranging, and include raw materials, food and other harvested inputs used in production, provision of freshwater, watershed protection, coastal habitats for off-shore fisheries, flood control, storm protection, climate stabilization, and similar services.

In the wealth accounting framework adopted here, the resource allocation mechanism of the economy may not be optimal or even efficient, so it is possible that ecosystem conversion

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not be optimal or even efficient, ecosystem conversion may not take place so that the difference between the price of developed land and the capitalized value of ecosystems is just equal to marginal cost of conversion.

[12] These adjustments to NDP for ecological capital are similar to those for environmental resource stocks derived by Mäler (1991). It appears that, although ecosystems generate a wide variety of complex goods and services, the actual rules for determining how the direct and indirect benefits of ecological capital should be accounted for in NDP are no different than for any stock that generates both affects human welfare directly or indirectly via supporting or protecting economic production.

may be taking place even though the capitalized value, or "price", of developed land is actually less than the capitalized value of ecosystems. In which case, as we have discussed, NDP should be adjusted for the depreciation in ecological capital that occurs as it is converted to less valuable developed land. But if ecosystems are an important component of natural capital, and if we want to adjust NDP to account for real depreciation in this form of natural wealth, then we need to find a way of, first, measuring such assets, and second, valuing the various benefit flows that they generate (Barbier 2008, 2011a and 2011b). The purpose of the next several sections is to discuss how best to overcome these measurement challenges. Later in the chapter, the example of mangrove loss in Thailand is used to illustrate the practical application of (13) to measure NDP correctly.

Ecosystems as natural capital

If we are to view ecosystems as economic assets, and measure their economic depreciation in wealth accounting, then we need a way of measuring this form of "ecological wealth" (Mäler et al. 2008). One barrier to such an approach is that, in ecology, the concept of an ecosystem has been difficult to define or to measure quantitatively (O'Neill 2001; Pickett and Cadenasso 2002).

However, it is increasingly recognized that most ecological processes are influenced by the spatial extent, or landscape, that defines the boundary of the system.[13] Similarly, the various coastal and marine ecosystems that make up the land-sea interface located between watersheds, the coast and oceans could be designated in terms of distinct *seascapes* that define the boundaries between each type of system (Moberg and Rönnbäck 2003; Shackeroff et al. 2009). Thus, as shown by (Barbier 2008 and 2011a), through adopting *ecological landscape*, or *land area*, as the basic unit, characterizing the ecosystem as a natural asset is relatively straightforward. It also facilitates the examination of human transformation of an ecological landscape through land use conversion, leaving the residual land for ecological processes and habitat for species through relatively straightforward models of land use change. This then facilitates measurement of the physical depreciation of ecosystems, which is essential if we are to account for how stocks of such wealth change.

To illustrate why the landscape containing an ecosystem might serve as the basic unit for measuring changes in this natural asset, it is helpful to discuss a specific example, such as wetland ecosystems. These systems, which comprise coastal wetlands, freshwater swamps and marshes (including floodplains), and peatlands, amount to 6 to 8 million km^2 globally (Mitsch et al. 2009). The goods and services provided by wetlands are uniquely related to hydrological processes. For example, seasonal soil-water regimes, surface inundation and maintenance of water quality, critically determine wetland ecosystem structure and function, and thus influence the type ecosystem goods and services provided. Similarly, changes in water regime will affect different wetland services significantly, resulting in many possible trade-offs and synergies among these services within different wetland scenarios and water regimes. The consequence is that the ecosystem services provided by wetlands are driven by hydrology, and understanding

[13] See, for example, Bockstael (1996); O'Neill (2001); Perry (2002); Pickett and Cadenasso (1995) and (2002); Turner (2005); and Zonneveld (1989).

how changes in hydrological processes affect the delivery of these services is critical to determining the impact on human welfare (Brauman et al. 2007; Bullock and Acreman 2003; Emerton and Boss 2008; Mitsch et al. 2009).

Because the structure and functions of many wetlands can be uniquely defined by hydrological processes, it is possible to identify the spatial unit, or natural landscape, that is distinct to each type of wetland. In particular, different aspects of the hydrological system underlying wetlands and their services operate at different scales, e.g. surface inundation (flooding), water quality and biodiversity. Thus, as a wetland landscape varies in scale, due perhaps to conversion, draining or other human-induced disturbances, the impact on the provision of and synergies between wetland services can be substantial. Such a landscape approach is being increasingly used for assessing the cumulative effects of wetland loss and degradation, characterizing wetland boundaries and identifying restoration or mitigation opportunities (Bedford 1996 and 1999; Gwin et al. 1999; Mitsch and Gosselink 2000; NRC 1995; Simenstad et al. 2006). It follows that the various goods and services provided by a wetland will also be tied to, and thus defined by, its landscape extent; i.e., "wetland values depend on the hydrogeomorphic location in which they are found" (Mitsch and Gosselink 2000, p. 27).

If the hydrological-related services of wetlands are related to their landscape extent, then characterizing wetland ecosystems as natural assets is straightforward. In other words, as there are "reciprocal interactions between spatial pattern and ecological processes" (Turner 2005, p. 319), it is the spatially heterogeneous area of a wetland landscape that is the fundamental to its ability to provide various goods and services. It follows that, if for each wetland ecosystem we can define its corresponding landscape in terms of a quantifiable "land unit", which is defined as "a tract of land that is ecologically homogeneous at the scale level concerned" (Zonneveld 1989, p. 68), then we have a representation of the wetland ecosystem as a natural asset in the form of this unit of land, or ecological landscape.

However, even with a well-defined ecological landscape one must be careful to account for heterogeneous units within such a landscape and to avoid problems of double counting. For example, large-scale forested ecosystems can also contain wetlands, freshwater channels and rivers. Similarly, qualitative features of the landscape may significantly influence the ecological production of benefits. For example, the timber benefits of a forested landscape may depend not only on the overall size of the system but also the spatial distribution of trees across the landscape in terms age, size and species as well as variations in soil quality and nutrients (Mäler et al. 2008). The ability of vegetated coastal landscapes to attenuate storm surges and protect against damages not only varies considerably at the seaward edge as opposed to further inland but also is affected by coastal geomorphology, elevation and topography (Koch et al. 2009). Finally, in a "mixed" ecological landscape, then it may be difficult to determine how a particular ecosystem benefit arises from the landscape and to avoid problems of double counting. For example, outdoor recreation values may be enhanced by the diverse ecological features of a mixed landscape, including the presence of wetlands, forests and river channels. How to separate out the specific contribution to the value of recreation provided by each ecological component of the landscape may be problematic. Nor would it be correct to attribute the full recreational value to each of the wetland, forest and river components of the landscape.

Ecosystems and ecosystem services

There is much confusion over the relationship between ecosystems, their structure and functions, and the ecological services they generate that contribute to human welfare. Understanding such a relationship is essential in order to determine how the structure and functions of an ecosystem provide valuable goods and services to humans.

An *ecosystem* has the characteristics of a "system", in the sense that it includes an assemblage of organisms interacting with its associated physical environment in a specific place (O'Neill 2001; Pickett and Cadenasso 2002). Thus, within its prescribed area or location, an ecosystem comprises its abiotic (nonliving) environment and the biotic (living) groupings of plant and animal species, or communities. The biotic and abiotic components, and the interactions between them, are often referred to as the *ecosystem structure*.

Two important *ecosystem functions* are carried out in every ecosystem: biogeochemical cycling and flow of energy. Important processes of biogeochemical cycling include primary production (photosynthesis), nutrient and water cycling, and materials decomposition. The flow, storage and transformation of materials and energy through the system are also influenced by processes that link organisms with each other, such as the food web, which is made up of interlocking food chains. These food chains are often characterized by other important functions, such as pollination, predation and parasitism.

The structure and functions of an ecosystem provide valuable goods and services to humans. For example, some of the living organisms found in an ecosystem might be harvested or hunted for food, collected for raw materials or simply valued because they are aesthetically pleasing. Some of the ecosystem functions, such as nutrient and water cycling, can also benefit humans through purifying water, controlling floods, recharging aquifers, reducing pollution, or simply by providing more pleasing environments for recreation. These various benefits provided by an ecosystem via its structure and functions are now referred to as *ecosystem services*. As summarized in Box 1, the structure and functions of an ecosystem are not synonymous with its services. Ecosystem structure and functions describe the components of an ecosystem and its biophysical relationship regardless of whether or not humans benefit from them. Only if they contribute to human well-being do these components and relationships generate an "ecosystem service".[14]

Assessing the value of ecosystem goods and services

The idea that ecosystems provide a range of "services" that have value to humans is an important step in characterizing these systems as "natural capital". In order to view ecosystems as a special type of capital asset – a form of "ecological wealth" – then just like any other asset or investment in the economy, ecosystems must be capable of generating current and future flows of income or benefits. It follows that, in principle, ecosystems can be valued just like any other asset in an economy. Regardless of whether or not there exists a market for the goods and services produced by ecosystems, their social value must equal the discounted net present value

[14] For more discussion, see Barbier (2011a and 2011b); Bockstael et al. (2000); Boyd and Banzhof (2007); EPA (2009); Polasky and Segerson (2009).

(NPV) of these flows. However, for economists, the term "benefit" has a specific meaning. This section discusses how this concept of economic benefit should be applied to ecosystem goods and services as a guide to their correct economic valuation. In addition, the section outlines the main approach that is required to integrate the “ecological production” of ecosystem goods and services with “economic valuation” of these benefits.

As noted previously, the literature on ecological services implies that ecosystems are assets that produce a flow of beneficial goods and services over time. For example, a common practice in this literature is to adopt the broad definition of the MA (2005) that “ecosystem services are the benefits people obtain from ecosystems.” However, for economists, the term "benefit" has a specific meaning. According to Mendelsohn and Olmstead (2009, p. 326): "The economic benefit provided by an environmental good or service is the sum of what all members of society would be willing to pay for it." Consequently, some economists argue that it is misleading to characterize all ecosystem services as "benefits." As explained by Boyd and Banzhaf (2007, p. 619), "as end-products of nature, final ecosystem services are not benefits nor are they necessarily the final product consumed. For example, recreation is often called an ecosystem service. It is more appropriately considered a benefit produced using both ecological services and conventional goods and services." To illustrate this point, they consider recreational angling. It requires certain "ecosystem services", such as "surface waters and fish populations" but also "other goods and services including tackle, boats, time allocation, and access" (Boyd and Banzhaf 2007, p. 619). But other economists still prefer the broader perspective of the MA (2005), which equates ecosystem services with benefits. For example, Polasky and Segerson (2009, p. 412) state: "We adopt a broad definition of the term ecosystem services that includes both intermediate and final services," which they justify by explaining that "supporting services, in economic terms, are akin to the infrastructure that provides the necessary conditions under which inputs can be usefully combined to provide intermediate and final goods and services of value to society." Thus, unlike Boyd and Banzhaf (2007), Polasky and Segerson (2009) consider recreation to be an ecosystem service.

Economists do agree that, in order to determine society's willingness to pay for the benefits provided by ecosystem goods and services, one needs to measure and account for their various impacts on human welfare. Or, as Bockstael et al. (2000, p. 1385) state: “In economics, valuation concepts relate to human welfare. So the economic value of an ecosystem function or service relates only to the contribution it makes to human welfare, where human welfare is measured in terms of each individual’s own assessment of his or her well-being.” The key is determining how changes in ecosystem goods and services affect an individual's well-being, and then determining how much the individual is either willing to pay for changes that have a positive welfare impact, or conversely, how much the individual is willing to accept as compensation to avoid a negative effect.

The starting point in identifying ecosystem services and their values is the consensus economic view outlined above. As long as nature makes a contribution to human welfare, either entirely on its own or through joint use with other human inputs, then we can designate this contribution as an “ecosystem service”. In other words. as stated in Box 1, "ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations." Although it is acceptable to use “the term ecosystem service to refer broadly to both intermediate and final end services”, “in specific valuation contexts…it is important to identify whether the service being valued is an intermediate or a final service" (EPA 2009, pp. 12-3).

Following this approach, for example, recreation can be considered the product of an ecosystem "service". But, as pointed out by Boyd and Banzhaf (2007, p.619), the ecosystem provides only an "intermediate service" (along with "conventional goods and services") in the production of the final benefit of recreation and tourism. In estimating the value of this "intermediate" ecosystem service in producing recreational benefits, it is therefore important to assess only the effects of changes in the ecosystem on recreation, and not the additional influence of any human inputs. The same approach should be taken for those "final" ecosystem services, such as coastal protection, erosion control, nutrient cycling, water purification and carbon sequestration, which may benefit human well-being with or without any additional human-provided goods and services. Valuation should show how changes in these services affect human welfare, after controlling for the influence of any additional human-provided goods and services.

Although valuing ecosystem goods and services seems straightforward, in practice there are a number of challenges to overcome. These difficulties are key to understanding why there are still a large number of ecosystem goods and services that have yet to be valued or have very unreliable valuation estimates.

The most significant problem is that very few are marketed. Some of the products provided by ecosystems, such as raw materials, food and fish harvests, are bought and sold in markets. Given that the price and quantities of these marketed products are easy to observe, there are numerous value estimates of the contribution of the environmental input to this production. However, this valuation can be more complicated than it appears. Market conditions and regulatory policies for the commodity bought and sold will influence the values imputed to the environment input. For example, one important service of many estuarine and coastal ecosystems is that they serve as coastal breeding and nursery habitat for offshore fisheries. As many fisheries are exploited commercially, the inability to control fishing access and the presence of production subsidies and other market distortions can impact harvests, the price of fish sold, and ultimately, the estimated value of coastal habitats in supporting these fisheries (Barbier et al. 2002, Barbier 2007, Freeman 1991, Smith 2007).

However, the majority of ecosystem goods and services are not marketed. These include many services arising from ecosystem processes and functions that benefit human beings largely without any additional input from them, such as coastal protection, nutrient cycling, erosion control, water purification and carbon sequestration. In recent years, substantial progress has been made by economists working with ecologists and other natural scientists in applying environmental valuation methodologies to assess the welfare contribution of these services. The various non-market valuation methods employed for ecosystem services are essentially the standard techniques that are available to economists.[15] Later on in this chapter, we discuss these issues further. Nevertheless, what makes applying these methods to estimate the value of a non-marketed ecosystem service especially difficult is that it requires three important, and

[15] For example, Barbier (2007; 2011a and 2011b), Bateman et al. (2011); EPA (2009), Freeman (2003), Hanley and Barbier (2009), Mendelsohn and Olmstead (2009), NRC (2005) and Pagiola et al. (2004) discuss how these standard valuation methods are best applied to ecosystem services, emphasizing in particular both the advantages and the shortcomings of the different methods and their application.

interrelated, steps (Barbier 1994, 2011a and 2011b; EPA 2009; Freeman 2003; NRC 2005; Polasky and Segerson 2009).

The first step involves determining how best to characterize the change in ecosystem structure, functions and processes that gives rise to the change in the ecosystem service. For example, the change could be in the spatial area or quality of a particular type of ecosystem, such as a mangrove forest, marsh vegetation or watershed extent. It could also be a change in a key population, such as fish or main predator. Alternatively, the change could be due to variation in the flow of water, energy or nutrients through the system, such as the variability in tidal surges due to coastal storm events or the influx of organic waste from pollution upstream from estuarine and coastal ecosystems.

The second step requires tracing how the changes in ecosystem structure, functions and processes influence the quantities and qualities of ecosystem service flows to people. Underlying each ecosystem service is a range of important energy flow, biogeochemical and biotic processes and functions. For example, water purification by seagrass beds is linked to the ecological processes of nutrient uptake and suspended particle deposition (Koch et al. 2006; Rybicki 1997). However, the key ecological process and functions that generate an ecosystem service are in turn controlled by certain abiotic and biotic components that are unique to each ecosystem's structure. The various controlling components that may affect nutrient uptake and particle deposition by seagrass ecosystems include seagrass species and density, nutrient load, water residence time, hydrodynamic conditions and light availability. Only when these first two steps are completed is it possible to conduct the final step, which involves using existing economic valuation method to assess the impact on human well being that results from the change in ecosystem goods and services.

Figure 1 provides a visual summary of the key elements of this three-step approach. Human drivers of ecosystem change affect important ecosystem processes and functions and their controlling components. Assessing this change is crucial yet difficult. However, as NRC (2005, pp. 2-3) points out, "making the translation from ecosystem structure and function to ecosystem goods and services (i.e. the ecological production) is even more difficult" and "probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function." Similarly, Polasky and Segerson (2009, p. 422) maintain that "among the more practical difficulties that arise in either predicting changes in service flows or estimating the associated value of ecosystem services" include the "lack of multiproduct, ecological production functions to quantitatively map ecosystem structure and function to a flow of services that can then be valued."

Valuing nonmarket ecosystem goods and services

One of the fundamental challenges is that many important ecosystem goods and services are nonmarketed. These include many important services arising from ecosystem processes and functions, such as coastal protection, nutrient cycling, erosion control, water purification and carbon sequestration. In recent years substantial progress has been made by economists working with ecologists and other natural scientists on this "fundamental challenge" to improve the application of environmental valuation methodologies to nonmarket ecosystem services.

Nevertheless, a number of important challenges arise in applying these methods, which are reviewed in this section.

In the previous section, we discussed the three-step approach that is required to integrate the “ecological production” of ecosystem goods and services with “economic valuation” of these benefits, which was summarized visually by Figure 1. In recent years substantial progress has been made by economists working with ecologists and other natural scientists on this “fundamental challenge” to improve the application of environmental valuation methodologies to nonmarket ecosystem services. Nevertheless, a number of important challenges arise in applying these methods. To help our subsequent discussion of valuation issues, it is useful to look at a more detailed version of Figure 1 that emphasizes the economic valuation component of the latter diagram (see Figure 2).

As indicated in Figure 2, there are a number of different ways in which humans benefit from, or value, ecosystem goods and services. The first distinction is between the *use values* as opposed to *nonuse values* arising from these goods and services. Typically, use values involve some human “interaction” with the environment whereas non-use values do not, as they represent an individual valuing the pure “existence” of a natural habitat or ecosystem or wanting to “bequest” it to future generations. Direct use values refer to both consumptive and non-consumptive uses that involve some form of direct physical interaction with environmental goods and services, such as recreational activities, resource harvesting, drinking clean water, breathing unpolluted air and so forth. Indirect use values refer to those ecosystem services whose values can only be measured indirectly, since they are derived from supporting and protecting activities that have directly measurable values. For example, for wetlands, the indirect use values associated with ecosystems services include coastal protection, erosion control, flood protection, water purification, carbon sequestration, maintenance of temperature and precipitation, and habitat support for fishing, hunting and foraging activities outside the wetlands (Barbier 2007).[16]

Table 1 indicates the various nonmarket methods that can be used for valuing ecosystem goods and services. As shown in the table, the methods employed are essentially the standard nonmarket valuation techniques that are available to economists. However, the application of nonmarket valuation to ecosystem goods and services is not without difficulties. Here, we simply summarize some of the key issues.

First, the application of some of the valuation methods listed in Table 1 is often limited to specific types of ecological goods and services. For example, the travel cost method is used principally for those environmental values that enhance individuals’ enjoyment of recreation and tourism, averting behaviour models are best applied to the health effects arising from environmental pollution. Similarly, hedonic wage and property models are used primarily for assessing work-related environmental hazards and environmental impacts on property values, respectively.

In contrast, stated preference methods, which include contingent valuation methods and choice modelling, have the potential to be used widely in valuing ecosystem goods and services.

[16] Another component of value, *option value*, is commonly referred to as a nonuse value in the literature. However, option value arises from the difference between valuation under conditions of certainty and uncertainty, and is a numerical calculation, not a value held by people *per se*. See NRC (2005, ch. 6) for further discussion.

These valuation methods share the common approach of surveying individuals who benefit from an ecological service or range of services, in the hope that analysis of these responses will provide an accurate measure of the individuals' willingness to pay for the service or services. In addition, stated preference methods can go beyond estimating the value to individuals of single and even multiple benefits of ecosystems and in some cases elicit non-use values that individuals attach to ensuring that a preserved and well-functioning system will be around for future generations to enjoy. For example, a study of mangrove-dependent coastal communities in Micronesia demonstrated through the use of contingent valuation techniques that the communities "place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves' marketable products" (Naylor and Drew 1998, p. 488). Similarly, choice modelling has the potential to elicit the relative values that individuals place on different ecosystem services. A study of wetland restoration in southern Sweden revealed through choice experiments that individuals' willingness to pay for the restoration increased if the result enhanced overall biodiversity but decreased if the restored wetlands were used mainly for the introduction of Swedish crayfish for recreational fishing (Carlsson et al. 2003).

However, as emphasized by NRC (2005), to implement a stated-preference study two key conditions are necessary:

(1) the information must be available to describe the change in an ecosystem in terms of the goods and services that people care about, in order to place a value on those goods and services; and

(2) the ecosystem change must be explained in the survey instrument in a manner that people will understand and not reject the valuation scenario.

For many of the specific ecosystem goods and services listed in Table 1, one or both of these conditions may not hold. For instance, it has proven very difficult to describe accurately through the hypothetical scenarios required by stated-preference surveys how changes in ecosystem processes and components affect ecosystem regulatory and habitat functions and thus the specific benefits arising from these functions that individuals value. If there is considerable scientific uncertainty surrounding these linkages, then not only is it difficult to construct such hypothetical scenarios but also any responses elicited from individuals from stated-preference surveys are likely to yield inaccurate measures of their willingness to pay for ecological services (Bateman et al. 2009). Valuation workshop methods may, however, help in terms of conveying information about complex ecological goods, and investigating the effects on people's values of scientific uncertainty about linkages within the system (see, for example, Christie et al. 2006).

In contrast to stated preference methods, the advantage of production function (PF) approaches is that they depend on only the first condition, and not both conditions, holding (see Barbier 1994 and 2007; McConnell and Bockstael 2005). That is, for those ecological functions where there is sufficient scientific knowledge of how these functions link to specific ecological services that support or protect economic activities, then it may be possible to employ the PF approach to value these services. The basic modelling approach underlying PF methods, also called "valuing the environment as input", is similar to determining the additional value of a change in the supply of any factor input. If changes in the structure and functions of ecosystems affect the marketed production activities of an economy, then the effects of these changes will be transmitted to individuals through the price system via changes in the costs and prices of final good and services. This means that any resulting "improvements in the resource base or environmental quality" as a result of enhanced ecosystem services, "lower costs and prices and increase the quantities of marketed goods, leading to increases in consumers' and perhaps producers' surpluses" (Freeman 2003, p. 259).

An adaptation of the PF methodology is required in the case where ecological regulatory and habitat functions have a protective value, through various ecological services such as storm protection, flood mitigation, prevention of erosion and siltation, pollution control and maintenance of beneficial species (Barbier 2007; McConnell and Bockstael 2005). In such cases, the environment may be thought of producing a non-marketed service, such as "protection" of economic activity, property and even human lives, which benefits individuals through limiting damages. Applying PF approaches requires modelling the "production" of this protection service and estimating its value as an environmental input in terms of the expected damages avoided by individuals. However, PF methods have their own measurement issues and limitations when they are employed to value ecosystem goods and services.

For instance, applying the PF method raises questions about how changes in the ecological service should be measured, whether market distortions in the final goods market are significant, and whether current changes in ecological services may affect future productivity through biological "stock effects". A common approach in the literature is to assume that an estimate of ecosystem area may be included in the "production function" of marketed output as a proxy for the ecological service input. For example, this is the standard approach adopted in coastal habitat-fishery PF models, as allowing wetland area to be a determinant of fish catch is thought by economists and ecologists to proxy some element of the productivity contribution of this important habitat function (Barbier 2000 and 2007; Freeman 2003, ch. 9; McConnell and Bockstael 2005). In addition, as pointed out by Freeman (1991), market conditions and regulatory policies for the marketed output will influence the values imputed to the environmental input. For instance, in the previous example of coastal wetlands supporting an offshore fishery, the fishery may be subject to open access conditions. Under these conditions, profits in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer values are zero and only consumer values determine the value of increased wetland area. Finally, a further measurement issue arises in the case where the ecological service supports a natural resource system, such as a fishery, forestry or a wildlife population, which is then harvested or exploited through economic activity. In such cases, the key issue is whether or not the effects on the natural resource stock or biological population of changes in the ecological service are sufficiently large that these stock effects need to be modelled explicitly. In the production function valuation literature, approaches that ignore stock effects are referred to as "static models" of environmental change on a natural resource production system, whereas approaches that take into account the intertemporal stock effects of the environmental change are referred to as "dynamic models" (Barbier 2000 and 2007; Freeman 2003, ch. 9).

Finally, measurement issues, data availability and other limitations can prevent the application of standard nonmarket valuation methods to many ecosystem services. In circumstances where an ecological service is unique to a specific ecosystem and is difficult to value, then economists have sometimes resorted to using the cost of replacing the service or treating the damages arising from the loss of the service as a valuation approach. However, economists consider that the replacement cost approach should be used with caution (Barbier 1994 and 2007; Ellis and Fisher 1987; Freeman 2003; McConnell and Bockstael 2005; Shabman and Batie 1978). For example, a number of studies that have attempted to value the storm prevention and flood mitigation services of the "natural" storm barrier function of mangrove and other coastal wetland systems have employed the replacement cost method by simply estimating the costs of replacing mangroves by constructing physical barriers to perform the same services (Chong 2005). Shabman and Batie (1978) suggested that this method can provide a reliable valuation estimation for an ecological service, but only if the following conditions are met: (1) the alternative considered provides the same services; (2) the alternative compared for cost comparison should be the least-cost alternative; and (3) there should be substantial evidence that the service would be demanded by society if it were provided by that least-cost alternative. Unfortunately, very few replacement cost studies meet all three conditions.

However, one study that met these criteria for valuing an ecosystem service was the analysis of the policy choice of providing clean drinking water by the Catskills Mountains for New York City (Chichilinsky and Heal 1998; NRC 2005). Rather than value all the services of the Catskills watershed ecosystems; instead, it was sufficient simply to demonstrate that protecting and restoring the ecological integrity of the Catskills was less costly than replacing this ecosystem service with a human-constructed water filtration system. The total costs of building and operating the filtration system were in the range of $6 billion to $8 billion, whereas it would cost New York City $1 billion to $1.5 billion to protect and restore the natural ecosystem processes in the watershed, thus preserving the clean drinking water service provided by the Catskills. A second case study that also met the above criteria estimates the value of using wetlands for abatement of agricultural nitrogen load on the Baltic Sea coast of Sweden (Byström 2000). In this study, the replacement value of wetlands was defined and estimated as the difference between two cost-effective reductions of agricultural nitrogen pollution: one that uses wetlands for nitrogen abatement, and one that does not. The study showed that the use of wetlands as nitrogen sinks can reduce by 30% the total costs of abating nitrogen pollution from agriculture in Sweden.

Correcting wealth accounts for ecological capital

Overcoming measurement issues and challenges to determine the value of non-market ecosystem goods and services is an important, but there are additional considerations in using these values to correct wealth accounts for ecological capital. This section focuses on two important issues: double counting and accounting for special properties of ecosystems, such as ecological stability, resilience and collapse.

Recall that, as equation (13) above indicates, the net domestic product (NDP) of the economy should be adjusted for the value of the *direct benefits* provided by the current stock of ecosystem. But NDP should not be adjusted for any *indirect benefits* of this current stock through its support or protection of production in the economy. The reason for the latter omission is that it may create problems of double counting in the wealth accounts of an economy.

As discussed in the previous section and outlined in Table 1, the production function method is an important non-market valuation method of measuring the economic contribution of many ecosystem goods and services that affect human welfare indirectly through their support or protection of production activities, property or human lives. In other words, ecosystem services that arise from the regulatory functions of ecosystems, such as waste management, habitat support, storm protection, flood mitigation, groundwater recharge, often serve as *intermediate inputs* in economic production activities, which are in turn often marketed. Similarly, goods or products from ecosystems, such as harvested raw materials, water supplies, food, fiber and fuel, may themselves be marketed, or in turn are processed by industries into marketed products. But if these goods and services produced from the current stock of ecosystems serve as intermediate inputs into marketed production, then conventionally defined NDP will most likely already reflect their current contribution. To add to NDP the marginal value contribution to economic production of ecosystem goods and services that are intermediate inputs would result in double counting (Mäler 1991; Mäler et al. 2008; Vincent 2012).

For example, if a coastal marsh or mangrove serves as a nursery or breeding habitat for an off-shore commercial fishery, then this habitat will have an influence on current harvested and marketed output of the fishery. However, the harvested fish will already be included in conventional NDP of an economy, as it is a marketed product. Similarly, if the wetlands also protect coastal property from storm damages, the value of the latter assets already accounts for the storm protection value of the wetlands. In addition, if the wetlands themselves are a source of currently harvested food, fiber and raw materials, which are in turn sold commercially, then the NDP will already included these marketed products. In contrast, if any harvested wetland products are not marketed but support the subsistence needs of harvesting households, then the value of these ecological goods will not appear in conventionally measured NDP. Because they are consumed and not marketed, these products are essentially direct benefits to households. Finally, coastal wetlands may generate many other non-marketed ecosystem services that also directly influence welfare, such as filtering water pollution that affects human health, enhancing enjoyment of coastal areas and recreation and providing cultural benefits. Again, these current values of the wetlands are unlikely to appear in conventional NDP.

To summarize, to avoid double counting, the NDP of an economy should not be adjusted by including the value of any goods and services provided currently by ecosystems, if they serve as intermediate inputs in the production of marketed final goods and services. However, if ecosystem goods and services affect current production activities that are not marketed, such as raw materials, food, fibre and water that are consumed directly by households, then the value of these ecological contributions should be assessed and added to NDP.

However, as indicated by equations (13) and (14), this particular double counting problem does not arise when adjusting NDP to account for any capital revaluation in the economy that occurs when, say, ecosystems are converted to other land uses. In this case, the capitalized value of converted ecosystems must reflect the present value of *all* foregone future benefits of these ecosystems, whether they influence welfare directly or indirectly through production of marketed final goods and services.

Landscape losses and degradation of ecosystem processes and functions can also lead to unpredictable and sudden increases in the risk of ecological collapse, due to the presence of ecological thresholds and feedback effects. That is, large shocks or sustained disturbances to ecosystems lead to further interactions that can contravene ecological thresholds, causing the systems to “flip” irreversibly from one functioning state to another. Thus the *resilience* or *robustness* of an ecosystem – its ability to absorb large shocks or sustained disturbances and still maintain internal integrity and functioning – may be an important attribute determining the extent to which landscape conversion and ecosystem degradation affects the risk of ecological collapse.[17] Thus, one approach to accounting for the resilience property of ecosystems is to measure directly the wealth effects of resilience (Mäler 2008; Walker et al. 2010).

Box 2 summarizes the effort by Walker et al. (2010) to value ecosystem resilience for the Goulburn-Broken Catchment (GBC) in Southeast Australia. The GBC is prime agricultural land, most of which is used for dairy pasture. However, the agro-ecosystem is threatened by increased soil salinity due to rising water tables from removal of native vegetation. At the 2 meter (m) water table threshold, the system is in danger of flipping to a different regime dominated by degraded and salinized pasture. The authors estimate resilience as the distance from the current water table to the 2m threshold. Under normal climate conditions, a 0.5 m change in ecosystem resilience is valued at about $23 million, or around 7% of the total wealth of the GBC in 1991. Under drier climatic conditions, resilience is worth $28 million, or 8.4% of total wealth.

This example from Australia of valuing of ecosystem resilience suggests that this economic contribution can be considerable. In such highly productive ecosystems supporting economic activity, regime shift can be catastrophic. Or to put it differently, the value of avoiding regime shift by maintaining or enhancing the resilience of ecosystems can be a sizable component of the total economic wealth generated by these systems.

A case study: adjusted NDP and mangrove loss in Thailand

To summarize, although the previous sections indicate the important issues and challenges that arise when attempting to value ecosystem services and account for their contributions to wealth, nonetheless significant progress has been made in recent years. For some major ecosystems, we may be very close to implementing the methodology of adjusting NDP to reflect ecological values as well as the depreciation or appreciation in these key natural assets.

The purpose of this section is to provide an example of wealth accounting with a case study from Thailand involving mangrove loss. The case study illustrates the two adjustments to NDP due to ecological capital: the value of the *direct benefits* provided by the current stock of ecosystems, and any capital revaluation that occurs as a result of ecosystem conversion to other land uses. Unfortunately, estimating the wealth effects of ecosystem resilience is beyond the scope of this case study.

Thailand is estimated to have lost around a third of its mangroves since the 1960s, mainly to shrimp farming expansion and other coastal development (FAO 2007a; Spalding et al. 2010). During this period, real GDP per capita in Thailand has increased five-fold (World Bank 2011). A measure of the adjusted NDP, taking into account human and natural capital loss since 1970, is constructed. Based on estimates of four mangrove ecosystem benefits – collected products, habitat-fishery linkages, storm protection and carbon sequestration – the methodology of adjusting NDP for the value of ecosystems is also included as an illustration.

In 1961, Thailand was estimated to have around 368,000 hectares (ha) of mangroves in 1961 (see Figure 3). Mangrove deforestation proceeded swiftly in the 1970s and 1980s, but

[17] See, for example, Dasgupta and Mäler (2003); Elmqvist et al. (2003); Folke et al. (2004); Levin (1999); Levin and Lubchenco (2008); Perrings (1998); Scheffer et al. (2001); Walker et al. (2004).

since 2000, the area of mangroves seems to have stabilized around 240,000 to 250,000 ha. The main cause of mangrove loss in Thailand is attributed to conversion to shrimp aquaculture (Aksornkoae and Tokrisna 2004). The main reason for the slowdown in mangrove loss is that many of the suitable sites for establishing shrimp farms in the Gulf of Thailand have been deforested, whereas the mangrove areas on the Andaman Sea (Indian Ocean) coast are too remote and less suitable for shrimp farms (Barbier and Cox 2004).

Box 3 outlines the valuation estimates that are used for accounting for the current benefits of mangroves as well as their capitalized values for Thailand over 1970 to 2009. The four principal ecosystem goods and services are the role of mangroves as natural "barriers" to periodic damaging coastal storm events, their role as nursery and breeding habitats for offshore fisheries, their ability store carbon, and the exploitation of mangrove forests by coastal communities for a variety of wood and non-wood products. As outlined in Box 3, these four benefits of mangroves in Thailand have a constant 2000 US$ capitalized value of $21,443 per ha. As the main activity responsible for mangrove conversion in Thailand has been shrimp aquaculture, the capitalized value (in 2000 US$) of this alternative use of mangrove ecosystems is $1,351 per ha. Note that, because the capitalized value, or “price”, of mangroves converted to shrimp farming is less than the capitalized value of mangroves, the NDP of Thailand should be adjusted for this depreciation in mangrove capital.

However, not all the current benefits of mangroves impact welfare directly, but may do so only through support or protection of economic activity and property. That is certainly the case for storm protection benefits of mangroves, which are estimated through an expected damage approach that determines their value in terms of protecting economic property (Barbier 2007). As this benefit is already accounted for in the current market values of property, to avoid double counting, the NDP of the Thai economy should not be adjusted to include the benefit of storm protection provided by the current stock of mangroves. Similarly, a survey of four Thai villages from two coastal provinces indicates that only 12.4% of the value of collected wood and non-wood products from mangroves and 5.3% of the value of coastal fishery harvests can be attributed to subsistence production (Sarntisart and Sathirathai 2004).[18] Thus, the NDP should be adjusted only for these subsistence contributions of these two benefits of the mangroves in Thailand.

Using the data from Box 3, Table 2 depicts the per capita wealth accounting estimates for Thailand's mangroves from 1970 to 2009. Average annual mangrove loss has fallen steadily in every decade since the 1970s (see also Figure 3). Nevertheless, because around a third of the mangrove area has been deforested from 1970 to 2009, whereas Thailand's population has nearly doubled over this period, the current per capita benefits of mangroves has halved since the 1970s, from $0.57 to $0.28 per person.[19] In the 1970s, when mangrove loss in Thailand was at its highest, mangrove depreciation amounted to $2.26 per person, whereas by the 2000s, it had fallen to only $0.03 per capita. The result is that the net value of mangroves per capita in Thailand, which is the total value less mangrove depreciation, was actually negative in the 1970s

[18] The four villages are Ban Sam Chong Tai and Ban Bang Pat of Phang-nga Province, and Ban Gong Khong and Ban Bkhlong Khut in Nakhon Si Thammarat Province.

[19] According to World Bank (2011), in 1970 Thailand's population was 36.9 million and grew steadily to 68.7 million in 2009.

and 1980s, averaging -$1.69 and -$0.76 per person respectively. However, in the 1990s and 2000s, the net value was slightly positive, averaging $0.11 and $0.22 respectively.

Table 3 depicts an approximate estimate of adjusted net domestic product (ANDP) per capita for real changes in reproducible, human and natural capital for Thailand over 1970 to 2009. ANDP is GDP less consumption of fixed capital and natural resource depletion, plus education expenditure and net values of mangrove depletion. The latter estimate is based on the net value of mangroves from Table 2. Since the 1970s, both consumption of fixed capital and natural resource depreciation have increased significantly in Thailand. The value of expanding human capital, as proxied by education expenditures, has also increased, and because of the slowdown in mangrove loss, the net value of this ecological capital has gone from a negative to a positive contribution to NDP. Overall, the value of mangroves and expanding human capital has not kept pace with reproducible capital depreciation and natural resource depletion in Thailand. As a consequence, adjusted net domestic product per capita in Thailand has remained consistently below GDP per capita since the 1970s. As shown in Figure 4, since 1990 the gap between GDP and ANDP per capita in Thailand has widened significantly.

To summarize, because many of the benefits provided by the current stock of mangroves in Thailand arise through supporting or protecting marketed production and property, these benefits should already be included in the GDP estimates for Thailand. However, any adjusted NDP measure does need to take into account the current direct benefits provided by mangroves in the form of carbon sequestration, habitat and breeding ground services that support any fishery harvests consumed by coastal households and mangrove products that also comprise subsistence consumption. On the other hand, all future mangrove benefits are lost as a result of mangrove conversion, which has been substantial in Thailand since the 1970s. The substantial mangrove depreciation that occurred in the 1970s and 1980s meant that the net value of mangroves was actually negative in these decades. Although mangrove deforestation and thus its capital depreciation has slowed since, the net value of mangroves per capita, as an indicator of its contribution to the wealth of Thailand, is still extremely low. Thus, the Thailand mangrove case study not only provides an illustration of the adjusted NDP methodology for ecological capital but also illustrates how significant loss of this capital can influence its net value in wealth accounts.

Conclusion

This chapter has explored the methodology and the challenges of including ecosystem goods and services in a wealth accounting framework. Following the approach developed by Dasgupta (2009), which is elaborated further in the chapters by Dasgupta (2012) and Perrings (2012), it is shown how this framework can be extended to incorporate ecosystem and their valuable goods and services. The approach developed here requires, first, recognizing ecosystems as a component of natural capital, or *ecological capital*, and second, measuring these important assets in terms of the land area, or *ecological landscape*, which defines their boundaries.

Such an approach clarifies how we should value and include changes in ecological capital in wealth accounting, which can be proxied by the net domestic product (NDP) of an economy provided that this indicator accounts for the depreciation of all forms of capital – reproducible, human and natural capital. There are two main adjustments to NDP of the economy that result, if ecological capital is also to be considered.

First, we should adjust NDP to include the value of the various goods and services provided by the current stock of ecosystems that derives from direct impacts on welfare. These direct ecosystem benefits might include the value of ecosystems in providing non-market recreational, educational and scientific benefits, their value in terms of natural heritage or bequests to future generation or the value of ecosystems in reducing harmful pollution and assimilating waste that affect human welfare and health directly. In addition, ecological capital protects or supports current economic activity and property. These indirect ecosystem benefits are broad ranging, and include raw materials, food and other harvested inputs used in production, provision of freshwater, watershed protection, coastal habitats for off-shore fisheries, flood control, storm protection, and managing climate. However unlike direct benefits to current well-being, these indirect benefits should not be included as additional values in any measure of an economy's NDP, as they are likely to already be reflected in the prices of final marketed goods and services.

Second, conversion of ecological capital to other land uses requires a further adjustment to GDP to reflect any capital revaluation as a result of this land use change. As the resource allocation mechanism of the economy may not be optimal or even efficient, ecosystem conversion may be taking place even though the capitalized value, or “price”, of developed land is actually less than the capitalized value of ecosystems. In which case, GDP should be adjusted for the depreciation in ecological capital that occurs as it is converted to less valuable developed land. The capitalized value of converted ecosystems must reflect the present value of *all* foregone future benefits of these ecosystems, whether they influence welfare directly or indirectly through production of marketed final goods and services.

The main challenges of applying such an approach is that there are still a large number of nonmakreted ecosystem goods and services that have yet to be valued or have very unreliable valuation estimates. Measurement issues, data availability and other limitations can prevent the application of standard nonmarket valuation methods to many ecosystem services. Fortunately, some progress is being made, due to the growing collaboration between economists, ecologists and other natural scientists in determining how the ecological production of key goods and services translate into economic valuation of these benefits.

For some major ecosystems, we may be very close to implementing the methodology advocated in this chapter of adjusting GDP to reflect ecological values as well as the depreciation or appreciation in these key natural assets. Using the example of mangroves in Thailand, this chapter illustrates how such an approach might be applied. The case study is able to show how valuation estimates from existing studies could be used for accounting for the current direct benefits of mangroves as well as their capitalized values for Thailand over 1970 to 2009. The per capita value of mangroves net of depreciation in Thailand was actually negative in the 1970s and 1980s due to mangrove conversion to development activities, and principally shrimp aquaculture. The net value of the wealth contribution of mangroves per person was positive but very small in the 1990sand 2000s, only $0.11 and $0.25 respectively. In comparison, in the 2000s, reproducible capital depreciation was $280 per person, natural resource depletion of energy, minerals and forest was $79 per capita, and human capital increased by $109 per person. Thus, the case study demonstrates that accounting for the economic contributions and deprecations of mangrove capital is an important, albeit relatively smaller, component of the key capital adjustments that occur in Thailand's economy.

But perhaps the more important lesson to be learned from the example of adjusting Thailand's wealth accounts for mangrove current benefits and depreciation is that it illustrates that the challenges of including ecosystem services in a wealth accounting framework can be overcome.

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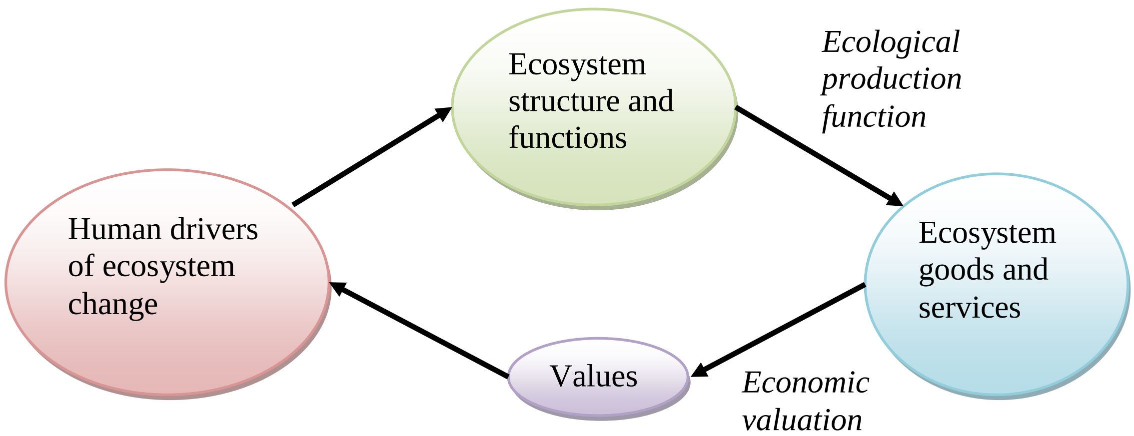
Box 1. Ecosystem functions and services

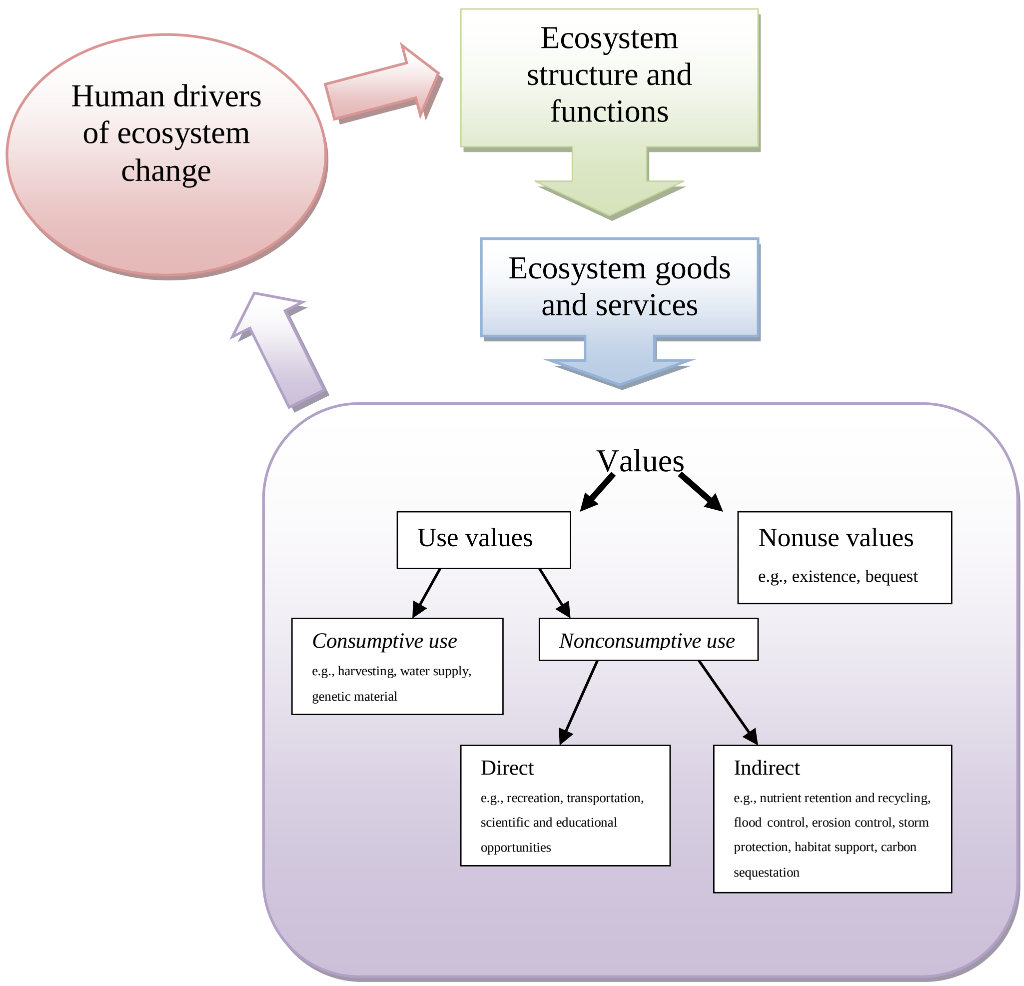
"…the term 'ecosystem' describes a dynamic complex of plant, animal, and microorganism communities and their non-living environment, interacting as a system. Ecosystems encompass all organisms within a prescribed area, including humans. Ecosystem functions or processes are the characteristic physical, chemical, and biological activities that influence the flows, storage, and transformation of materials and energy within and through ecosystems. These activities include processes that link organisms with their physical environment (e.g., primary productivity and the cycling of nutrients and water) and processes that link organisms with each other, indirectly influencing flows of energy, water, and nutrients (e.g., pollination, predation, and parasitism). These processes in total describe the functioning of ecosystems….Ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations. Ecosystem processes and functions contribute to the provision of ecosystem services, but they are not synonymous with ecosystem services. Ecosystem processes and functions describe biophysical relationships that exist whether or not humans benefit from them. These relationships generate ecosystem services only if they contribute to human well-being, defined broadly to include both physical well-being and psychological gratification. Thus, ecosystem services cannot be defined independently of human values." (EPA 2009, p. 12)

Figure 1. Key interrelated steps in valuing ecosystem goods and services



Source: Adapted from NRC (2005, Figure 1-3).

Figure 2. Economic valuation of ecosystem goods and services



Source: Adapted from NRC (2005, Figure 7-1).

Table 1. Various nonmarket valuation methods applied to ecosystem services

Valuation method[a]	Types of value estimated	Common types of applications	Ecosystem services valued
Travel cost	Direct use	Recreation	Maintenance of beneficial species, productive ecosystems and biodiversity
Averting behaviour	Direct use	Environmental impacts on human health	Pollution control and detoxification
Hedonic price	Direct and indirect use	Environmental impacts on residential property and human morbidity and mortality	Storm protection; flood mitigation; maintenance of air quality
Production function	Indirect use	Commercial and recreational fishing; agricultural systems; control of invasive species; watershed protection; damage costs avoided	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation
Replacement cost	Indirect use	Damage costs avoided; freshwater supply	Drainage and natural irrigation; storm protection; flood mitigation
Stated preference	Use and non-use	Recreation; environmental impacts on human health and residential property; damage costs avoided; existence and bequest values of preserving ecosystems	All of the above

Notes: [a]See Barbier (2007), Bateman et al. 2011; EPA (2009), Freeman (2003), Hanley and Barbier (2009), Mendelsohn and Olmstead (2009), NRC (2005) and Pagiola *et al.* (2004) for more discussion of these various nonmarket valuation methods and their application to valuing ecosystem goods and services.

Source: Adapted from NRC (2005), Table 4-2**.**

Box 2. The value of ecosystem resilience in the Goulburn-Broken Catchment of Southeast Australia

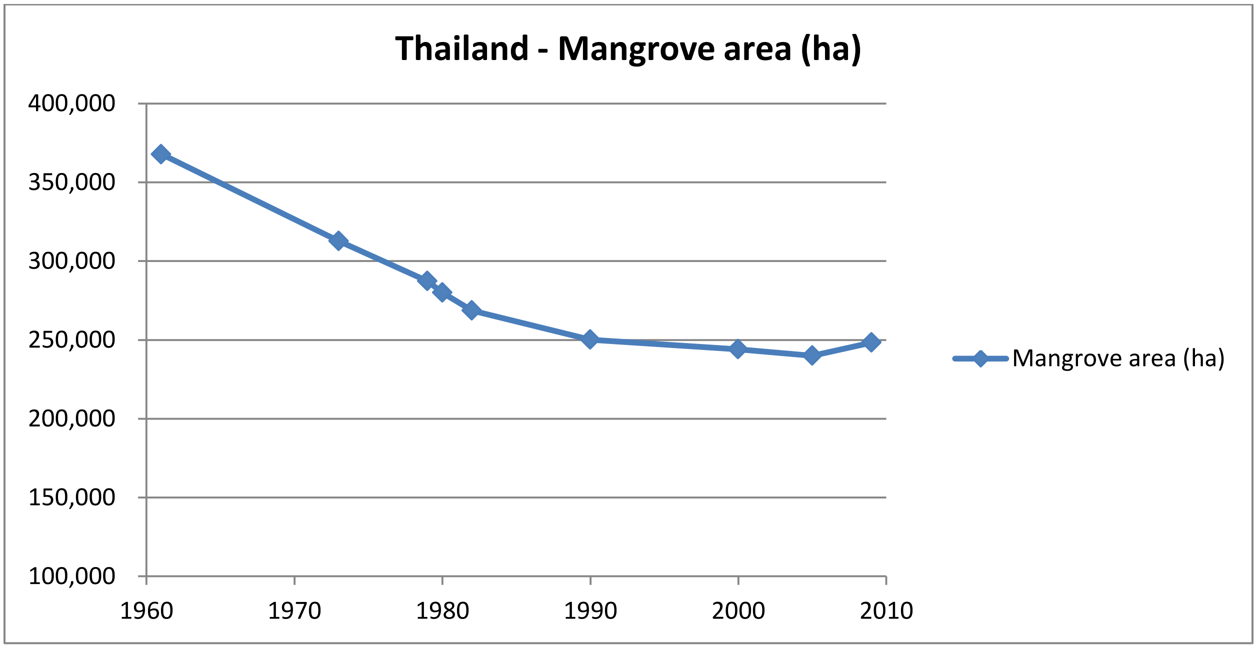
Using the inclusive wealth framework of Arrow et al. (2003), Mäler (2008) shows that it is possible to add a "resilience stock" to the measure of an economy's wealth. Resilience is interpreted as the probability of the system transitioning to another state (regime). That is, the closer to the threshold, the lower the stock of resilience, and the higher is the probability that the system will flip to the alternative regime. The real value, or shadow price, of the resilience stock is the expected change in future social welfare from a marginal change in resilience today. This value changes as the likelihood of crossing the threshold into the alternative regime increases.

Walker et al. (2010) apply this approach to the Goulburn-Broken Catchment (GBC) in Southeast Australia. The GBC includes 300,000 ha in irrigation, of which 80% is for dairy pasture. However, the removal of native vegetation for agriculture has led to rising water tables and increased soil salinity. Once the water table rises above 2 meters (m), however, pasture land is radically changed, and the agro-ecological system shifts to a different regime dominated by degraded and salinized soil. Thus, the resilience of the GBC system is measured by the distance from the water table to the 2m threshold, and this indicator determines the probability that the system will shift from the non-saline to saline regime. To demonstrate the impact of resilience on the inclusive wealth of the GBC, Walker et al. assume that all other economic assets are constant and only the stock of resilience changes. Between 1991 and 2001, they calculate that the resilience stock increased by 0.5 m due to a water table fall from 3 to 3.5 m. They estimate the value of this change in resilience under two different climate regimes: normal versus drier rainfall and evaporation conditions. The results are depicted in the table below. Under normal climate conditions, the 0.5 m change in ecosystem resilience is valued at about $23 million, or around 7% of the total wealth of the GBC in 1991. Under drier climatic conditions, resilience is worth $28 million, or 8.4% of total wealth.

Climate scenario	**Change in wealth from 1991 to 2001 form 0.5 m change in the resilience stock**	**Share of 1991 inclusive wealth**
Normal conditions	$22,852, 650	7.0%
Dry conditions	$28,558,360	8.4%

Source: Walker et al. (2010, Table 2).

Figure 3. Estimated Mangrove Area, Thailand 1961-2009



Sources: FAO (2007b) and Spalding et al. (2010).

Box 3. Valuation Estimates Used in Accounting for Mangrove Wealth, Thailand

As indicated in equation (13), the net domestic product (NDP) of an economy must be adjusted for the direct benefits to current well-being provided by ecosystems, $U_N N$, and any capital revaluation that occurs as ecosystems are converted by land use change for development, $\left(v^D - v^N\right)c$. Mangrove ecosystems in Thailand provide four essential goods and services. These are the role of mangroves as natural "barriers" to periodic damaging coastal storm events, their role as nursery and breeding habitats for offshore fisheries, their ability store carbon, and the exploitation of mangrove forests by coastal communities for a variety of wood and non-wood products. Estimates of the value of all four benefits exist for Thailand.

For example, the value of coastal protection from storms is based on a marginal value per ha of damages avoided (in 1996 US $) of $1,879; over a 20-year time horizon and a 10% discount rate this yields a net present value (NPV) of $15,997 per ha (Barbier 2007). The value of habitat-fishery linkages is based on a net value per ha (in 1996 $, assuming a price elasticity for fish of -0.5) of mangrove habitat of $249; over a 20-year time horizon and a 10% discount rate this yields a NPV of $2,117 per ha (Barbier 2003). The value of wood and non-wood products is based on net income per has from mangrove forests to local community (updated to 1996$) of $101; over a 20-year time horizon and a 10% discount rate this yields a NPV of $864 per ha (Sathirathai and Barbier 2001). Chmura et al. (2003) estimate permanent carbon sequestration by global mangroves of 2.1 metric tons per ha per year, and World Bank (2011) values unit carbon dioxide damage at $20 per ton of carbon (1995 US $), which yields an annual value (in 1995 US$) of $42 per ha for carbon sequestration. Over a 20-year time horizon and a 10% discount rate this yields a net present value (NPV) of $413 per ha. These values are converted to 2000 US$ using the GDP deflator for Thailand (World Bank 2011). As a result, mangroves in Thailand have a constant 2000 US$ capitalized value, v^N, of $21,443 per ha.

As the main activity responsible for mangrove conversion in Thailand has been shrimp aquaculture, the capitalized value of this activity is used for v^D. The net present value (NPV) per ha for the commercial net returns to shrimp farming over a 20-year time horizon and 10% discount rate is based on (Sathirathai and Barbier 2001), which when updated to 1996 US $, amounts to a value of $9,632 per ha. However, many of the inputs used in shrimp pond operations are subsidized, below border-equivalent prices, thus increasing artificially the private returns to shrimp farming. Without these subsidies, the resulting economic net returns to shrimp farming result in a NPV of $1,220 per ha. When converted to 2000 US$ using the GDP deflator for Thailand (World Bank 2011), the capitalized value of mangroves converted to shrimp farms is $1,351 per ha. Because the capitalized value, or “price”, of mangroves converted to shrimp farming is less than the capitalized value of mangroves, or $\left(v^D - v^N\right) < 0$, then the NDP of Thailand should be adjusted for this resulting capital depreciation in mangrove land.

Table 2. Wealth Accounting for Mangrove Capital, Thailand 1970-2009

		Average annual values per capita (constant 2000 US$)						
	Average annual mangrove loss (ha)	Storm protection	Habitat-fishery linkage	Wood & nonwood products	Carbon sequestration	Total value of mangroves	Mangrove depreciation	Net value of mangroves
1970-79	4,676	-	0.11	0.10	0.36	18.59	2.26	-1.69
1980-89	2,980	-	0.08	0.07	0.25	13.00	1.16	-0.76
1990-99	610	-	0.06	0.06	0.20	10.47	0.21	0.11
2000-09	97	-	0.05	0.05	0.18	9.28	0.03	0.25

Notes: As storm protection value is based on expected damages to economic property, it is assumed that this benefit is already accounted for in the current market values of property. Current habitat-fishery linkages benefits are based only the imputed subsistence value, which based on a survey of four Thai coastal villages, is approximately 5.3% of total household income (Sarntisart and Sathirathai 2004, Tables 6.3 and 6.4). Current wood and non-wood product benefits are based only the imputed subsistence value, which based on a survey of four Thai coastal villages, is approximately 12.4% of total household income (Sarntisart and Sathirathai 2004, Tables 6.3 and 6.4).

Table 3. Wealth Accounting, Thailand 1970-2009

	Average annual values per capita (constant 2000 US$)					
	GDP	ANDP	Consumption of fixed capital	Natural resource depletion	Education expenditure	Net value of mangroves
1970-79	617	544	89	13	30	-1.7
1980-89	956	852	130	19	46	-0.8
1990-99	1,793	1,563	296	20	86	0.1
2000-09	2,291	2,041	280	79	109	0.3

Notes: GDP = Gross Domestic Product

ANDP = Adjusted Net Domestic Product, or GDP less consumption of fixed capital and natural resource depletion, plus education expenditure and the net value of mangroves (estimated in Table 2).

Natural resource depletion is the sum of net forest depletion, energy depletion, and mineral depletion. Net forest depletion is unit resource rents times the excess of roundwood harvest over natural growth. Energy depletion is the ratio of the value of the stock of energy resources to the remaining reserve lifetime (capped at 25 years). It covers coal, crude oil, and natural gas. Mineral depletion is the ratio of the value of the stock of mineral resources to the remaining reserve lifetime (capped at 25 years). It covers tin, gold, lead, zinc, iron, copper, nickel, silver, bauxite, and phosphate.

Source: World Bank (2011), except for net value of mangroves, which is from Table 2.

Figure 4. GDP and ANDP per capita, Thailand 1970-2009