Monetary Green Accounting and Ecosystem Services

Ing-Marie Gren

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NIER
Kungsgatan 12-14
Box 3116
SE-103 62 Stockholm
Sweden
Phone: 46-8-453 59 00, Fax: 46-8-453 59 80
E-mail: ki@konj.se, Home page: www.konj.se

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*Ing-Marie Gren**

Abstract: The point of departure in this paper is that monetary green accounting aims at serving as indicator of wealth changes, sustainable use of natural capital, and performance of environmental policy. It is then investigated how wealth changes and sustainable development of natural capital can be estimated by means of ecosystem services. These services are defined as outputs from natural capital. The value of changes in natural capital, or wealth change, is thus measured as the value of impacts on current and future production of ecosystem services. It is then shown how this measure can be used as an indicator of sustainable use of the aggregate natural capital, and also how it can be applied efficient environmental policies. An empirical demonstration is made to the calculation of wealth changes to Swedish forests, agricultural landscape, wetlands, air quality, and coastal and marine ecosystems. The demonstration shows that the net welfare contribution from these natural capital assets during the period under study is positive, but that the use of the assets is unsustainable.

JEL classification: D6, D9, O4, Q2

*National Institute of Economic Research and Beijer International Institute of Ecological Economics, Stockholm, Sweden. E-mail: Ing-Marie.Gren@konj.se

1. Introduction

In this paper, the perceived purpose of monetary green accounts is threefold; to serve as *i)* wealth measurement, *ii)* indicator of sustainability, *iii)* decision support for environmental purposes. This is a point of departure in several theoretical papers, although there has been an ongoing debate on the appropriate wealth measurement of environmental changes (see e.g. Kriström and Heal 2001 for a review). In most of the theoretical literature, the environment is usually described as pollutant flows and stocks. In practice, however, many environmental impacts on utility occur through ecosystems' provision of ecosystem services, such as food, biodiversity, and recreational services. The purpose of this paper is to derive monetary green accounting systems where the value of changes in natural capital is derived from their production of ecosystem services. An empirical demonstration of the accounting system is made to some Swedish natural capital assets.

Following a long tradition, this paper relates wealth to society's capital asset, which includes all types of capital. Although there has been a debate among environmental economists on the proper theoretical basis for measurement of environmental wealth changes, the literature on income and wealth has converged to a common agreement on the natural capital stock as the basis for obtaining appropriate welfare measures (e.g. Heal and Kriström, 2001). This capital base reflects the future capacity of society to produce human well-being. For example, current status of forest ecosystem signals its ability to produce timber, biodiversity, and recreational services in the future. Changes in wealth can then be represented by genuine investment, which, in turn, is estimated by means of accounting prices of natural capital (e.g. Dasgupta and Mäler, 2000, 2001).

The theoretical underpinnings for measuring value of changes in natural assets by means of accounting prices is relatively well established (e.g. Arrow et al. 2002; Dasgupta and Mäler, 2001). Studies with explicit consideration of ecosystem services, i.e. the yield from natural capital, are, however, rare. The few that exist relate environmental services to pollution, and not to natural capital assets (e.g. Hamilton, 1996). The empirical estimates of accounting prices for green accounting system are mostly lacking, although there exist a few studies (Aniar, 2002; Ferreora and Vincent, 2002; Hamilton 2000; Vincent, 2001).

The paper is organised as follows. First, a simple theoretical model of capital assets and production of ecosystem services and market goods is presented, which is the basis for the derivations of indicators of wealth, sustainable development, and policy performance. Next, the suggested accounting system is applied to Swedish forests, wetlands, air quality, and marine and coastal ecosystems. The paper ends with some tentative conclusions.

2. The model

The concrete interpretation of natural capital is that it consists of a variety of ecosystems, such as lakes, wetlands, forests, agricultural landscape, and coastal water. Each of these ecosystems produces a number of outputs, so called ecosystem services. Several of these ecosystem services, e.g. fish and timber, have been known for a long time by mankind and are also subjected to market transactions. Others, like pollutant sequestration and recreational values, have received less attention as ecosystem outputs and are not traded on markets. Most ecosystems produce both market and non-marketed services. Examples are forests, which produce, among others, timber, recreational values, pollutant sequestration, and biodiversity. All these services except for timber are also produced by wetlands, and coastal waters, which also generate food.

For simplicity, all marketed goods and services are suppressed in the compounded good Q, and ecosystem services are represented by the single compounded service E. Both types of goods use natural capital, S, as a production factor. The marketed good and ecosystem service also need man-made capital, K, and emit pollutants, N, as by-products, which are treated as inputs into production of all marketed goods. For example, most goods use energy as inputs which generate emissions of carbon dioxide, sulphur dioxide, nitrogen oxide and so forth. Ecosystem production of non-marketed goods and services is also affected by pollutants. For example, carbon sequestration of forests and nutrient cleaning by wetlands depend positively on pollutant concentration in air and water respectively. The production functions are then written as $Q=Q(K,S,N)$, and $E=E(S, N)$. Except for the relation between E and N (which is discussed below), the production of market goods and ecosystem services are assumed to be increasing in all its arguments.

However, pollutants may also have a negative effect on production of ecosystem services, which is assumed to occur through their impact on *S*. In general, this effect is negative, but for certain pollutants such as nutrient, a positive impact may occur up to some pollution level. Beyond this pollutant level, bifurcation may occur and the characteristics of the ecosystem can be changed so it turns into another type of ecosystem. One example is provided by the Laholm Bay at the west of Sweden, which was heavily polluted by nitrogen during 1980s. The vegetation of large sea area bottoms vanished and species like cray fish became extinct. Such changes usually imply non-convexities and difficulties in assessing values of the ecosystems (see e.g. Mäler, 2000), but this is disregarded in the sequel.

The change in *S* during time is thus determined by its own growth, ecosystem management, and pollutant deposition, $g(S, N)$, where it is assumed that $g_S \geq 0$, and $g_N \leq 0$. Ecosystem changes can also occur from deliberate ecosystem management for the purpose of increasing harvests of marketed services. Examples are reforestation and cultivation of fish and mussels Ecosystem management is made at the cost $h(S)$, which is assumed to be increasing and convex in *S* For given initial stocks, K^o, and S^o, the change in these two types of assets is written as

$$\dot{K} = Q - C - \rho K - h(S) \tag{1}$$

$$\dot{S} = g(S, N) \tag{2}$$

where ρ is the capital depreciation rate, and C is consumption. A non-renewable resource, such as oil and coal, is represented by Hotelling type model $g(S) = -dS$, where dS is the extraction of the resource.

A key practical issue is how to define *S* and its relation to ecosystem services. For example, the production of pollutant cleaning of wetlands depends on concentration of pollutant in water inflow, (*N*), and the wetland stock as measured by its area (*S*). Carbon sequestration by forest is a function of atmospheric concentration of carbon dioxide (*N*) and biomass growth (*S*). In both these cases, $E_N \geq 0$ and $E_S \geq 0$.

Recreational and biodiversity services by forests or wetlands vary for different types of these ecosystems and also by their area coverage. Recreational values of coastal waters for bathing are probably also dependent on a water quality parameter such as sight depth. It is unclear if nutrients in one period affects sigh depth in subsequent periods. If not, recreational values depend only on N, and, since nutrients decrease sigh depth, $E_N \leq 0$. When water quality is a stock parameter, there is a negative correlation between N and S, which, in turn, reduces the coastal water's capacity to produce recreational values.

Utility in society is determined by consumption of both marketed and non-marketed goods and services, i.e. Q, and E. In addition, pollutants affect utility directly through its impact on health. For example, nitrogen dioxides may generate respiratory problems for some people. The utility function is then written as $U=U(Q,E,N)$, which is assumed to be non-decreasing in all its arguments except N.

3. Wealth, sustainable development and environmental policy

Welfare is determined by current and discounted future streams of utility, which is written as

$$W = \int_{t=0}^{\infty} U(Q, E, N) e^{-\theta t} dt \tag{3}$$

where θ is the utility discount rate. Assuming a time autonomous problem and given K^o, (3) can also be written in terms of initial stock parameters as $\widehat{W} = \widehat{W}(K, S, \varphi)$ where $S \in \Re^k$, and φ is a resource allocation mechanism which describes the institutional set up for allocating resources among goods and services, see Dasgupta and Mäler 2000 and 2001 for the definition and derivation of this mechanism.

3.1 Wealth change and national accounts

The value of a change in wealth during a period of time is now defined as

$$\frac{d\widehat{W}}{dt} = v\dot{K} + v^s \dot{S} \tag{4}$$

where

$$v = \frac{d\widehat{W}}{dK},\ v^s = \frac{d\widehat{W}}{dS}$$

Following Mäler and Dasgupta (2000) and (2001), v and v^s are interpreted as the capital resources' accounting price, which reflects the increase in welfare from a marginal change in the capital resource in question. Assuming that φ represents an optimal resource allocation mechanism, expressions for v and v^s can be derived from maximising (4), which, from the Hamiltonian,

$$H = U(C,E,N) + v(Q - C - \rho K - h(S)) + v^s g(S,N) \tag{5}$$

gives the first-order conditions as

$$U_C - v = 0 \tag{6}$$

$$U_E E_N + U_N + vQ_N + v^s g_N = 0 \tag{7}$$

$$\dot{v} = v(\theta - Q_K + \rho) \tag{8}$$

$$\dot{v}^s = v^s(\theta - g_S) - U_E E_S - v(Q_S - h_S) \tag{9}$$

According to (6), the trade off between current and future consumption occurs where marginal utility of current consumption equals the shadow price of capital. Similarly, optimal use of pollutants is determined where marginal benefit from production of marketed and non-marketed goods and services equals marginal cost. The latter includes direct disutility from emissions and indirectly through the impact on ecosystem production capacity.

The accounting price of the natural asset in period t, $v^s(t)$, is found from (6) and the first order differential (9), which gives

$$v^s(t) = \int_0^{\infty} (U_E E_S + U_C(Q_S - h_S) e^{-(\theta - g_S)(\tau - t)} d\tau \tag{10}$$

The accounting price of the natural asset in time t is thus the discounted streams of current and future net utility from marketed and non-marketed goods and services of a marginal change in *S(t)*. The future values of these services are then discounted by the utility discount rate, or pure time, discount rate plus the change in growth rate of the stock induced by the change in stock, which could be either positive or negative. When the change in the growth rate is positive, the discounting of future net utility is decreased as compared to when $g_S=0$. The latter is valid for a non-renewable resource, which thus is discounted only by the utility discount rate. For a renewable resource, g_S can be either positive or negative depending on the stock level. Usually, increased stock enhances growth at relatively low stock levels, but at larger levels a further increase in the stock may imply a reduction in growth.

Changes in wealth are partly captured by changes in net domestic product in utility terms, which, in turn, can be represented by the Hamiltonian in (5) under highly restrictive conditions (see e.g. Heal and Kriström, 2001, for a discussion of NDP and intertemporal welfare as expressed by the Hamiltonian). From (5) we have that all changes in market goods and services are captured by NDP expressed in utility terms. Corrections should then be made with regard to non-marketed ecosystem services. The corrected net domestic product, NDP^C, is then written as

$$NDP^C(t) = NDP(t) + \mu\left(U(N,E) + \int_0^\infty U_E E_S e^{-(\theta - g_S)(\tau - t)} dt \dot{S}\right) \tag{11}$$

where $\mu=1/U_C$. According to (11), the correction includes current utility from pollutants and ecosystem services, and change in future utility from ecosystem services caused by the period's change in the stock of natural capital.

3.2 Sustainable development

Since the publication of the Brundtland report (World Commission, 1987), there have been many attempts to define sustainability both in theoretical and operational terms. One may, for example, require single capital goods to be used in a sustainable way, which would require that the capital stocks are non-decreasing over time. However, this would be impossible even without anthropogenic influence since ecosystems are subjected to evolutionary processes, which may be destructive. Another approach, which is followed in this paper, is to require welfare to be non-declining over time, which, from (5), implies that

$$\frac{d\widehat{W}}{dt} = v\dot{K} + v^{s}\dot{S} \quad \geq 0 \qquad (12)$$

Sustainability thus implies that the total value of changes in the capital stocks is non-declining. The value of a stock unit is then determined by its accounting price, which, in turn reflects discounted current and future streams of net utility from a marginal change in the capital stock. The accounting price thus reflects the production potential of the capital base. When this production potential declines, it can not provide the same welfare for future as for current generation. Current generation's use of the resources is then unsustainable.

3.3 Environmental policy

When there exists non-marketed goods and service, the non-regulated market price of goods and services are likely to be incorrect. This can be seen from conditions (6)-(9) and the associated derivations of optimal natural asset accounting prices. At an unregulated market, the welfare impacts of non-traded ecosystem services will not be included in the prices. This means that the price of pollutants as inputs is too low, and the price of capital assets is also likely to be low since some ecosystem services have zero prices. For a policy, which internalises all impacts, the optimal emission and natural asset taxes, ι^{N} and ι^{S} respectively, can be derived from (6)-(9), which gives

$$\iota^{N}(t) = \mu(U_{E}E_{N} + U_{N} + \int U_{E}E_{S}g_{N}e^{-(\theta-g_{S})(\tau-t)}d\tau) \qquad (13)$$

$$-\iota^{S}(t) = \mu\int U_{E}E_{S}e^{-(\theta-g_{s})(\tau-t)}d\tau \qquad (14)$$

The optimal emission tax corresponds to the negative impact on consumption and ecosystem provision of services, and the natural assets is paid its value of marginal product with respect to non-marketed ecosystem services.

The derivation of these taxes relies on the assumption that the markets for goods, services, capital, and natural assets are competitive. Since it is quite likely that this will not hold for all markets, further corrections need to be made which reflect the impact of market power (see e.g. Baumol and Oates, 1988; Aronsson, 1998). For a more general analysis of the role of institutional design for accounting prices of capital assets, see Dasgupta and Mäler (2000) and (2001).

4. An empirical demonstration of a Swedish green accounting system

In the simple theoretical model above only pollutant emissions are assumed to produce environmental impacts. However, non-market environmental impacts occur also through other activities, such as certain types of land uses. Further, an economy is equipped with several natural capital assets, where each asset produces a number of marketed and non-marketed ecosystem services. Theoretically, such extension can be made to the theoretical model presented in Chapter 3. In this chapter, simple examples of environmental impacts are presented for both pollution activities and land uses, and also for ecosystem producing several ecosystem services.

Ideally, empirical dynamic models are available with numerical presentations of utility function, production functions for goods and ecosystem services, and equations of motions of the capital assets. Unfortunately, these data requirements are not met in a satisfactory way for Swedish natural capital assets. Therefore, the calculations made in this chapter shall be regarded only as examples of how wealth measurements of natural capital assets can be calculated and used.

A first practical difficulty is to choose relevant natural capital stocks and their measurements. Different types of classifications are suggested by UN (2002). The exemplification in this paper follows UN ecosystem classification, which distinguishes between two broad classes: aquatic and terrestrial ecosystems. Air quality is also classified as an ecosystem. It seems, however, meaningless to calculate the value of air quality as an input into production of ecosystem services since it is essential for all life on earth. On the other hand, value of changes in air quality shall be calculated, which is demonstrated in this chapter.

In the sequel, calculations are made for forests, agricultural landscape, coastal and marine waters, wetlands and air. Except for air, recreational values are calculated for all ecosystems. Carbon dioxide sequestration values are estimated for forests, and nitrogen sink values for wetlands. Monetary estimates are made for health impacts of changes in air quality. The exemplified accounting system follows the theoretical presentation in Chapter 3, which then includes wealth and sustainability measurements, corrections of the Swedish NDP, and derivations of environmental taxes. The calculations are made for 1999, and a common assumption for estimation of natural capital assets' production value of ecosystem services is that the real discount rate is 3 per cent.

Estimates of monetary values of ecosystem services underlie all calculations, and are presented in the appendix. Pollutant sequestration values of both forests and wetlands are calculated as cost savings from avoided cleaning from higher cost measures. For carbon sequestration by forests, a general equilibrium model is used to calculate society's cost for reducing emissions corresponding to the carbon sequestration of Swedish forests. Nitrogen sink values for Swedish wetlands are obtained from a survey of Swedish studies. Benefit transfers from other studies are also used for obtaining recreational values.

Choices and quantification of a relevant stock variables for the chosen ecosystems is not a self evident matter. For forests, two stock variables are used. Carbon sequestration depends, among other things, on biomass growth, which therefore is applied as a stock variable for this service, while area of forests is a more relevant stock variable for recreational values. Land area is also used as stock variable for wetlands and agricultural landscape. However, recreational values of coastal and marine waters are determined by sight depth and oxygen content respectively, which then are applied as stock variables for these ecosystems.

The production of services from all included ecosystems varies with respect to time and spatial allocation. Both allocation and supplied quantity of services change over the ecosystems' succession stages. In an elongated country like Sweden, there are also relatively large climatic variations among different parts of the country, which affect the supply of ecosystem services. However, due to lack of data, a simple a spatial division of only wetlands have been possible to make, which is divided into a northern and southern part.

Given all assumptions, which are more fully elaborated in the appendix, the calculated consumption and investment values are as presented in Table 1.

Table 1: Value of environmental services and natural capital investment in 1999, billions of SEK

Natural capital	*Pollutant cleaning*		*Recreation*		*Health*	*Total*
	Cons	*Invest*	*Cons*	*Invest*		
Forest	8.8-16.2	2.7-5.4	18	-14.3		15.2-25.3
Agr. landscape			7.3	-1.6		5.7
Coast. and mar. water			1.0	-7.4		-6.4
Wetlands	0.4-10.7	0.01-0.7	5.5-35.1	0.1-0.3		6.0-46.8
Air					-8	-8
Total	*9.2-26.9*	*2.71-6.3*	*31.8-61.4*	*-23.1*	*-8*	*12.6-63.4*

Source: See Appendix

Bot the consumptive and investment values show large variations depending on assumptions, mainly with respect to the relation between stock change and associated impacts on production of ecosystem services. Recreational values seem, however, to account for the major art of both total consumptive values and net investment. Due to the decreases in forest areas and to degradation of coastal and marine ecosystems, genuine natural capital investment (gross investment minus degradation) is negative and varies between –16.8 to –20.4 billions of SEK.

4.1 Wealth impacts and sustainable use of Swedish natural capital

According to (4) a measurement of wealth is obtained by multiplying the accounting price of a stock with its change during the period of time under study. This is obtained by summing the investment values in Table 1. The net wealth change varies between –16.8 and 20.4 billions of SEK and is thus negative. The negative investment in recreational values of forests, agricultural landscape, and coastal and marine ecosystems exceeds the positive investment in pollutant cleaning values.

The sustainability criterion requires a non-negative change in total welfare from changes in natural capital during any period. Then, the production potential of the capital base is increasing over the studied period, and vice versa. A sustainable change in wealth may thus include reductions in some capital stocks if this is compensated for by an increase in other resource stocks. When considering only natural capital, we allow for compensating increases for some declining assets only among the natural capital. The figures presented for 1999 in Table 1, is thus only a partial estimate of sustainable change in wealth during 1999.

According to the examples presented in Table 1, forests, agricultural landscape, coastal and marine ecosystems show a declining production potential while the production capacity of forests and wetlands are increasing. The value of the decrease exceeds that of the increase, and the total use of natural capital in 1999 is thus non-sustainable. It should be kept in mind, however, that only a fraction of ecosystems and their services are included in the calculations. Furthermore, strong assumptions underlie the calculations that are made and calculated value may be an underestimate for forests and agricultural landscape since only part of the value of the replaced areas are included. Conversion of land into wetlands, and from forests into agricultural land and vice versa are accounted for, but not other type of land conversions. On the other hand, overestimates are made due to the lack of ecosystem management costs.

4.2 Environmental assets changes and national accounts

The impact on NDP from natural capital assets occur through their provision of both marketed and non-marketed consumption of ecosystem services as well as investment in natural capital. Assuming a linear relation between utility and consumption, the value of non-marketed ecosystem services for consumption is measured by their monetary measurement of marginal utilities, which are used for deriving accounting prices of natural capital.

For an open economy like Sweden, there is a need for making a distinction between national income and national product. The concept NDP refers to national product, and as such it measures the sum of value added all production in Sweden regardless of who enjoys the benefits from the produced goods and services. National income, on the other hand, is the sum of all incomes obtained from Swedish activities regardless of where these are located. In this paper, corrections are made of both NDP and NNI.

The NDP concept includes all products and non-marketed ecosystem services as consumption and/or investment. Pollutants as inputs into production are then regarded partly as imports and partly as home made. The transboundary emissions to other countries use their assimilation capacity, which is regarded as imports. The price of pollutants corresponds to the marginal damage in these countries. However, the estimation of Swedish pollutant impact on ecosystem service production and associated effects on utility would require similar type of data for these countries as for Sweden. This is not available, and the correction of NDP presented in Table 3 is therefore made by assuming that the value per pollutant is the same for export/imports as for pollutant deposition/emission in Sweden.

Except for marine fishery, it is assumed that all recreational values are caused and consumed by Swedish citizens. The national product versus income concept then affects mainly carbon dioxides and nitrogen dioxides. In NDP the entire value of forest as carbon sink is included, but only export values of nitrogen dioxides. Total deposition of nitrogen dioxides on the Swedish territory is 454 000 tons and total Swedish emission are 267 thousand tons. The Swedish transboundary pollution corresponds to 71 per cent of total emissions. Assuming that the value per ton is the same for Swedish deposition and emissions, total value of Swedish nitrogen dioxide emissions is –4.7 billion, of which the import value is –3.3. Since the value of exported nitrogen dioxides (deposition on Swedish territory) is –6.6, the net impact is a net inflow of –3.3. Similarly, approximately 40 per cent of the fish from non-Swedish waters, and 0.2 billions of SEK from recreational fishery is then regarded as incomes from other countries.

When constructing NDI, all negative impacts on Swedish citizens from nitrogen dioxides are included, but only a small fraction of the value of Swedish forest as carbon sink. Assuming that the latter Swedish value corresponds to the country's share of total global emission, only 0.003 is the carbon sink value is included. Since the value of carbon sink is relatively high, the correction of NDP is larger than that of NDI, see Table 2.

Table 2: Corrections of Swedish NDP and NNI in 1999

	Billions of SEK	% of NDP[1]	% of NNI[1]	% of net investment[1]
NDP:				
Consumption	33.1-80.4	1.8-4.5		
Investment	-17-(-20.4)	-0.9-(1.1)		-20.1-(-24.9)
Total	16.1-60.0	0.9-3.3		
NNI:				
Values to other countries	8.8-16.2			
Values from other countries	-3.1			
Total	4.2-40.7		0.3-2.3	

[1] In 1999, NDP amounts to 1797 billions of SEK, NDI to 1776, and net investment to 82 (SCB, 2002)

According to this simple example, there is thus a net increase in NDP, which varies between 0.9 and 3.3 per cent of NDP. There is also a net increase in NNI, which is smaller than that in NDP. It is also interesting to note that net investment could be reduced by approximately ¼ when considering investment in natural capital.

4.3 Environmental taxes and compensation payments

Remember from the introduction that one purpose of the wealth measurement of environmental assets is to serve as a decision tool for environmental policy. This can be done in at least three ways: as indicator for taking actions, and as basis for designing and evaluating environmental policy. The first way is to use the suggested wealth and sustainability measurement as a signal for need of acting against negative investment in some or several natural capital assets. For example, reverting the negative investment in coastal and marine water systems.

When a decision of actions is taken and environmental targets are formulated, it remains to determine how measures and environmental policy instruments leading to the action targets should be designed. This is far from a trivial issue, and it has been analysed in environmental economics for almost 100 years. In principle, the design of efficient policies follows two steps. The first is to identify the technological domain of measures, and estimate associated cost. For example, investigation of all possible measures reducing the pollutant load to a recipient, such as implementation of catalysts in cars and/or changing land uses for creation of pollutant sinks. Ideally, cost effectiveness analyses have been made which is also used for deriving emission charges for various sources. The cost effective charges should then vary with respect to location of the source and timing of emission (see e.g. Baumol, 1988).

The derived unit values of nitrogen and ecosystems presented in 4.2 can be used for determining the charge for deposition of nitrogen at the coastal and marine waters and also compensation payments for land uses. As shown by eqs. (13)-(14), the efficient pollution charge is determined by both flow and investment impacts. If nitrogen oxides pay its marginal damage cost on health, this would imply a deposition charge of approximately SEK 18/kg nitrogen dioxide deposition on the Swedish territory, regardless from where the emission originates. Since Swedish emission account for 17 per cent of total deposition on its own territory, total payment of the damage in Sweden requires international agreement on nitrogen dioxide charges. Further, nitrogen dioxide charges should also be paid according to the disinvestment in marine and coastal ecosystems. When considering only the negative investment in Swedish coastal and marine ecosystems, and assuming that all nitrogen load to the Baltic Sea countries contributes equally much, the nitrogen deposition charge should then correspond to SEK 9.8/kg nitrogen deposition. Total load is then calculated as 800 thousand tons of nitrogen (Gren and Wulff, 2003), and the damage cost is 7.9 billions of SEK (see table A5).

The nitrogen emission charges from different sources are then determined by their impacts on these two recipients. For example, the charge on Swedish emissions is calculated as the share of nitrogen dioxides that is deposited on the Swedish territory (0.29) times the deposition charge of SEK 18/kg nitrogen dioxides on the Swedish terrestrial territory, plus the nitrogen deposition share on the Baltic Sea (0.2) times the Baltic Sea deposition charge of SEK 9.8/kg nitrogen. This gives a total emission charge of approximately SEK 5.7/kg nitrogen dioxides. Emission charges for other countries with NO_2 deposition on the Swedish territory and/or the Baltic Sea should be calculated in the same way, but the charge levels may differ due to differences in impacts.

Efficient compensation payments are derived from (14), and they are determined by the change in the stock variable and impacts on utility from the associated change in the supply of ecosystem services. Calculations of these values are shown in the appendix, which gives the compensation payments presented in Table 3.

Table 3: Compensation payment and charges on Swedish natural capital and nitrogen emission

	Compensation payment	Charge
Nitrogen emission		SEK 5.7/kg NO_2
Forest	SEK4415/ha + SEK 293/m^3 biomass growth	
Agricultural landscape	SEK 2650/ha	
Wetlands	SEK 9273/ha in north SEK 17025/ha in south	

Source: See Appendix

Forest managers are given two types of payments, per ha for recreational values and per m^3 for carbon sequestration. These incomes are of the same magnitude as the forest sector's contribution to NDP (Statistic Sweden, 2002). Wetland compensation payments also include recreational values and pollutant sink values. A spatial division is made between southern and northern Sweden due to differences in environmental damage from nitrogen, and, hence, different nitrogen sink values. The compensation payments for agricultural landscape cover recreational values and correspond to 1/3 of the sector's value added.

5. Summary and conclusions

The purpose of this paper has been to investigate conditions for an appropriate welfare indicator of changes in ecosystems, which aim at evaluating: *i*) welfare changes, *ii*) sustainable resource use, *iii*) correction of net national product, and *iv*) environmental policy. The analytical model used for deriving the indicator follows the literature with one exception, the explicit consideration of ecosystem services. Natural capital is commonly treated as an externality from production of market goods and services, but here natural capital is instead treated as an input into production of ecosystem services. This difference has a minor impact on the theoretical models, but is has several significant practical implications.

The theoretical analysis shows that all purposes of the indicator can be fulfilled by means of accounting prices of natural assets. This price gives a unit value of an asset's value of current and future marketed and non-marketed ecosystem services. A welfare improving change in a natural asset, or group of assets, occurs when the stock change as evaluated at the accounting price is positive. As has been shown Mäler and Dasgupta (2001), accounting prices can also be used for deriving welfare indicator of changes in natural assets also under conditions of non-optimal prices. This is a conclusion of much practical relevance, since the indicators are derived from existing, probably non-optimal, market prices and estimated values of non-marketed ecosystem services.

An indicator of sustainable use of natural capital assets is obtained from the summation of values of change in all assets. Values of some change may be negative and others can be positive. A sustainable use of all assets occurs when the sum of all values of changes is positive. Then, the capacity of future production of marketed and non-marketed goods (in constant prices) is non-decreasing.

Correction of NDP or NNI with regard to non-marketed ecosystem services is made with regard to both consumption and investment values. The latter is captured by the welfare and sustainability measure of future values of changes in natural capital assets. The consumptive values reflect utility from current use of natural capital as inputs into production of non-marketed ecosystem services. The same type of information is thus needed for estimating both consumption and investment values of natural capital.

The fulfilment of the fourth purpose of the paper, i.e. to quantify an indicator useful for environmental policy making, may need more data than is required for the achievement of the other three purposes. Since policies often are directed towards production sectors in the economy, information is required on the relation between the sectors' activities and their environmental impact. If these impacts occur through pollutant emission, a difficulty emerges from the determination of relations between emission sources and their impacts on various ecosystems. Further, for small countries like Sweden, most of the pollutant deposition within the national territory is caused by foreign emission sources, so the space of action may be limited.

Although theoretical, the analysis thus generates some conclusions of practical relevance. First, it is not the pollutants as such that is subjected to monetary valuation, but instead their environmental impacts. Second, the monetary valuation of these impacts is then concerned with the valuation of ecosystem services, which can be produced by any ecosystem. Thus, one should not directly value ecosystems as such, but instead ecosystem services independent on how they are produced. Ecosystem valuation can then be done if there are numerical estimates of ecosystem production functions. Third, since the focus is changed from emission sources to ecosystems, other classes of data is required for the welfare indicator, which include information on changes in the natural assets, relation between stock size and direct and indirect provision of ecosystem services, and monetary valuation of each ecosystem service.

The suggested method for welfare measure, sustainability indicator of assets changes, correction of NDP and NNI, and derivation of efficient environmental charges were demonstrated for a few Swedish ecosystem services: recreational values, pollutant sink values, and health impacts from nitrogen dioxide pollution. Recreational values are provided by forest, wetlands, agricultural landscape, marine and coastal waters. Forests and wetland also act as pollutant sink by sequestration of carbon and nitrogen respectively. Given all caveats associated with finding data, the net result points at an unsustainable use of the ecosystems under the year of study, 1999. On the other hand, the correction of NDP implies an increase, which varies between 0.9 and 3.3 per cent depending on assumptions with respect to the ecosystems' production of ecosystem services. This empirical result differs from other corrections of Swedish NDP, which instead result in a decline of NDP (Ahlroth, 2000). This is due to the difference in focus, which in Ahlroth and many other empirical studies is on pollutant emission, which enters directly into the utility function. The negative impact on utility from pollutants is then subtracted from conventional NDP. This paper also allows for a negative impact on utility from pollutant, but the main focus is on ecosystems as inputs in production of ecosystem services. This can generate a positive utility from production of non-marketed ecosystem services, which increases conventional NDP. Pollutants can reduce the ecosystems' production capacity, but ecosystem service production must not be negative. Negative impacts on conventional NDP occur only from direct disutility of pollution and disinvestment in natural capital.

The empirical demonstration clearly pointed at the difficulties in finding appropriate data, in particular on the relations between natural asset status and production of ecosystem services. These relations are characterised by spatial and dynamic heterogeneity, which, however, is true also for many marketed goods and services. Although there is currently much less information on the shape of production function for ecosystem services than for marketed goods, this lack of data could in principle be reduced by use of statistical methods. Such methods have been applied during decades for estimating market good production functions. This implies a focus on ecosystem services as outputs with various ecosystems as inputs. Most of the valuation literature so far has the reverse focus, i.e. on the valuation of ecosystems. Although results from such studies have been used for the empirical demonstration in this paper, it is difficult to obtain information on substitution or complementary impacts among ecosystems in producing similar types of outputs, such as recreational values.

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Appendix: Calculations of values of environmental services for differecnt Swedish natural capital assets

Forests

Forests provide a variety of values for society such as timber, biodiversity, carbon uptake etc. This was recognised in early 1990's as documented by Hultkrantz (1992). In this simple example, only the values of forest as carbon sink and for recreational purposes are calculated. The sink capacity for a given area is mainly determined by the forest growth, which, in turn, depends on a number of factors such as climate, type of forest trees, soil, etc. Depending on forest management – harvesting and plantation – sink capacity can be increasing, decreasing or unchanged over time. However, assuming that we have obtained an appropriate estimate of change in forest area and associated impact on carbon sink capacity, the value of this depends on costs of alternative measures for reducing emissions of green house gases, and also of the target for emission reductions.

In 1999, the total CO_2 uptake of Swedish forests was approximately 27 millions ton, which corresponds to almost half of total carbon dioxide emission in the same year (SOU, 2002). The uptake is also more than Swedish obligations of a 4 per cent reduction of the 1990 emissions level, which corresponded to a 2.7 millions ton reduction of CO_2 emissions in 1999. The value of this sink capacity depends on alternative ways of reducing carbon emission. If the only alternative is to reduce Swedish emission from energy combustion in all sector, the marginal impact on GDP would be SEK 1.2/kg, and the total flow value 16 200 millions of SEK (Östblom, 2002). However, the cost of Swedish emission reductions would be reduced if emission trading could be made with outside sources. The marginal impact at the same level of emissions would then be 0.65 SEK/kg, which corresponds to 8.8 billions of SEK.

Carbon sequestration is determined by several factors, such as forest biomass growth, forest management, and atmospheric concentration of carbon dioxide. In order to calculate the value of investment we use biomass growth as a stock variable, which amounts to approximately 30 millions of m^3, or 10 per cent of standing volume (SCB, 2002). Net increase in biomass growth in 1999 is estimated to 1 per cent [2], and the discounted carbon sink value of this increase would then be either 2.7 or 5.4 billions of SEK depending on policy arena.

Recreational values include a number of activities, such as hunting, picking of mushrooms and berries, sporting and walking. A survey of studies with estimates of such recreational values from Swedish forests is made by Jämttjärn (1996). The average value per person and

[2] This is a 5 year average increase in the period 1996-2000 and 1997-2001 (SLU, 2003, web-site)

year amounts to SEK 3000, which corresponds to SEK 4415/ha forest. The variation is, however, large among studies. The average total recreational value amounts to 18 billions of SEK, which corresponds to almost half of the forest sector's contribution to GDP in 1999.

In order to find a recreational investment value for forests it seems reasonable to use another stock variable than for carbon sequestration. Instead, change in area of forest land is applied as a measure of stock changes. The accounting price of forest is then found by assuming that the value of forest is the same irrespective of regional location. However, most of the reviewed valuation studies in Jämttjärn (1996) are made for areas with relatively high visiting frequency, and a relevant estimate of stock change would require investigations of regional changes in forests. Since this is not available, it is simply assumed that half of the estimated value per ha, i.e. SK 2208/ha, corresponds to the accounting price in 1999. The forest area change during this yeas is –194 000 ha, which implies an investment of –14.3 billions of SEK.

In summary, the calculated carbon dioxide sink value and recreational values from Swedish forests are divided among consumption and investment components as shown in Table A1.

Table A1: Carbon sink and recreational values from Swedish forests, billions of SEK 1999.

Ecosystem services	*Consumption*	*Investment:* *Acc. price*	*Stock change*	*Total*
Carbon dioxide sink	8.8-16.2	9-18/$10^6 m^3$ growth	0.3 $10^6 m^3$	11.5-21.6
Recreation	18	0.736 10^{-4}/ha	-194 000	3.7
Total	26.8			15.2-25.3

In 1999, the total GDP contribution from the agricultural and forest sectors amounts 10 41.1 billions of SEK and gross investment to 1.7 billions of SEK (Statistics Sweden, 2002). The agricultural sector accounts for 18.2 billions of SEK and the forest sector for 22.9 (Agricultural Statistics, 2002a). The simple estimate presented in Table A1 thus seems to be significant in relation to the sector's GDP contribution of market goods.

Agricultural landscape

The agricultural landscape provides a number of non-marketed ecosystem services from its mix of various land uses for grazing, cereal production etc. Marked transitions from one land type to another, such as ditches, are usually rich of biodiversity. Further traditionally managed agricultural landscapes provide scenic beauty, which can generate recreational values.

Two studies have estimated the value of Swedish agricultural landscape (Drake, 1994; Hasund, 1998). Drake estimates the willingness to pay for the agricultural landscape in general, and Hasund focus on the valuation of landscape elements. Common to both studies is the positive willingness to pay for landscape preservation. According to Drake, there is a large difference in estimated value per ha depending on type of agricultural landscape, from approximately SEK 500/ha to 4800. Assuming an average value of SEK 2650 and an area of agricultural land of 280 000 ha (Agricultural Statistics, 2002b), gives a total recreational value of 7.3 billions of SEK.

The investment value of agricultural landscape is found by multiplying the accounting price of agricultural landscape with regard to only its recreational values. Further, due to lack of information on spatial allocation of landscape changes, it is simply assumed that the annual value corresponds to half of the average value of agricultural landscape, i.e. SEK 1325 /ha, which gives an accounting price of billion 0.442×10^{-4}/ha. Multiplying this accounting pirce with the change in the area of arable land during 1999, -37000 (Agricultural Statistics, 2002b), the recreational investment value amounts to –1.6 billions of SEK. When comparing the flow and investment components of recreational values with the agricultural sector's total contribution to GDP of 18.2 billions of SEK it is noted that the recreational values correspond to approximately 1/3 of the marketed value added.

Coastal and marine ecosystems

Coastal and marine ecosystems produce different ecosystem services such as food (fish), recreational values from bathing, sailing, bird watching, biodiversity, and act as sinks for down streams pollution. In the following, only the value of recreation is calculated for coastal ecosystems.

Recreation values from Swedish coastal waters have been estimated by Sandström (1996). This was done by means of the travel cost method to recreational sites mainly for bathing along the south Swedish coast, and sight depth was used as a index of water bathing quality. The estimated result range between SEK 0.2 or 0.5 billions per year depending on choice of parameter for sight depth variable. Here, the average of SEK 0.4 billions per year in 1999 years prices is used.

In order to estimate a stock impact, an appropriate indicator of the coastal ecosystems' production capacity is needed. On such candidate is oxygen content, which indicates the impacts on water quality, such as sigh depth, from occurrences of algae and other oxygen consuming species. During 1999, the oxygen content in Laholm Bay decreased by approximately 12 percent, from 3.4 ml/l to 3.0 ml/l (SEPAa, 2003). Assuming that the recreational value of bathing is linearly related to oxygen content and that Laholm Bay is representative for the Swedish coasts, the value in year 1999 is billion 3.92 ml/l. A stock change of –0.4 then implies an investment of –1.6 billions of SEK.

Similar to coastal ecosystem, marine ecosystems provide a multitude of various services to society. Through centuries they have been used as resources for food and transports. Degradation still occurs from oil spills and over fishing. Further, eutrophication damages from nutrient loads have occurred since late 50's. In this simple example, we consider only recreational fishery from marine ecosystems.

Recreational fishery is defined as fishing performed with rod, reel and similar hand equipment and/or nets, creels and similar with an explicit recreational purpose where the fish landed is not sold. The total harvest from recreational fishery was, in 1999, 58 200 tons (Vredin-Johansson, 2002). Out of this, 34500 tons was caught in Swedish waters, and the rest in Baltic Proper, a basin in the Baltic Sea. A number of fish species are harvested, such as eel, fishes of prey, cod, salmon, flatfish, mackerel, herring and lobster. The single most important fish specie is cod, which account for approximately 15 per cent of total catch (Vredin-Johanson, 2002). In 1999, the total gross value of recreational fishery amounts to 621 millions of SEK, which corresponds to 55 per cent of the production value from the fishery sector.

A stock change in recreational fishery during 1999 is obtained by relating recreational fishery to oxygen content in the Baltic Proper. This decreased during 1999 due to the content of hydrogen sulphide, which increased from –1.8 ml/l to –2.3 (SEPA, 2002b). The accounting price at the –1.8 level is billion 11.5 /ml/l, and the investment during 1999 is then 5.8 billions of SEK.

In summary, the consumption and investment values of recreational values from coastal and marine fisher are shown summarised as in Table A2

Table A2: Recreational values of coastal and marine ecosystems, billions of SEK in 1999

	Consumption	*Investment:* *Acc.price*	*Stock price*	*Total*
Coastal water	0.4	3.92 ml/l oxygen in Kattegat	-0.4	-1.2
Marine water	0.6	11.5 ml/l oxygen in Baltic Proper	-0.5	-5.2
Total	1.0			-6.4

Wetlands

At a global scale, Sweden has one of the largest proportions of wetlands within its territory. Approximately 1/5 of total land is covered by wetlands. According to the Swedish EPA, wetland " …. is such land where water is, during a large portion of the year, just below, in line with, or just above the ground". Wetlands are among the most biodiversity rich ecosystems, and provide therefore a variety of ecosystem services (Mitsch and Gosselink, 1998). Examples of ecosystem services are recreational values, food, pollutant cleaning, and biodiversity. Since 1970s a number of wetland valuation studies have been made in different parts of the world. Recreational values and pollutant cleaning have mostly been valued. This is also the case for the six different studies valuing Swedish wetlands in monetary terms (Svensson, 2003). Subsequent calculation of the monetary value of changes in Swedish wetlands is based on these studies and also on Svensson (2003) for estimation of changes in wetland capital and impacts on ecosystem services.

During the period 1998-2002 the area of wetlands increased by 1400 ha/year due to the subsidy payment by the Swedish Board of Agriculture and by municipalities. There is also an ongoing degradation of wetlands from peat extraction and forest drainage, which varies between 670 and 933 ha per year. However, the provision of ecosystem services and, hence, their valuation is highly dependent on the location of the wetland site. Therefore, it might be misleading to simply value the net increase of 467 and 730 ha per year. The main increase of wetlands has occurred in the southern part of Sweden while the decrease has taken place in the north.

The six Swedish valuation studies have been made for wetlands in south Sweden. Four of these have estimated the value of wetland nitrogen abatement, which depends on the abatement capacity and costs of alternative abatement measures. Since nitrogen loads affect eutrophication in coastal waters of southern Sweden, it is simply assumed that this service is attributable only to the enlargement of wetlands in south Sweden of 1400 ha, and not to the decline which is assumed to have occurred only in the north. Depending on wetland abatement capacity, alternative measures and abatement targets the abatement value range between SEK 500 and 75000 per ha and year. The lower value assumes cost savings of SEK 5/kg N abatement with a capacity of 100 kg abatement whereas the highest value assumes SEK 150/kg N abatement and an abatement capacity of 500 kg. Two Swedish wetland studies investigated other values of wetlands, which range between 2500 and16 051 per ha and year.

The consumption value is obtained by multiplying the area of wetlands, 1471000 ha in North Sweden and 714 000 in south, with the unit values. The estimated consumption and investment values are then as presented in Table A3.

Table A3: Nitrogen cleaning and recreational values of Swedish wetlands, billions of SEK in 1999.

	Consumption	*Investment* *Acc.price*	*Stock change*	*Total*
Nitrogen abatement	0.4-10.7	0.017×10^{-4} – 0.0005/ha	1 400	0.4 –11.4
Recreation	5.5 – 35.1	0.083×10^{-4} – 0.000535/ ha	599	5.6 – 35.4
Total	5.9 – 45.8			6.0 – 46.8

Air

Air acts as a media of transport of many pollutants. As such, it affects the functioning of most ecosystems, and thereby environmental services such as yield on arable land and timber production. These indirect impacts are captured through an appropriate calculation of the value of changes in these assets, where air quality is more a cause than a source of asset changes. Therefore, monetary estimates of changes in air quality in this section capture only the direct impacts on utility

The direct impacts of air quality changes are those on human health. There are several air pollutants that can affect health. In Huhtala and Samakovlis (2003) an example of how to value health impacts from nitrogen dioxide emissions in urban areas is presented. The valuation of health effects are divided into two components: disutility from air pollution and productivity impacts.

Calculations are first related to a unit increase in the concentration of nitrogen dioxide and then translated to the yearly deposition in Sweden. According to Samakovlis et al (2003), a unit (μ/m^3) increase in the monthly average of nitrogen dioxide results in a 3 percent increase in respiratory-related restricted activity days (RRADs) in Sweden. This unit increase results in 885 727 extra RRADs per year in Sweden. Of these RRADs, 28 percent are so called minor RRADs lasting one day, and 62 percent are major RRADs. Transfer of results from international contingent valuation studies of willingness to pay to avoid the disutility for this amount of minor and major RRADs amount to SEK 498 millions. It is assumed that these estimates include only experienced discomfort and not labour productivity impacts. These are instead assumed to be loss of one work day for one major RRAD and 10 % of a work day for minor RRAD. This gives a productivity loss of SEK 312 millions.

In total, the disutility and productivity impact related to a unit increase in NO_2 amounts to SEK 809 millions. Assuming a linear relationship between deposition and concentration, this corresponds to a marginal damage value of SEK 29 per kilogram NO_2, or a total flow effect of SEK 13 billion for the 1999 deposition of NO_2 (454 453 ton) in Sweden.

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